US Army Corps of Engineers
Walla Walla District

Lower Snake River Water Quality And Post-Drawdown Temperature And Biological Productivity Modeling Study

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FOREWORD

This document is the product of the US Army Corps of Engineers' (Corps) efforts to involve the region in the development of the *Lower Snake River Juvenile Salmon Migration Feasibility Report/Environmental Impact Statement (FR/EIS)*. The Corps has reached out to regional stakeholders (states, tribes, Federal agencies, organizations, and individuals) for the input and development of various work products. This and various other products associated with the development of the EIS were authored and developed by these regional stakeholders and contractors. Although the Corps has acquired this document as part of its EIS process, the opinions and/or findings expressed herein do not necessarily reflect the official policy or position of the Corps. The Corps will review and incorporate information from these products into the analysis and development of the Draft FR/EIS.

In addition, this analysis is only one part of the overall EIS. For a true analysis of the implications of any of the study alternatives, costs and benefits of all the components of the analysis must be considered, but without any individual component taken out of context.

This document is being released for **information purposes only**. The Corps will not be responding to comments at this time. The formal comment period will coincide with the release of the Draft FR/EIS.

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1.0 - INTRODUCTION

This report has been developed to partially fulfill the requirements of Delivery Orders 007 and 008 of contract number DAC2W68-96-D-0003 between the U.S. Army Corps of Engineers and Normandeau Associates. The project is a part of a larger effort to evaluate potential changes in the biology and water chemistry of the Lower Snake River associated with the proposed removal of the four dams operated by the Corps on the Lower Snake. The Lower Snake River is a biologically productive system. This project involved monitoring the limnology and primary productivity of the Lower Snake River system from the Idaho/Washington border to the confluence of the Snake and the Columbia Rivers. The monitoring program was designed to encompass the 1997 growing season. These data are now being used to support a water quality modeling effort. Monitoring results, coupled with historic data are being compared to results from a water quality model (WQRRS) that will make projections of the biological productivity of the system under potential future free flowing conditions.

The monitoring program included measurement of physical water quality parameters such as transparency, conductivity and temperature, water chemistry including concentrations of nutrients, dissolved oxygen, cations and anions, pH, alkalinity, chlorophyll a and measures of the biological community including BOD, primary productivity estimates for both phytoplankton and attached benthic algae, and zooplankton biomass. The program included eight sampling events conducted under the range of water flows experienced during the 1997 growing season.

The model chosen for the simulation of productivity in the Lower Snake was WQRRS. This model has been extensively tested, contains a relatively complete set of ecological and water quality variables, allows the simulation of temperature by the heat budget method and contains the hydrodynamic routines to allow simulation of both river and reservoir segments within the same integrated model. This allows the projection of free-flowing conditions without the use of a different hydrodynamic model, as other models would require. The major state variables to be simulated during this effort include temperature, dissolved nutrients, phytoplankton, benthic algae, dissolved oxygen, zooplankton, benthic macroinvertebrates and three functional groups of fish. The model will be calibrated and verified using water quality data from pre-dam years. Calibration and verification data will be selected from years that spanned a range of hydrometeorological conditions (*e.g.*, normal, dry, wet) if possible. Model predictions will be compared to field data and predictions of state variables generated by other appropriate empirical approaches. Simulations of the free-flowing condition will include simulation under normal, dry and wet years.

This report is organized in three major sections. Section 3.0 encompasses the field program for 1997 including a discussion of all measures of water quality, periphyton primary productivity and phytoplankton primary productivity. Section 4.0 includes a discussion of the development of an empirical index that may be used to predict primary productivity for the Lower Snake River system. Section 5.0, to be included in future drafts, will include a discussion of the modeling effort for the system.

2.0 - BACKGROUND ON PROJECT

The Snake River is a major tributary river to the Columbia River traveling 1000 miles and draining a basin of 109,000 square miles encompassing parts of Montana, Wyoming, Idaho, Oregon and Washington. The Lower Snake River, which is the focus of this study encompasses the portion of the Snake from a point just upstream of it's confluence with the Clearwater River near Lewiston, Idaho downstream to it's mouth at the Columbia River near Pasco, Washington. The Lower Snake system consists of four impoundments formed by four hydropower dams over roughly 140 river miles. The four hydropower and navigation dams from upstream to downstream are Lower Granite, Little Goose, Lower Monumental and Ice Harbor. Each dam is approximately 100 feet high and each resultant impoundment is roughly 40 miles long. The characteristics of each of the dam/reservoir systems are summarized in Table 2-1. The combined power output of the 24 generating units on the Lower Snake is in excess of 3 million kilowatts.

Table 2-1
Characteristics of the Lower Snake River Dams

Dam	Year Completed	Height of Dam (feet)	River Mile	Impoundment Area (Acres)
Ice Harbor	1961	103	9.7	9,200
Lower Monumental	1969	103	41.7	6,590
Little Goose	1970	101	70.0	10,025
Lower Granite	1975	105	107.0	8,900

The Lower Snake River has become an important recreational and commercial resource since the dams and associated locks have been in place. The river is currently used year round to ship petroleum products upstream to Lewiston and Clarkston and to ship grain downstream from central and eastern Washington. The Lower Snake has become a focal point for local recreation offering picnicking, fishing, boating and other water based recreation. Withdrawals from the Lower Snake provide irrigation for agriculture.

The fisheries of the Lower Snake River have been the subject of extensive research in recent years. The river is known for it's salmon and steelhead fishery but also provides habitat for healthy populations of bass, trout, crappie, perch, catfish, shad and sturgeon. Recently, questions have been posed regarding passage of anadromous species of salmonids both upstream and downstream at each of the Lower Snake projects as well as the transit time of smolts downstream. This has led to the proposal and evaluation of alternative strategies to increase the success of the salmonids returning upstream and the survival of the smolts traveling downstream. One alternative proposed is to breach the four dams and return the Lower Snake to a free flowing condition. Evaluation of this alternative has provided the impetus for the field program described in this report. Results of the field program will be used, in part, to build a water quality model of the Lower Snake that can be used to evaluate potential changes in primary and secondary biological productivity anticipated from returning the Lower Snake from a series of four impoundments to a free flowing river.

3.0 - WATER QUALITY SAMPLING PROGRAM

3.1 - INTRODUCTION

The sampling program described in the following discussion was developed to characterize biological productivity in the Lower Snake River system. The data collected in 1997 as a part of this effort will be compared with both historical data from the Lower Snake and a water quality model of the Lower Snake system under free flowing conditions. The purpose of the modeling effort is to compare biological productivity in the existing Snake River system with productivity to be expected in the system if the four hydropower and navigation dams were to be breached. The sampling program described below included collection of data in free flowing sections of the Columbia River to provide surrogate estimates of water quality parameters for free flowing sections of the Lower Snake under the proposed unimpounded conditions in the future. This section is divided into several subsections. Section 3.2 includes a discussion of the database compiled as a part of this project. This database includes historic data that are relevant to the biological productivity modeling effort on the Lower Snake. Sections 3.3 and 3.4 describe the sampling stations and the timing of each sampling event. Section 3.5 describes the method for collecting the data. Section 3.6 describes and presents results of the limnological, phytoplankton productivity, and attached benthic algal productivity sampling programs. For ease in presentation, many of the data tables and figures have been placed in Appendices.

3.2 - REVIEW OF EXISTING DATA

In 1997, the National Marine Fisheries Service (NMFS) constructed a limnological database using a relational database management system, Microsoft Access version 2.0 (MS Access). Initially, the database contained information from limnological samples and physical data collected and analyzed by NMFS (selected samples were also analyzed by subcontractors to verify NMFS analyses). Additional limnological data, obtained from documented studies, existing datafiles, and samples collected by the Water Research Center (Washington State University) during 1994-1997 field seasons, was later added to the original database. The purpose of this document is to serve as an introductory guide to possible future users of the MS Access limnological database system. Limnological information from several sources has been included in this database. For each source, the following list provides the years in which data was collected, the sites from which data was collected, and a general description of the types of data collected:

National Marine Fisheries Service (NMFS)

Years:	1994 to 1995
Sites:	Various sites along John Day and Lower Granite reservoirs
Data:	Water Quality (physical and chemical parameters) Benthic invertebrates Zooplankton Fishes

Water Research Center, Washington State University

Years:	1994 to 1997
Sites:	Various sites along the Upper Columbia, Snake (above river mile 129), and Clearwater Rivers; along tributaries of the Snake River, and along McNary, Ice Harbor, Little Goose, Lower Monumental, and Lower Granite reservoirs
Data:	Water Quality (physical, chemical, and biological parameters) Zooplankton Phytoplankton

Water Quality Report, Snake River, Washington-Idaho

Years:	1970 to 1972
Sites:	Various sites along the Snake (above river mile 129) and Clearwater rivers and along Ice Harbor, Little Goose, and Lower Granite reservoirs
Data:	Water Quality (physical, chemical, and biological parameters) Phytoplankton

Water Quality Report, Snake River, Washington-Idaho

Years:	1975 to 1977
Sites:	Various sites along the Snake (above river mile 129) and Clearwater Rivers and along Ice Harbor, Little Goose, and Lower Granite reservoirs
Data:	Water Quality (physical, chemical, and biological parameters) Phytoplankton Zooplankton

United States Geological Survey (USGS) Files

Years:	1970's to Mid-1990's
Sites:	Snake River Mile 2.2 (Burbank), Snake River Mile 6.6 (Below Ice Harbor Dam), Snake River Mile 106.5 (Below Lower Granite Dam), Snake River Mile 132.9 (Clarkston, Washington), Clearwater River Mile 0.2
Data:	Water Quality (physical, chemical, and biological parameters) Phytoplankton (limited to the Burbank site only)

Environmental Protection Agency (EPA) File

Years:	1970's to Mid-1990's
Sites:	Snake River Mile 139
Data:	Water Quality (physical, chemical, and biological parameters)

Appendix L provides a description of the structure of the MS Access database and general guidelines for use.

3.3 - FIELD COLLECTION SAMPLING STATIONS

The sampling sites were chosen to fall into three broad categories; those designated as primary productivity sites, those scheduled for limnological evaluation, and those scheduled for analysis of attached benthic algae productivity. A total of thirteen sampling sites were selected as primary productivity sites and represented free-flowing sections and reservoir portions of the Clearwater River, Lower Snake River, and the Columbia River. An additional eight limnological locations were also included because they were either: 1) sampled during the 1994 - 1995 study, but were not part of the primary productivity investigation; or 2) supplementary locations that provided data useful for long-term monitoring purposes and concomitant research. Thirteen sites were chosen for determination of attached benthic algae productivity. These sites, a brief description of each, and the reasons for their inclusion are presented in Figures 3.3-1 and 3.3-2.

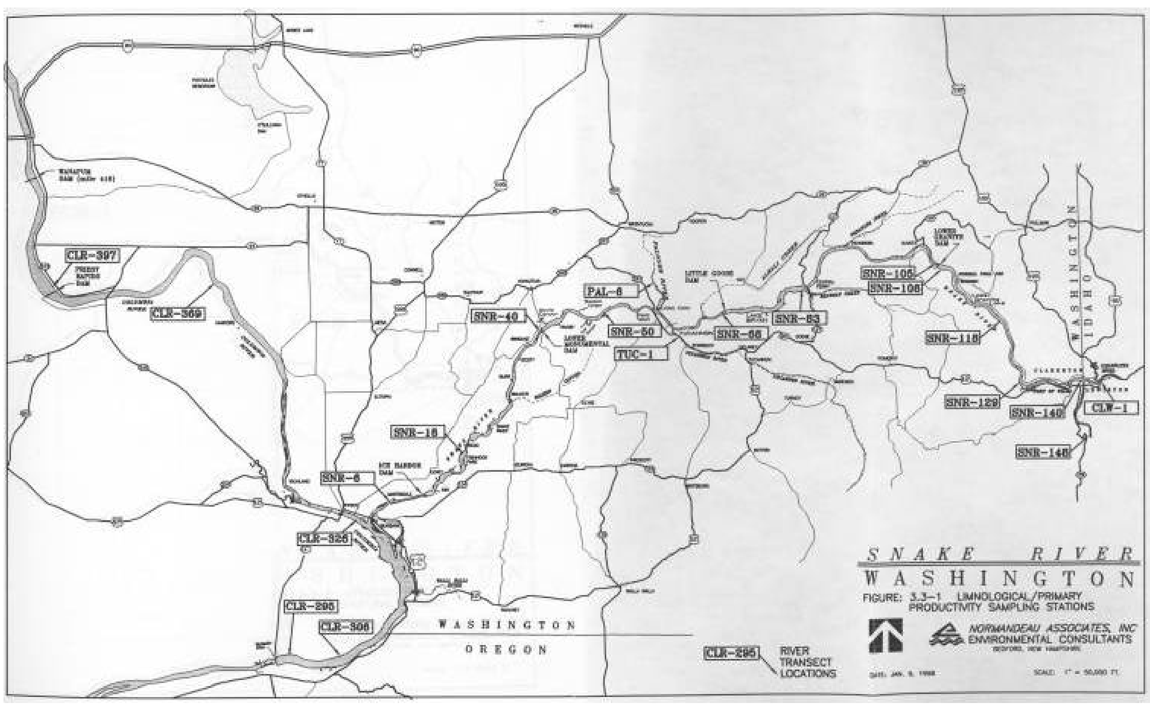

Figure 3.3-1. Limnological/Primary Productivity Sampling Stations

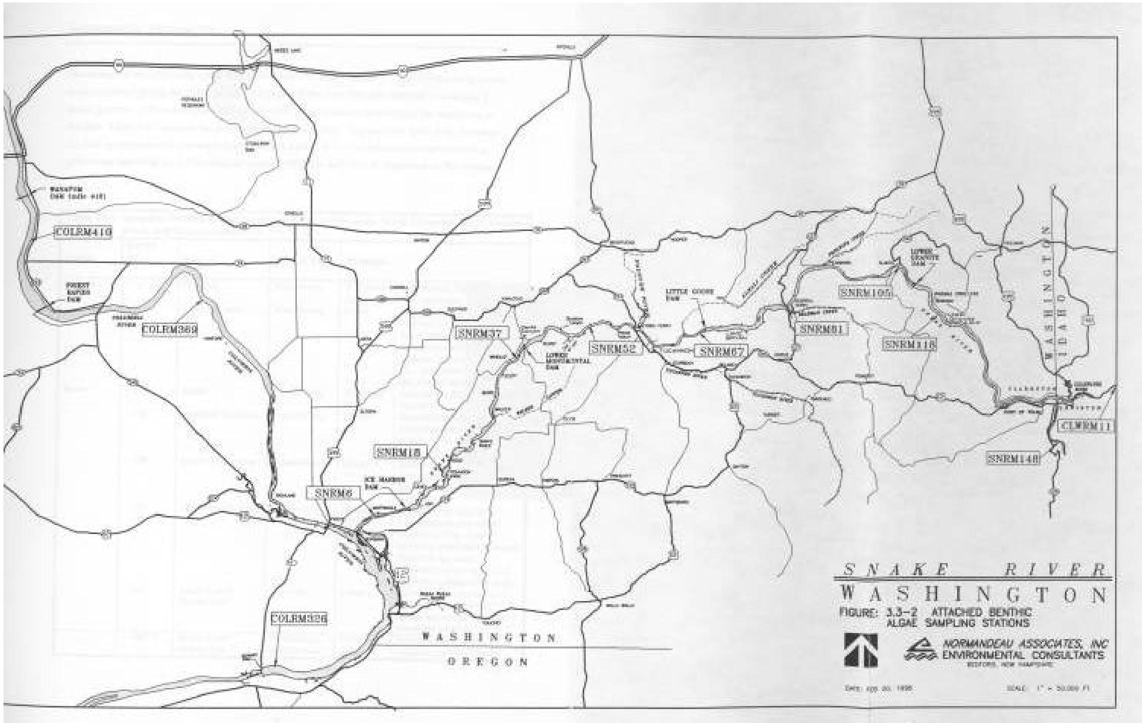

Figure 3.3-2. Attached Benthic Algae Sampling Stations

Table 3.3-1
Sampling Stations in the Clearwater River, Lower Snake River, Columbia River, Palouse River, and Tucannon River

River	River Mile	Reach	Reach Type	Purpose	
Clearwater	~1	Spaulding	Free-flowing	PP/Limno/ABA	Free-flowing Clearwater River, little controlled
	~1	Lewiston, Idaho	Free-flowing	Limno	Free-flowing Clearwater River before it merges with the Snake River. Included in previous studies and compliments the upstream primary productivity site. Also useful for eliciting any changes between stations.
Snake	~148	Asotin	Free-flowing	PP/Limno/ABA	Free-flowing Snake River, little controlled
	~140	Lewiston/ Clarkston	Free-flowing	Limno	Free-flowing Snake River used in previous studies. Analogous benefits as the Clearwater 1 station.
	~129	Lower Granite Pool	Transition zone	Limno	Visited in previous studies, and represents the transition between riverine and lacustrine environments

Snake	~118	Lower Granite Pool	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.
	~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
	~106/ 105	Below Lower Granite Dam	Free-flowing/reservoir mix	PP/Limno/ABA	Hybrid of free-flowing/reservoir; but more riverine
	~83/ 81	Little Goose Pool	Reservoir	PP/Limno/ABA	Only station that has consistently been sampled in Little Goose pool, and was included in the primary productivity study
	~68/ 67	Below Little Goose Dam	Free-flowing/reservoir mix	PP/Limno/ABA	Hybrid of free-flowing/reservoir, but more riverine
	~52/ 50	Lower Monumental Pool	Reservoir	PP/Limno/ABA	Snake River impoundment
	~40/ 37	Below Lower Monumental Dam	Free-flowing/reservoir mix	PP/Limno/ABA	Hybrid of free-flowing/reservoir, but more riverine
	~18	Ice Harbor Pool	Reservoir	PP/Limno/ABA	The only site that has routinely been sampled in the Ice Harbor Pool
	~6	Below Ice Harbor Dam	Free-flowing/reservoir mix	PP/Limno/ABA	Hybrid of free-flowing reservoir, but more riverine
Columbia	~410/ 397	Priest Rapids Pool	Reservoir	PP/Limno/ABA	Impoundment and unconfounded by pollution inputs
	~369	Hanford Reach	Free-flowing	PP/Limno/ABA	True free-flowing river with similar gradient to the lower 140 miles of the Snake River
	~326	McNary Pool	Transition zone	Limno/ABA	Similar to RM-129 on the Snake River, in that it is at the upper end of the McNary pool and in the transition between riverine and lacustrine environments. Also upstream from the confluence of the Snake River.

Columbia	~306	McNary Pool	Reservoir	Limno	Impoundment receiving Snake River flows and sampled in the past by USACE.
	~295	McNary Pool	Reservoir	PP/Limno	The closest station to McNary Dam that has traditionally been sampled by USACE.
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.
	~1	Tucannon	Free-flowing	Limno	Has less discharge than the Palouse River, but water quality unknown at beginning of study.
PP = Primary Productivity Sampling LIMNO = Limnological Sampling ABA - Attached Benthic Algae Sampling					

3.4 - TIMING OF FIELD COLLECTIONS

Flow conditions in the Lower Snake are variable and cannot be predicted accurately within a calendar year. Flow, however, is critical to the primary productivity of the Lower Snake River system. For this reason, the 1997 monitoring program was designed with flow as the principal determinant of sample timing rather than a strict calendar based schedule. Eight full sampling events were completed during the 1997 season. The timing of the field trips was intended to encompass a broad spectrum of flow conditions through the 1997 growing season to maximize the usefulness of the data. Table 3.4-1 presents the dates of each sampling event. Approximate mean daily discharge for each sampling station and event is presented in Table 3.4-2. This information was obtained as provisional data from the US Geological Survey (USGS) or the USACE, depending on the location:

- CLW-11 and CLW-1: USGS gauging station at Spaulding, Idaho (Station 13342500).
- SNR-148 and SNR-140: USGS gauging station at Anatone, Washington (station 13334300).
- SNR-129 through SNR-6: USACE total discharge data for Lower Granite Dam, Little Goose Dam, Lower Monumental Dam, and Ice Harbor Dam.
- CLR-397 and CLR-369: USGS gauging station below Priest Rapids Dam, Washington (station 12472800).
- CLR-326: USGS gauging station below Priest Rapids Dam, Washington (station 12472800) plus data from the gauging station on the Yakima River at Kiona, Washington (station 12510500).

- CLR-306 and CLR-295: USACE total discharge data for McNary Dam.
- PAL-6: USGS gauging station at Hooper, Washington (station 13351000).
- TUC-1: USGS gauging station at Starbuck, Washington (station 13344500).

Table 3.4-1
Dates of Each Sampling Event

Sampling Event	Dates
1	June 2, 1997 - June 12, 1997
2	June 28, 1997 - July 3, 1997
3	July 14, 1997 - July 19, 1997
4	July 28, 1997 - August 1, 1997
5	August 11, 1997 - August 15, 1997
6	September 8, 1997 - September 15, 1997
7	September 22, 1997 - September 26, 1997
8	October 6, 1997 - October 10, 1997

Table 3.4-2
Approximate Mean Daily Discharge at Each Sampling Station During a Given Field Event

System/ Site	Sampling Event										
	1a (kcfs)	1b (kcfs)	2a (kcfs)	2b (kcfs)	3 (kcfs)	4 (kcfs)	5 (kcfs)	6a (kcfs)	6b (kcfs)	7 (kcfs)	8 (kcfs)
Clearwater											
CLW-11 CLW-1	62.30	--	28.00	--	18.00	28.03	23.40	3.58	--	4.22	7.20
Snake											
SNR-148	178.10	--	110.10	--	62.50	58.70	51.90	26.40	--	40.20	33.80
SNR-140		--		--					--		
SNR-129	191.40	--	108.90	--	54.10	61.30	49.10	28.70	--	40.10	40.50
SNR-118	191.40	--	108.90	--	54.10	61.30	49.10	28.70	--	40.10	40.50
SNR-108	191.40	--	108.90	--	54.10	61.30	49.10	28.70	--	40.10	40.50
SNR-106	183.40	--	110.10	--	62.50	58.70	51.90	30.50	--	40.20	42.00
SNR-83	176.50	--	110.20	--	64.90	58.60	53.30	28.00	--	37.60	40.30
SNR-68	171.40	--	101.10	--	62.60	55.90	48.00	26.70	--	40.20	39.90
SNR-50	182.90	--	109.80	--	66.20	61.40	50.40	28.90	--	42.50	42.10
SNR-40	175.00	--	107.30	--	68.70	62.70	58.60	28.80	35.80	42.70	37.90
SNR-18	170.00	--	103.70	--	72.30	64.70	61.10	28.00	32.60	43.10	38.10
SNR-6	177.00	--	113.70	--	69.50	60.50	51.20	29.90	30.60	37.70	39.40

Columbia											
CLR-397	325.00	--	179.00	--	238.00	135.00	137.00	68.30	--	137.00	138.00
CLR-369	327.00	--	181.00	--	225.00	153.00	153.00	86.70	--	139.00	139.00
CLR-326	334.90	--	204.74	--	242.17	128.36	159.91	115.96	--	126.17	151.75
CLR-206	517.90	--	302.60	317.10	302.20	192.90	210.70	124.90	--	165.00	164.70
CLR-295	517.90	--	302.60	317.10	302.20	192.90	210.70	124.90	--	165.00	164.70
Tributaries											
PAL-6	0.778	0.530	0.286	--	0.218	0.141	0.090	0.107	--	0.132	ND
TUC-1	0.333	0.280	0.172	--	0.115	0.102	0.077	???	--	ND	ND
ND: Data not available from USGS at this time.											

Each sampling sequence was completed within a one week period. This approach provided a synoptic view of the system and broad coverage of parameter variability, yet also required two teams with matching equipment that were assigned to a given set of stations (Table 3.4-3).

Table 3.4-3
Station Responsibilities for Each Sampling Team During A Typical 5-Day Sampling Sequence

Day	Team A	Team B
1	CLW-1, -11	SNR-108, -118, -129
2	SNR-140, -148	SNR-83, -106
3	CLR-397	SNR-50, -68 Palouse River Tucannon River
4	CLR-369	SNR-18, -40
5	SNR-6, CLR-326	CLR-295, -306

3.5 - METHODS

3.5.1 - Limnological and Primary Productivity Sampling

The water samples collected by the WRC were gathered by a variety of sampling equipment, depending on the section of the river and subsequent analyses. Discrete water samples from the reservoir and major river sites were retrieved with 4-L discrete point water samplers, while the ones from the Palouse River and Tucannon River were retrieved as sub-surface grab samples. In either case, individually labeled sample bottles were rinsed three times before filling. The phytoplankton and zooplankton populations were collected from the photic zones at the sites using an integrating sampler and vertical tows with a plankton net, respectively.

Samples that required preservation were treated in accordance with *Standard Methods for the Examination of Water and Wastewater* (American Public Health Association, 1992), hereafter referred to as *Standard Methods*, unless otherwise noted. After collection, the sample bottles were placed in ice chests, packed with ice, and transported to the Environmental Engineering Laboratory at Washington State University (WSU) where they were kept in a 4- to 6- °C temperature control room.

Physicochemical Parameters

Alkalinity/Inorganic Carbon

Samples for alkalinity/inorganic carbon evaluations were collected at each depth where a primary productivity incubation occurred. Zero headspace was maintained to prevent changes in the carbonate equilibrium.

Total alkalinity was evaluated titrimetrically (Standard Methods 2320 B) with 0.02N H_2SO_4 to an end-point of pH 4.5. The pH of each titrated sample was determined using a Corning Model 130 pH Meter fitted with a Ross combination electrode. Alkalinity values were converted to milligrams inorganic carbon following the table developed by Saunders *et al.* (1962).

Anions and Cations

Samples for Ca, Mg, Na, and K determinations were collected in 125-mL polyethylene plastic (PP) bottles and preserved with reagent grade HNO_3 to pH <2. Water samples for subsequent Cl, SiO_2, and SO_4 analyses were retained in a 500-mL PP bottle. These samples were not preserved, but were refrigerated.

Calcium, magnesium, sodium, and potassium were analyzed by LBB2 Analytical Services in the Department of Chemistry at WSU using a JY-24 Sequential ICP Atomic Emission Spectrophotometer following the manufacturer recommendations.

Chloride analyses were completed on an Alpkem RFA-300 using two ranges. The higher range of 0.5 to 25 mg/L (Alpkem RFA Method A303-SO90-02) was used for most of the Snake River and lower Columbia River samples. The low range encompassed 0.1 to 5 mg/L (Alpkem RFA Method A303-SO9-00) and was used for the samples collected at the upper Columbia River and Clearwater River stations.

The concentrations of molybdate-reactive silica in the June through August water samples were determined on an Alpkem RFA-300 (Alpkem RFA Method A303-S220), while the ones collected in September and October were analyzed with a Technicon AutoAnalyzer (Technicon Industrial Method 105-71W). A dilution manifold was used on the Technicon to increase the working range from 10 mg/L to 50 mg/L. Because of this modification, the latter samples were analyzed at a rate of 30/hr rather than 40/hr as specified in *Standard Methods*.

Sulfate concentrations were determined using a Technicon AutoAnalyzer II (Technicon Industrial Method 18-71W).

Biochemical Oxygen Demand (BOD)

Samples for BOD- determinations were collected as integrated samples from the photic zone at the primary productivity stations, as well as subsurface samples at the Palouse River and Tucannon River stations, in 1-L PP bottles.

BOD5 (Standard Methods 5210 B) was determined using standard 300-mL BOD incubation bottles with glass stoppers and plastic seal caps. The samples were air incubated in a Fisher Scientific incubation cabinet at 20 +/- 1 °C. Initial and subsequent dissolved oxygen measurements were completed with a YSI Model 52 Dissolved Oxygen Meter utilizing a YSI Model 5905 stirring BOD probe.

Selected samples scheduled for BOD_{20} determination (Standard Methods 5210 C) were resealed and placed back in the incubation cabinet. The dissolved oxygen concentrations in these samples were measured after 20 days.

Dissolved Oxygen, pH, Temperature, and Specific Conductance

Dissolved oxygen concentrations, pH, temperature, and specific conductance were determined in situ at each station. Vertical profiles at the reservoir stations were obtained with a Hydrolab Surveyor 3 or 4 at 1-m increments within the first 10 m and at 5-m intervals thereafter. Data collection at the riverine sections followed the same schedule, except where total depth or current velocity dictated fewer data points. A Hydrolab was also utilized at the Palouse River and Tucannon River stations to obtain those field data sets.

Light

Light attenuation in the water column was evaluated using a Secchi disk and a submarine photometer. The depths of the photic zones, where 1% of the photosynthetically available radiation remained, was determined with a Kahlsico Model 268WA360 underwater irradiameter or a Li-Cor Model LI-1000 data logger with an underwater spherical quantum sensor.

Nitrogen and Phosphorus

The concentrations of various nitrogen and phosphorus species were evaluated for each sample location. At the reservoir stations, 1 m, mid-depth, and near-bottom samples were collected. In riverine sections, samples from a depth of 1 m were retrieved. The samples for NH_3-N were kept in 60-mL PP bottles. Total nitrogen and total phosphorus samples were placed in 250-mL or 300-mL Pyrex glass bottles. An aliquot of the contents of each glass bottle was filtered and acidified to pH <2 with H_2SO_4 for subsequent determination of orthophosphorus and NO_2-N + NO_3-N.

Each water sample was analyzed for three forms of nitrogen: NO_2-N + NO_3-N, NH_3-N, and total nitrogen-N. NO_2-N + NO_3-N was determined colorimetrically using a cadmium catalyst reduction column (Standard Methods 4500-NO_3 F; Technicon Industrial Method 100-70W/B). NH_3-N was determined colorimetrically using the automated phenate method (Standard Methods 4500-NH_3 G; Technicon Industrial Method 154-71W). Total nitrogen was evaluated by digesting 5 ml of the water sample in media containing sodium hydroxide and potassium persulfate in an autoclave at 121° C and 22 psi for 30 minutes. This method is accurate, precise, and yields a total persulfate nitrogen value that is equivalent to total Kjeldahl nitrogen plus NO_2-N + NO_3-N (D'Elia *et al.*, 1977). After the digestion procedure, the samples were analyzed for nitrates using the method identified above.

Two forms of phosphorus were determined: orthophosphorus and total phosphorus. To separate these forms, part of each water sample was filtered through a 0.45 um Millipore filter (HAWG 047). An aliquot of the filtered sample was analyzed for orthophosphorus on a Technicon AutoAnalyzer II using the automated ascorbic acid reduction method (Standard Methods 4500-P F; Technicon Industrial Method 155-71W). Total phosphorus was determined by utilizing the same peroxodisulfate oxidation method used for total nitrogen (Valderrama, 1981; Ebina *et al.*, 1983; Hosomi and Sudo, 1986). The digested phosphorus samples were analyzed using the orthophosphate Technicon method with appropriate matrix modifications.

Sediment Oxygen Demand

Dynamic sediment oxygen demand was estimated for three reservoir sites twice during the field season. Seven 2.36-cm by 1.03-cm butyrate tubes were collected by divers at SNR-50, SNR-123, and SNR-132. The divers pushed the cores into the sediments until they were flush with the sediment surface and then capped each end. The cores were stored in plastic buckets with lids and covered with water during transport to the lab. Since the biological decay of organic sediment is a long-term process of months or years, preservation (other than keeping them stored in a 4° C room) or holding times were not required (Normandeau, 1992).

The cores from an individual site were placed collectively in a sealed container that had a volume of about 0.03 m^3. Three chambers were operated simultaneously and each one was connected to a Masterflex variable-speed pump operating at 500 ml/min. The entire set-up was located in a 20° C temperature control facility. Dissolved oxygen concentrations in the containers were checked at least twice a day until the remaining concentration was less than 2 mg/L. Duplicate ambient water blanks were also run in BOD bottles in conjunction with the larger containers, and the results were subtracted from the oxygen depletion value from the chambers during the calculations.

Total Suspended Solids

Water samples for determining total suspended solids concentrations were collected following the same schedule established for the nitrogen and phosphorus series. Samples were retained in 1-L cubitainers. Total suspended solids (Standard Methods 2540 D) were determined by filtering a well-mixed aliquot through a pre-washed and weighed glass-fiber filter with a mean pore diameter of 1.2 um. Each filter was placed in a pre-weighed aluminum weighing dish and dried overnight at 105° C. Following a cool-down period in a desiccator, the dish and filter were reweighed and the difference in weight constituted suspended solids.

Turbidity

Turbidity samples were obtained from the same container used for Cl, SiO_2, and SO_4. A Hach 2100AN Turbidimeter was utilized in the laboratory. Instrument calibration was checked using Formazin standards that were appropriate for the turbidity levels of the samples.

Biological Parameters

Chlorophyll a

Water samples for chlorophyll a determinations were retrieved from the photic zones with a 3.8-cm diameter by 8-m long integrating sample tube. If the photic zone in a riverine section was greater than the total depth, a sample was taken to within 0.5 - 1.0 m from the bottom. The water samples were stored in brown 500-mL Nalgene bottles and preserved with 0.5 ml of a saturated $MgCO_3$ solution (American Society of Testing and Materials, 1980), and stored on ice.

The spectrophometric method (Standard Methods 100200 H) was utilized to determine the concentration of chlorophyll a, corrected for phaeophytin. A given aliquot of water was filtered through a 0.45-um Millipore membrane filter (HAWP 047). The filter was then placed in a PetriSlide and stored in a -20° C freezer. Chlorophyll a was extracted from the algal cells by sonicating the filter in 90% buffered acetone with a Heat Systems W-385 Sonicator and centrifuging for 20 minutes at 3,000 RPM in a Sorvall RC 5C Plus centrifuge. Supernatant from each tube was pipetted into a 5-cm cylindrical glass cuvette and absorbances were measured with a Beckman DU-65 spectrophotometer.

Phytoplankton

Algal samples were collected following the same regime established for the chlorophyll a samples. The water samples were transferred to 250-mL PP bottles and preserved with 2.5-mL Lugol's solution.

Phytoplankton identification and enumeration were completed by BSA Environmental Services, Inc. Cell numbers of all identified phytoplankton taxa (autotrophic non-flagellated eukaryotes, all flagellated eukaryotes, and cyanobacteria) were quantified on a per milliliter basis using the Utermohl method (Lund *et al.*, 1958). An appropriate volume of sample was allowed to settle at least 24 hours in Utermohl chambers. Recently viable cells were included in the analyses as determined by the respective physiological characteristics of eukaryotes (the presence of chloroplasts and other organelles) and prokaryotes (pigmented cells). At least 400 units (colonies, filaments, unicells) were enumerated at a magnification of at least 600x from each sample using a Wilovert inverted microscope equipped with phase-contrast. The abundance of large, rare taxa was estimated by scanning the entire chamber at a lower magnification. For samples with common colonies or filaments, the counts include several thousand cells since total cell numbers of multi-cell units (colonies, filaments) were quantified.

In colonies with extremely small cells (*e.g.*, Microcystis), cells were enumerated from a small representative area of the colony containing at least 100 cells. For common filamentous taxa, the total cells per filament were determined by first quantifying the cell number within a known length (*e.g.*, 100 um). This process was repeated for 25 filaments of each abundant filamentous taxon, and subsequently used to calculate the mean number of cells per filament length for that taxon. This quantity was applied to measurements of the length and width of each filament that was encountered so that the total number of cells per filament could be estimated.

Primary Productivity

Primary productivity was evaluated using ^{14}C-$NaHCO_3$ at up to five depths, depending on total depth and current velocity, within the photic zone at the designated sites. Two light bottles and one dark bottle were used at each depth and allowed to incubate for approximately 2 to 4 hours during midday. At the reservoir stations, one set of three incubation bottles that represented a given depth was secured radially to a bottle holder. These holders were attached to a nylon-coated aircraft cable and suspended in the water column from a float. At the riverine productivity stations, the BOD bottles from a given depth were incubated in the same manner as those at the reservoir stations if a quiescent location was available. Otherwise, the three bottles from each depth were placed in a Plexiglas tube that was mounted inside a steel cage to protect the glass bottles from debris.

When the scintillation vials were returned to the laboratory, the septa caps are removed under a fume hood and the containers were left open for 24 hours. Ten milliliters of Insta-Gel XF was then added to each vial, the vials were capped, and the contents vigorously mixed. The vials were allowed to rest for approximately one day in the dark to prevent false readings due to chemoluminescence. As a last step, they were placed in a Packard Tri-Carb 2200CA scintillation counter to determine the amount of activity, as disintegrations per minute, in each vial. Counts were made using normal mode, efficiency tracing, and the low-level count mode (Packard Instrument Company, 1988).

Gross primary productivity in mg $^{12}C/m^3$/hr was calculated using the formula given by Britton and Greeson (1987). Hourly rates were extrapolated to daily rates by applying the method proposed by Vollenweider (1965).

Zooplankton

Zooplankton samples were collected from the photic zones of lotic and lentic sites during each sampling sequence. Vertical plankton tows were completed with plankton nets that had a 153-um mesh net and a 75-um mesh in the bucket.

Zooplankton identification and enumeration were also completed by BSA Environmental Services, Inc. The volume of sample concentrate was measured and three 1-mL aliquots were examined at 100x on a Wilovert inverted microscope equipped with phase contrast. If the total tally was less than 200, additional aliquots, up to a maximum of 10 ml, were examined. Taxonomic identification followed Ruttner-Kolisko (1977), Edmundson (1959), and Pennak (1989).

Biomass estimates were based on established length/width relationships (Dumont *et al.*, 1975; McCauley, 1984; Lawrence *et al.*, 1987). The lengths, or the lengths and widths, of each species encountered was measured from a composite sample formed by pooling approximately 5 ml from each sample for that date. The number of specimens examined was equal to 25 for common species, and lesser for more rare taxa. For cladocerans, the length was measured from the tip of the head to the end of the body (shell spines excluded). For copepods, the length was determined from the tip of the head to the insertion of the caudal ramus. The length of rotifers was measured from the tip of the head to the end of the body (spines, toes, etc. excluded). In accordance with McCauley (1984), biomass was computed for the appropriate number of individuals for each sample location and the arithmetic mean biomass was multiplied by the species abundance to produce a species biomass for each sample.

3.5.2 - Attached Benthic Algae Productivity

Algae Incubation and Retrieval

At each site, 12-18 bottom-set, unglazed, ceramic tiles were incubated for fourteen-day intervals from July to October 1997. Tiles were 0.150 m x 0.150 m (0.0225 m^2) and were attached to bricks using steel clips and nylon ties, so that tiles were incubated at approximately 8-10 cm off the river bottom with a net algae incubation area of 0.0218 m^2. Six tiles were incubated at each of the two depths, 0.75 m and 1.5 m. At reservoir sites, six tiles were additionally incubated at 3.0 m depth attached to a steel cage resting on the river bottom.

From mid-August through October, mylar substrate was used in addition to the ceramic tiles. Three mylar replicates were suspended in the water column near shore at a depth of 1.5 m. The 0.150 m x 0.150 m mylar squares were suspended by submerged buoys that were anchored to bricks on the river bottom. This construction provided an incubation area of 0.225 m^2, which was horizontal and free-floating in the water column. This resulting undulating motion permitted algal colonization without the interference of excess sedimentation.

Artificial substrates were retrieved after fourteen days. Shallow tiles and mylar were retrieved by swimmers, while the 3.0 m cage was retrieved by lifting the cage out of the water column by a rope attached to the cage. Care was taken not to disturb the sediment and algal community on the surface of the tiles. The top surface of the tiles were immediately scraped of accumulated algae and sediment with a razor blade, rinsed with distilled water, then brushed and rinsed a second time into 50 ml centrifuge tubes. Samples were stored on ice until filtering that evening on Whatman GF/C filters. Filters were stored in the dark, on ice until processing for chlorophyll analysis and biomass.

Attached benthic algal communities were also measured on natural substrates. Two types of sampling techniques were used. On the first and last run, July 7 to 21, 1997 and October 1 to 15, 1997, freshly scrubbed medium sized rocks were incubated at 1.5 m depths adjacent to the bottom set tiles for a comparison of natural versus artificial substrates. Rocks were removed after fourteen days and a 0.0016 m^2 surface of the rock was inscribed, scraped, brushed, and rinsed as described above. The second technique involved dredging rocks from 0 - 10 m depths using a Ponar dredge. Rocks were processed as above to determine chlorophyll and biomass levels on natural substrates and to delineate the empirical depth of the photic zone.

Additionally, on February 20 - 22, 1998, rocks were dredged at the incubation sites in order to determine the increased substrate area over and above the assumed planar area available for algal growth. Using a Peterson dredge, rocks were obtained from three depth zones: 0.25 m - 1.0 m, 1.0 m - 2.0 m, and 2.0 m -3.5 m to estimate the substrate at each one of the incubation depths of 0.75 m, 1.5 m, and 3.0 m. Of the thirteen study sites, data was obtained from Snake RM 148 (Asotin), Clearwater RM 11 (Spaulding), Snake RM 118 (Lower Granite Reservoir), Snake RM 105 (Below Lower Granite Dam), Snake RM 81 (Little Goose Reservoir), Snake RM 67 (Below Little Goose Dam), Snake RM 52 (Lower Monumental Reservoir), Snake RM 37 (Below Lower Monumental Dam), and Snake RM 6 (Below Ice Harbor Dam).

At each depth, 6 - 12 dredges were taken. The number of dredges at each site was recorded and samples were classified according the Wentworth Classification Scale (Wentworth in Hynes, 1970). Small pebbles (less than 32 mm in diameter), gravel, sand, and silts obtained from the dredges were classified and then discarded. Larger pebbles, cobbles, and small boulders were immediately marked with chalk to record the level of embeddedness. Rocks were then placed on paper and the perimeter of the unembedded portion of the rock traced onto the paper to determine the planar area of the rock. The area was then determined using a Numonics Corporation Model 1224 electronic planimeter. The non-embedded surface of the rock was then covered with tinfoil shaped to fit the rock. The tinfoil was carefully removed from the rock, cut with scissors in order to measure the flattened surface area of the tinfoil and taped to paper. The paper was Xeroxed, and the area of the tinfoil was measured by planimetry.

The total surface area of the rock above the sediment surface (as determined by the tinfoil) was divided by the planar area of the rock (as determined by the tracing) for a ratio of the actual surface area of the rock substrate available for algal colonization at the site, An average of the ratios for each dredge was calculated for each depth to determine the surface area coefficient. Dredges that resulted in small pebbles, gravel, silt and sand were incorporated into the average and the ratio of increased surface area to planar area was determined to be 1. In the case of dredges that failed to obtain any substrate (where mechanical failure of the dredge was rejected), it was assumed that the substrate was larger than the dredge and could not be picked up. In this case, ratios of successful dredges which had obtained cobbles and boulders at that depth were averaged and used as the ratio for that dredge. (Note: The determination of large boulder substrate was made by the operator of the dredge. The various types of substrate was usually apparent by the dredge performance).

The ratios obtained from the dredges were then multiplied by the mean chlorophyll *a* and biomass (ash-free-oven dry weight) at each of the eight sites. However, in the main results portion of the report, productivity values for the sites were calculated based on a flat river bottom.

Physical Measurements

At each site, light, temperature, solar radiation and substrate composition were monitored. Light penetration was determined using a standard 20 cm Secchi disk and a Montedoro Whitney submarine photometer calibrated with an Extech photometer. Vertical extinction coefficients were determined by the following equation:

$$VEC = \frac{\ln I_1 - \ln I_2}{z_1 - z_2}$$

Where: I_1 = Light intensity at depth 1
I_2 = Light intensity at depth 2
z_1 = Depth 1
z_2 = Depth 2

Solar radiation measurements were taken every hour during the day with a handheld Extech photometer. Temperature and dissolved oxygen measurements were determined with a YSI Model 55 and Model 57 Temperature and Dissolved Oxygen Probe. Substrate composition at the site was classified according to the Wentworth Classification Scale (Wentworth, in Hynes 1970).

Chlorophyll Analysis

After freezing and holding for a maximum of three weeks, individual filtered algal samples were split in equal halves in the lab in order to analyze both chlorophyll and biomass on each sample. Each filter half was then processed for either chlorophyll or ash-free-oven-dry weight (AFODW) and oven-dry weight (ODW). All lab techniques for the determination of chlorophyll and biomass were according to Standard Methods for the Examination of Water and Wastewater (1992).

Monochromatic chlorophyll *a* and trichromatic chlorophyll *a*, *b*, and *c* were determined by the spectrophotometric method using the Beckman 40 Spectrophotometer. Algae samples were macerated in a 40 ml Wheaton glass tissue grinder with 90 percent acetone solution. Following grinding, samples were refrigerated and allowed to steep 24 hours in the dark at 4°C. Prior to analysis, samples were centrifuged for 15 minutes at 1,500 rpm and the chlorophyll extract decanted, volume measured, and placed in the spectrophotometer. Monochromatic chlorophyll *a* and phaeophytin were determined by the addition of 0.1N HCL before samples were reread at 750, 665, and 664 nm wavelengths. In the discussion of this report, monochromatic chlorophyll *a* will be emphasized because it is a more representative value of the functional chlorophyll *a* of the ABA community.

Biomass Determination

Oven-dry weight was determined by drying samples at 104o C in pre-weighed aluminum tares for 15 hours. Tares were cooled in a desiccator for 45 minutes prior to weighing. Samples were dried at 500 °C for 1 hour, rewet, and dried again at 104 °C for 15 hours. Samples were allowed to cool in a desiccator for 45 minutes and then reweighed for the determination of AFODW. Due to the large amount of sediment which collected on the tiles at some sites, an estimate of sedimentation was obtained by the weight of the inorganic material left after drying in the oven at 500 °C, rewetting, and drying at 104 °C.

Autotrophic Index

The Autotrophic Index (AI) was also calculated. The AI is determined by dividing chlorophyll a (mg/m^2) by the AFODW (g/m^2) of each sample. Large AI values (greater than 200) indicate heterotrophic associations or possibly reduced water quality in the periphyton community and is an indicator of the nature of the periphyton community between sampling sites. Both monochromatic and trichromatic chlorophyll a values were used in the AI calculations. The trichromatic AI values are lower than the monochromatic calculations due to the phaeophytin in the sample. Although both data is given in the tables, the trichromatic AI is the historically preferred value and will be emphasized in the discussion of this report.

Primary Productivity

Primary productivity was estimated at Snake RM 148, 118, and Columbia RM 410 and 369. Primary productivity measurements were estimated using the dissolved oxygen technique (APHA, 1992). This technique involves determining the amount of oxygen produced in a closed compartment by the algal community over a defined time period of photosynthesis. Calculations then determine the amount of carbon fixed per hour *via* photosynthesis after determining oxygen produced per unit substrate per hour.

Mylar substrates ($0.225m^2$) which had been incubated for 14 days were inserted into 1-gallon clear ziploc bags containing 1.5 l of water. Initial dissolved oxygen concentrations were lowered by bubbling in compressed nitrogen gas. (Note: On the first primary productivity run in early August, ceramic tile substrates were used. Mylar was later chosen over the tiles due to the heavy sediment accumulation on the tiles. When tiles were utilized, both the large amount of sediment and the algal community were incubated.) Bags were incubated at 1.5 m depth for 3 hours during the two time intervals: 1100 - 1400 and 1400 - 1700.

A total of six attached benthic algae chambers were used at a site, three replicates for gross and net oxygen production estimates and three additional chambers incubated in the dark at the 1.5 m depth to measure respiration. Two additional phytoplankton chambers were used, one light and one dark, to compensate for phytoplankton photosynthesis and respiration at the 1.5 m depth. Phytoplankton chambers were filled with 1.5 l of water and incubated at 1.5 m for three hours adjacent to the light and dark attached benthic algae chambers. Oxygen evolution from phytoplankton production was subtracted from ABA production values so that ABA primary productivity values would not be inflated.

On October 19, 1997, the 14 °C and dissolved oxygen techniques to estimate rates of ABA primary productivity were compared at Columbia RM 369. The two techniques were compared side by side in a joint effort between Washington State University, who performed the 14 °C analysis, and the University of Idaho, following the above dissolved oxygen procedure.

Community Composition

Qualitative algae samples were obtained from both natural and artificial substrates. On the artificial substrates, samples were obtained by scraping the fourteen-day algal growth from the side of the tile, the brick that supported the tiles, or the buoys attached to the artificial substrate. On natural substrates, samples were obtained by randomly selecting rocks at the shoreline, 0.75 m and 1.5 m depths. Samples were then stored in 50 ml centrifuge tubes with Lugol's iodine solution. At the end of the field season, samples were identified in the lab to Genus.

3.5.3 - Diel Dissolved Oxygen Survey

Changes in dissolved oxygen and temperature were measured at Snake RM 148 on Sept. 17 -18 from 2015 - 0830 and on Oct. 1-2 from 1130 - 0730. In addition, dissolved oxygen and temperature were measured at Snake RM 118 on Sept. 18-19 from 2130 - 1030 and on Oct. 2-3 from 1000 - 0800. Measurements were taken every 2 hours at 2.5 m depth at both the free flowing and reservoir sites.

3.6 - RESULTS

The following presentation of the project results is based on the following methods and assumptions:

1. Lower Snake River refers to the stretch of that river below the confluence of the Snake and Clearwater Rivers, and therefore encompasses stations SNR-129 through SNR-6. Upper Snake River stations refers to SNR-140 and SNR-148 (not truly upper Snake River in the drainage basin context, but upper relative to the other sites utilized in this study). Similarly, upper sites in the Columbia River refer to those above the confluence with the Lower Snake River (*i.e.*, CLR-397, CLR-369, and CLR-326).

2. For several of the explanations that follow (*e.g.*, field measurements, nutrients, and ions), values or concentrations obtained at a depth of 1 m are used for making comparisons. This approach was used since it simplifies the analyses and because it is a reasonable course of action due to the well-mixed nature of the systems. Any important differences that occurred with depth are noted separately.

3. Median values are used extensively to make comparisons rather than means since the data sets are often not normally distributed. However, just because two medians are different does not necessarily mean that a statistically significant difference exists between them, and efforts will be made to make that clear where necessary.

4. Similarly, with the relatively small number of samples collected per field event from the Clearwater River and upper Snake River, the accompanying box plots are intended to display trends in the data rather than statistically significant differences.

Box plots are a useful graphical technique for displaying batches of data (Figure 3.6-1). The technique is based on order statistics (ordering the data points from low to high value) and the plot itself is constructed from several values from the ordered data set. To understand the box plots, a few definitions are needed.

- The *median* of the batch is marked by the center horizontal line and splits the ordered batch of numbers in half.

- The *lower and upper hinges* comprise the edges of the central box and split the remaining halves in half again.

- *Hspread* is comparable to the interquartile range or midrange. It is the absolute value of the difference between the values of the two hinges.

- The *inner fences* are defined as:

  lower fence = lower hinge - (1.5 Hspread)

  upper fence = upper hinge + (1.5 Hspread)

- The *outer fences* are defined as:

  lower fence = lower hinge - (3 Hspread)

  upper fence = upper hinge + (3 Hspread)

- The *whiskers* show the range of values which fall within 1.5 Hspreads of the hinges. They do not necessarily extend to the inner fences.

- Values outside the inner fences are plotted with asterisks. Values outside the outer fences are plotted with empty circles.

- The *notch* is a way of putting confidence intervals on the median. If the intervals around two medians do not overlap, then we are confident at about the 95% level that the two medians are significantly different. If the intervals do overlap, then the medians cannot be considered significantly different, even if the medians appear different.

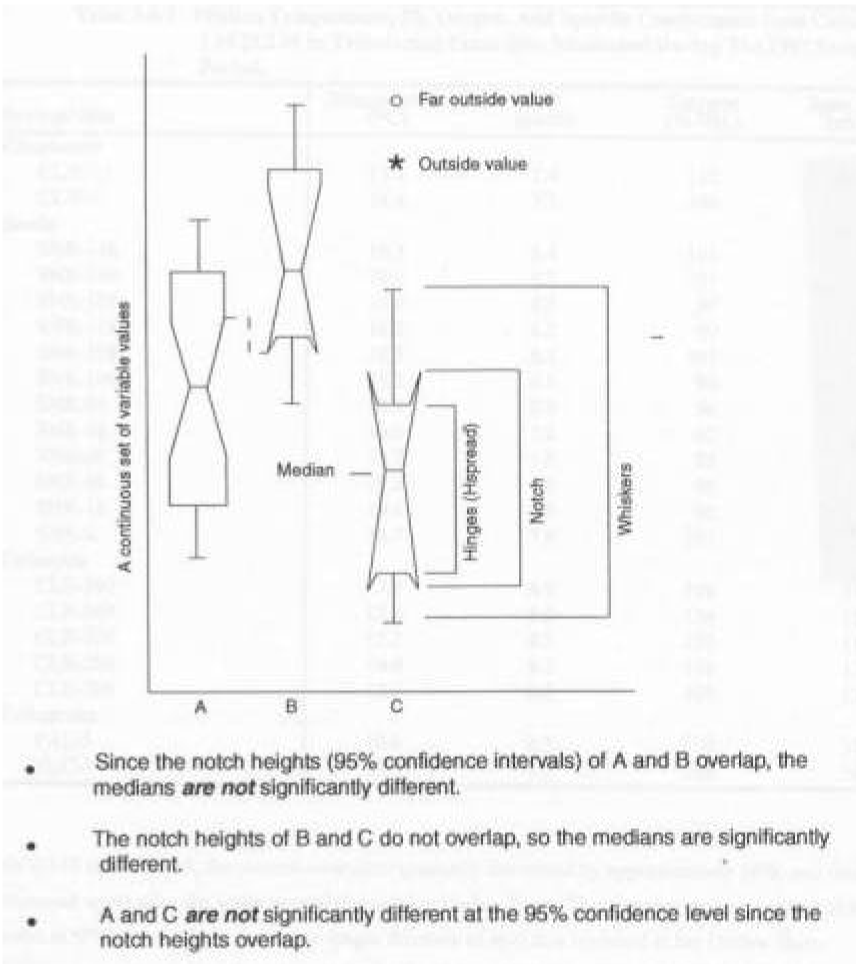

Figure 3.6-1. Configuration of box plots and their relationships

3.6.1 - Physical Parameters

Dissolved Oxygen

The amount of oxygen dissolved in water is dependent on several physical, chemical, and biological parameters. Median percent dissolved oxygen conditions in the Clearwater River ranged from 102 to 106% for the entire sampling season (Table 3.6-1). This was slightly higher than the 101% median calculated for the Snake River at SNR-140 and SNR-148; possibly due to the generally cooler water in the Clearwater River and the its' shallower depth that may lead to more reaeration. Between SNR-148 and SNR-18, the percent saturation gradually decreased by approximately 10%, and then increased again after the water passed through Ice Harbor Dam. The elevated oxygen content of the water at SNR-6 was likely due to the longer duration of spill that occurred at Ice Harbor Dam. Spilling water at the other three lower Snake River dams did not appear to affect the oxygen saturation of the water. All of the Columbia River sites had median oxygen saturation ranging from 103% at CLR-326 to 108% at CLR-397, and were not statistically different from one another.

Table 3.6-1
Median Temperature, pH, Oxygen, and Specific Conductance Data Collected at 1 M (0.2 M in Tributaries) From Sites Monitored During the 1997 Sampling Period

System/Site	Temperature (°C)	pH (Units)	Oxygen (% Sat.)	Spec. Cond. (*u*S/cm)
Clearwater				
CLW-11	13.4	7.4	102	20
CLW-1	14.4	7.7	106	27
Snake				
SNR-148	19.7	8.3	101	248
SNR-140	19.9	8.2	101	246
SNR-129	19.0	8.3	99	147
SNR-118	18.2	8.2	99	146
SNR-108	19.5	8.4	101	140
SNR-106	18.2	8.1	98	141
SNR-83	19.9	8.2	96	135
SNR-68	19.0	7.8	92	140
SNR-50	19.7	7.9	92	143
SNR-40	19.8	7.9	90	147
SNR-18	19.8	7.9	90	147
SNR-6	19.7	7.9	101	138
Columbia				
CLR-397	17.0	8.0	108	117
CLR-369	17.4	8.0	104	119
CLR-326	17.7	8.2	103	116
CLR-306	19.0	8.2	106	132
CLR-295	18.6	8.2	105	131
Tributaries				
PAL-6	19.6	8.7	103	377
TUC-1	19.3	8.5	108	148

Between the beginning of the sampling season and the end, the percent oxygen saturation at the sites did change (Figure 3.6-2). At the Clearwater River stations, the amount of oxygen in the water initially decreased slightly from trip one to trip two, but was still close to 100%. During July, the percent saturation increased and reached about 120% by late July; presumably as a result of the water released from Dworshak Reservoir. For the remaining four sampling events the percent saturation gradually decreased, but still remained close to 100%; suggesting very little oxygen depletion. At the two upper sites on the Snake River, maximum oxygen conditions were noted on the first trip in June during peak flow conditions when they reached 119% and 114% at SNR-148 and SNR-140, respectively. After that time, the percent saturation remained close to, and usually slightly greater than, 100%. The unexpectedly low readings measured during trip five could be correct or the result of a faulty sensor.

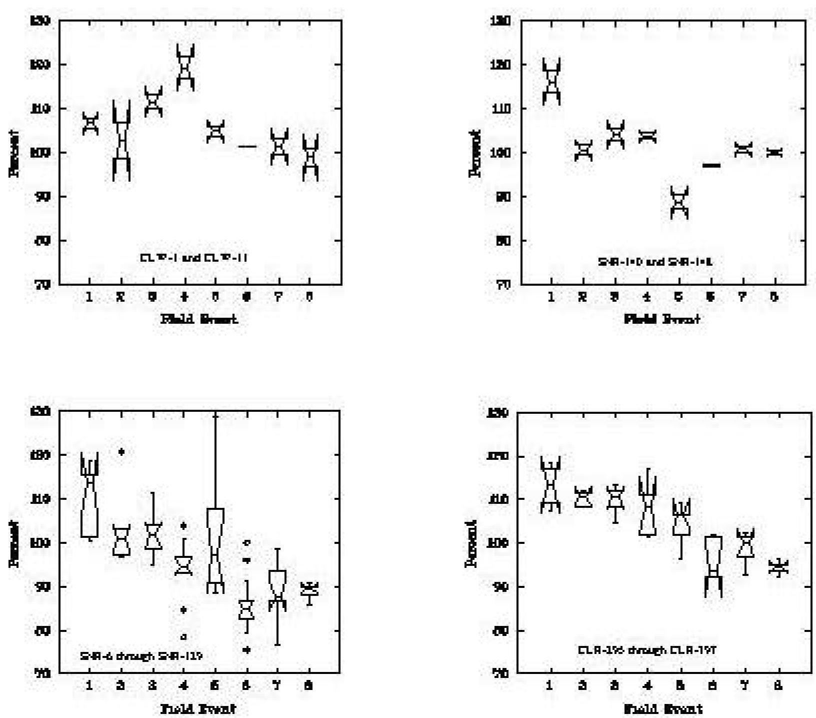

Figure 3.6-2. Box plots of percent oxygen saturation near the surface at the Clearwater River, Snake River, and Columbia River sampling stations during each field event.

The remaining sampling stations on the Lower Snake River and on the Columbia River followed similar patterns with regard to oxygen saturation. Median values exceeding 110% were observed at both systems in the beginning of June. Following the first field event, the percent saturation decreased, though not linearly, through the sixth trip in mid September and then either increased or remained at about the same level for the remainder of the sample period. Two points regarding this trend should be noted. First, this pattern shows several similarities to the hydrographs, which also decreased through mid September and then subsequently rose. Second, with the exception of the early June field trip, the values obtained for the combined sites on the Columbia River were significantly greater that the ones determined for the Lower Snake River sites.

Differences in oxygen concentrations between surface waters and those near the sediments were usually small. At most of the deep reservoir sites, where near-bottom oxygen depletions would most likely occur, the change in oxygen concentration between the top and bottom was usually less than 1 mg/L. The notable exceptions were recorded in the beginning of September in Lower Granite Lake and Lake Bryan (Little Goose Pool). At that time, differences in oxygen concentrations of 2.5, 3.8, and 5.6 mg/L were measured at SNR-83, SNR-108, and SNR-118, respectively. These depletions may reflect the oxygen demand of organically rich sediments in parts of the reservoirs, and is supported by the results of the SOD experiments, which will be discussed later.

The percent oxygen saturation in the Palouse River and the Tucannon River was very comparable to those determined for all of the other monitoring sites. Calculated medians for the Palouse River and Tucannon River were 103% and 108%, respectively. The lowest percentages (93% in both systems) occurred in June, while the most saturated conditions were noted in September (106% at PAL-6 and 118% at TUC-1).

Temperature

The surface water temperatures in the Clearwater River did show some variability within the sampling region (Table 3.6-1). The median temperatures of the Clearwater River for the entire sampling period were 13.4°C at CLW-11 and 14.4oC at CLW-1. These values were statistically lower than those determined for any other site. The median, as well as the range of, water temperatures between SNR-148 and SNR-6 did vary, but the differences were not statistically significant and did not exhibit any definite trends. The measurements from the Columbia River did show a slight warming trend from CLR-397 where the median temperature was 17.0°C to CLR-295 where the analogous value was 18.6°C. These values are less than the overall median of 19.2 °C obtained for the Lower Snake River, but not statistically different using the available data.

The temperatures recorded at the sampling stations did follow a season trend, although the expected pattern in the Clearwater River and part of the Lower Snake River was slightly modified by water released from Dworshak Reservoir. The lowest surface temperatures observed during the sampling period occurred at the beginning of June. At that time, Snake River and Columbia River water temperatures were in the 12 to 14°C range while the ones in the Clearwater River were slightly cooler at just under 9°C

(Figure 3.6-3). As the season progressed, water temperatures at all of the sites increased, and surface temperatures at the upper Snake River stations, as well as the ones on the Columbia River, peaked at 20 to 23°C during August. The temperatures in the Clearwater River were also increasing through the third field event in mid July. When the fourth field event was commenced during the latter part of July, relatively cool metalimnetic water was being released from Dworshak Reservoir at a rate of 22.5 kcfs, and this led to the temperature depressions illustrated in Figure 3.6-3 for trips four and five. By the time the sixth field event was completed, releases from Dworshak Reservoir had been reduced to 1.4 kcfs and the median temperature of the Clearwater River at the sample locations rose to about 20°C. Thereafter, the water temperatures in this river decreased as fall approached.

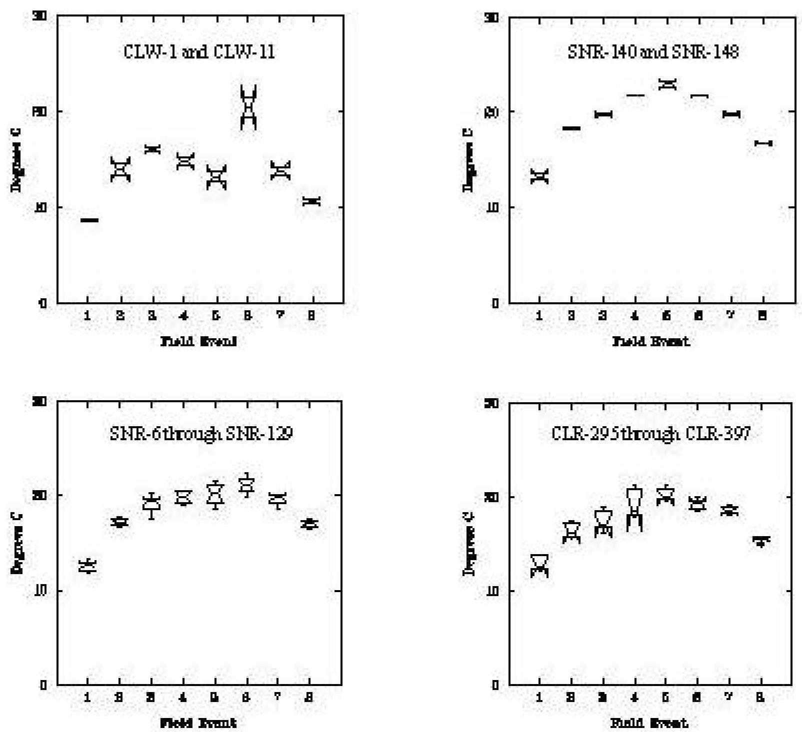

Figure 3.6-3. Box plots of surface temperatures at the Clearwater River, Snake River, and Columbia River sampling stations during each field event

The mid-summer temperature reduction in the Clearwater River was noticeable through a large segment of the Lower Snake River, although with decreasing intensity as the distance downstream increased (Figure 3.6-4). The identical phenomenon has been observed previously during summer releases from Dworshak Reservoir, and has been used to stimulate salmonid migration.

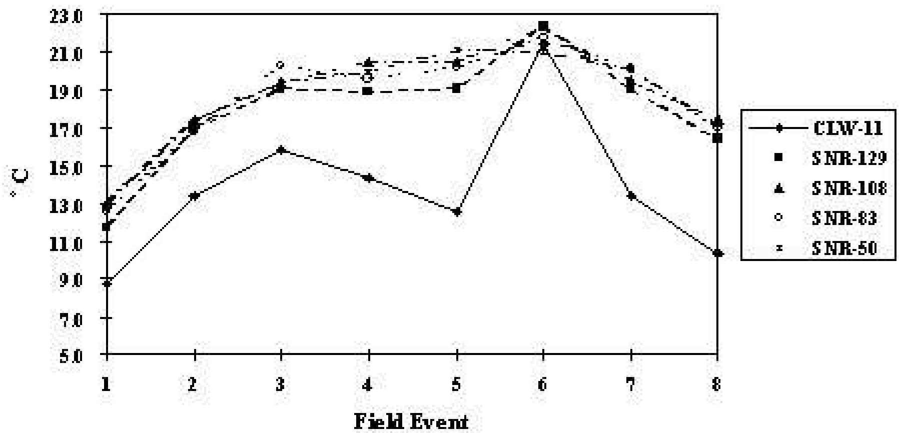

Figure 3.6-4. Surface temperatures in the Clearwater River and selected stations along the lower Snake River during each field event

None of the reservoir stations were thermally stratified during the summer. The greatest temperature difference between surface and bottom waters occurred at SNR-108, the deepest reservoir station, in mid August when the lower stratum was 4.7°C colder than the upper layer. The remaining reservoir locations on the Lower Snake River and Columbia River did not have more than a two to three degree temperature difference throughout the water column at any time.

Median seasonal temperatures in the Palouse River and Tucannon River were very similar to each other, and to those in the Lower Snake River (Table 3.6-1); however, the ranges of values were greater. Maximum temperatures in the Palouse River and Tucannon river reached 22.9°C and 25.0°C, respectively, at the end of July and were higher than any values from the other systems.

Minimum temperature in these tributaries were also lower than those in the Lower Snake River, reaching 10.4°C in the Palouse River and 12.3°C in the Tucannon River on 8 October.

Specific Conductance

The major differences in specific conductance observed between sites were again those associated with the Clearwater River. The season-wide median values for CLW-1 and CLW-11 were 26.5 and 20.0 *u*S/cm, respectively (Table 3.6-1). The analogous values for the upper Snake River sites were approximately an order of magnitude greater, with 246 *u*S/cm at SNR-140 and 248 *u*S/cm at SNR-148. The median values for the remaining Lower Snake River sites were intermediate between those documented in the Clearwater and Snake Rivers, ranging between 135 and 147 *u*S/cm; the differences between these stations were not significant. The median values for the stations that bracketed the Columbia River sites were 117 *u*S/cm at CLR-397 and 131 *u*S/cm at CLR-295. The differences between the three up-river sites and the two below the confluence were due to the influence of the Snake River, and were most notable for the site at CLR-306.

Specific conductance at the stations between SNR-129 and SNR-6 were, of course, influenced by the conductivity values, as well as discharge, in the upstream reaches. The combined median value for these sites was about 68 *u*S/cm in early June; rose to approximately 131*u*S/cm by mid July; leveled off during the fourth sampling event; continued to increase slightly by mid August; and then, as the flow augmentation from Dworshak Reservoir was curtailed, escalated rapidly to 363*u*S/cm by the beginning of October.

The sampling stations in the Columbia River also displayed increases in conductivity with time, but the changes were much less than those seen in the Snake River (Figure 3.6-5); due, in part, to the fact that the mean discharge in the Columbia River decreased by 66% between June and October while average flows in the Snake River at Ice Harbor declined by 79% during the same time period. At CLR-397, for example, the difference between the first and last conductivity measurement was only 35 *u*S/cm. Modest increases of 19 and 7 *u*S/cm occurred at CLR-369 and CLR-326, respectively, while the station at CLR-306 showed a similar increase to the one at CLR-397.

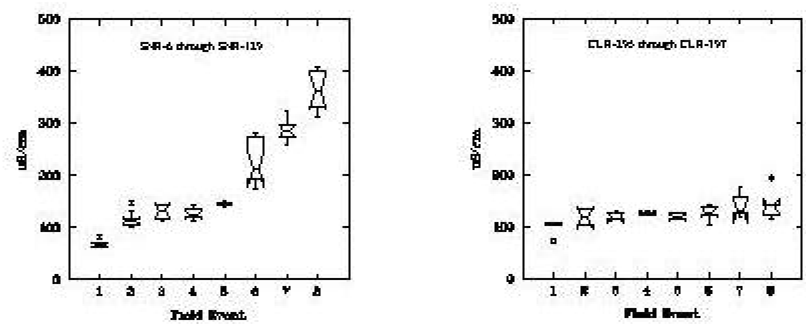

Figure 3.6-5. Box plots of specific conductance from 1 m at the lower Snake River and Columbia River sampling stations during each field event

The specific conductance measurements from the Palouse River and Tucannon River revealed significant differences. The seasonal median in the Tucannon River was 148 *u*S/cm, and the range went from 91 *u*S/cm to 163 *u*S/cm; the lowest values occurred at the beginning of June and reached a plateau by the first part of September. All of these values were easily within the range of those determined at the other monitoring stations. At PAL-6, the seasonal median was 377 *u*S/cm, the lowest instantaneous measurement was 230 *u*S/cm on 4 June, and the greatest single value was 389 *u*S/cm recorded on 8 October. As such, the Palouse River almost always had the highest recorded conductivity during any sampling event, reflecting the greater concentrations of dissolved ions in that system.

pH

The trends in pH values observed during the study were more subtle than the ones seen for the parameters previously evaluated. The Clearwater River was slightly different than the other sites since the lowest median pH values were determined for CLW-1 and CLW-11 (Table 3.6-1). The median pH (7.4) calculated for CLW-11 was significantly less than those determined for the Snake River sites, and reflect the lower buffer capacity of that system. The values measured at the stations from SNR-148 to SNR-83 were essentially the same, averaging slightly more than 8 pH units. The sites between SNR-68 and SNR-6, however, had noticeably lower pH's; median values were less than 8 at each of the five sites. The departures from the upriver sites were not necessarily statistically significant, but they do tend to correspond to the reduced primary productivity at these sites which will be mentioned again in a later section. Median values at the Columbia River stations did not differ markedly from each other, and ranged from 8.0 at CLR-397 and CLR-369 to 8.2 at the three down-river stations.

The pH values recorded for the Palouse and the Tucannon Rivers were slightly higher than comparable measurements from the Snake or Columbia Rivers. The median seasonal pH at PAL-6 was 8.7, and only 0.2 units lower at TUC-1. Maximum and minimum values in these two tributaries were also greater than in the main stems, ranging from 8.1 to 9.3 in the Palouse River and from 7.8 to 8.8 in the Tucannon.

Light

Since light attenuation in the water column is dependent on the type and quantity of dissolved or suspended material, it is not surprising to see a correlation with suspended solids, and thus discharge, and transparency. The smallest VEC's (Vertical Extinction Coefficient), and thus the deepest photic zones and Secchi disk depths, were determined for the upper Columbia River site (Table 3.6-2; Figure 3.6-6). These values increased slightly between CLR-397 and CLR-326, and then rose successively at CLR-306 and CLR-295. Maximum VEC's at the up-river site ranged from 1.06 n/m in the spring to 0.36 n/m in the fall, while the range at CLR-295 was between 1.55 n/m and 0.88 n/m. The VEC's evaluated for the Clearwater River were greater than those described for the Columbia River, ranging from 1.49 n/m to 0.51 n/m at CLW-11. It is interesting to note that, although the correlation is not perfect, the VEC's at the Clearwater River stations appeared to decline during the period when the amount of water released from Dworshak Reservoir was at a maximum. The overall median VEC's calculated for the Snake River stations were higher than those in the other rivers, and during spring runoff the stations at SNR-148 and SNR-140 had the highest computed VEC's, 3.25 n/m and 4.07 n/m, respectively. The below-dam stations at SNR-106, SNR-68, and SNR-40 generally had higher VEC's and shallower photic zones than the reservoir sites immediately upstream or downstream. This difference was very likely due to the increased turbulence encountered below the dams.

Table 3.6-2
Median Secchi Disk Depths, Photic Zone Depths, and VEC's For the Sites Monitored During the 1997 Sampling Period

System/Site	Secchi Disk (m)	Photic Zone (m)	VEC (n/m)
Clearwater			
CLW-11	*	*	0.65
CLW-1	3.1	*	0.72
Snake			
SNR-148	*	*	0.93
SNr-140	2.5	5.5	0.84
SNR-129	1.9	4.8	0.96
SNR-118	1.7	4.9	0.95
SNR-108	1.8	4.9	0.93
SNR-106	1.2	4.0	1.15
SNR-83	1.5	4.9	0.95
SNR-68	1.1	3.3	1.39
SNR-50	1.4	3.9	1.21
SNR-40	1.2	3.8	1.22
SNR-18	1.4	4.1	1.13
SNR-6	1.2	4.0	1.14
Columbia			
CLR-397	3.5	8.7	0.53
CLR-369	3.5	8.7	0.53
CLR-326	2.7	8.1	0.58
CLR-306	1.7	5.8	0.79
CLR-295	1.5	4.4	1.06

*Some of the observed or calculated values were greater than the depth of the stream.

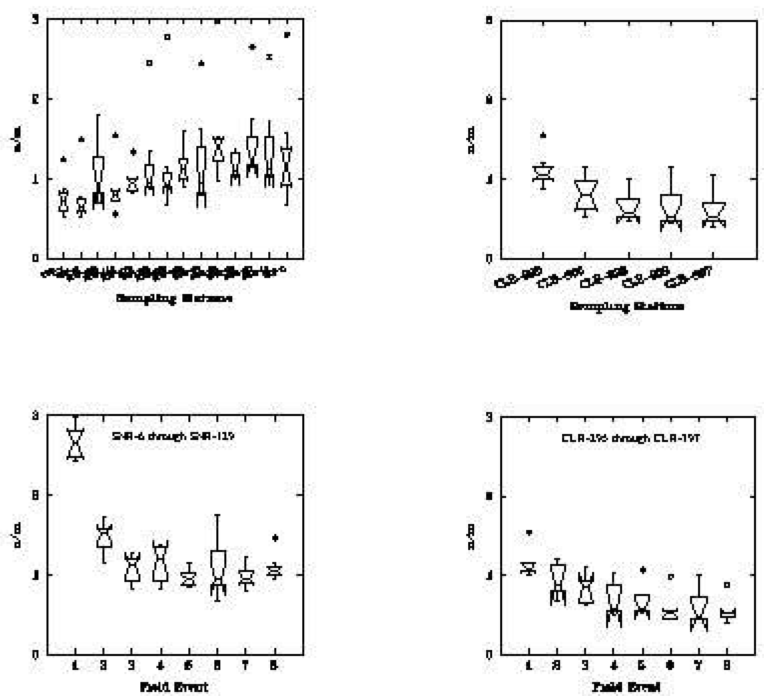

Figure 3.6-6. Box plots of vertical extinction coefficients at the Clearwater River, Snake River, and Columbia River sampling stations

3.6.2 Water Chemistry

Phosphorus

Since orthophosphorus (OP) is the soluble fraction of the phosphorus species, it is not surprising that the distribution of concentrations mimics NO_3 dynamics in several ways (Table 3.6-3). The Clearwater River and the Upper Columbia River sites were very similar with respect to the concentrations of this nutrient; beginning with about 0.006 mg/L in the spring, declining to 0.001 mg/L, or less, in mid summer, and then increasing slightly to 0.003 mg/L in the fall. The Upper Snake River stations had higher concentrations than the sites in the lower segment during trips two through seven, with values ranging from 0.013 mg/L to 0.059 mg/L. The greatest concentrations were again observed during the fall sampling events. The Lower Snake River sampling locations were found to have median concentrations that ranged from 0.013 mg/L to 0.023 mg/L. The two highest median values were for SNR-18 and SNR-40, but it does not appear as though this difference was statistically significant. It should also be noted that the concentrations in the Lower Snake River did not go below the detection limit of 0.001 mg/L during any of the sampling events.

Table 3.6-3
Median Values for Selected Phosphorus and Nitrogen Parameters For the Samples Collected at 1 m (0.2 m at the Tributaries) During the 1997 Sampling Period

System/Site	Ortho-P (mg/L)	Total-P (mg/L)	TP as OP (%)	NO3 (mg/L)	Inorg-N as N03* (%)	Total-N (mg/L)	TN:TP (Ratio)
Clearwater							
CLW-11	0.002	0.010	25	0.03	83	0.08	9
CLW-1	0.003	0.012	33	0.03	69	0.10	8
Snake							
SNR-148	0.023	0.067	54	0.35	97	0.65	11
SNR-140	0.022	0.063	52	0.33	97	0.52	11
SNR-129	0.015	0.045	51	0.18	96	0.33	11
SNR-118	0.014	0.042	44	0.19	95	0.31	9
SNR-108	0.014	0.036	36	0.13	94	0.31	10
SNR-106	0.014	0.046	31	0.16	90	0.37	10
SNR-83	0.013	0.043	33	0.16	94	0.33	9
SNR-68	0.015	0.045	42	0.18	88	0.35	9
SNR-50	0.015	0.040	41	0.17	93	0.34	9
SNR-40	0.023	0.040	53	0.18	96	0.41	10
SNR-18	0.021	0.041	49	0.18	96	0.40	10
SNR-6	0.016	0.049	36	0.17	96	0.41	9
Columbia							
CLR-397	0.002	0.010	18	0.07	88	0.15	15
CLR-369	0.002	0.013	13	0.07	90	0.18	14
CLR-326	0.002	0.013	13	0.08	89	0.17	13
CLR-306	0.005	0.022	23	0.11	91	0.26	11
CLR-295	0.004	0.025	20	0.13	93	0.24	10
Tributaries							
PAL-6	0.024	0.143	22	1.38	99	2.15	14
TUC-1	0056	0.082	66	0.21	96	0.35	4

*Percentages estimated since some of the NH3 concentrations were less than the instrument detection limit

Total phosphorus (TP) concentrations, like total nitrogen, were lowest in the Clearwater River and the Columbia River, and greatest in the Snake River (Figure 3.6-7). Median concentrations at CLW-1, CLW-11, CLR-326, CLR-369, and CLR-397 were all between 0.010 mg/L and 0.013 mg/L, although the initial spring values were higher at the Columbia River sites; 0.028 mg/L as opposed to 0.018 mg/L at the Clearwater River sites. The concentrations determined at SNR-140 and SNR-148 were usually greater than those in either of the other two major rivers with values ranging from 0.034 mg/L in July to 0.111 mg/L in June. The stations in the Lower Snake River were not significantly different from one another with regards to TP, but did exhibit the familiar U-shaped trend of intermediate concentrations (0.051 to 0.081 mg/L) in the spring, low concentrations (0.018 to 0.032 mg/L) in late July, and highest concentrations (0.059 to 0.103 mg/L) in the fall.

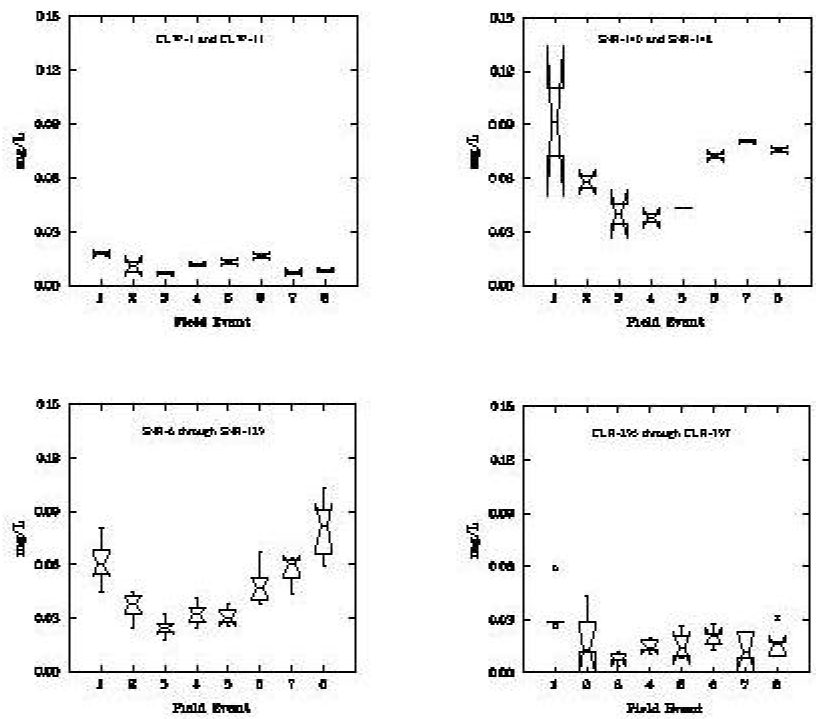

Figure 3.6-7. Box plots of total phosphorous concentrations from 1 meter at the Clearwater River, Snake River, and Columbia River sampling stations

Total phosphorus concentrations in the Palouse River and Tucannon River were greater than those determined for the main river sections. The seasonal median concentration in the Tucannon River was 0.082 mg/L while the Palouse River was almost twice as high at 0.143 mg/L. These median values were based on data points that ranged from 0.062 mg/L to 0.212 mg/L in the Tucannon River and 0.100 mg/L to 0.287 mg/L in the Palouse River. In both cases, low concentrations occurred in October while the peak values were noted during the spring freshet.

When the percentage of the total phosphorus that was orthophosphorus was compared, some noticeable differences were apparent between the rivers. At the three upper Columbia River sites, median values ranged from 13% to 18%, and only exceeded 30% on two occasions at CLR-397. The Clearwater River had somewhat higher percentages, with median values close to 40% through mid July, declining to between 8% and 15% during the next three sampling events, and then approaching the original values during the remaining two field trips. Percentages in the Upper Snake River were below 20% at the beginning of the season, but rose to 77% by September, and then declined slightly. The trend in the Lower Snake River was essentially the same. Although the first field trip showed a higher percentage (24%), and trips four, five, and six were lower, possibly due to the influence of the Clearwater River.

The TN:TP ratios in the Clearwater River and Snake River were typically close to 10, while those on the Columbia River and Palouse River averaged between 13 and 17. The lowest ratios were calculated for the Tucannon River, which ranged from 2 to 8 and had a median of 4.

Nitrogen

The concentration distributions of the nitrogen series, as well as the phosphorus series, share several similarities to those already discussed. Of the soluble inorganic nitrogen species evaluated, NO_3+NO_2 nitrogen (here after referred to simply as NO_3) was the primary component, often comprising greater than 90% of the total (Table 3.6-3). The lowest median concentrations and the lowest detected concentration, 0.01 mg/L, were associated with the Clearwater River. Furthermore, NO_3 concentrations in this river did not vary appreciably during the study period and did not exceed 0.05 mg/L. In contrast, the concentrations at the upper Snake River sites were higher than those encountered at the Lower Snake River stations or in the Columbia River. Values ranged from 0.12 mg/L in early June to 0.87 mg/L in early October (Figure 3.6-8). The sampling stations in the Lower Snake River did not differ markedly from one another with respect to NO3 concentrations. Median values ranged from 0.13 mg/L to 0.19 mg/L, and there did not appear to be a rising or declining trend as the distance down stream increased. Concentrations, however, remained relatively constant from early June through mid August and then rose rapidly as discharge decreased; the increase that was observed at the SNR-140 and SNR-148 was evidently moderated by the seasonally large volume of water in the Clearwater River. The Columbia River monitoring sites exhibited NO_3 concentrations that were intermediate between those in the Snake River and Clearwater River (Table 3.6-3). The stations at CLR-369 and CLR-397 generally had lower values than the sites downstream, and only showed a slight increase at the end of the season.

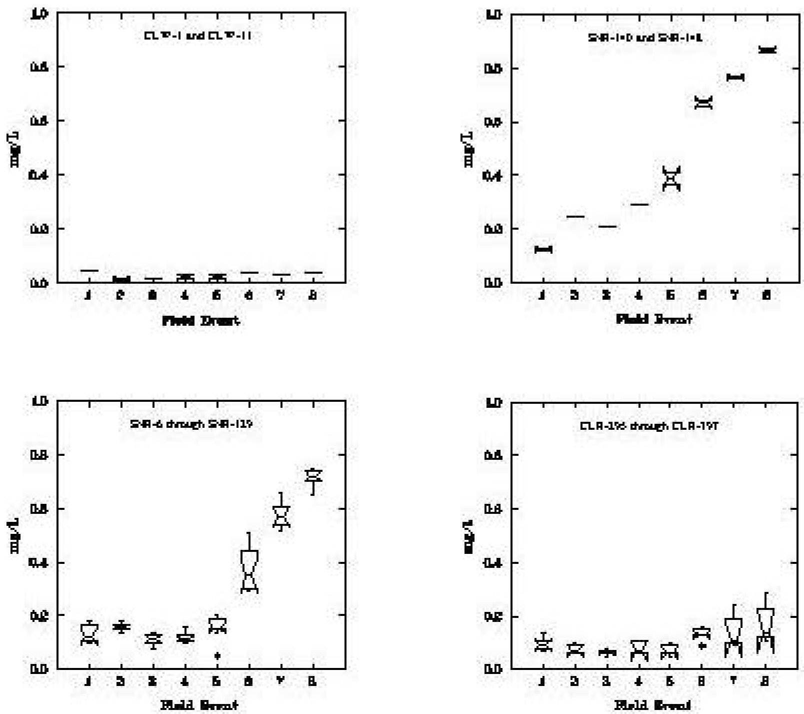

Figure 3.6-8. Box plots of nitrite plus nitrate concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

The Palouse River and Tucannon River had very distinct NO_3 distributions. The Palouse River had the highest NO_3 concentration measured, 2.69 mg/L, which was recorded in June. The concentrations decreased until mid August when they reached 0.30 mg/L, but then rose again to 1.75 mg/L by early October. The Tucannon River also had it’s peak concentration, 0.43 mg/L, in early June and declined to 0.11 mg/L in mid August, but only rose to 0.21 mg/L at the end of September.

Total nitrogen concentrations followed some of the same trends set by NO3 in the Snake River, but not necessarily in the Clearwater and Columbia Rivers (Figure 3.6-9). The Clearwater River did have the lowest median concentrations, 0.08 mg/L and 0.10 mg/L at CLW-11 and CLW-1, respectively, but the highest concentrations (0.16 mg/L) were determined during spring runoff and the lowest occurred in October. The reason for this trend is that samples for total nitrogen analyses are digested as part of the analytical process. Nitrogenous compounds associated with elevated suspended solid concentrations in spring would, be digested by this process.

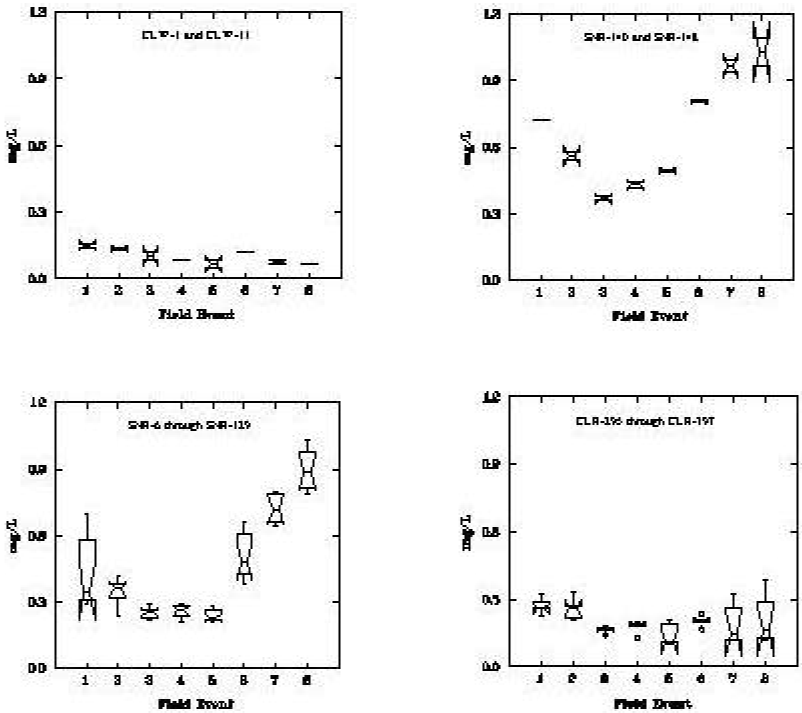

Figure 3.6-9. Box plots of total nitrogen concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

The Upper Snake River sites again had the highest overall concentrations of total nitrogen (Table 3.6-3) and displayed seasonal variation. At SNR-148, for example, 0.72 mg/L observed in the spring, declined to 0.38 mg/L by mid July, and then quickly rose to 1.09 mg/L by October. In the Lower Snake River, concentrations were slightly lower than those in the reaches above the confluence of the Clearwater River, and there appeared to by a slight, but not significant, increase in concentration from upstream to downstream. In the Columbia River, however, concentrations did increase noticeably from Priest Rapids Dam (CLR-397) to McNary Dam (CLR-295). The three sites in the main stem above the confluence generally had high concentrations ranging from 0.23 to 0.26 mg/L in the spring and 0.12 to 0.16 mg/L in the fall. The two sites below the confluence showed the influence of the Snake River concentrations with spring-time values of approximately 0.30 mg/L, declining by almost one half during the summer, and then rising again as fall approached.

Major Ions

The major anions and cations that were evaluated as part of the study included Ca, Mg, Na, K, Cl, SiO_2, and SO_4 (Table 3.6-4). With the exception of SiO2, the concentrations of these ions (Figures 3.6-10 through 3.6-15) followed the same pattern established by specific conductance with: 1) low concentrations in the Clearwater River; 2) comparable concentrations in the Lower Snake River and Columbia River, with slightly elevated values below the confluence; 3) higher concentrations in the Palouse River for virtually every ion evaluated; and 4) increasing concentrations as the season progressed and the flows diminished.

Table 3.6-4
Median Values For Selected Ions, Suspended Solids, and Turbidity Collected at 1 M (0.2 M at the Tributaries) From the Sites Monitored During the 1997 Sampling Period

System/Site	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	Cl (mg/L)	SiO_2 (mg/L)	SO_4 (mg/L)	Susp. Solids (mg/L)	Turbidity (NTU)
Clearwater									
CLW-11	--	--	--	--	--	--	--	2	--
CLW-1	3.0	0.7	1.4	0.5	0.4	10.6	1.6	2	2
Snake									
SNR-148	--	--	--	--	--	--	--	6	--
SNR-140	29.7	8.2	14.1	2.6	7.4	13.4	21.7	5	2
SNR-129	16.4	4.5	8.5	1.7	4.5	11.9	12.4	5	2
SNR-118	17.0	4.7	8.1	1.6	4.4	12.0	12.2	5	2
SNR-108	16.9	4.6	7.8	1.6	4.3	11.9	12.3	6	3
SNR-106	--	--	--	--	--	--	--	5	--
SNR-83	17.0	4.4	7.9	1.5	4.3	12.1	11.8	5	3
SNR-68	--	--	--	--	--	--	--	4	--
SNR-50	--	--	--	--	--	--	--	5	--
SNR-40	--	--	--	--	--	--	--	5	--
SNR-18	16.4	4.4	8.0	1.6	4.5	12.4	11.4	4	4
SNR-6	--	--	--	--	--	--	--	6	--
Columbia									
CLR-397	--	--	--	--	--	--	--	2	--
CLR-369	-	--	--	--	--	--	--	2	--
CLR-326	17.9	4.4	2.5	0.8	1.0	5.7	7.9	5	2
CLR-306	17.4	4.4	4.3	1.1	2.1	7.5	8.8	4	2
CLR-295	17.7	4.5	4.7	1.1	2.2	7.9	9.0	6	3
Tributaries									
PAL-6	37.4	14.8	25.9	5.1	11.4	22.8	19.0	21	19
TUC-1	16.6	6.5	6.2	2.8	1.9	40.6	3.5	9	2

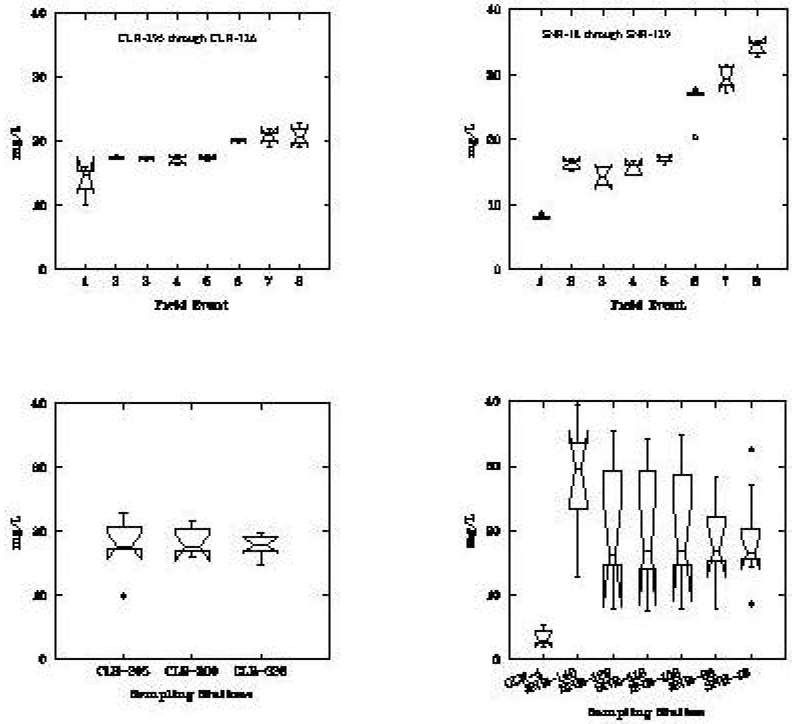

Figure 3.6-10. Box plots of calcium concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

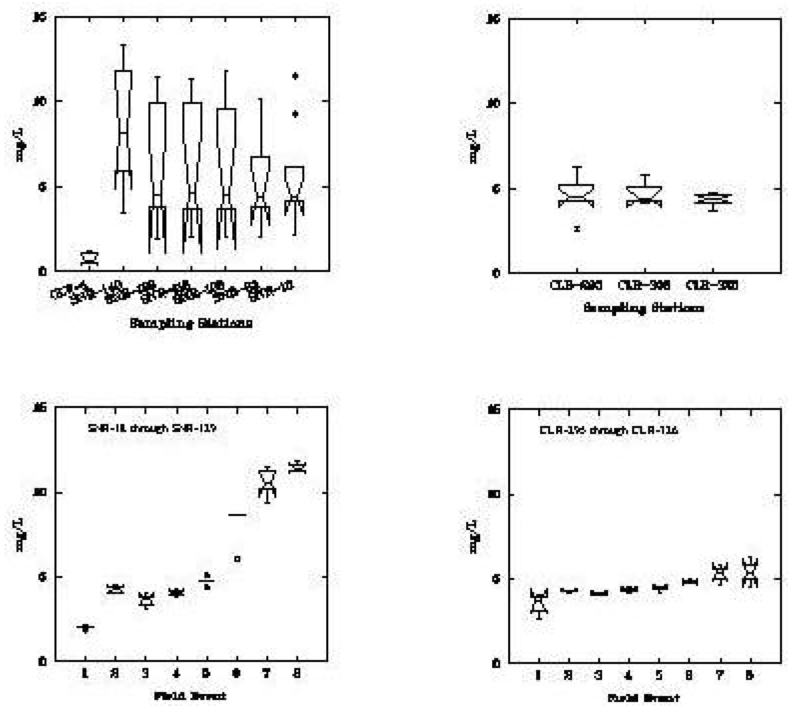

Figure 3.6-11. Box plots of magnesium concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

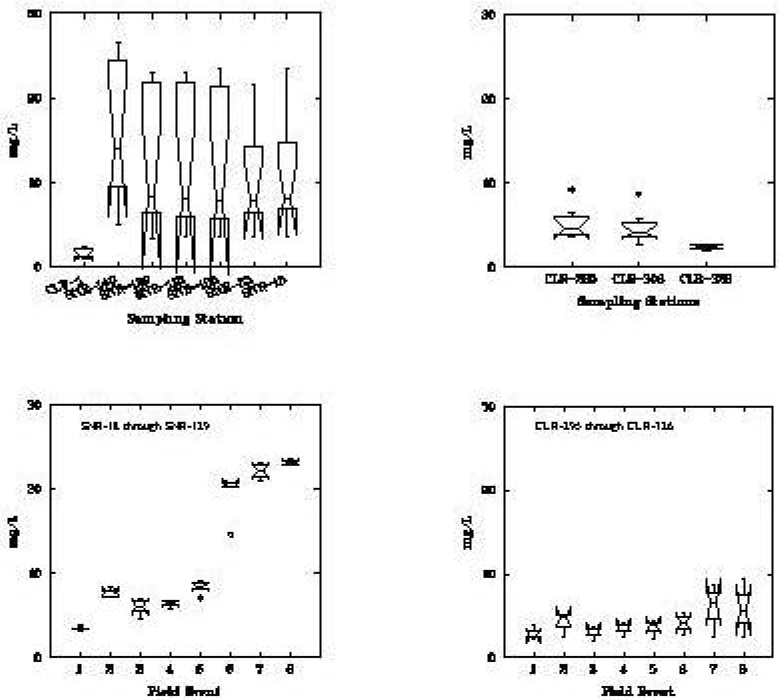

Figure 3.6-12. Box plots of sodium concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

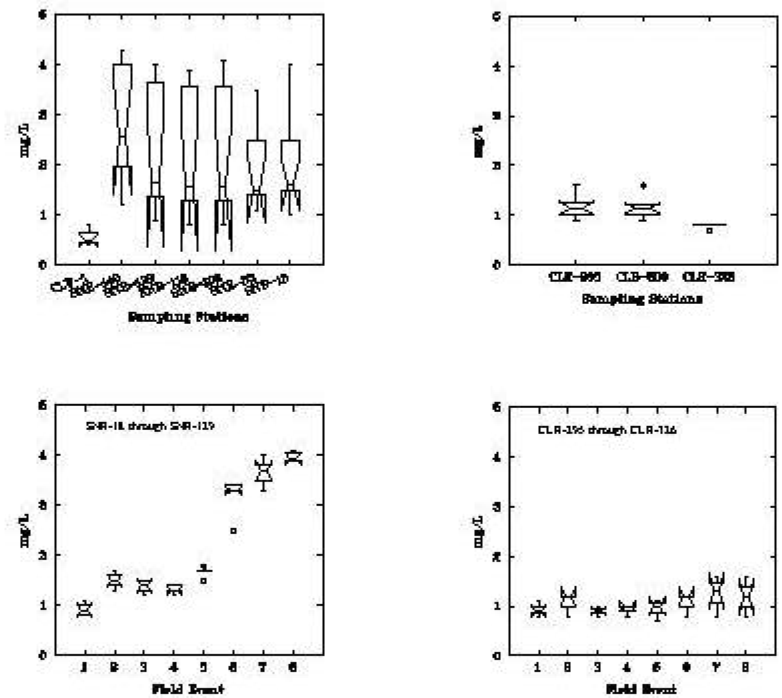

Figure 3.6-13. Box plots of potassium concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

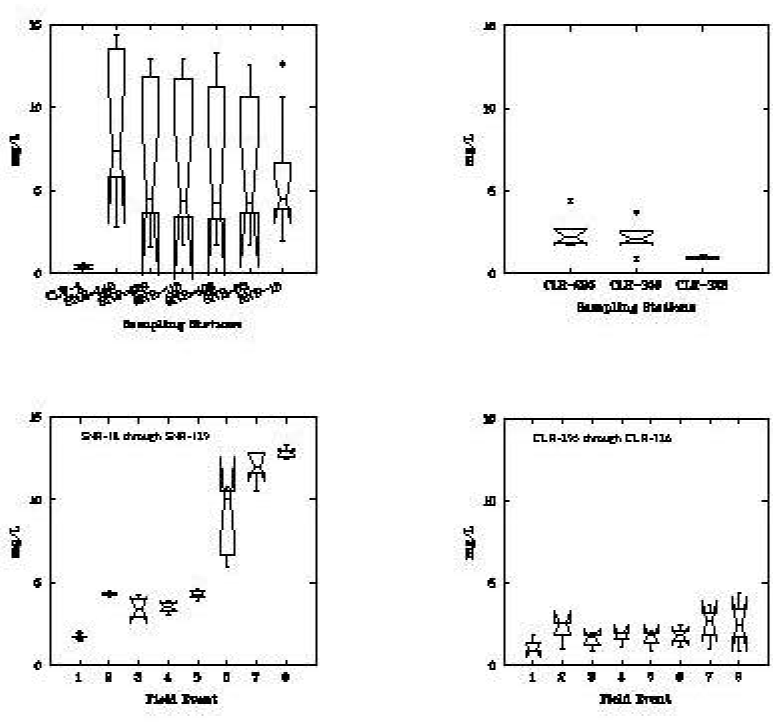

Figure 3.6-14. Box plots of chloride concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

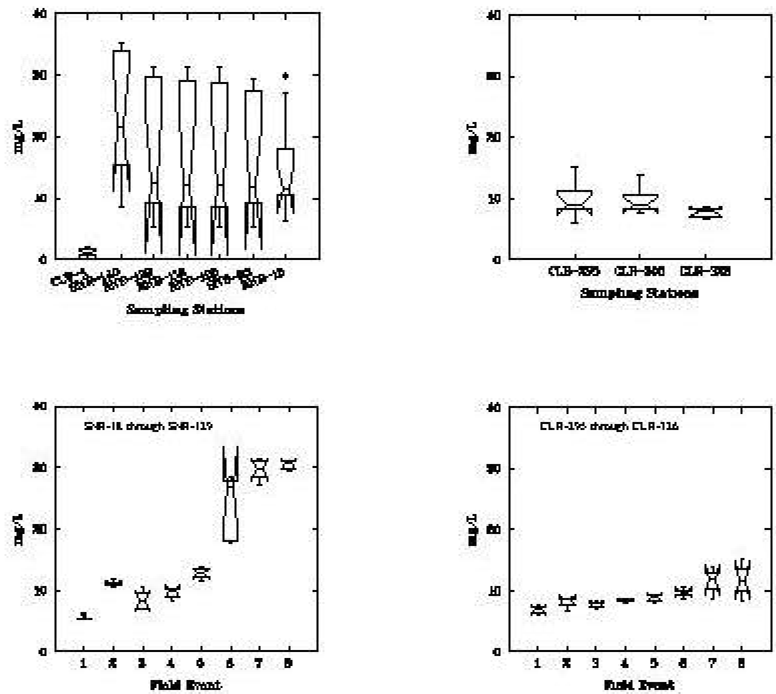

Figure 3.6-15. Box plots of sulfate concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

The spatial and temporal patterns for the SiO2 concentrations differed in several ways from those determined for the other ions. First, the seasonal median concentration in the Clearwater River was 10.6 mg/ L (Table 3.6-4). This value was not significantly less than those determined for the majority of the Snake River stations, which ranged from 11.9 to 13.4 mg/ L. Second, the median concentrations in the Columbia River ranged from 5.7 to 7.9 mg/ L, and were statistically lower than the ones from the same time period in the Snake River (Figure 3.6-16). Third, seasonal concentrations in the Lower Snake River followed a now familiar pattern; decreasing slightly from early June to mid July, increasing marginally in late July and August, and then rising substantially for the remainder of the field season. Conversely, in the Columbia River the concentrations were greatest (approximately 10 mg/ L) in early June, declined by early July, and then remained in the 6 to 8 mg/ L range through early October.

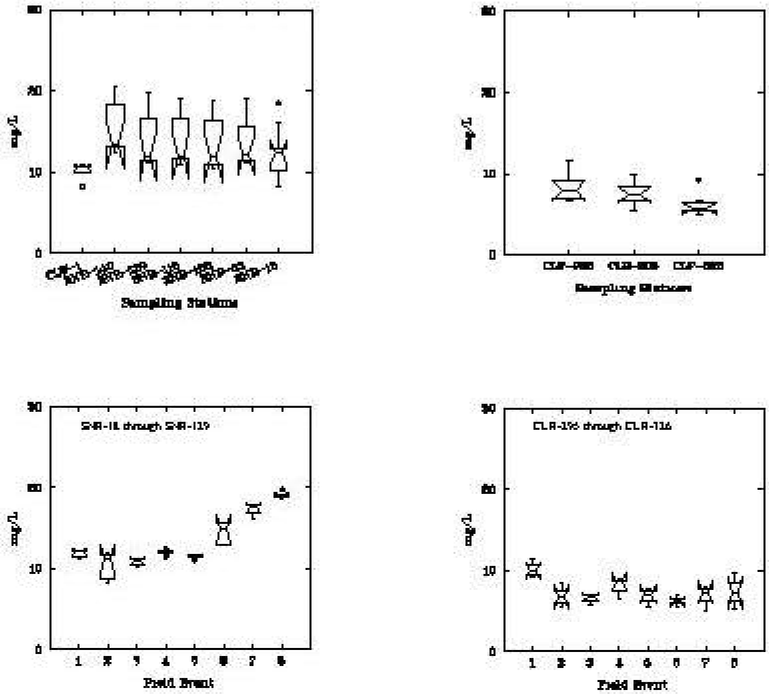

Figure 3.6-16. Box plots of silica concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

Biochemical and Sediment Oxygen Demand

Five day biochemical oxygen demand (BOD5) was evaluated on samples collected from the photic zones at sites where primary productivity was determined, and at CLR-326. When the data for the entire sampling season is plotted, there is a trend of decreasing BOD5 values traveling downstream from CLW-11 and SNR-148 to SNR-6 (Figure 3.6-17). The median concentrations at CLW-11 and SNR-148 were 2.0 mg/L and 1.7 mg/L, respectively, and approximately 1.0 mg/L from SNR-83 to SNR-6. The Lower Snake River values increased slightly during the second series of field trips and then declined for the remainder of the season; the exception was the mid-August period (Trip 5) when the median concentration was about two times greater than it had been during the previous trip (Figure 3.6-17). There also appears to be a very slight downstream decrease in the Columbia River, where a median concentration of 1.6 mg/L was calculated for CLR-397 compared to 1.3 mg/L at CLR-295. The seasonal pattern in this river was similar to the one in the Lower Snake system, albeit without the mid-August increase.

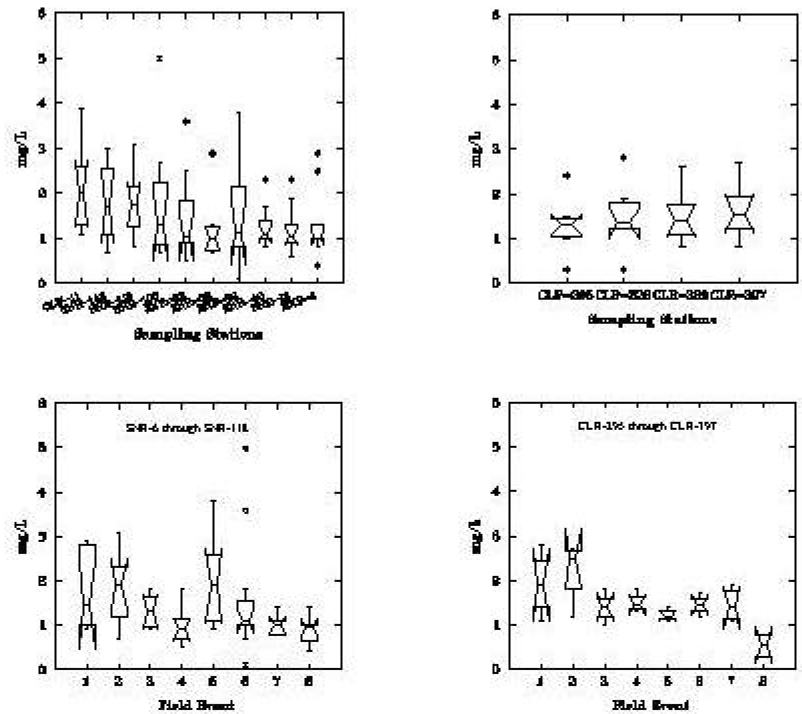

Figure 3.6-17. Box plots of BOD concentrations from the photic zones at the Clearwater River, Snake River, and Columbia River sampling stations

The results of the BOD_5 analyses completed for the Tucannon River were very similar to those described for the Lower Snake River and Columbia River, but quite different from those for the Palouse River. The concentrations in the Tucannon River ranged from 0.5 mg/L to 3.3 mg/L, and the median was 0.9 mg/L. In the Palouse River, the minimum value was 1.7 mg/L while the maximum was 10.7 mg/L, giving a median of 3.6 mg/L.

Sediment oxygen demand was determined twice during the project at three locations; SNR-50, SNR-123, and SNR-132. The results for the samples collected at any site during the same sampling period were very similar, but there was a noticeable difference between the first and second set of cores. The oxygen depletion for the cores collected during August showed a very minor increase moving downstream, with values ranging from 0.8 $gm/m^2/d$ at SNR-132 and SNR-123, to 0.9 $gm/m^2/d$ at SNR-50. The October samples provided the same trend, albeit with higher depletion rates; 1.9 $gm/m^2/d$ at SNR-132, 2.1 $gm/m^2/d$ at SNR-123, and 2.2 $gm/m^2/d$ at SNR-50. The larger values observed for the second set may have been due to the slower flow rates and accumulation of organic matter at the sediment/water interface.

Suspended Solids and Turbidity

Within the region studied, the concentrations of suspended solids tended to be lowest in the Clearwater and Upper Columbia River sites and approximately equal at the remaining sites when the entire data set is evaluated (Table 3.6-4). Furthermore, there was a general increase in concentrations at the downstream stations in the Columbia River and a slight, though not as noticeable, augmentation at the four Lower Snake River sites.

The suspended solids concentrations followed the same temporal pattern at all of the sites; greatest values in the spring during peak runoff and diminishing synchronously with discharge (Figure 3.6-18). The stations on the Clearwater River, as well as those at CLR-369 and CLR-197, had the lowest maximum values of 16, 10, and 8 mg/L at CLW-11, CLR-369, and CLR-397, respectively. Peak spring-time concentrations at SNR-148 reached 65 mg/L, and averaged 28 mg/L in the surface waters of the Lower Snake River during the same time interval. Concentrations were usually less than 5 mg/L by mid to late summer, with possible exceptions at SNR-6, SNR-108, and CLR-295 which were slightly greater.

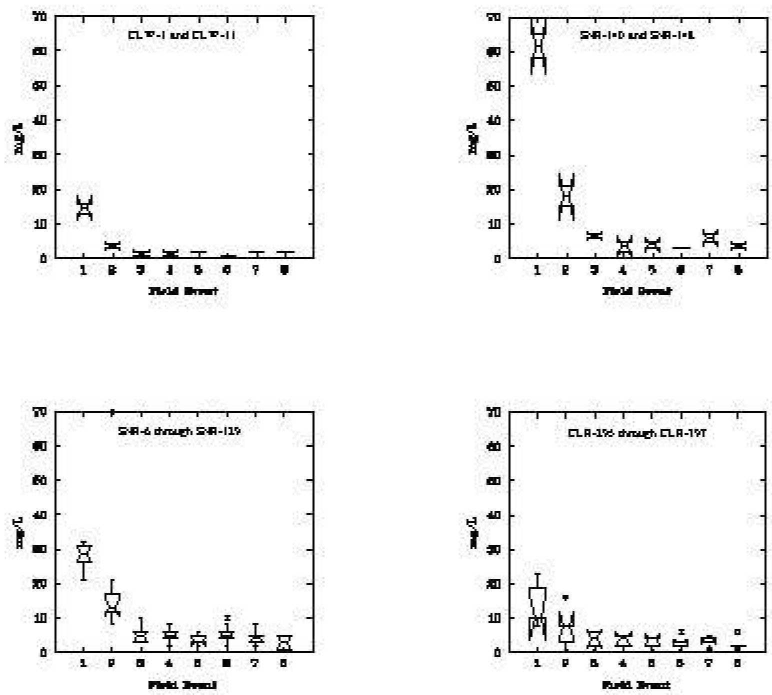

Figure 3.6-18. Box plots of suspended solids concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

The suspended solids concentrations in the Palouse River and Tucannon River were frequently greater than those at the other sampling sites during the same sampling events. The Palouse River, in particular, had quite high concentrations (1,035 mg/L) in early June, and the lowest values were 13 mg/L. The Tucannon River had the second highest spring runoff concentration (130 mg/L), but remained below 10 mg/L after mid-August.

Chlorophyll a

The concentrations of chlorophyll *a* determined for the sites on the Clearwater River and Snake River had greater spatial and temporal variability than those from the Columbia River. The station at CLW-11 had the lowest median concentration (3.13 *u*g/L), while the site at SNR-108 was found to have the highest median value (8.74 *u*g/L) within the Lower Snake River system (Figure 3.6-19). Furthermore, there was a decline in the median concentration of chlorophyll *a* from SNR-83 (7.35 *u*g/L) to SNR-18 (3.20 *u*g/L). Though not universal among the stations, maximum concentrations were often observed during the first or second sampling event, while minimum values occurred between mid-August and mid-September. Thus, the median concentration at the Lower Snake River sites during the second sampling event was about 9.5 *u*g/L and had declined significantly to about 4.5 *u*g/L by early September.

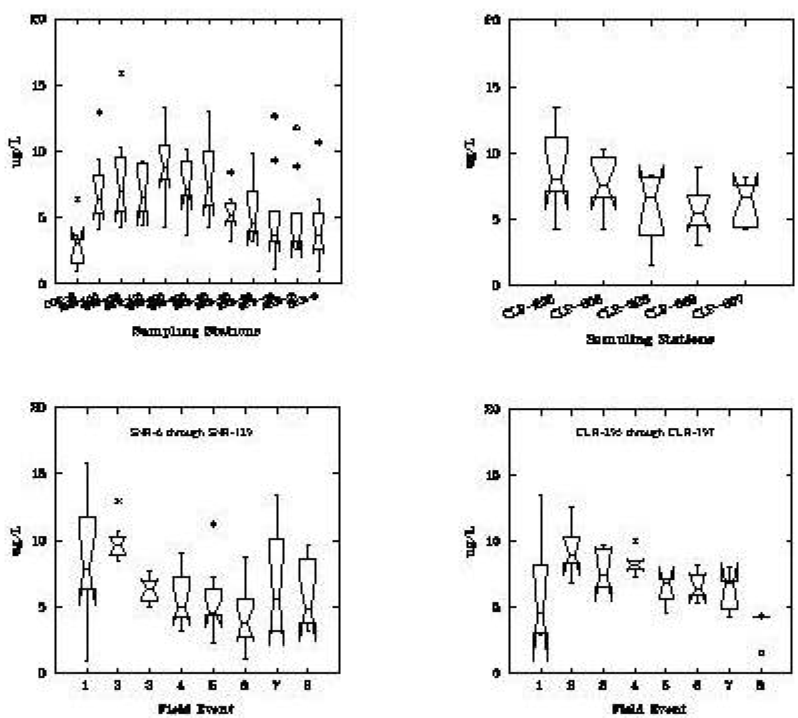

Figure 3.6-19. Box plots of chlorophyll *a* concentrations from 1 m at the Clearwater River, Snake River, and Columbia River sampling stations

Chlorophyll *a* concentrations at the Columbia River sites are not significantly different from the Lower Snake River values when the entire sampling season is considered. The median concentration at CLR-397 was 6.72 *u*g/L and gradually increased to 8.01 *u*g/L at CLR-295. This spatial difference was minor and not great enough to be considered significant, even though the maximum values were recorded for CLR-295 (13.43 *u*g/L) and CLR-306 (10.35 *u*g/L). The seasonal trend at the Columbia River locations was similar to the one noted for the Lower Snake River; a median value of 4.5 *u*g/L during the initial sampling event, an increase to 8.90 *u*g/L by trip two, and a gradual reduction to 4.27 *u*g/L by the end of the season (Figure 3.6-19).

3.6.3 - Biology

Phytoplankton

The phytoplankton communities in each of the river systems exhibited spatial and seasonal patterns that, to some extent, resembled the ones established by some of the inorganic parameters. Total biovolume was significantly lower at the CLW-11 site than it was at most of the other Snake River and Columbia River stations (Table 3.6-5; Figure 3.6-20). The biovolumes determined for SNR-148 displayed a wide range of values, but overall they were not significantly different from those determined for the Lower Snake River sites. Biovolume appeared to increase slightly from SNR-129 to SNR-106, declined sharply at SNR-83, and remained constant for the remainder of the Lower Snake River. There was also a slight increase in biovolume downstream from CLR-397 in the Columbia River. The general temporal pattern was to have low algal biovolume during peak runoff, a significant increase by the end of June at many of the stations, followed by a decrease to seasonal lows in August; a pattern that departs from the trend of increasing biovolume as the season progresses that is often observed in other aquatic systems. Following this minimum period, the phytoplankton population increased again at most stations during late September before declining once more, or staying at about the same level, in October.

Table 3.6-5
Median Biovolume and Average Percent Composition
Of the Dominant Phytoplankton Divisions at Each Monitoring Station During the 1997 Sampling Season

System/Site	Median Biovolume (um^3/ mL)	Bacillariophyta (Percent)	Chlorophyta (Percent)	Cryptophyta (Percent)	Cyanophyta (Percent)	Dinophyta (Percent)
Clearwater						
CLW-11	180,719	78	1	2	13	5
Snake						
SNR-148	578,980	92	3	2	1	2
SNR-129	490,967	88	3	4	1	4
SNR-118	519,242	90	3	3	1	4
SNR-108	639,820	85	2	7	1	5
SNR-106	738,062	95	1	2	0	1
SNR-83	419,559	87	3	6	0	4
SNR-68	484,236	88	3	4	1	4
SNR-50	332,289	88	1	6	2	2
SNR-40	323,697	87	2	5	0	6
SNR-18	359,894	88	1	7	0	4
SNR-6	360,339	86	1	5	0	8
Columbia						
CLR-397	465,433	91	2	2	1	4
CLR-369	488,886	91	1	4	0	4
CLR-326	520,467	86	3	4	1	5
CLR-306	626,033	90	2	3	1	4
CLR-295	674,104	91	2	3	1	4

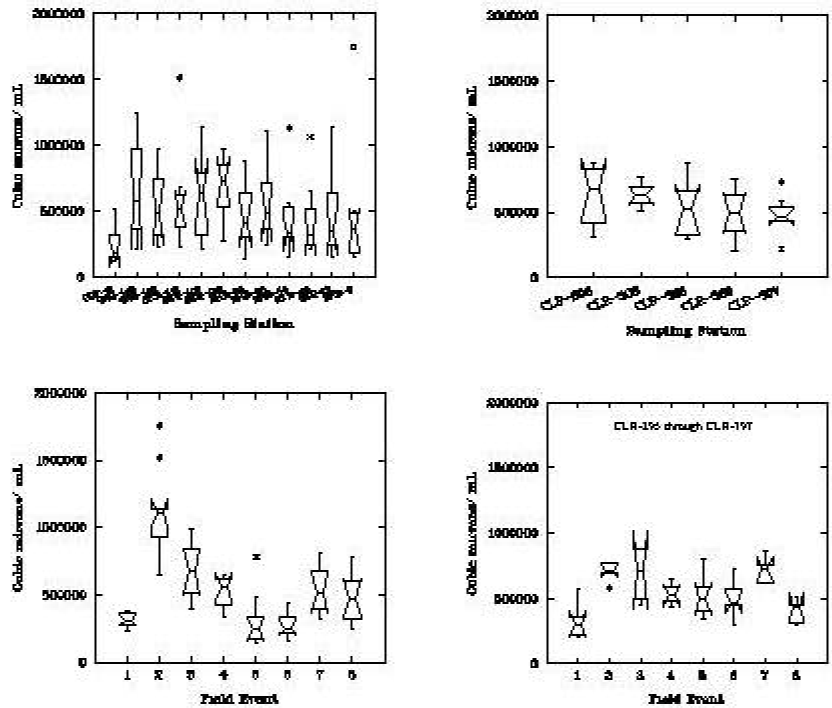

Figure 3.6-20. Box plots of total phytoplankton biovolume from the photic zones at the Clearwater River, Snake River, and Columbia River sampling stations

The composition of the phytoplankton community in the Clearwater River showed some similarities to those in the other river systems, but also several dissimilarities. The median biovolume for all of the algae combined was lower at CLW-11 than at any other station (Table 3.6-5), displaying a relatively small peak in the spring, reduced values during the summer, and a resurgence in September and October (Figure 3.6-21); however, their percent contribution (Figure 3.6-22) and species composition did change during the year. *Cyclotella meneghiniona, Melosira granulata,* and *Melosira islandica* were not only the major diatoms present in early June, but combined to occupy about 71% of the total biovolume with 81,456, 71,436, and 76,167 *u*m^3/mL, respectively. These three species were not consistently present during the remainder of the season, but did intermittently represent 10% to 15% of the total biovolume during mid season. *Cyclotella meneghiniona* reached a second maxima in early October when their biovolume rose to 157,412 *u*m^3/mL (85 cells/mL). At the same time, *Melosira granulata* and *Melosira islandica* declined to relatively low values of 15,233 *u*m^3/mL (42 cells/mL) and 37,766 *u*m^3/mL (111 cells/mL). The diatoms that were consistently present during mid season included *Asterionella formosa, Cymbella* spp., and *Eunotia* spp. *Asterionella formosa* was present (25 to 47 cells/mL) at the beginning of the season, but did not reach their peak until mid July when they claimed almost 34% (42,653 *u*m^3/mL and 68 cells/mL) of the total biovolume. Their population declined after mid summer, they were not identified in September, but were again present in early October at 53 cells/mL (30,678 *u*m^3/mL). *Cymbella* spp. and *Eunotia* spp., on the other hand, were minor components of the algal community in the early part of the season and became more prevalent after July. *Cymbella* spp. Reached a summer maxima of 24,278 *u*m^3/mL (61 cells/mL). Although more *Cymbella* spp. were present in the fall, their percent contribution to the total biovolume was lower; 7.4% versus 14.8% due to the greater diversity and total biomass of diatoms during October. *Eunotia* spp. showed a rapid population surge in late July to 25 cells/mL, increased to 61 cells/mL by mid-August, declined along with the other diatoms in early September to 11 cells/mL, and then reached a seasonal high of 115 cells/mL (123,771 *u*m^3/mL or 37% of the total biovolume) in late September. Two final diatoms that were periodically plentiful were *Epithemia* spp. in late June, and *Fragillaria crotonensis* in late July and early September. The relatively large fraction of Cyanophytes in the Clearwater River set it apart from any other station monitored. *Anabaena* spp. were present during all but one of the sampling events, with notable peaks in mid July (29,716 *u*m^3/mL or 23.4% of the total), mid August (34,762 *u*m^3/mL or 18.5% of the total), and early September (83,925 *u*m^3/mL or 48.0% of the total). The Cryptophyta, primarily as *Rhodomonas minuta*, were also observed in almost all of the samples, but their contribution to the total biovolume was fairly small, ranging from less than 1% in September to almost 6% in late June. *Ceratium hirundinella*, *Gymnodinium caudatum*, and *Peridinium inconspicuum* were the most frequently detected Dinophytes, but their population also was sporadic and almost never exceeded 5% of the total biovolume. The exception was *Gymnodinium caudatum*, which comprised 18.3% of the total biovolume in late June, despite the fact they only numbered 7 cells/mL. Chlorophytes were only identified during two sampling trips; once in June, when *Scenedesmus bijuga* was found at less than 1 cell/mL; and again in September, when *Scenedesmus* spp. combined to number 84 cells/mL, or 10% of the total biovolume.

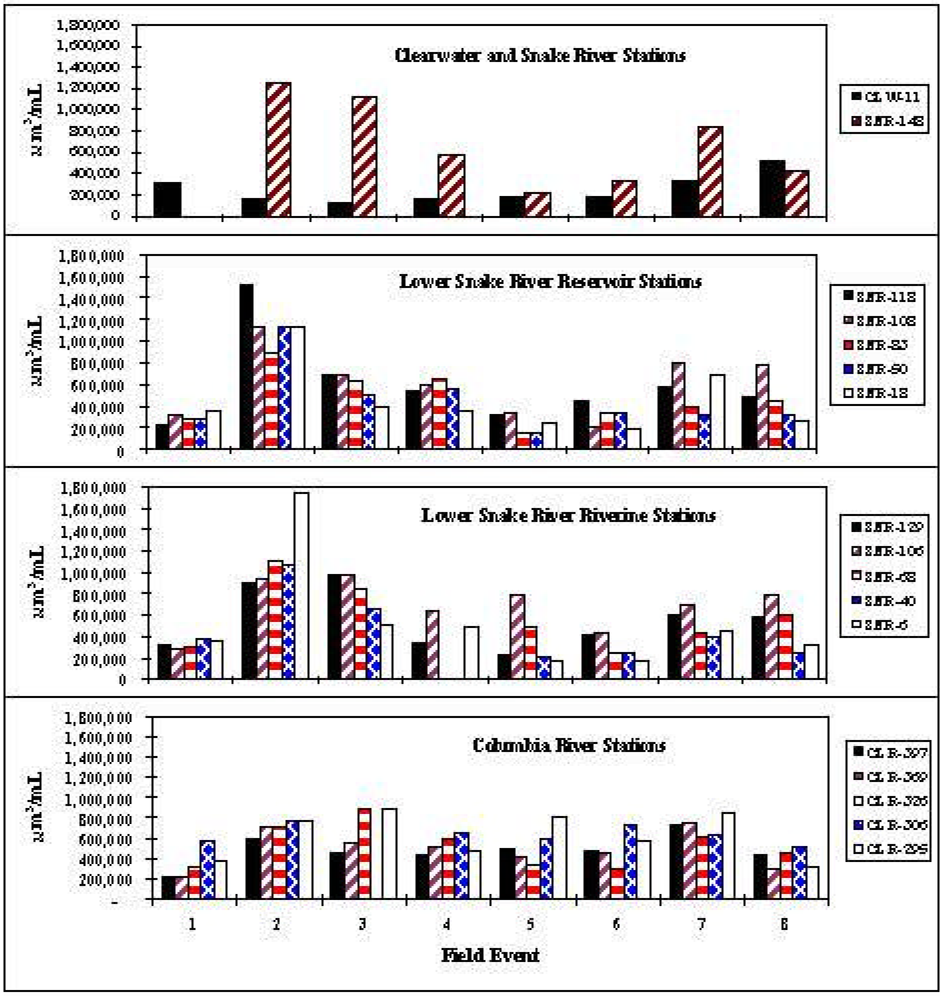

Figure 3.6-21. Total phytoplankton biovolume from the photic zones in the Clearwater River, Snake River, and Columbia River monitoring stations

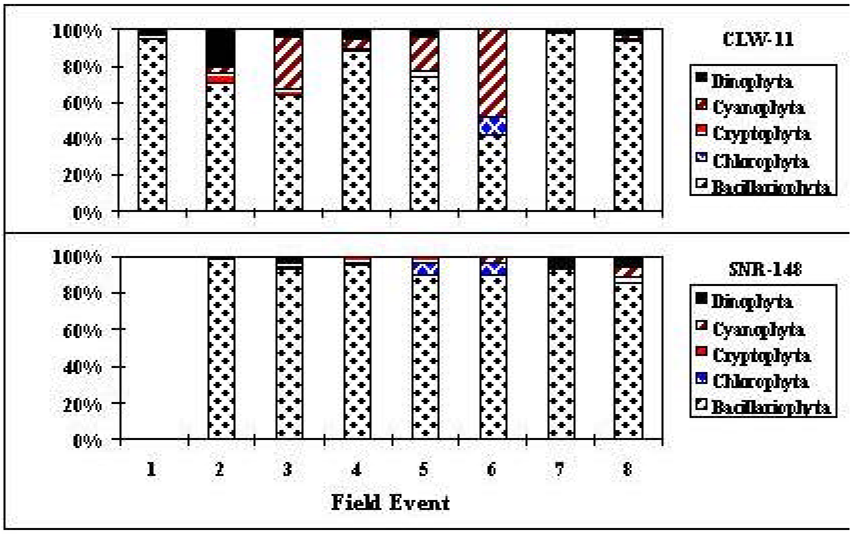

Figure 3.6-22. Percentages of the dominant phytoplankton divisions at CLW-11 and SNR-148 during each field event

The phytoplankton community in the inflowing Snake River differed from the one just detailed for the Clearwater River in several respects. First, the median biovolume was almost four times greater at SNR-148 than at CLW-11 (Table 3.6-5). Second, the Bacillariophytes almost always comprised greater than 90% of the total biovolume, with characteristic peaks in the spring and fall, and a seasonal minimum in August (Figure 3.6-22). *Cyclotella meneghiniona* was identified in all of the samples analyzed, and it was typically the dominant diatom. Spring maximums reached 624 cells/mL (898,769 um^3/mL or 72.0% of the total biovolume) and 365 cells/mL (770,079 um^3/mL or 69.2% of the total) in June and July, respectively, and a fall peak of 206 cells/mL (396,068 um^3/mL or 47.7% of the total biovolume) in late September. *Melosira islandica* was often the second most prevalent diatom. However, unlike *Cyclotella meneghiniona* their density was relatively low (66 to 101 cells/mL) in the spring. Their biovolume did increase during July and August, when they comprised 16% to 17% of the total biovolume, and in late September at 503 cells/mL (170,847 um^3/mL or 20.6% of the total biovolume). *Fragillaria crotonensis* were present at the beginning of the field season, but as in the Clearwater River they developed to their fullest extent by late July when they numbered 239 cells/mL (76,505 um^3/mL or 17.2% of the total biovolume). *Asterionella formosa* were only identified through mid-July, when they peaked at 169 cells/mL, which was only 106,634 um^3/mL, or less than 10% of the total biovolume. *Cymbella* spp. was another group that occurred in early summer (45 cells/mL or 24,278 um^3/mL in late June), attained a local maximum in the summer (99 cells/mL or 50,714 um^3/mL in late July), and was marginally represented in the fall (13 cells/mL or 7,998 um^3/mL in October. *Eunotia* spp. and *Epithema* spp. were both late season species, and individually occupied approximately 30,000 um^3/mL in September--still less than 5% of the total biovolume. One of the reasons that the Bacillariophytes comprised a greater percentage of the total algal biovolume in the Snake River as compared to the Clearwater River was the virtual absence of Cyanophytes. The greatest abundance of blue-greens was reached in October, when *Anabaena* spp., *Chroococcus* spp., and

Oscillatoria spp. combined to total 21,083 *u*m^3/mL, but that was still only 5% of the total algal biovolume during that sampling event. Even during early September, when *Anabaena* spp. reached 83,925 *u*m^3/mL in the Clearwater River, they only occupied 10,170 *u*m^3/mL, or 3.1% of the total in the Snake River. The Dinophytes, *Glenodinium armatum* and *Gymnodinium caudatum*, were noted in the mid-July samples at 11 and 5 cells/mL, respectively. The same two species, along with *Glenodinium palustre*, were absent until they reappeaed in late September, but still did not individually exceed 4% of the total biovolume. The Cryptophyte, *Rhodomonas minuta*, was ubiquitous and sometimes numerous (up to 130 cells/mL), but, as was the case with the Dinophytes, never exceeded 4% of the total biovolume. Finally, the Chlorophytes were omnipresent in the Snake River. *Pediastrum duplex* attained 74 cells/mL (14,218 *u*m^3/mL or 4.3% of the total) in early September, while *Scenedesmus* spp. reached 79 cells/mL (12,772 *u*m^3/mL or 1.5% of the total) later that month; otherwise, the densities of the green algae were minor.

The Bacillariophytes dominated the total biovolume at each station within the Lower Snake River segment both on a trip-by-trip basis and for the project period as a whole, and the temporal pattern and species composition of the communities reflected the pattern established at SNR-148 (Figure 3.6-22). *Cyclotella meneghiniona, Fragillaria crotonensis*, and *Melosira islandica* were the three diatom species that were consistently present, and either singularly or combined comprised the greatest proportion of the total biovolume (Figures 3.6-23 and 3.6-24). *Cyclotella meneghiniona* was, by far, the species with the greatest biovolume, displaying early-summer maximums, mid-summer lows, and fall secondary peaks. With the exception of SNR-6 during the second field event, the early summer biovolume appeared to be greatest at the stations between SNR-118 and SNR-68, and declining after SNR-50. Late June/ early July values ranged from 437,706 um3/mL at SNR-18 to 1,056,180 *u*m^3/mL at SNR-118. Summer minimums ranged from 5,655 *u*m^3/mL at SNR-18 *u*m^3/mL in August to 102,983 *u*m^3/mL at SNR-106 in late July. The second seasonal maximum reached 138,116 *u*m^3/mL at SNR-40 and 519,458 *u*m^3/mL at SNR-108 during early October. *Melosira* spp., mainly *Melosira islandica* but also including *Melosira granulata*, also showed a slight spatial trend. The stations between SNR-129 and SNR-68 experienced maximum seasonal biovolumes for this genus between mid-to-late summer, with values ranging from 137,947 *u*m^3/mL at SNR-83 to 331,987 *u*m^3/mL at SNR-106. The populations at SNR-50, SNR-40, and SNR-6 displayed early July maximums ranging from 186,053 *u*m^3/mL at SNR-40 to 366,923 *u*m^3/mL at SNR-6. Of the three primary diatoms, *Fragillaria crotonensis* was the smallest component. Biovolume was greatest during mid July at most stations, with peak values ranging widely from 45,319 *u*m^3/mL at SNR-83 to 195,424 *u*m^3/mL at SNR-50. *Asterionella formosa, Diatoma* spp., and *Cymbella* spp. were secondary diatoms that did not attain the biovolumes of those previously mentioned, but are still worthy of mention. Asterionella formosa was a spring to early summer form that was rare during the second half of the project, with maximum station values ranging from 40,446 *u*m^3/mL at SNR-18 to 159,598 *u*m^3/mL at SNR-118. *Diatoma* spp., primarily *Diatoma moniliformis*, did not show a consistent temporal pattern with station peaks ranging from 18,619 *u*m^3/mL at SNR-18 during early October to 125,887 *u*m^3/mL at SNR-50 in early September. Of the secondary Bacillariophytes, *Cymbella* spp. was the least prevalent and appeared to be more numerous during July and early August. Maximum station biovolumes spanned the entire Lower Snake River region, reaching 48,556 um^3/mL at SNR-6 in early July and 43,549 um^3/mL at SNR-118 during late July. The lowest site specific value was again seen at SNR-18 where the population only attained 7,236 um^3/mL at the same time when peak volume was measured at SNR-6.

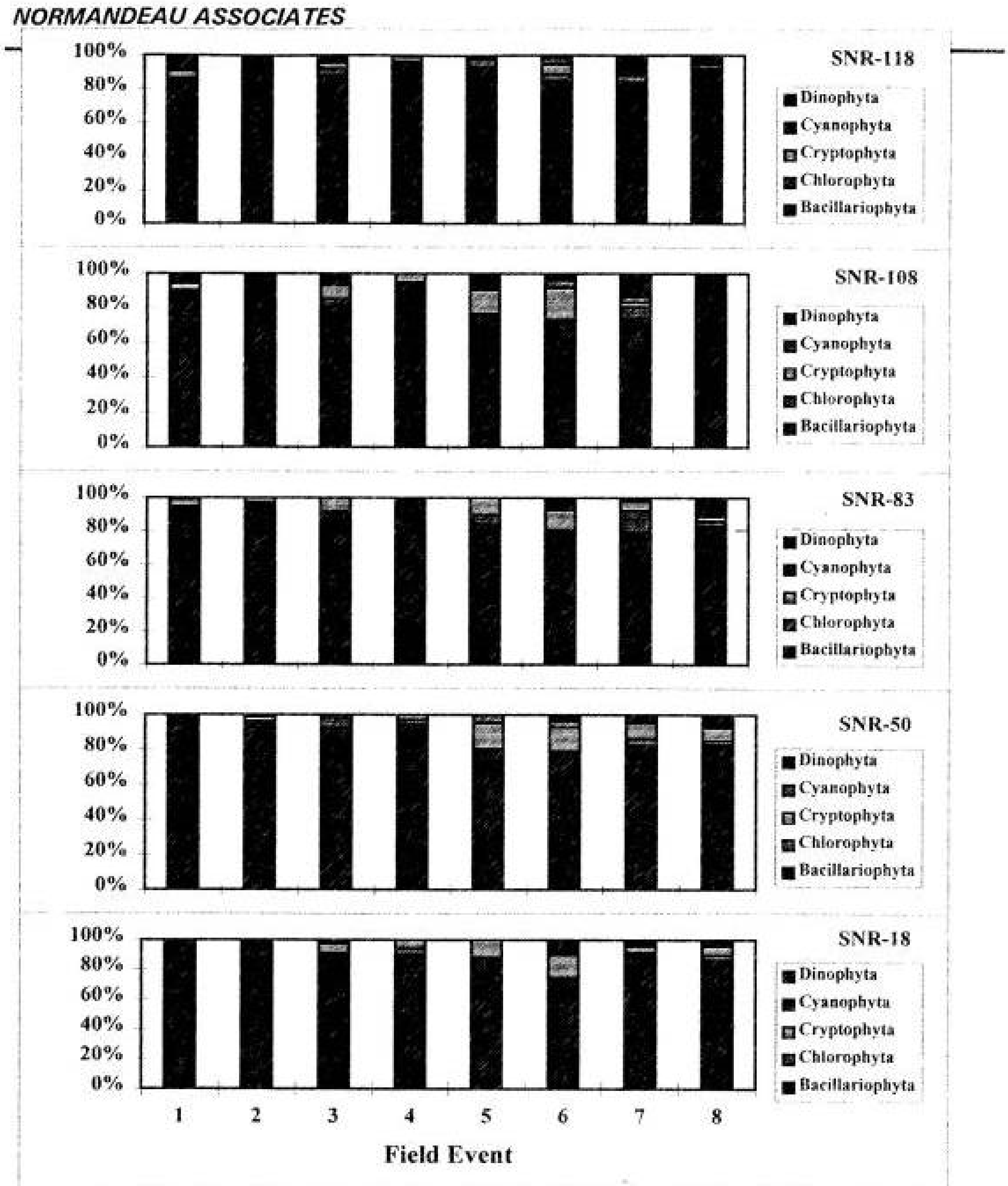

Figure 3.6-23. Percentages of the dominant photoplankton divisions at the reservoir stations on the lower Snake River during each field event

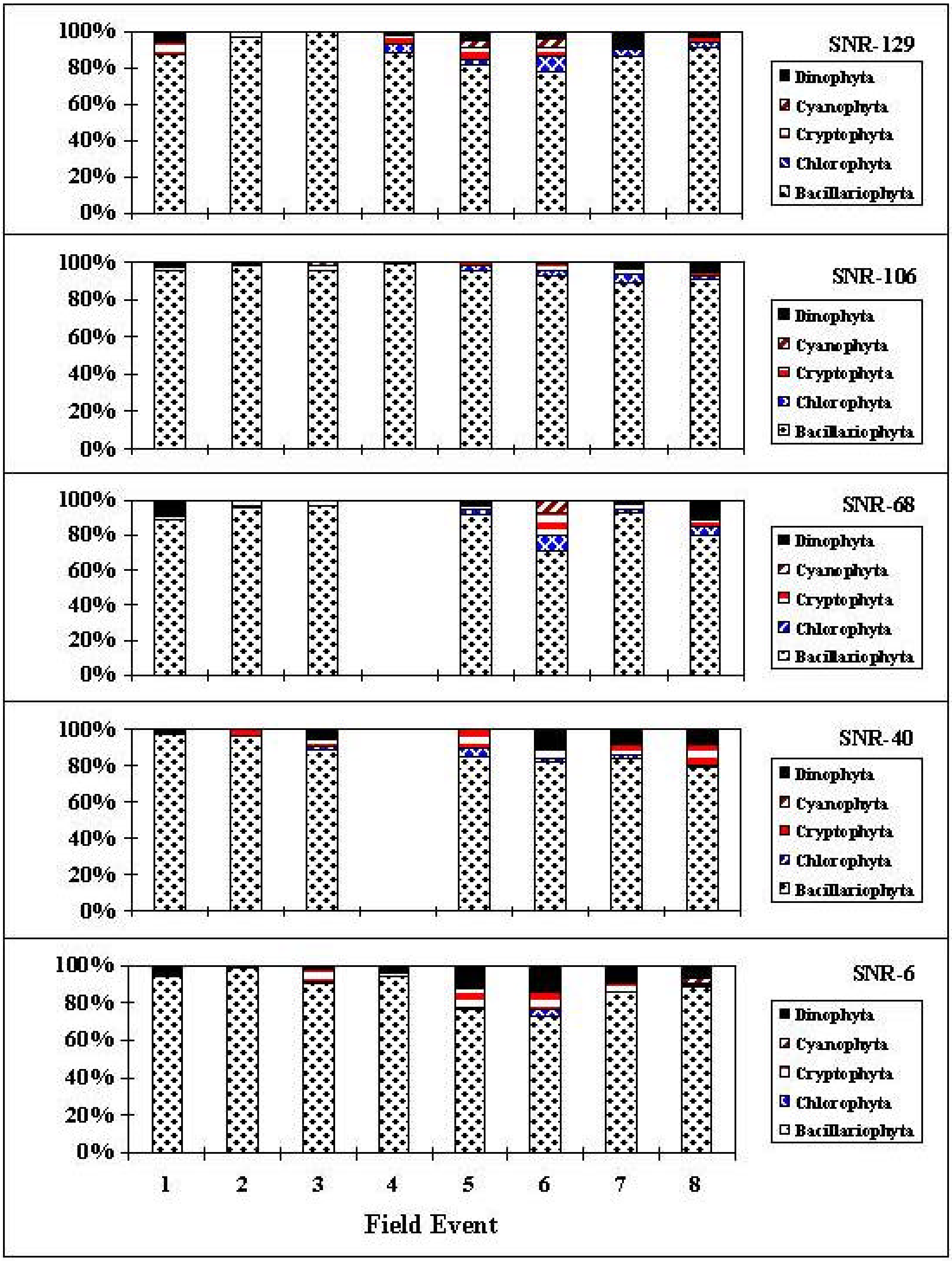

Figure 3.6-24. Percentages of the dominant phytoplankton divisions at the riverine stations on the lower Snake River during each field event

The remaining algal divisions, Cryptophyta, Dinophyta, Chlorophyta, and Cyanophyta typically occupied less than 5% of the total biovolume in the Lower Snake River, but there were exceptions. *Rhodomonas* spp., primarily *Rhodomonas minuta*, were ubiquitous at all of the sites, but seasonal influence was not the same throughout the system. For example, the largest seasonal median biovolume was calculated as 32,636 *u*m^3/mL for SNR-108, and this site also had the greatest maximum daily value of 49,030 *u*m^3/mL during mid July. However, the upstream station at SNR-118 had the lowest median seasonal biovolume of 8,433 *u*m^3/mL, and on the same day as the samples were collected at SNR-108 the biovolume was only 24,373 *u*m^3/mL. The remaining stations had biovolumes that were intermediate between these two values, and in most cases maximums occurred during July. *Glenodinium armatum, Glenodinium palustre, Gymnodium caudatum*, and *Peridinium inconspicuum* were the most commonly identified Dinophytes. Maximum densities for this division were more frequent in the late summer and early fall, although peaks during early June did also occur. *Glenodinium armatum* and *Glenodinium palustre* were either co-dominant or alternately dominant. For example, during the seventh sampling event the biovolume of *Glenodinium armatum* was 36,390 *u*m^3/mL at SNR-108, but on the same day at SNR-118 *Glenodinium palustre* occupied 38,041 *u*m^3/mL. These were the highest instantaneous values obtained during the study, and SNR-108 and SNR-118 did have seasonal medians of 7,313 *u*m^3/mL and 7,961 *u*m^3/mL, respectively, but median biovolumes ranged from less than 3,000 *u*m^3/mL to over 8,000 *u*m^3/mL. The population of *Gymnodium caudatum* was very sporadic with many samples containing no specimens, to the collection of 44,536 *u*m^3/mL at SNR-68 in early October. *Peridinium inconspicuum* were observed less frequently than *Glenodinium* spp. but more often than *Gymnodium caudatum*. Highest densities were almost always found to occur during late September and early October, with instantaneous biovolumes ranging from 5,000 *u*m^3/mL to 74,183 *u*m^3/mL, but more typically they were close to 20,000 *u*m^3/mL. The Chlorophytes were observed less frequently than the Dinophytes, usually during mid- to late-summer, and their percent contribution to the total biovolume typically only ranged from 1% to 3%. *Scenedesmus* spp. was identified in virtually all of the samples and did reach 74 cells/mL (14,000 *u*m^3/mL or 4% of the total) on 23 September at SNR-83, but densities were usually less than 20 cells/mL. *Pediastrum duplex* and *Ankistrodesmus falcatus* were also noted occasionally, but more often than not they were not identified in the samples. *Anabaena* spp. and *Chroococcus* spp. were the only blue-green algae identified, and that was primarily at the stations between SNR-129 and SNR-106 during September. At SNR-118 on 7 September, for example, *Anabaena* spp. numbered 198 cells/mL (11,108 *u*m^3/mL or 2.5% of the total) and *Chroococcus* spp. density was 158 cells/mL (12,536 *u*m^3/mL or 2.8% of the total); but even relatively low percentages such as these were the exception rather than the rule.

The algal populations in the Columbia River did not vary as much seasonally as they did in the Snake River, and the large early summer pulses were absent. During the project, the median biovolume of the algae in the Columbia River was fairly equivalent to the values determined in the Lower Snake River (Table 3.6-5). Two important differences, however, were that 1) the early spring pulses of diatoms were much lower in the Columbia River, especially at the upper sites; and 2) the percent biovolume attributable to the Bacillariophytes (Figure 3.6-25) was distributed more evenly among the various species.

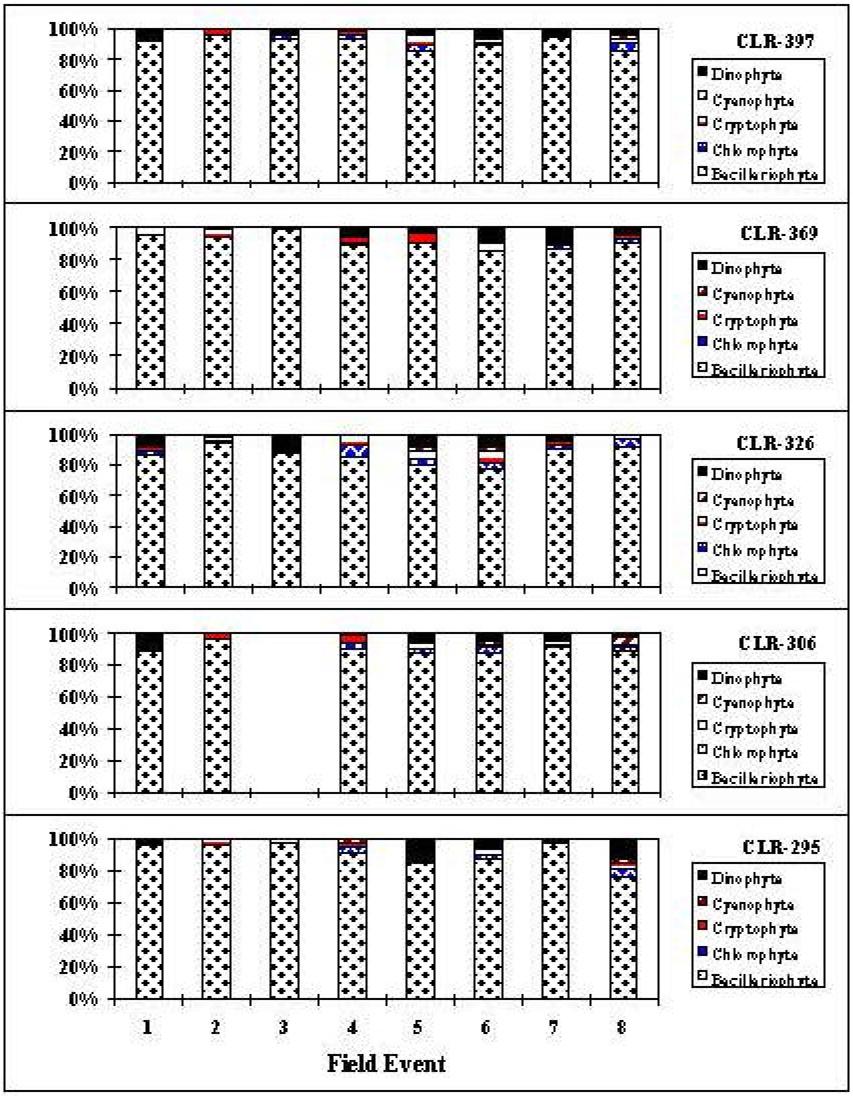

Figure 3.6-25. Percentages of the dominant phytoplankton divisions at the stations on the Columbia River during each field event

The diatoms *Cyclotella meneghiniona, Fragillaria crotonensis*, and *Melosira islandica* were all found in the Columbia River, and at times they were the dominant species, but *Asterionella formosa, Diatoma* spp., and *Synedra* spp. were more prevalent than they were in the lower Snake River. For example, the early season biovolume of *Cyclotella meneghiniona* was 97,494 *u*m^3/mL (16.7% of the total biovolume) at CLR-397 during the second sampling event, but reached 441,768 *u*m^3/mL (57.5% of the total) at CLR-295 during the same trip; the latter biovolume, however, was still less than half of what was determined at SNR-118 during the same period of time. The *Fragillaria crotonensis* population rose at all of the stations from early-summer low values to mid-summer maximums. The highest density was recorded at CLR-369 on 31 July, when this species reached 690 cells/mL (270,884 *u*m^3/mL or 42.3% of the total). During the same sampling event, biovolumes of 164,520 *u*m^3/mL (38.3% of the total) and 111,711 *u*m^3/mL (23.4% of the total) were determined at CLR-397 and CLR-295, respectively. Densities declined at all stations as the season progressed. *Melosira* spp. was also present in the Columbia River throughout the entire study, with maximum densities in September or October. The greatest instantaneous density was determined for CLR-397 in late September, when *Melosira* spp. reached 608 cells/mL (213,558 *u*m^3/mL or 29.2% of the total). This concentration was the greatest determined for this genus at the three upper Columbia River sites, but the season median biovolumes at all three sites were very close, ranging from 52,825 *u*m^3/mL at CLR-397 to 56,490 *u*m^3/mL at CLR-369. In contrast, the seasonal median at CLR-295 was 107,281 *u*m^3/mL, while the instantaneous maximum density on 26 September was 768.8 cells/mL (262,951 *u*m^3/mL or 31.7% of the total). *Asterionella formosa* attained maximum densities in the Columbia River during early July, as they typically did in the Snake River, but the values were considerably higher. For example, this species reached 457 cells/mL (283,341 *u*m^3/mL or 39.7% of the total) at CLR-326 on 2 July, and only 8% less at CLR-397. Furthermore, they continued to be a part of the algal community for the remainder of the study period; albeit at reduced numbers, but regularly exceeding 30,000 *u*m^3/mL and even reaching secondary maximums in the late summer of 175,000 *u*m^3/mL. At the two remaining stations on the Columbia River, *Asterionella formosa* was, of course, present, but the greatest density achieved was 307 cells/mL (193,273 *u*m^3/mL or 22.0% of the total) at CLR-295 on 18 July. The percent contribution of this species to total biovolume was usually less than this during the first three field events at the two downstream stations, primarily as a result of the relatively large influx of *Cyclotella meneghiniona* from the Snake River. *Diatoma* spp., primarily *Diatoma moniliformis*, was another group that was much more prevalent in the Columbia River. Seasonal biovolume medians at the three upper sites ranged from 32,981 *u*m^3/mL at CLR-326 to 41,321 *u*m^3/mL at CLR-397, while the analogous calculated value for CLR-295 was 43,675 *u*m^3/mL. In contrast, seasonal medians at the lower Snake River stations were usually less than 4,000 *u*m^3/mL, although SNR-50 was an exception with 20,724 *u*m^3/mL. Similarly, *Synedra* spp., mainly *Synedra ulna*, had median biovolumes of less than 2,000 um^3/mL in the lower Snake River, but ranged from 11,074 um^3/mL to 14,268 um^3/mL at the upper Columbia River sites; median values at the two lower Columbia River stations were slightly less than 5,000 um^3/mL. Three additional Bacillariophytes that were a minor component of the diatom community, though more prevalent in the Columbia River than in the Snake River, were *Navicula* spp. (mainly *Navicula radiosa*), *Nitzchia* spp., and *Surirella ovata*.

The Cryptophyta, Dinophyta, Chlorophyta, and Cyanophyta divisions in the Columbia River did not differ markedly from those in the Lower Snake River. *Glenodinium* spp., *Gymnodium caudatum*, and *Peridinium inconspicuum* were again the main Dinophytes, and although they appeared to be present more consistently in the Columbia River, their sporadic presence in the samples prevents any definitive statement regarding differences. Median seasonal biovolumes for the Cryptophyte *Rhodomonas* spp. in the Columbia River ranged from 11,500 um^3/mL at CLR-397 to 17,841 um^3/mL at CLR-326, which is well within the range of values calculated for the Lower Snake River stations. Similar correlations can be made for the *Chlorophytes Scenedesmus* spp. and *Peridinium duplex* and the Cyanophyte *Anabaena* spp.

Zooplankton

The number and total biomass of the zooplankters identified in the Clearwater River and Snake River samples were comparatively low, but there were also noteworthy differences between the two systems. The median biomass in the Clearwater River was about one half of what it was in the Snake River when the entire sampling period is considered (Table 3.6-6), but on a trip-to-trip basis the differences were not consistent (Figures 3.6-26 and 3.6-27). In the beginning of June, the biomass in the Clearwater River was 0.614 *u*g/L; 100% Copepoda with a nauplii density of 0.764 individuals/L and a *Cyclopoid copepodid* population of 0.164 individuals/L. At the same time in the Snake River, the total zooplankton biomass was 2.983 *u*g/L - the highest value it reached. Sixty-four percent of this biomass was due to the Cladocerans *Alona quadrangularis* (1.584 individuals/L, *Bosmina longirostris* (0.594 individuals/L), and *Alona guttata* (1.585 individuals/L); this was also the only time that any Cladocerans were enumerated in SNR-148 samples during the entire project. Nauplii (1.188 individuals/L) and *Cyclopoid copepodid* (0.594 individuals/L) were the only Copepods present. *Brachionus calyciflorus* was a minor component of the community, comprising only 1% of the total biomass (0.594 individuals/L). By the end of June, the Copepods dominated the biomass in both systems, with the Clearwater River at 1.047 *u*g/L, compared to 0.581 *u*g/L in the Snake River. The higher value in the Clearwater River was associated with greatest diversity found in that river at any time, and included *Leptodiamtomus ashlandi* (0.045 individuals/L), *Calanoid copepodid* (0.158 individuals/L), *Cyclopoid copepodid* (0.407 individuals/L), *Diacyclops thomasi* (0.091 individuals/L), *Epischura* spp. (0.272 individuals/L), and nauplii (0.211 individuals/L). In the Snake River, on the other hand, *Cyclopoid copepodid* (0.067 individuals/L), *Diacyclops thomasi* (0.133 individuals/L), and nauplii (0.465 individuals/L) were the only representatives. For the remainder of the project, the total biomass in the both rivers was low, ranging from <0.001 *u*g/L to 0.457 *u*g/L. In the Snake River, 100% of the biomass was due to the Copepods (primarily

Cyclopoid copepodid, Diacyclops thomasi, and nauplii), with the exception of the late-September trip, when the rotifer *Keratella cochlearis* was the only species identified (0.045 individuals/L). Diversity in the Clearwater River was slightly greater during the summer. When the same three groups of copepods found in the Snake River were not responsible for 100% of the biomass, it was primarily due to the presence of *Bosmina longirostris* during mid July and mid August when a few (0.022 individuals/L and 0.029 individuals/L, respectively) were identified. Perhaps importantly, the only time that *Daphnia retrocurva* was identified in either river was during early October in the Clearwater River, when a population of 0.010 individuals/L comprised 100% of the total biomass (0.030 *u*g/L).

Table 3.6-6
Median Biomass and Average Percent Composition of the Zooplankton Community At Each Monitoring Site During the 1997 Sampling Season

System/Site	Median Biomass (*u*g/L)	Cladocera (Percent)	Copepoda (Percent)	Rotifera (Percent)
Clearwater				
CLW-11	0.126	4.9	95.1	0.0
Snake				
SNR-148	0.247	41.8	57.9	0.3
SNR-129	0.098	47.2	52.4	0.4
SNR-118	0.247	57.4	20.2	0.6
SNR-108	0.839	93.8	5.3	0.9
SNR-106	0.549	80.4	17.9	1.7
SNR-83	1.000	87.9	11.9	0.2
SNR-68	1.903	74.1	25.7	0.2
SNR-50	11.584	71.0	29.0	0.0
SNR-40	3.142	66.8	33.1	0.1
SNR-18	2.336	67.1	33.8	0.1
SNR-6	1.288	75.1	24.9	0.0
Columbia				
CLR-397	0.484	54.1	45.5	0.4
CLR-369	0.108	57.0	42.4	0.6
CLR-326	0.126	36.7	59.5	3.8
CLR-306	0.879	64.2	32.9	2.9
CLR-295	1.367	51.7	47.7	0.6

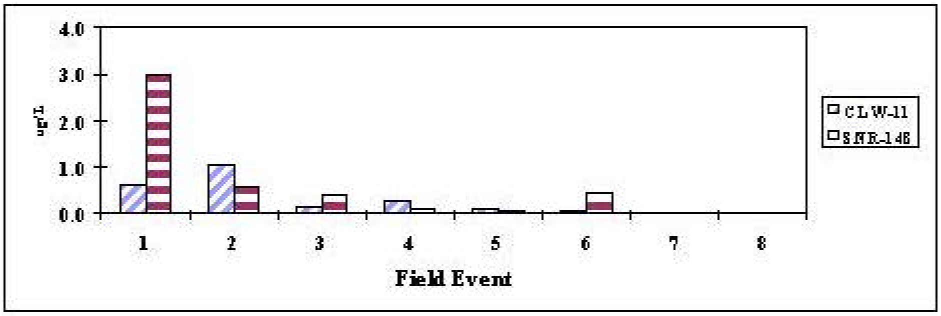

Figure 3.6-26. Total zooplankton biomass at CLW-11 and SNR-148 during each field event

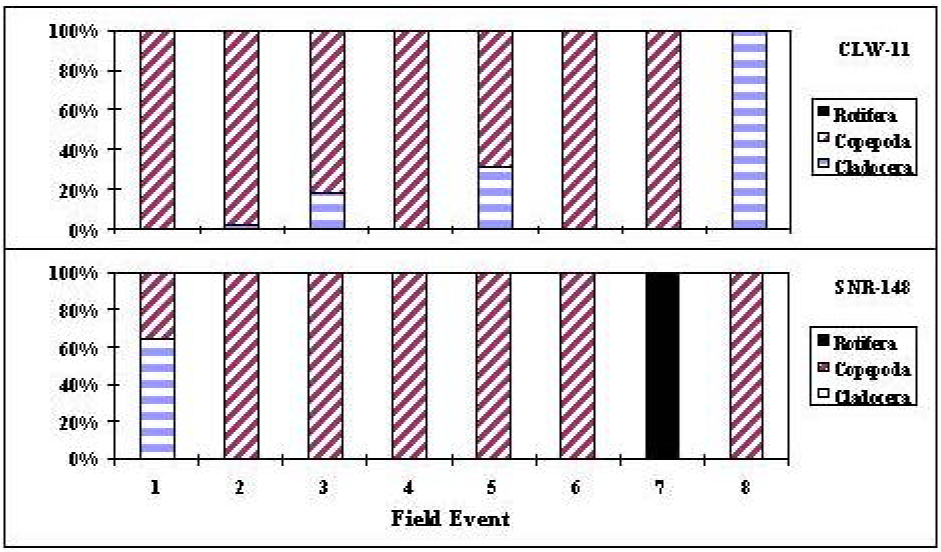

Figure 3.6-27. Percentage contribution of the major zooplankton divisions to total biomass at CLW-11 and SNR-148 during each field event

The number and dominant zooplankton species appear to change not only temporally, but also spatially within the Lower Snake River system. At all of the reservoir sites (SNR-118, SNR-108, SNR-83, SNR-50, and SNR-18), the seasonal trend was for the copepods, primarily *Cyclopoid copepodid, Diacyclops thomasi*, nauplii, and to a lesser extent *Leptodiaptomus ashlandi*, to dominate the total biomass at the beginning of the season (Figure 3.6-28). As spring turned into summer, the cladocera, mainly *Bosmina longirostris* and *Daphnia retrocurva*, became more numerous and reached initial maximal biomass (up to 98% of the total) between mid-August and mid-September. Rotifers such as *Brachionus calyciflorus, Keratella cochlearis, Polyarthra vulgaris*, and *Synchaeta pectinata* did average up to 30% of the zooplankton density at the stations upstream from SNR-83, only up to 10% downstream from Central Ferry, but because of their small size they only represented 1%, or less, of the biomass.

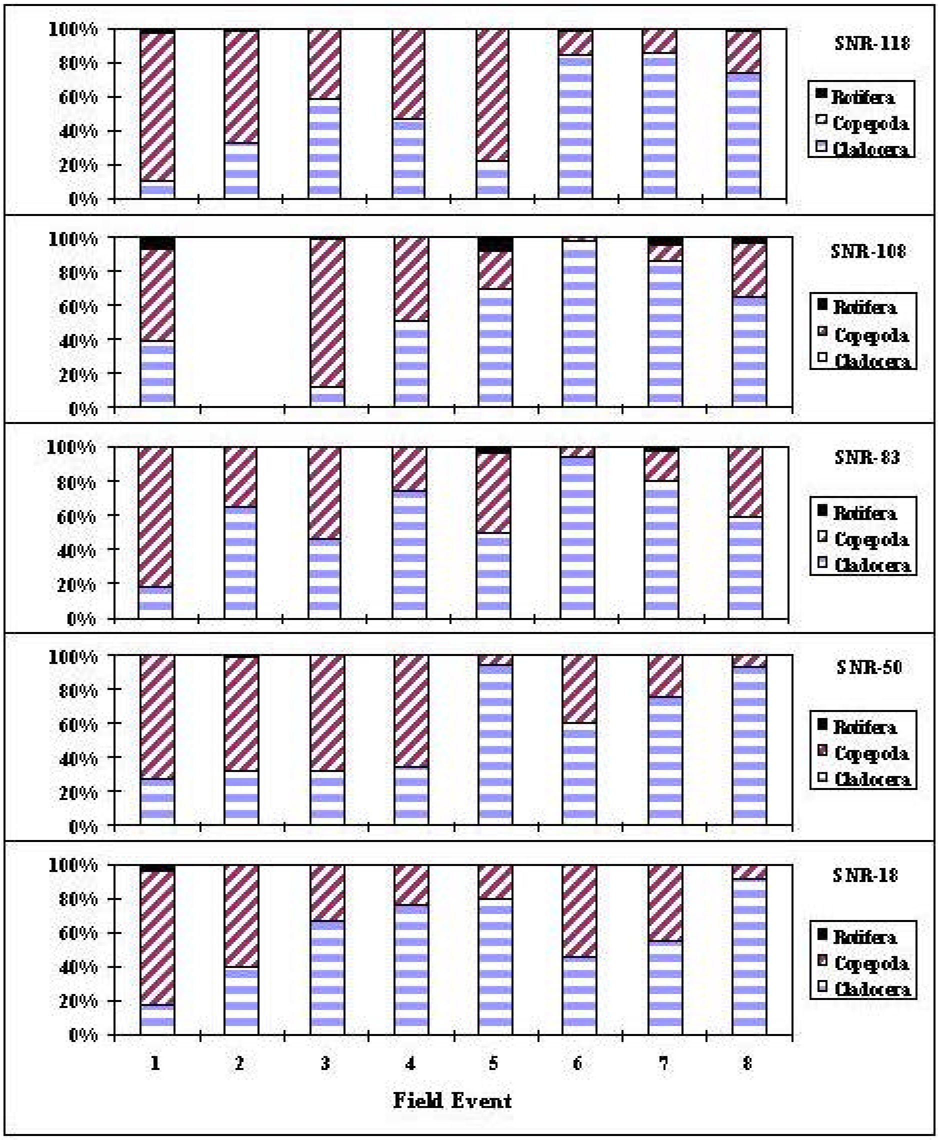

Figure 3.6-28. Percentage contribution of the major zooplankton divisions to total biomass at the reservoir stations on the lower Snake River during each field event

Of the reservoir stations (Figure 3.6-29), the one at SNR-118 had the lowest mean biomass (0.582 *u*g/L), while the one below Lyons Ferry had the greatest (16.354 *u*g/L). The reason for the comparatively high biomass at SNR-50 was the abundance of *Daphnia retrocurva*. This species was absent during the early June sampling event, at the same level or less through mid July, but began to flourish by the end of July when they were present at 4.37 individuals/L, or 12.46 *u*g/L. Their density and biomass more than doubled by mid-August, declined precipitously to 1.16 *u*g/L in early September, and then rebounded to 14.19 *u*g/L and 33.10 *u*g/L in late September and early October, respectively.

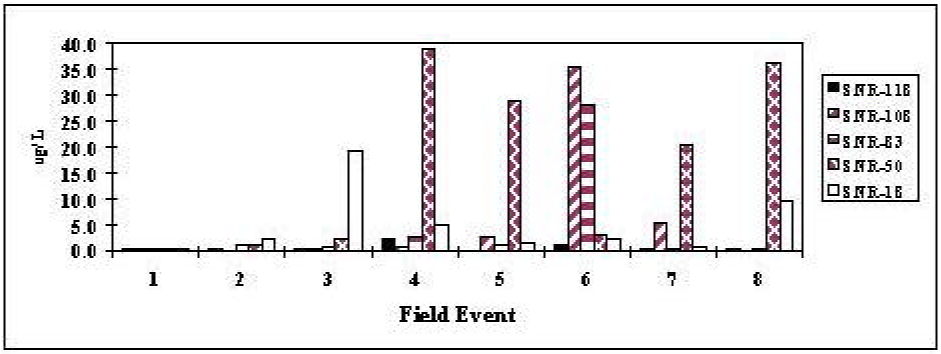

Figure 3.6-29. Total zooplankton biomass from the photic zones of the reservoir stations on the lower Snake River during each field event

Daphnia did attain relatively high densities of 9.57 individuals/L at SNR-108 and 7.34 individuals/L at SNR-83 in mid-August, but these were one-time maximums and lacked the consistency of the SNR-50 population. At SNR-18, the *Daphnia* population was not as numerous as noted at the station upstream, but they did have the greatest mid-July biomass of 12.18 *u*g/L, low to intermediate values during August and September, and increased to 8.25 *u*g/L in early October. The reason for the seemingly larger population of *Daphnia* at SNR-50 is unknown, but it could be that the area where the Palouse River enters the Snake River provides good habitat for these organisms. The other cladoceran that was consistently present at all of the reservoir stations was *Bosmina longirostris*. Biomass increased slightly going downstream, averaging 0.07 *u*g/L at SNR-108 to 0.29 *u*g/L at SNR-18. Maximum biomass ranged from 0.53 *u*g/L, 0.66 *u*g/L, and 0.68 *u*g/L at SNR-118, SNR-18, and SNR-83, respectively, during the first half of the sampling season, and reached 0.61 *u*g/L at SNR-50 during the second half. *Diacyclops thomasi* was one of the consistently dominant copepods at all of the reservoir stations. A system high density of 11.47 individuals/L (22.74 *u*g/L) was reached at SNR-50 during late July. The next closest density identified during any sampling event was 4.22 individuals/L (5.88 *u*g/L) at SNR-18 in mid-July. Otherwise, sampling event biomass ranged from 0.01 *u*g/L to 3.24 *u*g/L, with the lower values reported for the stations in Lower Granite pool. Nauplii were ubiquitous throughout the sampling season, and mean biomass ranged from 0.11 *u*g/L to 0.29 *u*g/L at the five reservoir stations.

The composition of the zooplankton community at the riverine sites on the Lower Snake River (SNR-129, SNR-106, SNR-68, SNR-40, and SNR-6) was essentially the same as that identified at the reservoir stations, but the density and biomass of those species was often noticeably different (Figures 3.6-30 and 3.6-31). Mean seasonal biomass, for example, was less at each below-dam station when compared to the reservoir site upstream (Table 3.6-6). This does not imply that density zooplankton and biomass was always less below the dams, but using *Daphnia retrocurva* as an example, when this species reached maximum concentrations at SNR-50 the corresponding biomass at SNR-40 was only 3 to 36% of upstream values. The same type of trend appeared to be true for *Diaphanosoma brachyurum, Cyclopoid copepodid*, and *Diacyclops thomasi*, with some variation between stations.

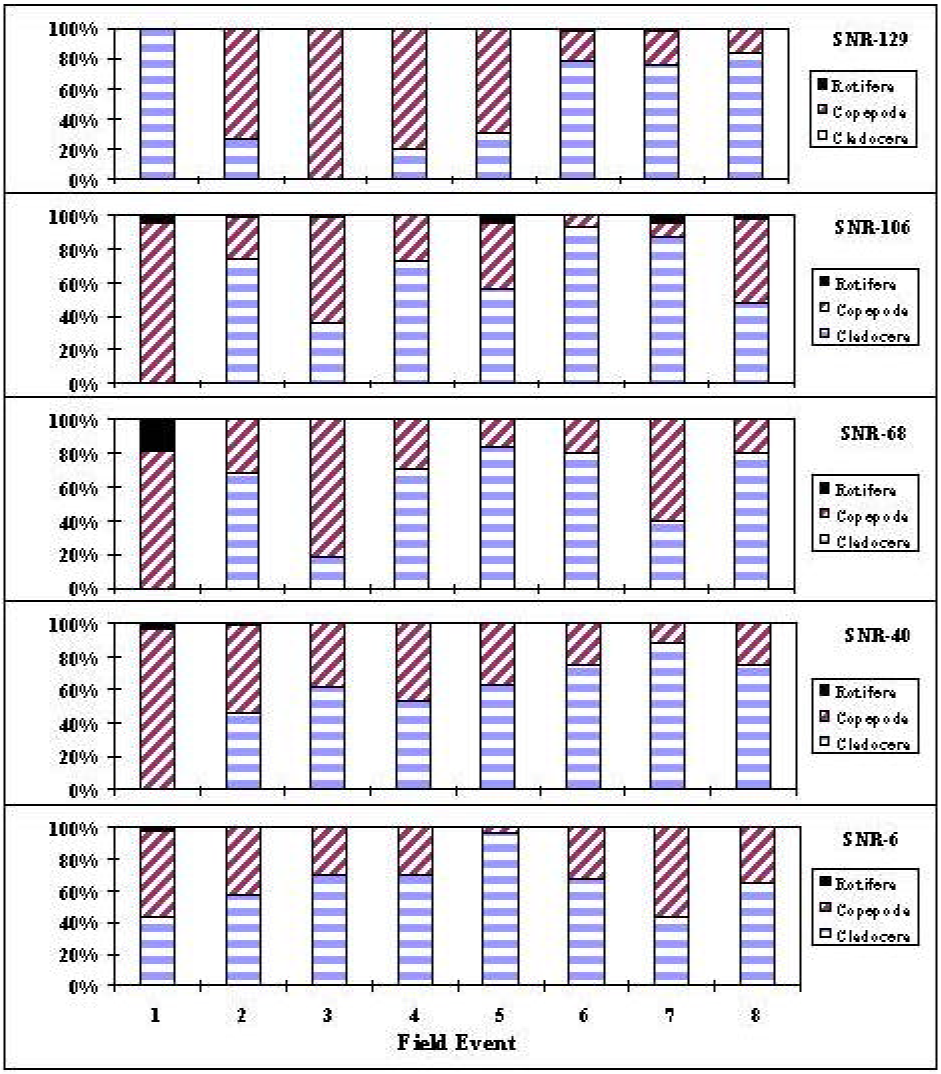

Figure 3.6-30. Percentage contribution of the major zooplankton divisions to total biomass at the riverine stations on the lower Snake River during each field event

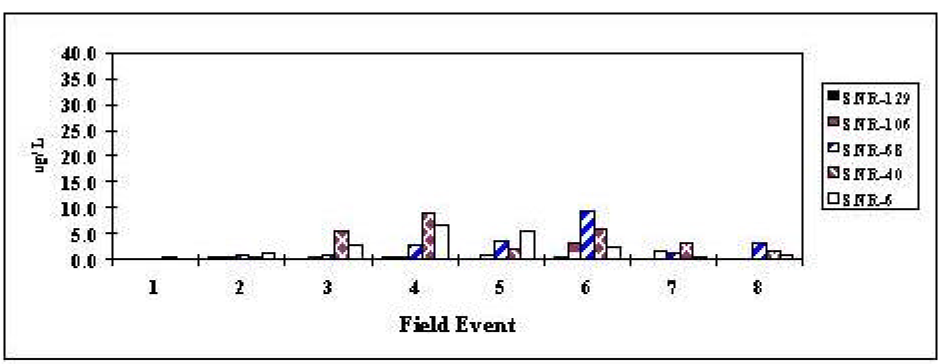

Figure 3.6-31. Total zooplankton biomass from the photic zones of the riverine stations on the lower Snake River during each field event

The zooplankton communities at the Columbia River sites showed some similarities, and several differences, with those on the Lower Snake River: average biomass and density increased with distance downstream; the relative abundance of copepods appeared to be greater; and, the average seasonal biomass of several species was less (Figures 3.6-32 and 3.6-33). The mean seasonal biomass at CLR-397 was 0.60 *u*g/L, which was virtually equivalent to that calculated for SNR-148, and only slightly higher than it was at SNR-129, but lower than at any of the sites between SNR-118 and SNR-6. The average biomass declined by about 62% at CLR-369, and an additional 35% reduction occurred at CLR-326. At the beginning of the season, the biomass determined for several species, and the community as a whole, were analogous to those in the Snake River, but from August through October density and biomass declined rather than increased. *Daphnia retrocurva*, a significant contributor to overall biomass in the Snake River, was absent in the samples collected at CLR-397 during the first two sampling events, reached a maximum of 0.056 *u*g/L by mid-July, and declined thereafter. *Diacyclops thomasi, Cyclopoid copepodid*, and nauplii were present at all three of the upstream Columbia River sites during most of the season, but reached maximum biomass (0.155 *u*g/L for Diacyclops, 0.080 *u*g/L for Cyclopoid, and 0.103 *u*g/L for nauplii) during the first half of the study. In each of these cases, the seasonal averages and peak values resembled corresponding numbers determined for SNR 129 more so than any other site on the Lower Snake River. The reason for the overall lower numbers in the Columbia River is not known, but may be related to system operations (*e.g.*, the timing of water releases from Lake Roosevelt in anticipation of spring runoff versus when that facility was used for storage) and possibly shorter hydrologic residence times. Below the confluence with the Snake River, the biomass of the zooplankton increased, but seldom attained values equivalent to those determined for several of the below-dam sites on the Snake River. *Daphnia retrocurva* biomass, for example, peaked at only 4.978 *u*g/L and 2.230 *u*g/L at CLR-306 and CLR-295, respectively, in late July. The one species that did not always follow the pattern of diminished presence in the Columbia River was *Bosmina longirostris*. The average seasonal biomass at CLR-397 was 0.30 *u*g/L, and did decline significantly at CLR-369 and again at CLR-326, but this mean was very comparable to those calculated for the lower Snake River sites, and in fact the peak biomass of 1.531 *u*g/L was the highest value determined for any station.

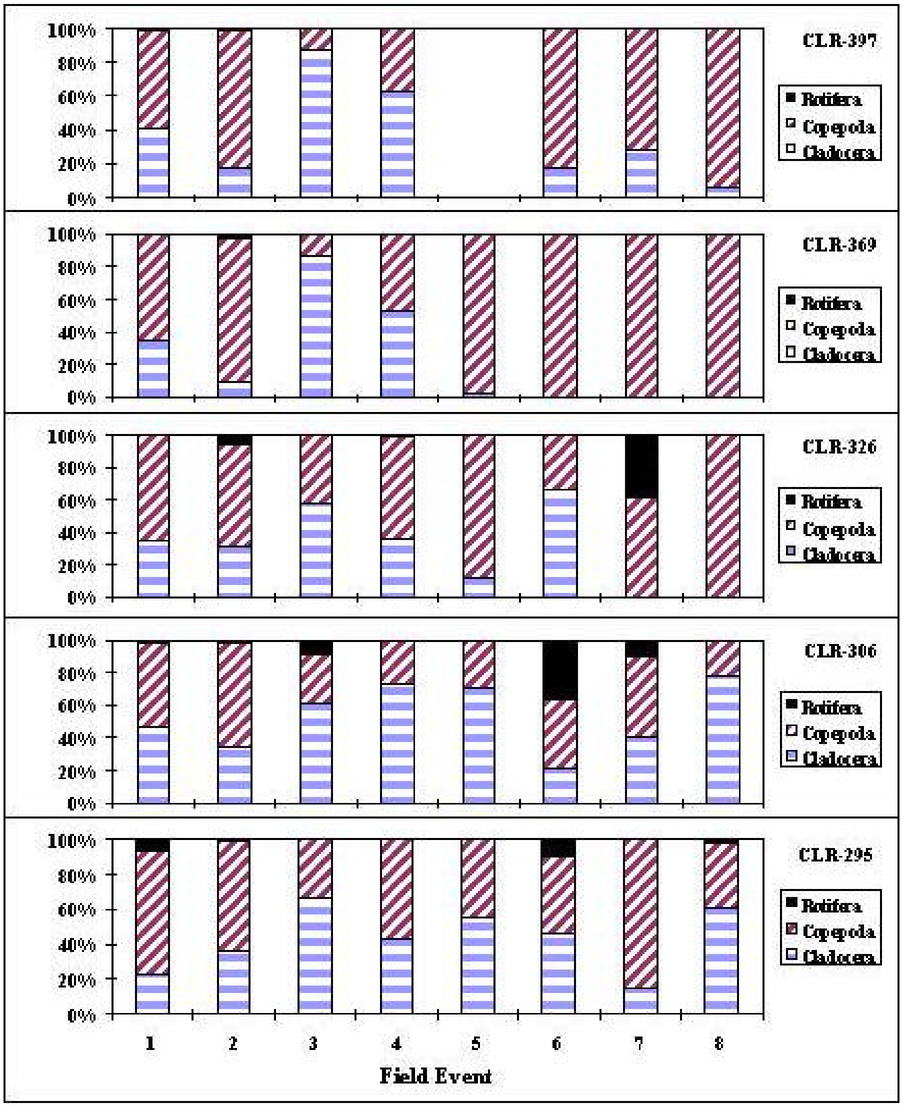

Figure 3.6-32. Percentage contribution of the major zooplankton divisions to total biomass at the stations on the Columbia River during each field event

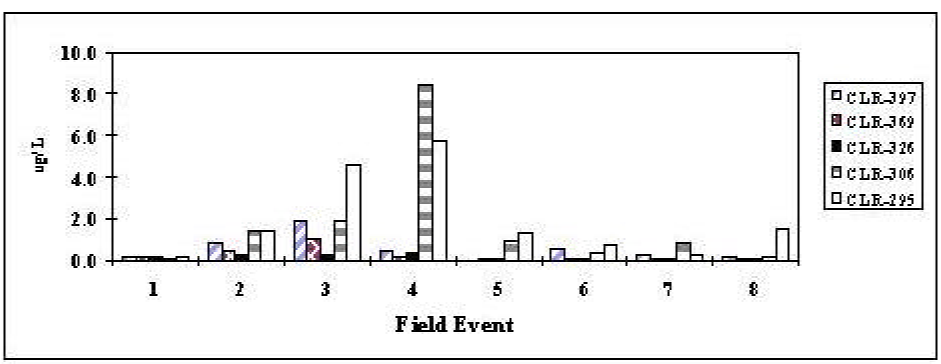

Figure 3.6-33. Total zooplankton biomass from the photic zones of the stations on the Columbia River during each field event

3.6.4 - Phytoplankton Primary Productivity

The rate of primary productivity is influenced by the interaction of all the parameters that were included in the scope of this investigation and discussed in the previous text. There are several ways available to present primary productivity data, four of which are shown in Table 3.6-7. The volume weighted hourly rate will be the primary focus of this discussion. The integrated daily rate is frequently utilized in limnology related literature, but because there was such a large difference between the depths of the photic zones in the Lower Snake River and the Columbia River, and that the depths of the photic zones often exceed the channel depths in the Upper Snake River and Clearwater River, the weighted hourly rate provides a more equitable comparison between systems However, integrated daily rates will be included in the Lower Snake River analysis when it will enhance the explanation.

Table 3.6-7
Median Photic Zone Primary Productivity Rates
For the Sites Monitored During the 1997 Sampling Period

System/Site	Volume Weighted Hourly Rate (mg $^{12}C/m^3/hr$)	Integrated Hourly Rate (mg $^{12}C/m^3/hr$)	Volume Weighted Daily Rate (mg $^{12}C/m^3/hr$)	Integrated Daily Rate (mg $^{12}C/m^3/hr$)
Clearwater				
CLW-11	1.97	8.6	18	86
Snake				
SNR-148	14.49	49.6	134	425
SNR-118	15.87	67.0	141	590
SNR-106	19.87	78.5	197	737
SNR-83	14.58	68.3	142	767
SNR-68	18.01	59.9	151	502
SNR-50	13.63	60.4	133	584
SNR-40	12.11	40.3	127	366
SNR-18	12.57	46.2	103	390
SNR-6	9.49	38.6	78	314
Columbia				
CLR-397	9.85	79.3	93	862
CLR-396	10.92	95.1	100	916
CLR-295	20.58	90.4	196	855

The rates of primary productivity at the Clearwater River CLW-11 station and the Snake River SNR-148 site were very different (Figure 3.6-34). The median seasonal hourly rates at CLW-11 and SNR-148 were 1.97 mg/m^3/hr and 14.49 mg/m^3/hr, respectively; a statistically significant difference. Even the ranges of values did not overlap. In the Clearwater River, the lowest rate was 0.13 mg/m^3/hr during the first field event and the highest was 4.00 mg/m^3/hr in mid-August. At SNR-148, the lowest rate was 9.21 mg/m^3/hr and the largest rate of 32.94 mg/m^3/hr was documented at the end of June.

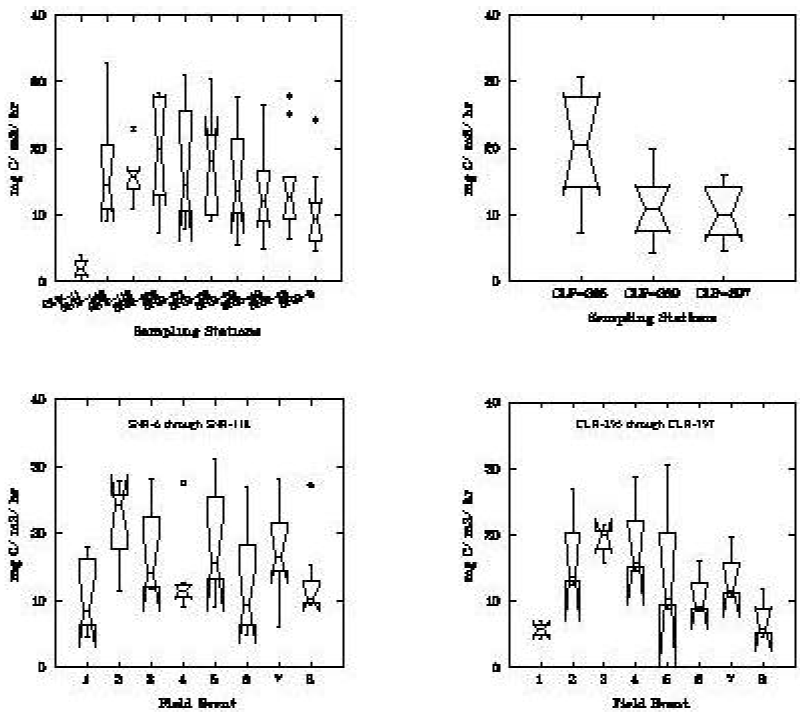

Figure 3.6-34. Box plots of volume-weighted primary productivity from the photic zones at the Clearwater River, Snake River, and Columbia River monitoring stations

There were some spatial and temporal differences in the rates of productivity at the Lower Snake River sites, but due to the spread of the data points these differences were not always significant. There was a slight increase progressing downstream from SNR-118 to SNR-106 with median rates rising from 15.87 mg/m^3/hr to 19.87 mg/m^3/hr. During six of the eight sampling events, the rates at SNR-106 were greater by 15% to 118%. But during the first and sixth evaluations they were 56% and 42% lower, respectively. On an hourly basis the seasonal median decreased to 14.58 mg/m^3/hr at the next downstream station, SNR-83. However, if the integrated daily rate is used then there was a very slight increase; 737 mg/m^2/day at SNR-106 and 767 mg/m^2/day at SNR-83. In either case, there did not appear any significant differences with respect to the rates of primary productivity between the two sites. Similarly, the rates at SNR-68 were very similar to the ones determined for SNR-83, but downstream from SNR-68 there was a gradual decline in productivity. The median value at SNR-68 was 18.01 $mg/m^3/hr$ while the median rate at SNR-6 was 9.49; a 47% decrease. There did not appear to be any measurable differences in the productivity rates at the reservoir sites versus the riverine stations as previously described for the zooplankton, but there was some variance between field events (Figure 3.6-34). The median rate for all of the Lower Snake River stations during the first field event was 8.46 $mg/m^3/hr$ and almost tripled to 24.31 $mg/m^3/hr$ three weeks later. The median rate was down to 14.35 $mg/m^3/hr$ by mid July, and declined further to 11.29 $mg/m^3/hr$ at the end of that month. The late-July depression could have been due to the greater volume of water contributed to the system from the Clearwater River, which had significantly lower productivity. Overall productivity increased slightly to 15.46 $mg/m^3/hr$ by mid-August, fluctuated during the next two field events, and then ended the season at 10.35 $mg/m^3/hr$.

Primary productivity at CLR-369 and CLR-397 was typically less than values determined in the Lower Snake River, but almost doubled at CLR-295 (Figure 3.6-34). The seasonal medians at the two Upper Columbia River stations were 9.85 $mg/m^3/hr$ and 10.92 $mg/m^3/hr$ at CLR-397 and CLR-369, respectively. By the time the water had traveled 74 miles and merged with the Snake River, the median seasonal productivity had increased to 20.58 $mg/m^3/hr$. Peak hourly productivity occurred in mid July at CLR-397 (15.84 $mg/m^3/hr$) and CLR-369 (19.90 $mg/m^3/hr$), and in mid August at CLR-295 (30.48 $mg/m^3/hr$).

3.6.5 - Benthic Algae (Periphyton) Primary Productivity

Snake and Columbia Rivers Overview

Snake River discharge at the beginning of the field season early in July, 1997 averaged 75,000 cfs dropping steadily by late August to minimal flows of 28,700 cfs before increasing again in late September to 40,000 cfs (USGS provisional data, Tacoma, WA, December, 1997). Transparency decreased from RM 148 with an average Secchi disk depth of 1.8 m decreasing to 1.2 m at RM 37, increasing in Ice Harbor Reservoir (RM 18) to 1.3 m, and then dropping again below the dam to 1.1 m at RM 6 (Table 3.6-8). Temperature showed a slight warming trend through the four reservoirs. There was a decrease from RM 148 to RM 118 from a mean of 19.9°C to 19.4°C. This is attributed to the mixing of the colder water from the Clearwater River which had a season long mean temperature of 15.1°C. Substrate of the lower Snake impounded reach was mainly fines, silts, sand, and embedded cobble. Upstream of the impoundments, RM 148 substrate composition was mainly sand, gravel, and medium embedded cobble.

Table 3.6-8
Mean Habitat Descriptors for Study Sites on the Lower Snake, Columbia, and Clearwater Rivers
July to October, 1997

Site	Secchi Disk	VEC	Photic Zone Depth	Temp (0-1.5 m)	Substrate Type
Snake RM 148 (ASO-A)	1.82 m	0.94	5.97 m	19.9°C	Sand, gravel, medium embedded cobble
Clearwater RM 11 (SPA-A)	2.99 m	1.07	3.93 m	15.1°C	Mostly cobble, some sand
Snake RM 118 (GRA-A)	1.73 m	0.92	5.14 m	19.4°C	Fines, silts, some sand, embedded cobble, woody debris
Snake RM 105 (GRA-B)	1.56 m	1.29	3.54 m	19.4°C	Fines, silts, sand
Snake RM 81 (GOO-A)	1.55 m	1.18	4.08 m	20.1°C	Fines, silts, some sand, entirely embedded cobble
Snake RM 67 (GOO-B)	1.30 m	1.18	4.26 m	20.0°C	Fines, silt, sand, and some gravel
Snake RM 52 (MON-A)	1.25 m	1.24	3.84 m	20.4°C	Fines, silt, and sand
Snake RM 37 (MON-B)	1.20 m	1.34	3.50 m	20.0°C	Riprap shoreline, sand, and silt
Snake RM 18 (ICE-A)	1.30 m	1.29	3.56 m	21.5°C	Increased sand, some gravel, and silt
Snake RM 6 (ICE-B)	1.10 m	1.22	3.87 m	20.4°C	Gravel, sand, and some silt
Columbia RM 410 (PRI-A)	3.35 m	0.56	9.28 m[a]	18.6°C	Riprap shoreline and sand
Columbia RM 369 (HAN-A)	3.98 m	0.51	11.13 m[b]	18.0°C	Cobble and sand
Columbia RM 326 (MNA-A)	1.50 m	1.17	4.47 m	21.5°C	Sand, fines, and silt

[a]Value obtained on October 18, 1997, and is not a season-long mean.
[b]Value obtained on October 19, 1997, and is not a season-long mean.

Columbia River discharge at Priest Rapids and Hanford averaged 212,000 cfs at the beginning of July, dropping to a low of 93,800 cfs by the first week of September, then increasing to 132,300 cfs by mid-October (USGS provisional data, Tacoma, WA, December, 1997). Flows at McNary Dam were higher due to the input of the Snake River. At the start of July, discharge was 301,860 cfs dropping to 111,100 by the end of September (US ACOE provisional data, Walla Walla, Washington, December, 1997). The highest transparency of Columbia River sites was observed at the free-flowing reach, RM 369, with a mean Secchi disk depth of 4.0 m, followed by Priest Rapids Reservoir (RM 410) with 3.4 m. By RM 326, transparency diminished with a mean Secchi of 1.5 m. Temperature was highest at McNary Reservoir with a mean of 21.5°C with cooler temperatures in the upper reaches of 18.1°C and 18.5°C at RM 369 and 410 (Table 3.6-8).

In the reservoirs of the lower Snake River, ABA chlorophyll *a* increased downstream from Snake RM 118 to Snake RM 18. Mean ABA chlorophyll *a* at RM 118 in Lower Granite Reservoir at 0.75 m was 36.1 mg/m^2 and 21.4 mg/m^2 at 1.5 m. Ice Harbor Reservoir (RM 18) had the highest mean chlorophyll *a* of 91.5 mg/m^2 and 50.4 mg/m^2 at the 0.75 m and 1.5 m depths. There was a slight decrease in chlorophyll *a* below Little Goose, Lower Monumental, and Ice Harbor Dams (Table 3.6-9). However, upstream of the impoundments, chlorophyll *a* was high at the free-flowing reach, RM 148. Mean chlorophyll *a* at RM 148 was 55.5 mg/m^2 and 40.7 mg/m^2 at the 0.75 m and 1.5 m depth (Table 3.6-9).

Table 3.6-9 Mean Chlorophyll *a*, Biomass, and Autotrophic Index for Study Sites on the Lower Snake, Columbia, and Clearwater Rivers July to October, 1997						
Site	**0.75 m Tile Substrate**			**1.5 m Tile Substrate**		
	Mono Chl A (mg/m^2)	**AFOD W (g/m^2)**	**AI**	**Mono Chl A (mg/m^2)**	**AFOD W (g/m^2)**	**AI**
Snake RM 148 (ASO-A)	55.53	10.01	234.67	40.71	8.26	220.68
Clearwater RM 11 (SPA-A)	15.21	2.20	204.44	12.99	5.81	389.62
Snake RM 118 (GRA-A)	36.07	17.24	604.43	21.35	23.64	529.60
Snake RM 105 (GRA-B)	39.91	19.14	252.53	29.68	10.47	501.84
Snake RM 81 (GOO-A)	38.10	36.99	685.02	25.17	9.08	355.39
Snake RM 67 (GOO-B)	35.36	13.88	291.82	23.35	9.76	258.19
Snake RM 52 (MON-A)	51.43	12.52	248.75	24.21	5.02	239.77
Snake RM 37 (MON-B)	44.43	7.52	187.44	40.51	8.27	177.61
Snake RM 18 (ICE-A)	91.46	21.77	236.11	50.38	13.12	346.38
Snake RM 6 (ICE-B)	22.70	8.76	250.56	24.92	5.34	248.18
Columbia RM 410 (PRI-A)	20.43	9.01	410.59	23.26	11.64	353.06
Columbia RM 369 (HAN-A)	10.60	1.55	187.69	32.52	5.82	146.60
Columbia RM 326 (MNA-A)	42.06	16.28	368.80	42.92	47.20	117.40

The Autotrophic Index decreased below each of the dams with a general decrease downstream through the study area. Highest values were obtained at Lower Granite RM 118 and Little Goose Reservoir RM 81 which consistently had the highest amount of sediment accumulation on the tiles. Mean AI at Lower Granite and Little Goose Reservoirs was 604.4 and 685.0 at the 0.75 m depth. This is in contrast to the AI below the impoundments at RM 105 and RM 67, which had AI values of 252.5 and 291.8 at the 0.75 m depth. Lowest AI values occurred below Lower Monumental Dam with AI values of 187.4 and 177.6 at the 0.75 m and 1.5 m depths. The AI then increased at Ice Harbor Reservoir to 250.6 and 248.2 (Table 3.6-9).

In the Columbia River, mean chlorophyll a at the 1.5 m depth increased for the upstream to the downstream stations. Priest Rapids (RM 410), had a mean chlorophyll *a* of 23.3 mg/m^2, followed by 32.5 mg/m^2 at RM 369, then increased sharply at RM 326 in the McNary reservoir to 42.9 mg/m^2. The AI at the 1.5 m depth decreased downstream from 353.1 at RM 410 to a low of 117.4 at McNary reservoir (Table 3.6-9).

Lower Snake River Impoundments

General Environmental and ABA Trends

The four impounded sites: Lower Granite (RM 118), Little Goose (RM 81), Lower Monumental (RM 52), and Ice Harbor (RM 18) reservoirs were very similar. Sites were comparable in temperature, light profiles, substrate, and sediment accumulation. Substrate consisted of fine particles, silt with some sand, and highly embedded cobble. Care was taken in placing and picking up tiles as not to stir up the fine substrate. The tiles at all four impounded sites accumulated a substantial amount of sediment during the fourteen-day incubation periods. Sediment deposition gradually increased throughout the summer, peaking with the lower flows during August. Although total sediment deposited on the tiles fluctuated, Little Goose consistently had the highest amounts of sediment accumulation throughout the field season. Even though some of these tiles had a large amount of sediment accumulation, there was still considerable algal growth on the tiles. The tiles had a matrix of attached benthic algae community growing on and in the deposited sediment layer and it was not possible to separate the algae from the sediment. Therefore, the entire tile was scraped and processed as described in Section 3.6.

Littoral substrate was comprised mainly of embedded cobble, fine particles, silt, and sand, along with large organic debris and detrital material. There was a slight shift in substrate composition downstream from Lower Granite to Ice Harbor. The upstream impoundments, Lower Granite and Little Goose were characterized by an increased amount of large organic debris, fines, and silts with some sand. Progressing downstream to Ice Harbor, there was a reduction in fines and silts with increased prevalence of sand and gravel.

Lower Monumental Reservoir differed from the three other sites due to the inflow of the Tucannon and Palouse Rivers upstream of the study site. These rivers carry a high amount of sediment from surrounding upland loess soils in the agricultural region of southeastern Washington. The Lower Monumental impounded study site also had lower transparency and higher VEC's than the upstream sites.

Of the four impoundments, Ice Harbor had the most recreational use. Along with the increased boating activity, two large parks that are fertilized and irrigated are located downstream of Lower Monumental Dam. In addition, proximity of agricultural use to the river increased downstream to Ice Harbor Dam.

Temperature was comparable between the four reservoir sites. Surface temperatures ranged from 17.6 - 19.8°C during the second week of July. Temperature peaked at 22.8°C during the first week of September (Appendix K).

Light attenuation varied between incubation periods. Although each dam acts as a settling basin with generally increasing water clarity downstream, Secchi disk measurements showed decreased light transparency from Lower Granite to Lower Monumental Reservoir, before showing a slight increase in light attenuation in Ice Harbor Reservoir. Secchi disk measurements ranged from a low of 0.8 m in Lower Monumental Reservoir to a high of 2.9 m in Lower Granite Reservoir (Appendix K).

Substrates were placed at three depths: 0.75 m, 1.5 m, and 3.0 m at the reservoir sites. The majority of attached benthic algal growth, as measured by monochromatic chlorophyll *a*, occurred in the 0.75 m to 1.5 m range. Growth on the 3.0 m substrates was minimal, and exhibited increased growth with decreased flows and increased light attenuation.

Overall, the general trend was of increased attached benthic algal growth downstream. Ice Harbor Reservoir had the largest ABA chlorophyll *a* values of the four impounded sites. Growth at Ice Harbor was also atypical in that its monochromatic chlorophyll *a* values peaked much earlier in the season during the August 7 to 21 incubation period. Chlorophyll *a* values at the 0.75 m depth at Ice Harbor were over 100 mg/m2 on three of the five incubation periods (Appendix K). At the most upstream impounded site, Lower Granite (RM 118), chlorophyll *a* values at 0.75 m were minimal during July and August, with a high of 17.2 mg/m^2, but increased in late September to an annual high of 73.4 mg/m^2 (Appendix K).

Biomass levels were variable at all sites, with fluctuating values at each depth over the sampling period. Biomass values are affected by algal growth, organic deposition, protozoans, and bacteria. Lower Granite exhibited the highest range of values ranging from 0.12 g/m^2 to 78.9 g/m^2 at the 1.5 m depth (Appendix K). The AFODW peak values coincided with the lower flows in mid-August prior to the chlorophyll *a* peak.

The Autotrophic Index decreased over time during the July to October sampling period and decreased downstream from Lower Granite to Ice Harbor Reservoir. Lower Granite ranged from 195.1 to 2,309.1 whereas Ice Harbor ranged from 123.7 to 280.9 (Appendix K).

Algal community composition was consistent throughout the four reservoirs. The communities were dominated by diatoms, blue-green, and filamentous green algae. Dominant species at all impounded sites included *Cymbella, Gomphonema, Fragillaria, Lyngbya, Aphanizomenon, Oscillatoria*, and *Rhizoclonium*. Other species which were present in the reservoirs included *Melosira, Tabelleria, Diatoma, Synedra*, and *Spirogyra*. The algal community on the tiles were more flocculent and easily disturbed as compared to the free-flowing reaches where the attached benthic algae was firmly attached to the substrate.

Lower Granite Reservoir (RM 118)

Of the four lower Snake Reservoirs, Lower Granite consistently had higher Secchi disk measurements and lower VEC's. The low Secchi of 1.1 m occurred on July 22, 1997 increasing to 2.9 m by October 16, 1997. Temperature during the first incubation period was 19.0°C climbing to a high of 22.2°C on September 4, and ending with 15.0°C by mid-October (Figure 3.6-35).

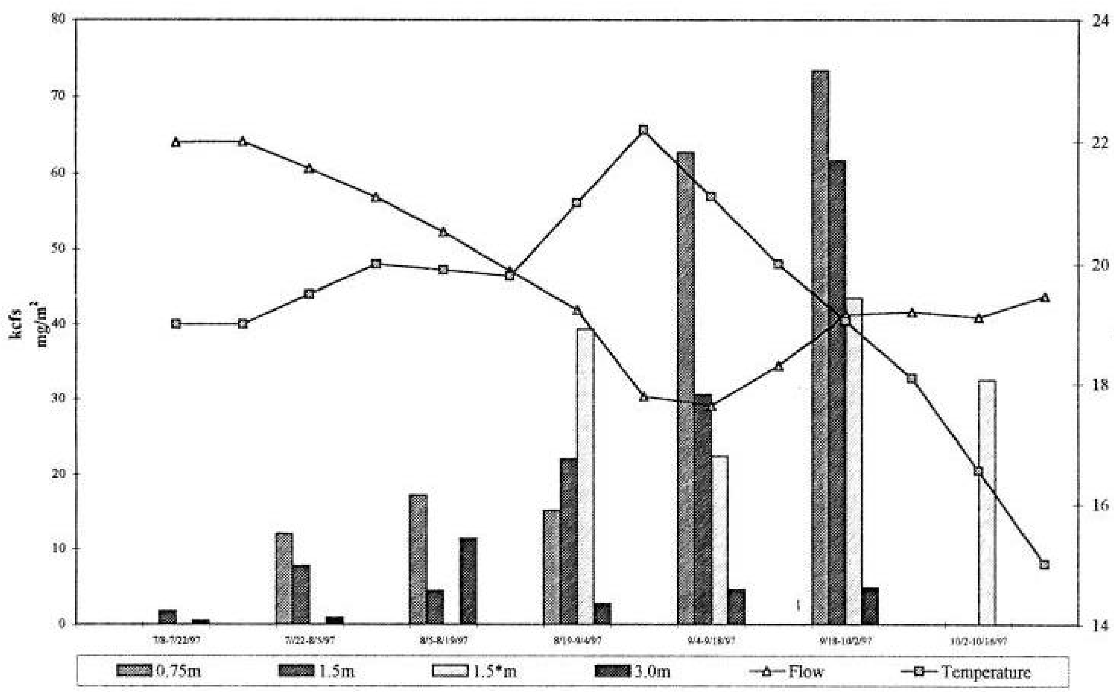

Figure 3.6-35. Attached benthic algae chlorophyll *a*, discharge, and temperature at Snake River Mile 118 (Lower Granite Reservoir), July to October, 1997 (*indicates incubation on Mylar substrate)

Substrate at the site consisted of medium-sized boulders, completely embedded cobble, silt, fine particles, and some sand. The fines were easily disturbed and resuspended into the water column. There was an abundance of organic matter at the site consisting primarily of logs, leaves, and detritus.

Natural substrates, mylar, and tiles were used at the site. The site was dredged for rocks on August 8 from 0.5 m to 1.5 m. Despite repeated attempts, deeper dredging was unsuccessful. Any rocks in the area were too large and highly embedded, and any attempts at retrieving rocks in deeper water were futile. Results of the shallow dredged samples revealed highest chlorophyll *a* values in the 0.5 m to 1.0 m samples. Natural substrate chlorophyll a at 0.5 m ranged from 87 to 107 mg/m^2, at 1.0 m values of 36 to 73 mg/m^2 and 1.5 m values of 11 to 25 mg/m^2 (Appendix K). Artificial substrates had peak ABA chlorophyll *a* from September 18 to October 2, as flows began to increase after the summer low (Figure 3.6-35). Algal growth at Lower Granite Reservoir was minimal throughout July and August, with the lowest chlorophyll *a* values of the four reservoir sites. Monochromatic chlorophyll *a* from July 8 to 22 was minimal, with 1.8 mg/m^2 at 1.5 m and 0.5 mg/m^2 at 3.0 m. However, by late September, chlorophyll *a* values climbed to a high of 73.4 mg/m^2 at 0.75 m, 61.5 mg/m^2 at 1.5 m, and 4.8 mg/m^2 at 3.0 m (Appendix K).

Biomass on the tiles primarily consisted of detritus, living algae, protozoans, and bacteria. Biomass values fluctuated but peaked earlier than chlorophyll *a* values during the August 19 to September 4 incubation period, thereby coinciding with increased temperature and decreased flow. The peak values of AFODW incubation values ranged from 9.1 g/m^2 at 0.75 m, 78.9 g/m^2 at 1.5 m (tile substrate), 19.3 g/m^2 on the mylar substrate at 1.5 m, to 1.8 g/m^2 at 3.0 m (Appendix K). The mylar values were consistently lower than the tiles due in part that the mylar were free-floating and moving, therefore did not accumulate much sediment.

The Autotrophic Index was very high. The general trend was of moderate to high AI values during July, peaking during August, and then declining in September and October. The lowest AI values of 230.2 at 1.5 m (mylar) occurred during the last incubation period from September 18 to October 2, and the highest value of 2,309.1 at 1.5 m occurred on August 19, 1997 (Appendix K).

Little Goose Reservoir (RM 81)

Transparency at Little Goose Reservoir was comparable to that of the upstream reservoir, Lower Granite. The low Secchi disk measurement of 1.1 m was recorded on July 23 and the high of 2.2 m on September 5. The VEC ranged from 1.3 to 0.8 on the above dates. Temperature during the season reached a high of 21.8°C on September 5 and lows of 18.0°C on July 9 and October 3 (Figure 3.6-36).

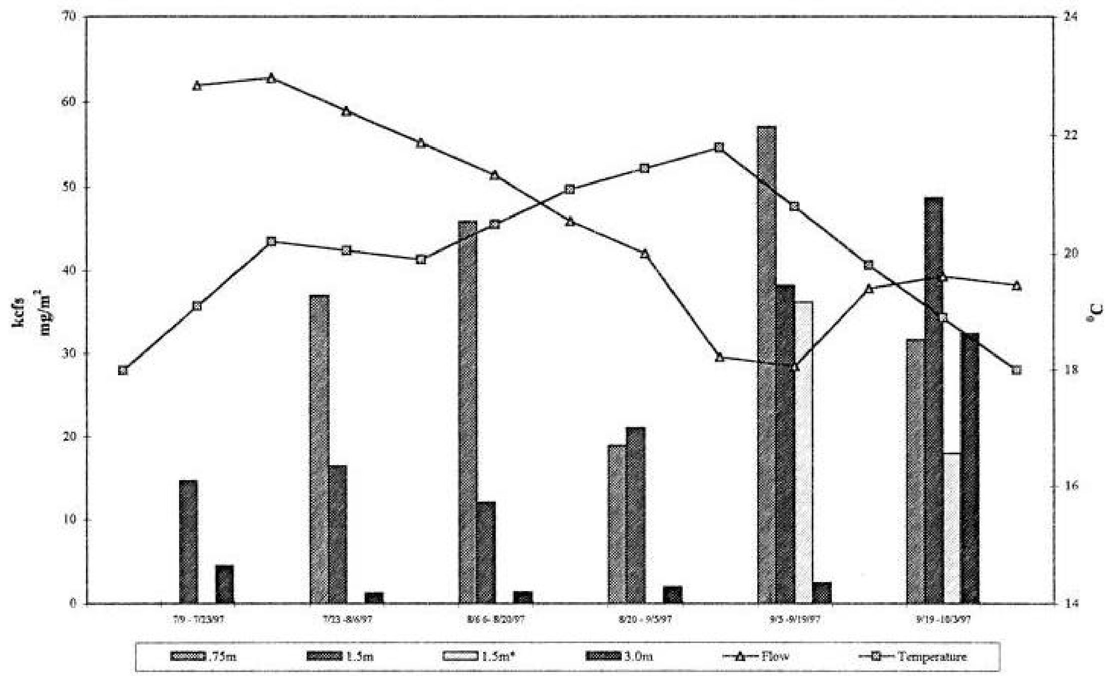

Figure 3.6-36. Attached benthic algae chlorophyll *a*, discharge, and temperature at Snake River Mile 81 (Little Goose Reservoir), July to October 1997 (*indicates incubation on Mylar substrate)

Substrate was similar in composition to Lower Granite Reservoir consisting of fines, silt, sand with completely embedded cobbles. Shoreline substrate also contained large partially embedded boulders. The fines were easily disturbed and settled on the tiles. However, Little Goose did not have the abundance of large organic debris as did Lower Granite Reservoir (RM 118).

Chlorophyll *a* values climbed steadily throughout the summer, except for sharp decline on the August 20 to September 5 incubation period. Water depth had increased and tiles were retrieved at 1.5 m, 2.25 m, and 3.5 m depths rather than their original set depths of 0.75 m, 1.5 m, and 3.0 m. Although Little Goose had more growth at the onset as compared to Lower Granite, it did not have the higher peak growth later in the season. Monochromatic chlorophyll *a* values ranged from 14.7 mg/m^2 at 1.5 m and 4.4 mg/m^2 at 3.0 m depths during July 9 to 23 to final values of 48.6 mg/m^2 and 32.4 at 1.5 m and 3.0 m depths (Figure 3.6-36). Although Little Goose had comparable light profiles and Secchi disk depths to Lower Granite, it consistently supported the highest growth at the deepest incubation depth of 3.0 m.

The AFODW values during the two incubation periods in July were low, ranging from 15.9 g/m^2 to 3.2 g/m^2 at 1.5 m and 1.1 g/m^2 to 0.9 g/m^2 at 3.0 m (Appendix K). Although values at each depth fluctuated, AFODW values were elevated from August to the beginning of October. There was a small decline as with the chlorophyll *a* values in the beginning of September when tiles were retrieved from the deeper water.

The AI values ranged from a low of 125.6 on August 20 to September 5 at the 1.5 m depth to a high of 1,105.7 at 3.0 m on September 5 to 19. The 3.0 m depth had the highest AI values throughout most of the season (Appendix K).

Lower Monumental Reservoir (RM 52)

Lower Monumental pool typically exhibited the lowest water transparency of the four reservoirs. Secchi disk depth ranged from 0.8 m (July 23) to 1.8 m on September 6. Surface temperature ranged from 17.6°C at the beginning of the study climbing to a high of 21.6°C by September 20 (Figure 3.6-37).

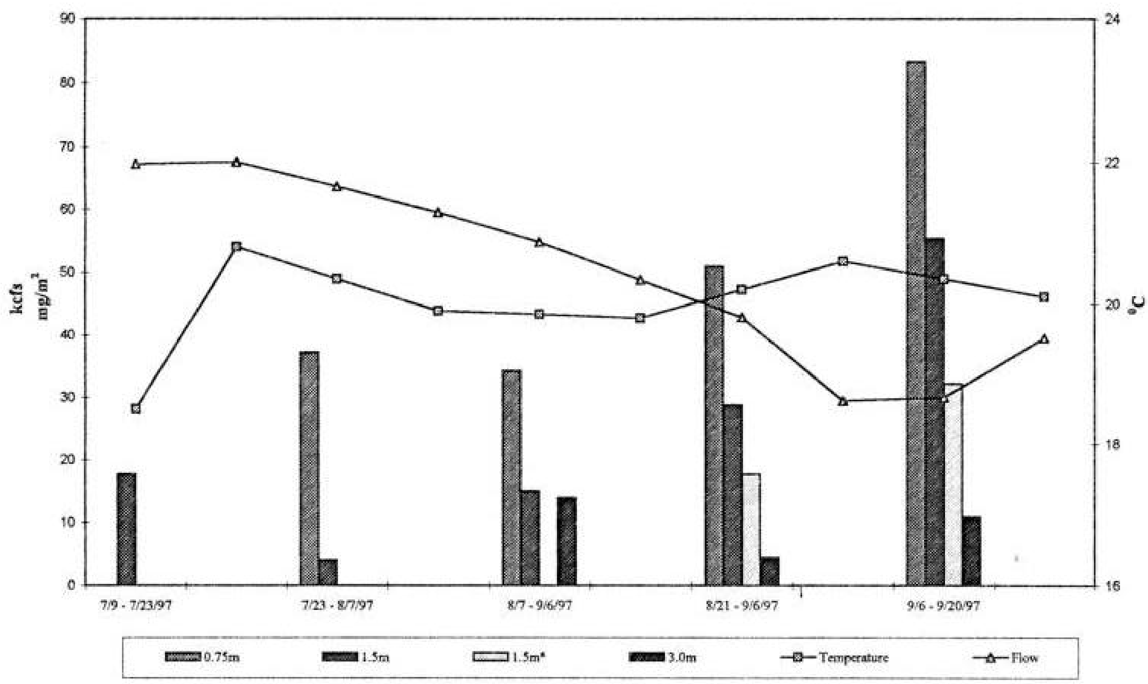

Figure 3.6-37. Attached benthic algae chlorophyll *a*, discharge, and temperature at Snake River Mile 52 (Lower Monumental Reservoir), July to September 1997 (* indicates incubation on Mylar substrate)

Substrate at the site was similar to Lower Granite and Little Goose in the prevalence of fines and silts. However, there was an increasing amount of sand, with large boulders along the shoreline above the waterline, and a decreased amount of large organic debris. The site still had detrital material, but large logs and woody debris were much reduced.

The large boulders at the shoreline had a high amount of algal growth. Therefore, tiles were set out at approximately 0.2 m depth to sample the shoreline area. Even with fluctuating water levels, shoreline growth on the tiles was immense with the highest of 80.2 mg/m^2 on August 7 to 21. Natural rock substrate sampled at the area from July 9 to 23 revealed high chlorophyll *a* values of 125.4 - 436.2 mg/m^2 (Appendix K).

Algal growth increased at 0.75 m and 1.5 m depths with the largest values during lowest discharge (Figure 3.6-37). Early summer monochromatic chlorophyll a values were at 1.5 m 17.8 mg/m^2 rising to 55.4 mg/m^2 by September 20. Values at 3.0 m fluctuated but were comparable to the high values found at Little Goose ranging from 4.4 mg/m^2 to 14.0 mg/m^2 despite the lower measured water transparencies in Lower Monumental Reservoir (Appendix K).

Biomass also fluctuated throughout the summer. The 3.0 m depth consistently had higher AFODW ranging from 16.9 - 53.4 g/m^2. The 1.5 m depth ranged from 1.2 g/m^2 to 10.9g/m^2 on the second and third incubation periods (July 23 to August 7, and August 7 to 21) whereas at the 0.75 m depth highest AFODW were during July 7 to 23 at 22.1 g/m^2 and a low of 6.8 g/m^2 during August 21 to September 6, 1997 (Appendix K).

The AI values were much lower than the upstream reservoirs at the 0.75 m and 1.5 m depth whereas at the 3.0 m depth, the AI values were much higher than the upstream sites. The lowest AI at 1.5 m was 84.3 from August 21 to September 6 and a high of 369.6 during the first incubation period. At the 3.0 m depth, values ranged between 1,009.8 and 1,976.8 (Appendix K).

Ice Harbor Reservoir (RM 18)

Water transparency at Ice Harbor was similar to Lower Monumental ranging from 0.9 m at the start of the field season to a high of 1.6 m on September 6, 1997. Temperature showed similar trends as the other reservoirs with an early season low of 18.5° C at the surface and a peak of 24.1° C in mid-August (Appendix K).

Substrate composition was similar to Lower Monumental, but there was a definite decrease in fine particles and an increase in sand and gravel as compared to the three upstream reservoirs.

Chlorophyll *a* values in Ice Harbor were the highest of the impounded reaches and did not exhibit the trend of high chlorophyll *a* values with decreased flow. In fact, chlorophyll *a* values dropped in late August and at the beginning of September when the other sites exhibited increased growth (Figure 3.6-38). The high chlorophyll *a* values at 0.75 m and 1.5 m occurred during August 7 - 21, 1997 with values of 133.7 mg/m^2 and 70.9 mg/m^2, respectively (Figure 3.6-38).

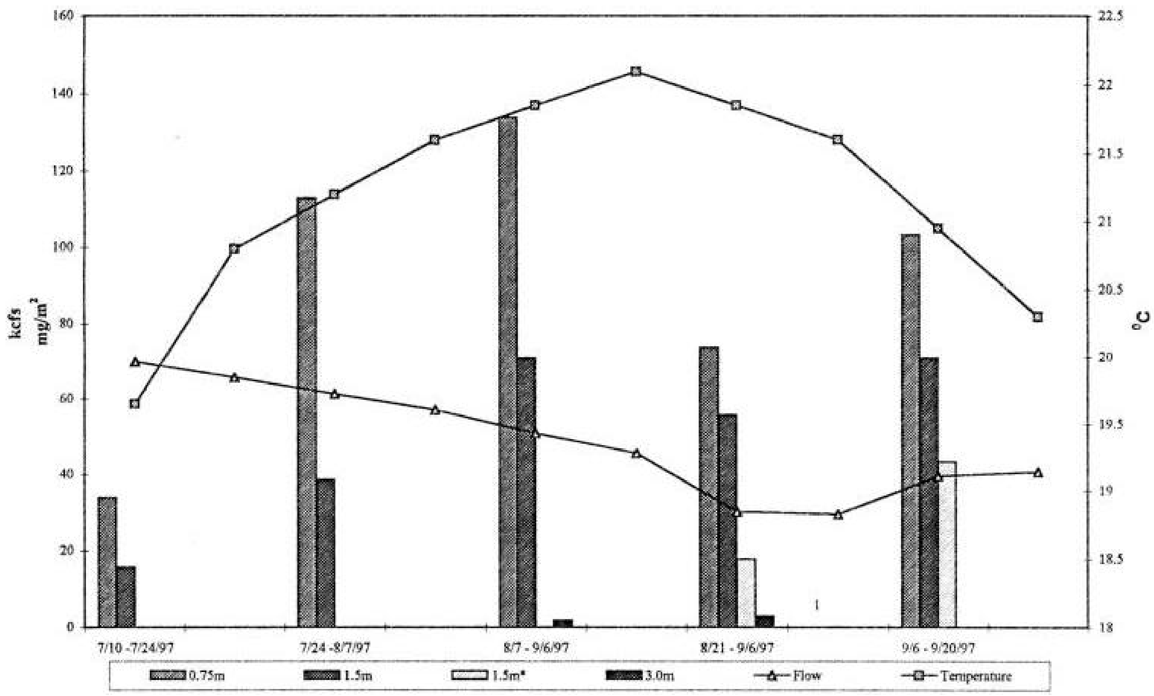

Figure 3.6-38. Attached benthic algae chlorophyll *a*, discharge and temperature at Snake River Mile 18 (Ice Harbor Reservoir) July to September 19997 (* indicates incubation on Mylar substrate)

The highest AFODW values were at the 0.75 m depth ranging from 10.9 g/m2 to 37.1 g/m2 while the only two measurements at the 3.0 m depth were 0.4 g/m^2 and 0.5 g/m^2 (Appendix K).

The AI gradually decreased throughout the sampling time frame (Appendix K). Starting AI values were 275.7 and 218.1 at the 0.75 m and 1.5 m depth falling to 176.5 and 123.7 during the last run from September 6 to 20, 1997.

Columbia River Impoundments

General Environmental and ABA Trends

Two impounded areas were studied on the Columbia River, Priest Rapids Reservoir (RM 410) and McNary Reservoir (RM 326). The differences between the two sites were considerable. The McNary reach differs from upstream impounded reaches on the Columbia because of the input of the Snake River and its location downstream from the urban population and industrial concentration in the Tri-Cities area as well as the Yakima River inflow.

Priest Rapids Reservoir (RM 410)

Relative to other impounded sites in this study, Priest Rapids was very clear with Secchi disk measurements ranging from 2.1 m on July 11 to 4.9 m on September 5, 1997 (Appendix K). The VEC's were considerably lower than those observed at McNary ranging from 0.4 to 0.8 (Appendix K). The chlorophyll *a* peaked when flows were lowest during the August 21 to September 6 incubation period with monochromatic chlorophyll a values at 1.5 m (tiles) of 51.5 mg/m^2 (Figure 3.6-39). Due to the increased photic zone depth, monochromatic chlorophyll *a* values in Priest Rapids Reservoir at the 3.0 m depth were generally higher than at McNary Reservoir, with a high of 11.0 mg/m^2 during the August 7 to 21 incubation period.

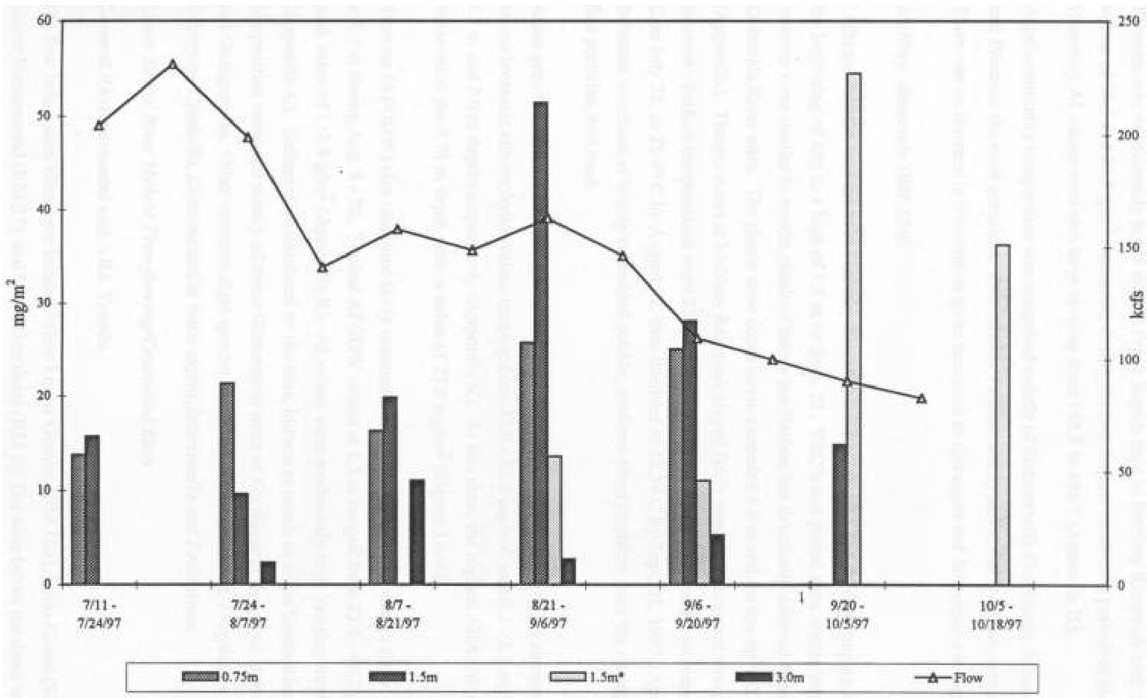

Figure 3.6-39. Attached benthic algae chlorophyll *a*, and discharge at Columbia River Mile 410 (Priest Rapids Reservoir) July to October 1997 (*indicates incubation on Mylar substrate)

Biomass was considerably lower than in Priest Rapids than in McNary Reservoir with the highest values of 20.5 and 23.7 g/m^2 occurring during the two August incubation periods at the 1.5 m depth. However, AI values were still large ranging from 165.9 to 489.7 (Appendix K).

Algal community composition was comprised mainly of diatoms with *Fragillaria, Melosira, Cymbella*, and *Diatoma* the most prevalent. *Asterionella, Pediastrum, Synedra*, and *Lyngbya* were also present. There was an increase in filamentous green taxa such as *Spirogyra* and *Microspora* in September.

McNary Reservoir (RM 326)

Light penetration was lower at McNary than at Priest Rapids with Secchi disk ranging from 1.2 m at the beginning of July to a high of 1.8 m on Sept. 21. VEC's and photic zone depths throughout the summer were similar to results obtained below Ice Harbor, but drastically differed from upstream Columbia River sites. The photic zone depth never exceeded 4.8 m and the lowest VEC was 1.0 (Appendix K). Temperatures at McNary Reservoir stayed fairly warm and constant throughout the summer. Surface temperatures were 22.0 to 22.8° C while at 1.5 m temperature increased from 20.9° C on July 25, to 22.4° C by August 22, then declined to 18.5° C by September 21, 1997 (Appendix K). Substrate consisted of highly embedded cobble, medium-sized boulders from the riprap shoreline, fine particles, and sand.

Algae growth was abundant and growth remained fairly constant throughout the summer with monochromatic chlorophyll *a* values ranging from 43.0 - 52.8 mg/m^2 and 0.3 - 2.9 mg/m^2 at the 1.5 m and 3.0 m depths, respectively (Appendix K). At this time, the highest ABA chlorophyll was observed at the 0.75 m depth, with a value of 73.9 mg/m^2 (Figure 3.6-40).

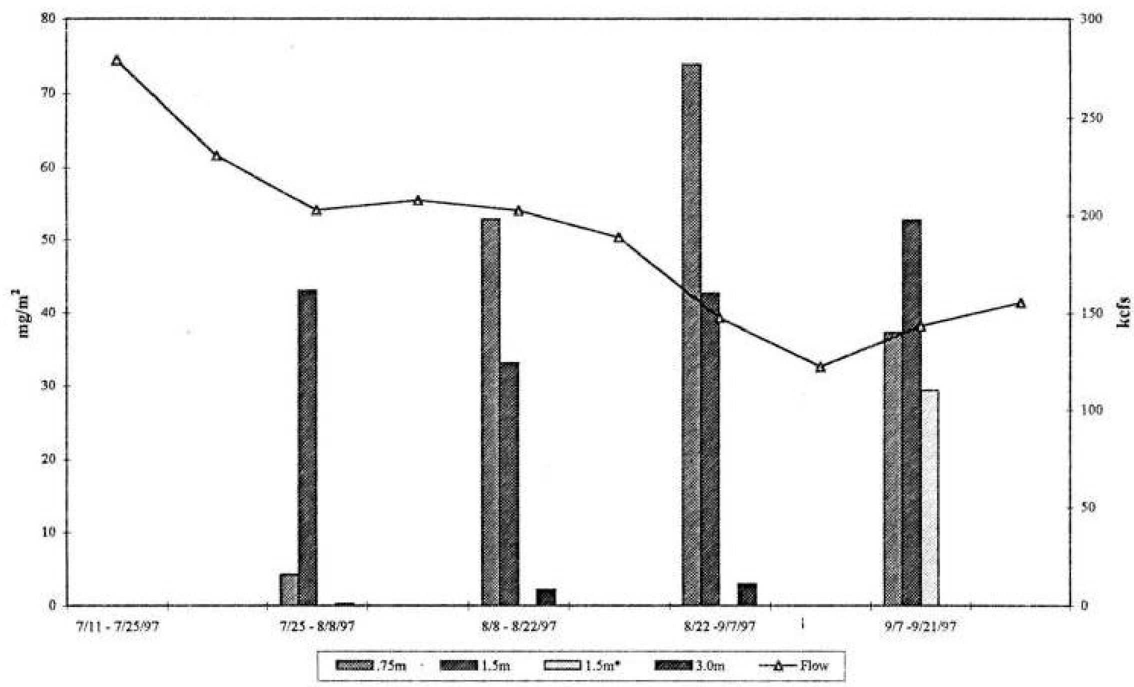

Figure 3.6-40. Attached benthic algae chlorophyll *a*, and discharge at Columbia River Mile 326 (McNary Reservoir) July to September 1997 (*indicates incubation on Mylar substrate)

Biomass (AFODW) also remained fairly constant throughout the summer except for an earlier peak at 1.5 m during August 8 to 22. Typical AFODW values at 1.5 m ranged from 23.0 - 48.7 g/m^2 with a peak value of 112.9 g/m^2 (Appendix K). The AI values were moderately high. ranging from 242 to 551 (Appendix K). Sediment accumulated on the tiles, but not as much as in the Snake River. Algal composition consisted mostly of dense filamentous mats of *Oscillatoria, Lyngbya, Aphanizomenon*, and *Oedogonuim*. Other common algal species consisted of *Gomphonema, Fragillaria, Melosira, Spirogyra, Cymbella, Diatoma* and, to some extent, *Asterionella* and *Pediastrum*.

Lower Snake River Hybrid Free-flowing/Controlled Sites

General Environmental and ABA Trends

The four below dam sites were located below Lower Granite (RM 105), Little Goose (RM 67), Lower Monumental (RM 37), and Ice Harbor dams (RM 6). The sites below the dams were still affected by sediment accumulation although to a lesser degree than the reservoir sites. However, sediment on the tiles decreased downstream with little deposition on tiles at RM 37 and RM 6. As with the change in sediment, there was a slight shift in substrate composition. At RM 105 and 67, substrate was very similar in composition to the sites above the dams, consisting of fine particles, silt, and some sand. However, by RM 37 and RM 6, the substrate shifted with an increased proportion of sand and gravel and less fine particles and silt.

Light attenuation was higher upstream below Lower Granite and Little Goose with Secchi disk measurements ranging from 0.9 - 2.5 m compared to 0.9 - 1.7 m below Lower Monumental and Ice Harbor (Appendix K).

Monochromatic chlorophyll *a* values were fairly similar between the four sites with highest growth in September. Of the four sites, Lower Granite had the highest chlorophyll *a* values of 58.0 mg/m^2 at 1.5 m and 83.1 mg/m^2 at the 0.75 m depth from September 18 to October 2 (Appendix K).

Below Lower Granite Dam (RM 105)

Water transparency fluctuated with a low Secchi disk measurement of 1.2 m on July 22 and a high of 2.5 m on September 4. Temperature was 18.2° C and 18.0° C degrees on July 7 and October 2, respectively, and reached a high of 21.8° C on September 4 at 1.5 m (Appendix K).

Substrate was comparable to the upstream reservoir site, Lower Granite Reservoir (RM 118). Substrate consisted of completely embedded cobble, larger partially embedded boulders and fine particles, silt, and sand. This site also had large organic debris - mostly woody and some leaf litter.

Monochromatic chlorophyll *a* values gradually increased in mid-August to values of 42.0 mg/m^2 and 34.1 mg/m^2 at 0.75 m and 1.5 m depths. From mid-August to mid-September, chlorophyll *a* values remained fairly steady, with a large increase during the last incubation period from September 18 to October 2 (Figure 3.6-41). Ending values at 0.75 m were 83.1 mg/m^2 on tile substrate. At 1.5 m, chlorophyll *a* was 58.0 mg/m^2 and 60.1 mg/m^2 on tile and mylar substrate, respectively (Appendix K).

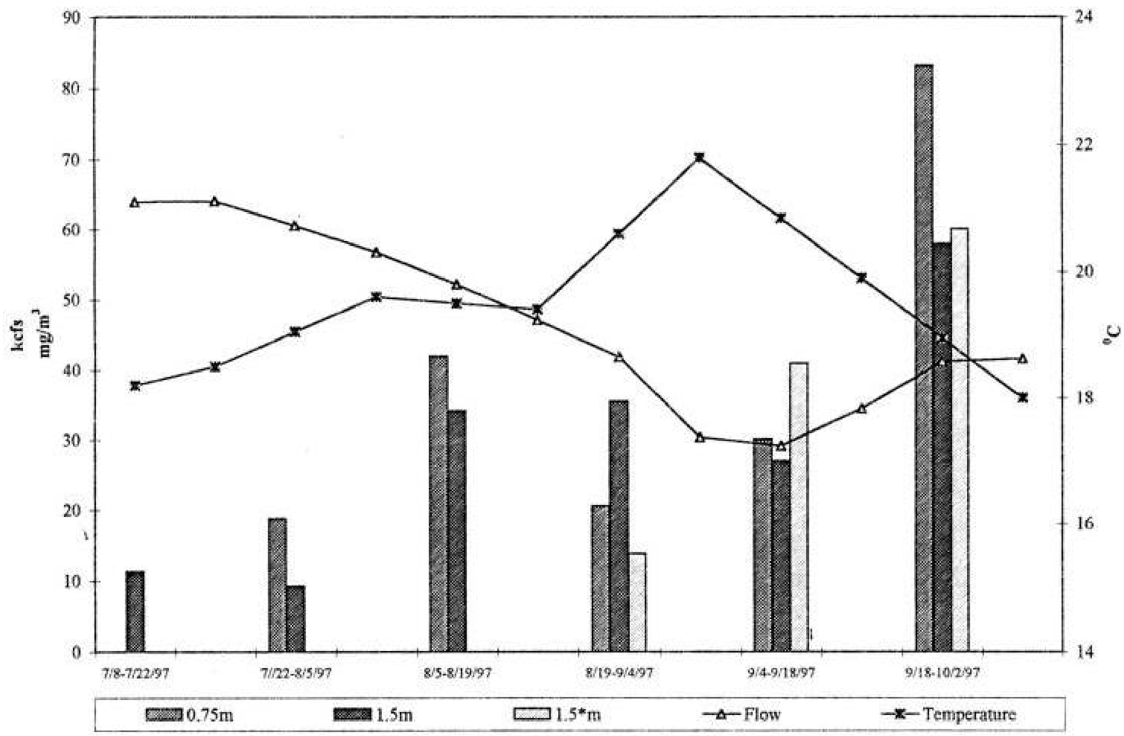

Figure 3.6-41. Attached benthic algae chlorophyll *a*, discharge, and temperature at Snake River Mile 105 (Below Lower Granite Dam) July to October 1997 (*indicates incubation on Mylar substrate)

Dredges for natural substrate from 0 to 10 m on August 5 revealed the majority of algal growth concentrated from 0.25 to 1.0 m with chlorophyll *a* values ranging from 194.6 to 104.9 mg/m^2. By 2.0 m, chlorophyll *a* values dropped to 13.4 mg/m^2, and by 3.3 m values ranged from 0.68 - 5.62 mg/m^2. One rock was obtained from a 10 m depth which had a monochromatic chlorophyll *a* of 1.93 mg/m^2 (Appendix K).

The AFODW values were minimal during the first two incubation periods with beginning values of 1.3 g/m^2 at the 1.5 m depth. The 1.5 m values peaked during the August 19 to September 4 incubation period with a high of 62.8 g/m^2 (Appendix K).

The Autotrophic Index followed the same trend as AFODW. Initial values were 109.4 with a large increase during Aug. 19 to Sept. 4 to 1,156.0 at 1.5 m. The AI values at the 1.5 m depth were consistently larger than the 0.75 m values during all but one incubation period. The shallow depth AI values ranged from 179.4 to 332.8 (Appendix K).

Below Little Goose Dam (RM 67)

The site below Little Goose Dam had relatively low water transparency with a Secchi range of 0.9 m on July 23 to 1.5 m on July 9. There was a general decline in VEC throughout the summer, beginning with 1.5 m on July 9 dropping steadily to a low on September 5 of 0.7 m. The VEC then increased on September 19 to 1.2 (Appendix K). Temperature remained fairly constant with an initial temperature of 18.1° C and a high of 20.9° C by September 5 at the surface (Appendix K).

Substrate exhibited decreased silt and fine particles and more sand, with less embedded cobble relative to other sites. The shoreline consisted of gravel and cobble with dense vegetation on the south shore.

Chlorophyll *a* values peaked during the last run at 37.6 mg/m^2 at 0.75 m, and 46.4 mg/m^2 and 47.9 mg/m^2 at the 1.5 m depth for tile and mylar substrates, respectively (Figure 3.6-42).

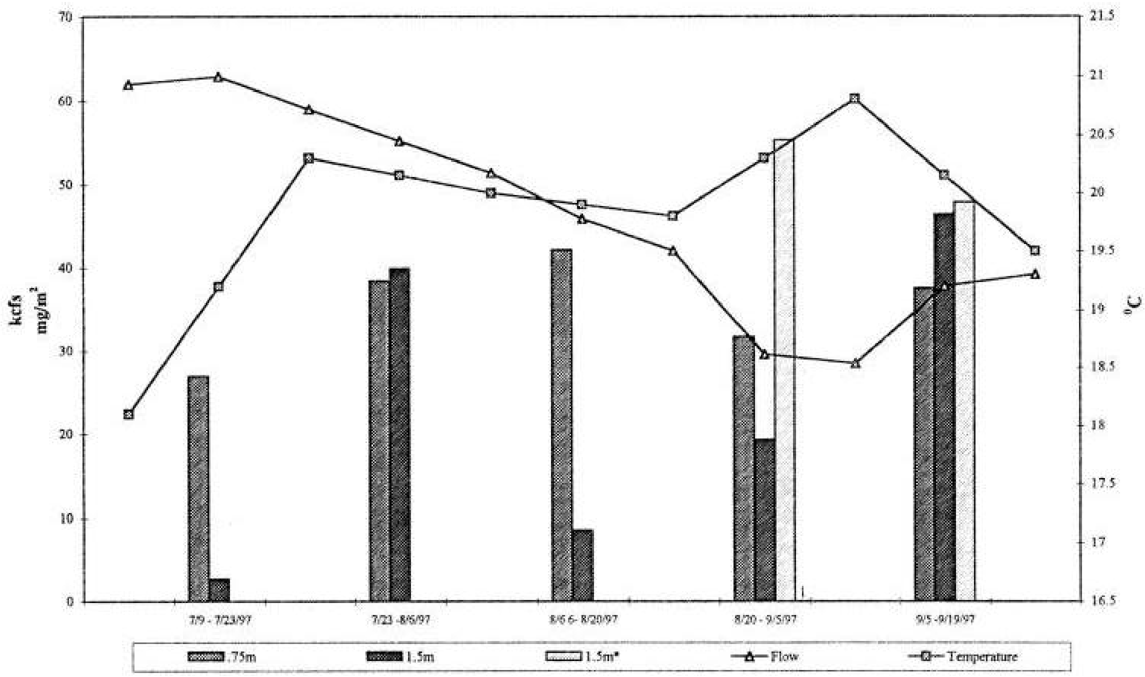

Figure 3.6-42. Attached benthic algae chlorophyll *a*, discharge, and temperature at Snake River Mile 67 (Below Little Goose Dam), July to September 1997 (*indicates incubation on Mylar substrate)

The AFODW values gradually decreased throughout the sampling period at the 0.75 m depth from 17.5 g/m^2 to 8.6 g/m^2. At the 1.5 m depth, initial values during the first three incubation periods from July 9 to August 20 increased slightly from 0.8 g/m^2 to 6.5 g/m^2. However, there was a large increase from August 20 to September 4 with an AFODW of 140.5 g/m^2 before decreasing to 29.0 g/m^2 from September 5 to 19 (Appendix K).

The Autotrophic Index varied from 120.7 to 427.5 at the 0.75 m depth on July 23 to August 6 and August 20 to September 5. At the 1.5 m depth, the mylar values were lower than the tiles during the last two incubation periods of August 20 to September 19 with values of 141.0 and 267.2. The tiles at 1.5 m ranged from a low of 220.4 at the beginning of the field season to a high of 498.0 during the last incubation period from September 5 to 19 (Appendix K).

Below Lower Monumental Dam (RM 37)

Transparency below Lower Monumental Dam gradually increased from July 10 to August 7 with Secchi measurements of 0.9 - 1.7 m. After August 7, the transparency declined to 1.1 m on September 20. Temperature remained constant throughout the field season. A slight increase was observed from July 10 to 24 with temperatures of 17.9° C to 20.2° C. Between July 24 and September 20, recorded temperatures ranged from 20.0 to 20.8° C at 1.5 m (Appendix K).

Substrate at the site consisted of riprap shoreline with fines, silt, and sand. There was little organic debris at the site.

Algal growth at 1.5 m peaked midway in the season with monochromatic chlorophyll *a* values beginning with 20.5 mg/m^2 increasing to 78.6 mg/m^2 during the August 7 to 21 incubation period (Figure 3.6-43). During this period, there was a large bloom of Rhizoclonium on both the tile substrate and the riprap.

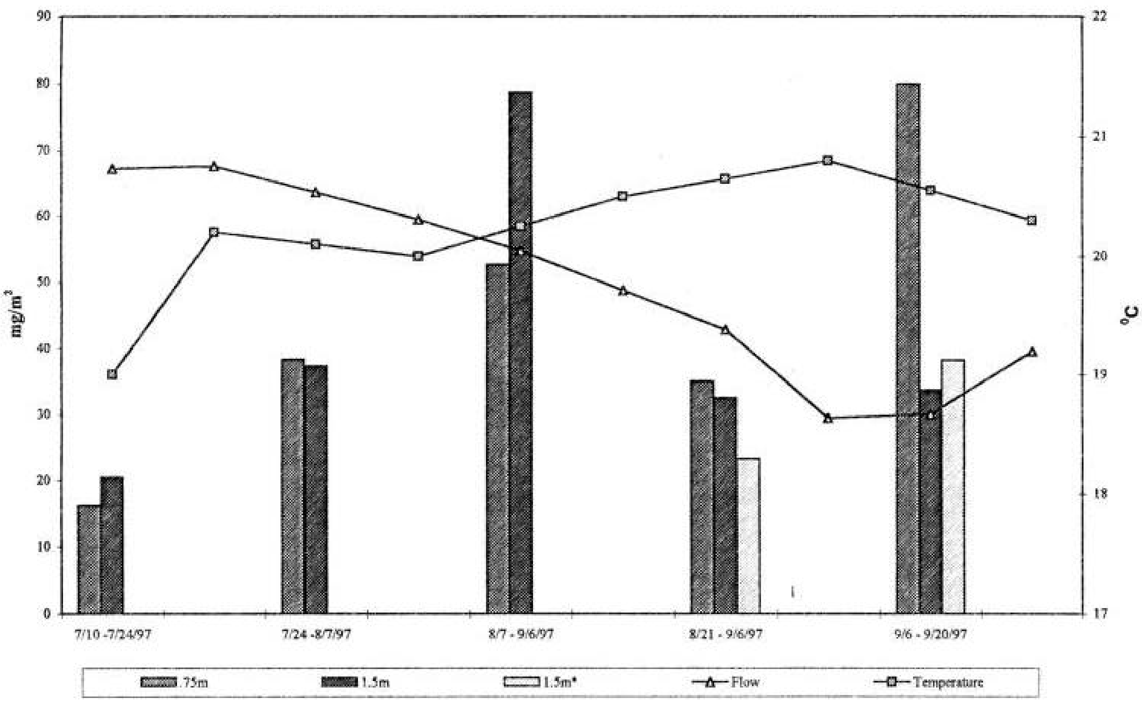

Figure 3.6-43. Attached benthic algae chlorophyll *a*, discharge, and temperature at Snake River Mile 37 (Below Lower Monumental Dam), July to September 1997 (*indicates incubation on Mylar substrate)

Natural substrate dredges from 0 to 8.0 m on August 21, revealed chlorophyll *a* values of 65.0 mg/m^2 at 2.0 m dropping steadily to 6.86 mg/m^2 by 6.0 m, and increased to 29.7 mg/m^2 at 7.0 m but then sharply decreased at 8.0 m to 2.0 mg/m^2 (Appendix K).

The AFODW peaked with chlorophyll at values of 11.6 g/m^2 and 17.8 g/m^2 at the 0.75 m and 1.5 m depths, respectively (Appendix K). The AI values fluctuated throughout the summer ranging from 106.8 to 270 at the 0.75 m depth and 128.9 to 242.0 at the 1.5 m depth (Appendix K).

Below Ice Harbor Dam (RM 6)

Light penetration was low with Secchi disk measurements ranging from 0.9 to 1.4 m throughout the summer. The VEC's decreased through the summer from 1.6 on July 11, 1997 to 1.0 on September 21, 1997. Temperature below the dam at the surface ranged from 18.8° C on July 11, to 21.2° C by August 22 and declined to 19.8° C by September 21.

Substrate was primarily sand, gravel, and embedded cobble. In comparison to the sites below Lower Granite and Little Goose, substrate composition was very different. There was a major decrease in the amount of silt and fine particles.

Chlorophyll *a* values remained fairly constant throughout the season, except for an increase during the August 22 to September 7 incubation period with monochromatic chlorophyll *a* values of 27.1 mg/m^2 at 0.75 m and 32.4 mg/m^2 and 37.5 mg/m^2 at the 1.5 m tile and mylar substrates. Early and late season chlorophyll *a* values ranged from 21.4 to 23.8 mg/m^2 at 1.5 m (Figure 3.6-44).

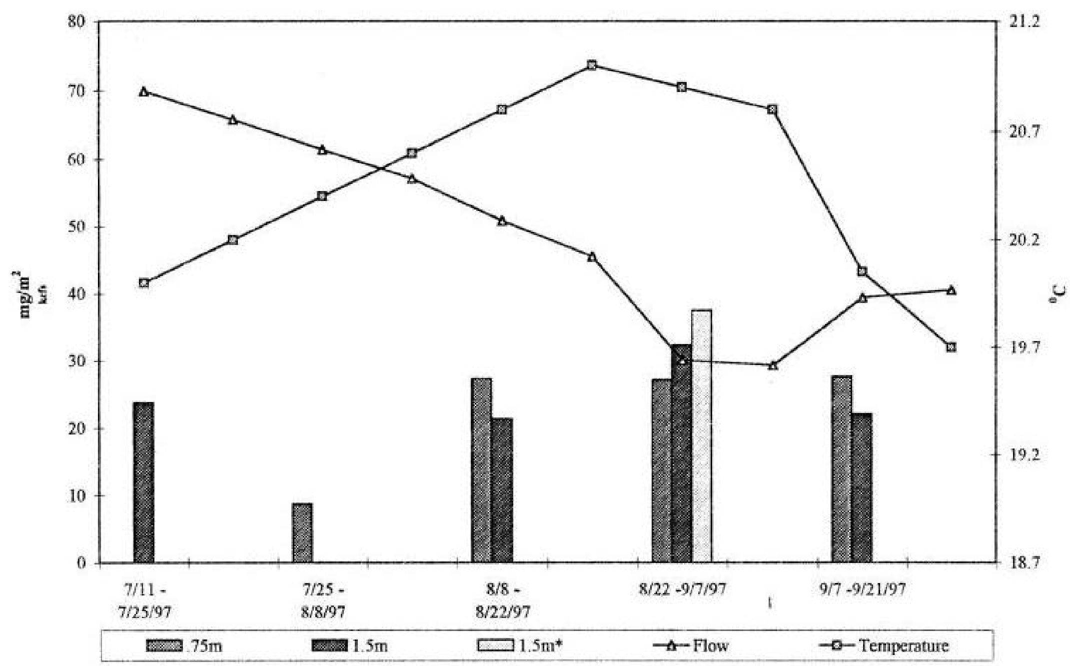

Figure 3.6-44. Attached benthic algae chlorophyll *a*, discharge, and temperature at Snake River Mile 6 (Below Ice Harbor Dam), July to September 1997 (*indicates incubation on Mylar substrate)

Dredges for rock substrate on August 22 from 2 to 8 m revealed chlorophyll *a* values of 196.6 mg/m^2 at 2.0 m dropping to 35.1 at 3.0 m. By 7 to 8 m, chlorophyll *a* values ranged from 3.0 to 5.7 mg/m^2 (Appendix K).

Biomass exhibited the same trend as chlorophyll *a*, with a high of 17.4 g/m^2 at the 1.5 m depth during the August 22 to September 7 incubation period. AI values were smallest during July 11 to 25, with 143.1 at the 1.5 m depth, increasing to a high of 342.3 at 1.5 m during August 8 to 22.

Free-flowing Sites

General Environmental and ABA Trends

Of the thirteen sites, only three were classified as truly free-flowing: the Snake River at Asotin (RM 148), the Clearwater River (RM 11), and the Columbia River at Hanford (RM 369). Although each water body is very different in flow, temperature, and light attenuation, all three sites did show similar trends in chlorophyll *a*, species composition, and biomass.

Snake River at Asotin (RM 148)

The free-flowing site of the lower Snake River was located upstream of both the confluence of the Clearwater River and the impounded region on the Snake River. River Mile 148 had high light attenuation late in the season with Secchi disk ranging from 1.2 to 2.6 m (Appendix K). Temperature increased throughout the summer with a high of 22.2° C at the beginning of September (Appendix K).

The substrate of the Asotin site was mostly sand with moderately embedded cobble. Between-tile variation was greatest at this site. It was observed that usually one quarter to one half of the tiles had very little algal growth, with a large amount of sand deposited on the tiles. The tiles with little sand deposition supported considerable algal growth. An additional problem that occurred at the site was vandalism and tile destruction during the first incubation period.

Peak chlorophyll *a* values occurred during the September 17 to October 1 incubation period with a high of 93.7 mg/m^2 at the 0.75 m depth, and 54.5 mg/m^2 and 65.5 mg/m^2 at the 1.5 m tile and mylar substrate. From October 1 to 15, 1997, monochromatic chlorophyll *a* values decreased at all depths except for the 1.5 m mylar, which had a high of 89.6 mg/m*2* (Figure 3.6-45).

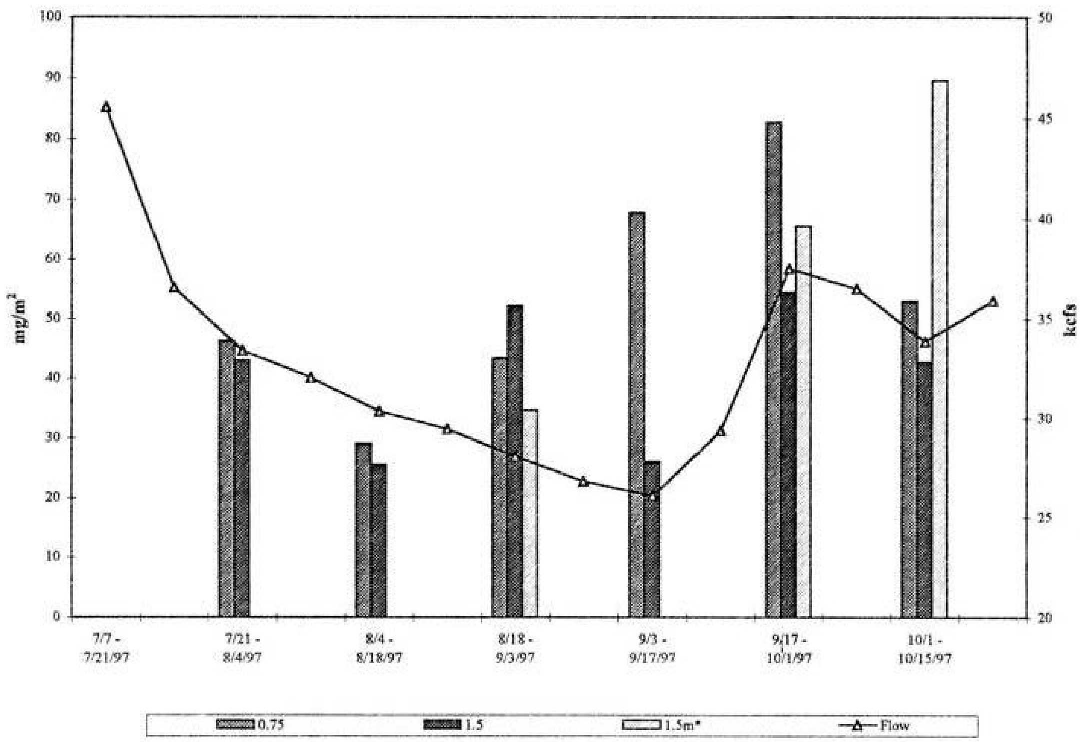

Figure 3.6-45. Attached benthic algae chlorophyll *a*, discharge, and temperature at Snake River Mile 148 (Asotin) July to September 1997

Extensive dredging of rocks at the 0.75 m and 1.5 m depths was employed on July 21 and August 4 due to the loss of tiles by vandalism. Chlorophyll *a* values on these substrates varied from 6.2 to 43.8 mg/m^2 at the 0.75 m depth and 38.4 to 72.8 mg/m^2 on July 21 (Appendix K). On August 4, dredging revealed that the majority of algal growth was concentrated in the 1.0 m and 1.5 m depth range. Mean chlorophyll values at 0.5 m were 53.4 mg/m^2 increasing to 127.1 mg/m^2 at 1.0 m, 147.9 mg/m^2 at 1.5 m and then dropping to 23.3 mg/m^2 at 3.0 m (Appendix K).

Biomass values were highest at 0.75 m throughout the season with AFODW of 4.4 to 15.2 g/m^2 (Appendix K). The AFODW at the 1.5 m depth decreased over time with the lowest AFODW values of 5.7 g/m^2 and 6.9 g/m^2 during peak monochromatic *a* levels. AI levels slowly decreased from 233. 4 to 190.6 at the 1.5 m depth (tiles). Lowest AI values occurred on the 1.5 m mylar substrates with values of 80.1 and 56.0 on the September 17 to October 1 and October 1 to October 15 incubation periods (Appendix K).

Clearwater River (RM 11)

Of all thirteen sites, the data collected at the Clearwater (RM 11) was most problematic. Fluctuating water levels due to the spilling at Dworshak Dam caused variable results. On the first run, tiles were swept downstream and lost. Throughout the summer, fluctuating water levels caused the tiles to be completely exposed or recovered from deep water. With the increased spilling, the water was considerably colder. Temperatures during high flows were 13 to 15° C, while during low flow (August 18 to September 10) the temperature was 15 to 20° C (Figure 3.6-46). Water transparency was moderate, decreasing with the increased flow. Secchi disk measurements ranged from 2.1 to 3.6 m. VEC's varied between 0.6 to 1.9 (Appendix K). Substrate at the site was mostly cobble.

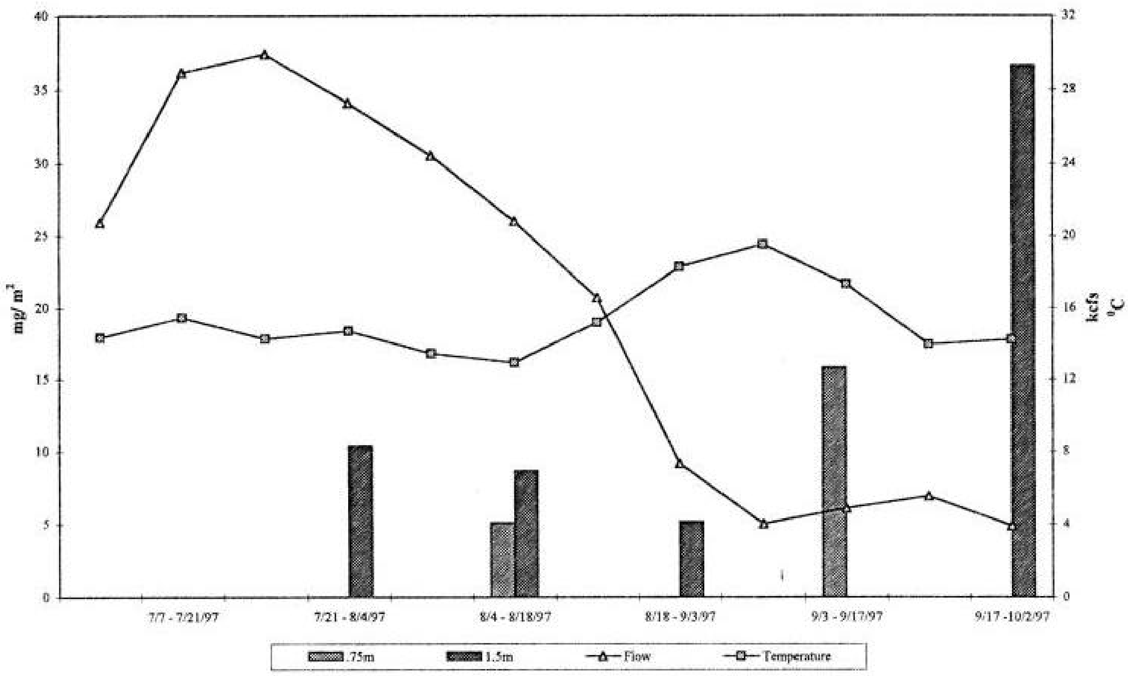

Figure 3.6-46. Attached benthic algae chlorophyll *a*, discharge, and temperature at Clearwater River Mile 11 July to September 1997 (*indicates incubation on Mylar substrate

Embeddedness varied between the margins of the stream and thalweg. Larger substrate was located in the thalweg and was highly embedded while smaller, less embedded cobble was located by the stream margins.

Chlorophyll *a* values were minimal throughout the July and August with a high of 10.4 mg/m^2 at 1.5 m on July 21 to August 4. However, by the September 17 to October 1 incubation period, monochromatic chlorophyll *a* increased to 36.63 mg/m^2 at 1.5 m depth. *Melosira* and *Cymbella* dominated the Clearwater River ABA.

Due to tile losses, natural rocks were dredged on July 21, August 4, and September 17. On July 21, chlorophyll *a* at 0.75 m ranged from 91.3 to 140.9 mg/m^2, whereas values at 1.5 m ranged from 53.0 to 164.2 mg/m^2. By September 17, chlorophyll *a* values at 1.5 m on natural substrate increased to a mean of 104.7 mg/m^2 (Appendix K).

The AFODW values were also minimal until the last incubation period from September 17 to October 1. Previously, values ranged from 1.0 g/m^2 to 3.4 g/m^2 from August 4 to September 17. However, during the bloom, AFODW increased to 12.6 g/m^2 at the 1.5 m depth.

Highest AI values occurred during the August 18 to September 3 incubation period. This was also the time of lowest flow, and tiles were retrieved in much shallower water on the September 3 than the previous incubation depth. Tiles set out at 1.5 m were recovered in 0.3 m of water, while the 0.75 m tiles were completely exposed on the riverbank. The AI values during the August 4 to 18 and September 3 to 17 incubation period remained between 196.8 and 212.0, whereas AI increased during the increased growth by late September with an AI value of 326.9 at 1.5 m (Appendix K). (Note: AI values during the August 18 to September 3 incubation period were elevated with values of 636.1. However, during this incubation tiles which had been incubated at 1.5 m were recovered at 0.3 m.)

Columbia River at Hanford (RM 369)

Fluctuating water levels were also a problem at the third free-flowing site, Columbia RM 369 at Hanford, were on a daily basis rather than a weekly basis as with the Clearwater River site. Shallow set tiles (0.75 m depth) were often discarded due to the fluctuating water levels. Substrate was a mixture of sand and cobble.

Temperature on July 25 was 17.8° C, which increased to 19.4° C by August 22 before declining to 14.9° C by October 19 at 1.5 m. Water transparency was the greatest of all thirteen sites. Secchi disk transparency was 1.5 m on July 11, increasing to a maximum of 5.5 m on September 7. After September 7, Secchi disk gradually declined with low values of 4.5 m and 4.6 m on October 4 and October 19.

Chlorophyll *a* values were lowest during July and August (Figure 3.6-47). Tiles were retrieved in 1.0 m of water and it is possible that the tiles were exposed at some time during the incubation period. Original chlorophyll *a* values during the first incubation period were 14.8 and 15.6 mg/m^2 at the 0.75 m and 1.5 m depths. However, by the August 22 to September 7 incubation period, monochromatic chlorophyll *a* values increased to 35.0 and 55.2 mg/m^2 at the 1.5 m depth on tile and mylar substrate. Values fluctuated during the next two incubation periods, but still remained relatively high (34.6 to 47. 4 mg/m^2), ending with a high of 67.8 mg/m^2 at the 1.5 m mylar on October 5 to 19 incubation period. Natural rocks dredged from 0.5 to 4.0 m on August 8, 1997 resulted in fluctuating chlorophyll *a* values from 0.5 to 3.1 m ranging from 6.50 to 44.9 mg/m^2. However, by 4.0 m in depth chlorophyll *a* dropped to 3.92 mg/m^2 (Appendix K).

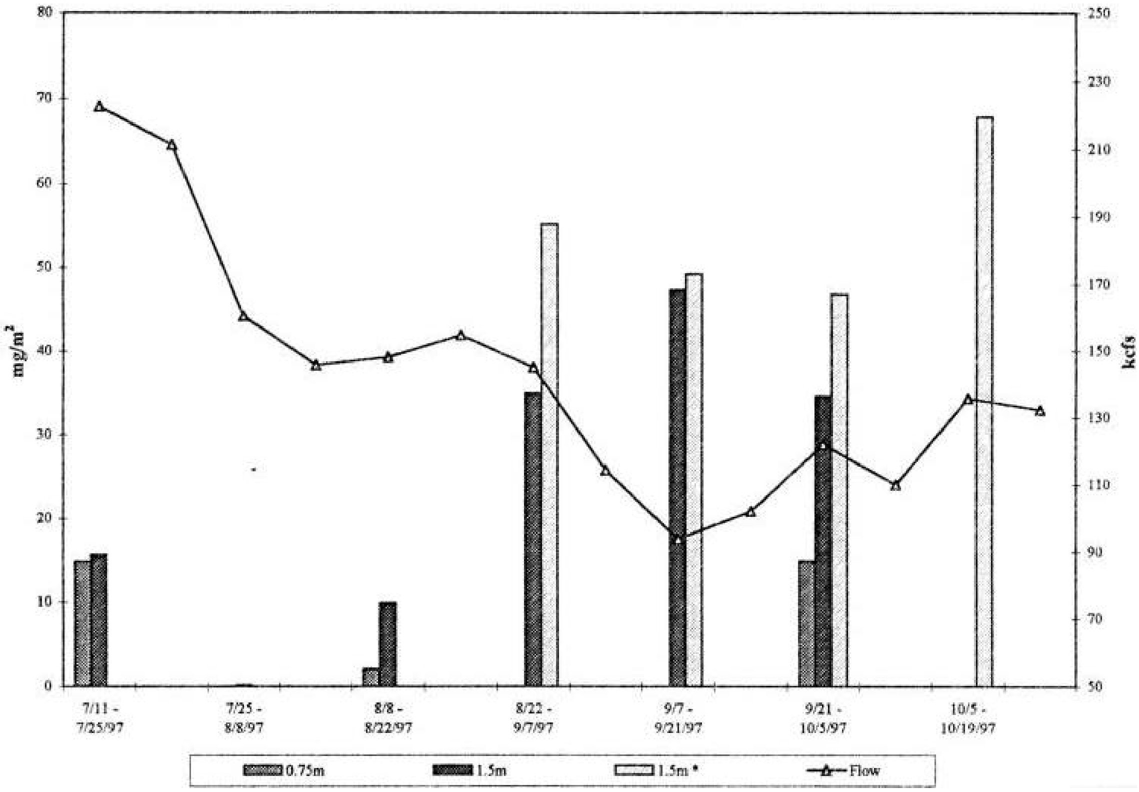

Figure 3.6-47. Attached benthic algae chlorophyll *a*, and discharge at Columbia River Mile 369 (Hanford) July to October 1997 (*indicates incubation on Mylar substrate)

The AFODW values peaked during the August 22 to September 7 incubation period, with a high of 14.1 and 8.2 g/m^2 at the 1.5 m depth on tile and mylar substrate.

The largest AI value at the 1.5 m depth of 2,432.4 occurred during the July 25 to August 8 incubation period in which 1.5 m tile retrieval was in 1.0 m of water. Not including the above incubation period, AI values decreased at the 1.5 m depth from 265.2 during July 11 to 25 to a low of 76.7 by October 5 to 19 (Appendix K).

Primary Productivity

Primary productivity estimates were obtained at two Snake River sites (RM 148 at Asotin and RM 118 in Lower Granite Reservoir) and two Columbia River sites (RM 369 at Hanford and RM 410 in Priest Rapids Reservoir). Data precision was low when using tile substrates. Tile substrates contained a high amount of sediment that was incubated in the chamber along with the attached benthic algae. Therefore, following this first productivity assay, free-floating mylar substrates were used for the primary productivity estimates for the remainder of the field season.

The lower Snake River at Asotin (RM 148) and the Columbia River at Hanford (RM 369) were most productive late in the season during the October primary productivity analysis. Chlorophyll *a* values during the October incubation periods at the free-flowing Asotin and Hanford sites were higher than the impounded sites on the Snake and Columbia Rivers.

Primary productivity estimates on October 1 and 15, remained high at the Snake RM 148 (Asotin) site averaging 110 and 89 mg/C/m^2/hr, respectively, during the mid-morning and late afternoon incubation periods. Snake RM 118 (Lower Granite Reservoir) was lower with mid-morning estimates of 27.4 and 21.8 mg/C/m^2/hr on October 2 and 16. Highest rates of photosynthesis at RM 118 were obtained earlier in the season on September 18 (Table 3.6-10).

Table 3.6-10
Primary Productivity Estimates (mg/C/m^2/hr) For Attached Benthic Algae Incubated at 1.5 M On Tile and Mylar Substrate for 14 Days at the Lower Snake River Study Sites. Substrate Incubation Sites Were Located in the Free-Flowing Reach at RM 148 (Asotin) And the Impounded Reach at RM 118 (Lower Granite Reservoir)

	RM 148 (Asotin)			RM 118 (Lower Granite Reservoir)				
Date	09/17/97	10/01/97	10/15/97	08/05/97	08/05/97	09/18/97	10/02/97	10/16/97
Substrate Incubation Period	09/03 to 09/17/97	09/17 to 10/01/97	10/01 to 10/15/97	07/22 to 08/05/97	07/22 to 08/05/97	09/04 to 09/18/97	09/18 to 10/02/97	10/02 to 10/16/97
Substrate	Mylar	Mylar	Mylar	Tiles	Buoy²	Mylar	Mylar	Mylar
Area of Substrate	.0225 m	.0225 m	.0225 m	.0218 m	.0238 m	.0225 m	.0225 m	.0225 m
Mono chlorophyll *a* (mg/m^2)	****	70.99	89.44	7.78	17.88	27.49	38.34	21.72
1100 to 1400	****	110.64	109.47	7.72	55.11	80.42	27.41	21.78
mg C/mg chl *a*	****	1.56	1.22	0.99	3.08	2.93	0.71	1.00
1400 to 1700	****¹	88.92	90.01	2.36	54.48	36.67	5.56	24.76
mg C/mg chl *a*	****¹	1.25	1.01	0.30	3.50	1.33	0.15	1.14

¹Mylar substrate lost.
²Due to the large sediment accumulation on the bottom-set tiles, free-floating buoys which had been incubated for 14 days at 1.0 m, were used in comparison to tile substrates. Tiles and buoys were incubated in side-by-side primary productivity chambers at the 1.5 m depth during the above timeframes. Following this analysis, free-floating 0.150 m x 0.150 m x 0.001 m mylar substrates were used for primary productivity estimates for the remainder of the field season.

Primary productivity estimates in the Columbia River followed chlorophyll *a* trends with the same trend of higher rates of photosynthesis in the free-flowing reach compared to the impounded reach. Primary productivity estimates on mylar substrate at Columbia RM 369 (Hanford) ranged from 102.7 mg/C/m^2/hr on September 21 decreasing to 94.3 mg/C/m^2/hr on October 19. In the impounded reach, Columbia RM 410 (Priest Rapids Reservoir), primary productivity estimates were lower, with a high of 48.2 mg/C/m^2/hr obtained on October 18 (Table 3.6-11). Comparison of the ^{14}C and dissolved oxygen techniques with Washington State University and the University of Idaho resulted in similar productivity rates for the attached benthic algal communities at Columbia RM 369 (Hanford). Mid-day productivity rates for dissolved oxygen were 94.3 mg/C/m^2/hr and 92.6 mg/C/m^2/hr for ^{14}C (Table 3.6-11).

Table 3.6-11
Primary Productivity Estimates ($mg/C/m^2/hr$) For Attached Benthic Algae Incubated at 1.5 M On Tile and Mylar Substrate for 14 Days at the Columbia River Study Sites. Substrate Incubation Sites Were Located in the Free-Flowing Reach at RM 369 (Hanford) And the Impounded Reach at RM 410 (Priest Rapids Reservoir)

	RM 369 (Hanford)			RM 410 (Priest Rapids Reservoir)		
Date	08/08/97	09/21/97	10/19/97	08/07/97	09/20/97	10/18/97
Substrate	Tiles	Mylar	Mylar	Tiles	Mylar	Mylar
Area of Substrate	.0218 m	.0225 m	.0225 m	.0218 m	.0225 m	.0218 m
Substrate Incubation Period	07/25 to 08/05/97	09/07 to 09/21/97	10/05 to 10/19/97	07/27 to 08/07/97	09/06 to 09/20/97	10/04 to 10/18/97
Mono chlorophyll *a* (mg/m^2)	0.19	49.19	67.79	9.59	11.00	36.30
1100 to 1400	1.50	102.67	94.32[1,3]	67.79	30.00	48.22[2]
mg C/mg chl *a*	7.89	2.09	1.39	7.07	2.73	1.33
1400 to 1700	2.89	58.20			18.90	
mg C/mg chl *a*	15.21	1.18			1.72	

[1]Incubation time: 1115-1145.
[2]Incubation time: 1230-1600.
[3]^{14}C ABA primary productivity estimates obtained by WSU using 1 square inch of mylar substrate during this incubation period was equal to 92.57 $mg/C/m^2/hr$.

Natural, Tile, and Mylar Substrate Comparisons

Ceramic tiles and freshly scrubbed natural rock substrate were incubated side by side at 1.5 m for 14 days at seven sites during the first incubation period in July. Retrieval of the large rock substrate was difficult, and our sample size ranged from 1 to 6 at each of the sites. At six of the seven sites, chlorophyll *a* and AFODW values on the natural rock substrate were higher (Table 3.6-12). Chlorophyll *a* on natural substrates averaged 1.8 times higher than on tile substrates; AFODW on natural substrates also averaged 1.8 times higher than on tile substrates. The difference in the amount of chlorophyll varied widely between the sites. At Lower Granite Reservoir, artificial substrate chlorophyll *a* and AFODW were 1.8 mg/m^2 and 1.0 g/m^2, respectively, compared to the natural substrate values of 16.1 mg/m^2 and 16.5 g/m^2. Below Lower Granite Dam at RM 105, however, chlorophyll *a* on tile substrate was 11.3 mg/m^2 compared to 19.2 mg/m^2 on natural substrate.

Table 3.6-12 Attached Benthic Algae Chlorophyll *a*, Biomass (AFODW), and Autotrophic Index For Tile and Natural Substrate Incubated at 1.5 m for a 14-Day Interval in July 1997 At the Lower Snake and Columbia River Study Sites						
Site	**1.5 M Tile Substrate**			**1.5 M Natural Substrate**		
	Mono Chl A	**AFODW**	**AI**	**Mono Chl A**	**AFODW**	**AI**
Snake RM 118 (GRA-A)	1.78	0.97	391.60	16.09	16.53	618.56
Snake RM 105 (GRA-B)	11.31	1.25	109.44	19.15	5.31	277.36
Snake RM 81 (GOO-A)	14.66	15.90	783.02	44.50	30.75	529.29
Snake RM 18 (ICE-A)	15.72	4.47	123.72	5.09	1.56	268.96
Snake RM 6 (ICE-B)	23.81	3.75	143.11	38.17	7.69	183.15
Columbia RM 410 (PRI-A)	15.71	6.81	348.71	26.78	14.25	358.94
Columbia RM 369 (HAN-A)	15.63	3.15	265.21	28.62	8.03	243.81
Columbia RM 326 (MNA-A)	--	48.67	--	--	67.42	--

Comparing tile and mylar substrate, chlorophyll *a* values in the impounded regions tended to be higher on the tile substrate and lower on the mylar substrate. However, in the free-flowing and the hybrid free-flowing/controlled reaches, chlorophyll *a* values were higher on mylar and lower on the tile substrate. At all sites, however, AI values on the mylar were smaller than the AI on tile substrate. This was due to the decreased amount of sedimentation and deposition of organic material.

Substrate Analysis

We sampled substrates on the lower Snake and Clearwater Rivers during an in-depth substrate analysis on February 20 to 22, 1998. The purpose was to develop a surface area coefficient to adjust planar surface area to actual surface area of substrate pieces of varying shapes. A general trend emerged of increased surface area for algal growth at the free-flowing and hybrid free-flowing reaches compared to the reservoir sites. The reservoirs were dominated by fines and sand along with cobbles, pebbles, and gravel (Table 3.6-13). Lower Granite (RM 118) and Little Goose (RM 81) Reservoirs had less cobble and boulders substrate than the free flowing and hybrid free flowing reaches. In addition, the amount of exposed hard surface area was reduced by the amount of cobble embeddedness. This was most evident by the comparison of the 0.75 m depth at Snake RM 52 and Snake RM 37. Both sites had similar substrate composition: 75% cobble, 13% boulder, and 13% sand and silt at Snake RM 52 and 81% cobble, 13% boulder, and 6% sand and gravel at RM 37. However, due to cobble embeddedness, the surface area coefficient was smaller in the reservoir at 1.16 as compared to 2.12 below the dam.

Table 3.6-13
Substrate Composition and Surface Area Coefficients for Algal Growth At the Lower Snake and Clearwater Rivers Attached Benthic Algae Incubation Sites

	Depth Zone		
	0.25 m to 1.0 m	**1.0 m to 2.0 m**	**2.0 m to 3.5 m**
Snake River Mile 118 (Lower Granite Reservoir)			
Substrate Composition	58% silt/sand 42% pebble	72% silt/sand 19% boulder 9% cobble	80% silt/sand 10% boulder 5% sand/gravel 5% cobble
Surface Area Coefficient	1.16	1.14	1.26
Snake River Mile 105 (Below Lower Granite Dam)			
Substrate Composition	66% cobble 56% pebble 11% gravel	69% cobble 26% pebble 5% gravel	50% silt/sand 40% boulder 10% cobble
Surface Area Coefficient	2.12	2.53	2.08
Snake River Mile 81 (Little Goose Reservoir)			
Substrate Composition	60% cobble 30% silt/sand 10% boulder	100% muds/fines	67% silt/sand 17% sand/gravel 17% boulder
Surface Area Coefficient	1.51	1.00	1.14

Snake River Mile 67 (Below Little Goose Dam)			
Substrate Composition	69% cobble 13% boulder 1% gravel/sand	65% cobble 10% boulder 10% silt/sand 5% gravel/sand	100% silt/sand
Surface Area Coefficient	1.87	1.8	1.00
Snake River Mile 52 (Lower Monumental Reservoir)			
Substrate Composition	75% cobble 13% boulder 5% sand/silt	35% cobble 36% sand/silt 7% gravel/sand 14% pebble 7% boulder	80% silt/sand 20% boulder
Surface Area Coefficient	1.31	1.57	1.2
Snake River Mile 37 (Below Lower Monumental Dam)			
Substrate Composition	81% cobble 6% sand/gravel 13% boulder	72% cobble 20% pebble 10% sand/gravel	56% cobble 25% silt/sand 13% boulder 13% sand/gravel
Surface Area Coefficient	2.02	1.48	1.97
Snake River Mile 18 (Ice Harbor Reservoir)			
Substrate Composition	*	*	*
Surface Area Coefficient	*	*	*
Snake River Mile 6 (Below Ice Harbor Dam)			
Substrate Composition	33% cobble 33% silt/sand 25% boulder	50% cobble 30% boulder 20% sand	38% sand 38% boulder 25% cobble
Surface Area Coefficient	2.04	1.5	1.65
Snake River Mile 148 (Asotin)			
Substrate Composition	38% boulder 31% cobble 23% sand 7% pebble	100% sand/gravel	20% sand/gravel 80% boulder
Surface Area Coefficient	2.4	1.00	2.18
Clearwater River Mile 12 (Spaulding)			
Substrate Composition	73% cobble 18% boulder 9% sand	78% cobble 22% boulder	83% cobble 17% boulder
Surface Area Coefficient	2.57	1.39	2.03
*No data available			

Highest surface area coefficients were found at the 0.75 m depth and decreased at the 1.5 m and 3.0 m depths at all sites, (with the exception of RM 105 which had higher ratios at the 1.5 m depth). Of the sites, the highest ratio was found at the Clearwater River site at 2.57 at 0.75 m and the lowest of 1.00 at RM 148 (Asotin) at the 1.5 m depth and RM 67 at 3.0 m.

Multiplying ratios by chlorophyll *a* and AFODW values will result in corrected primary productivity values for the reach (*i.e.*, true values per unit area of planar area). The mean corrected primary productivity values are summarized below in Table 3.6-14.

Table 3.6-14
Surface Area Corrected Mean Chlorophyll *A* and Biomass (AFODW) Values For the Snake and Clearwater Rivers

Site	Chl *a* (mg/m^2)*		AFODW (mg/m^2)*	
	0.75 M	1.5 M	0.75 M	1.5 M
Snake RM 148 (ASO-A)	133.27	40.71	24.02	8.26
Clearwater RM 11 (SPA-A)	39.09	18.06	5.65	8.07
Snake RM 118 (GRA-A)	41.84	24.33	19.99	29.96
Snake RM 105 (GRA-B)	84.61	75.09	40.58	26.49
Snake RM 81 (GOO-A)	57.53	25.17	55.85	9.08
Snake RM 67 (GOO-B)	66.12	43.66	20.95	17.57
Snake RM 52 (MON-A)	67.37	38.00	16.40	7.88
Snake RM 37 (MON-B)	89.74	59.95	15.34	12.23
Snake RM 6 (ICE-B)	46.31	37.38	17.87	8.01

*Values obtained from tile substrate incubated for 14 days.

Summary

The role of attached benthic algae in the primary productivity of large rivers has been addressed by a number of studies (Cushing, 1967; Luttenton *et al.*, 1986). It has been shown that large rivers can support a substantial benthic algal production (Wetzel, 1983). However, algal type and production depends on such factors as light, water chemistry, nutrient loading, substrate type, and water movements. Thus, the construction of dams on large rivers will be a major controller of the attached benthic algal community.

The attached benthic algae of the lower Snake River exhibited a seasonal trend of highest growth rates in September with decreased flow, increased water transparency, and increased water temperature. September chlorophyll *a* values ranged from 103.3 mg/m^2 at 0.75 m in Ice Harbor Reservoir to a low of 2.3 mg/m^2 at 3.0 m in Little Goose Reservoir. Due to low light attenuation in the Snake River (average Secchi disk measurement of 1.4 m), the greatest benthic algal growth occurred at 0.75 m provided that there was minimal wave action at the site, and the tiles were not exposed due to fluctuating water levels. In previous studies it has been shown that wave action can have considerable effects on algal colonization (Austin *et al*., 1981). Fluctuating water levels reduced ABA growth due to desiccation when exposed to the air or the reduced water depth made the tiles more prone to wave action. Wave action can limit algal growth by disrupting the tile substrate and depositing sand, gravel and other sediment on the substrate.

Assessment of attached algae on natural rock collected from deeper depths revealed that algal growth declined considerably between 3 to 4 m depth. There was increased water transparency and increased light attenuation in the Columbia River at RM 410 and RM 369, permitting increased algal growth at deeper depths.

In the lower Snake River impoundments, chlorophyll *a* values increased downstream from Lower Granite to Ice Harbor reservoir. The AI values decreased downstream from Lower Granite to Ice Harbor, with the decreased sedimentation and increased chlorophyll *a*. The AI values, however, remained high at all the impounded sites at the 3.0 m depth, where there was high organic matter, but algal growth was low at low light conditions, as determined by chlorophyll analysis.

In comparing the free-flowing versus the impounded reaches, the free-flowing sites supported a more firmly attached benthic algal community than the flocculent algae/sediment matrix found in the impoundments. The ABA growth at the free-flowing sites on the Snake and the Columbia Rivers was greater than at the impounded reaches from mid-August through October. The true free-flowing upstream reaches tended to have higher water transparency whereas downstream impounded reaches had lower transparency as a result of high phytoplankton levels. Earlier in the season, upstream water transparency was low due to high abiotic turbidity during the high flows at the free-flowing sites.

Substrate composition was a major difference between the two habitat types. Free-flowing substrate composition was mainly cobble and sand. Algal growth was high on the cobble substrate, but low on the sandy substrate. This was most evident at Snake RM 148 at Asotin with the high with-in site variability observed between the tiles. This concurs with the findings of Rier and King (1996) who found low periphyton biomass on abrasive substrates such as shifting sand. In the impounded reaches, however, fines, silts, sand, and entirely embedded cobble dominated the substrate composition. The benthic algal community grew on the fines and silt forming an algal matrix within the top layer of the sediment. Although algae growth was high, the fines and silts were easily disturbed and therefore provided for a less stable substrate that is less suitable for ABA growth as compared to the cobble substrate.

In comparison, the hybrid/free-flowing reaches tended to have more similar ABA chlorophyll *a* than the upstream impounded sites from July to mid-August. By September, ABA chlorophyll *a* values were higher above the dam in the impounded reach. The increased algal growth in the impounded reach versus below the dam was most evident at Ice Harbor. A major factor to the lesser algal growth would be the decreased water transparency below the dam in comparison with the upstream reservoir sites. The AI values below the dam were much smaller than the impounded reaches throughout the study. Each dam acts as a sediment trap, reducing the sediment load and organic debris that is carried downstream below each dam. Sedimentation on the tiles tended to inflate the AI.

Diel measurements revealed little difference between Snake RM 148 and Snake 118 in changes in dissolved oxygen and temperature patterns. However, primary productivity estimates using the dissolved oxygen technique revealed that there was increased ABA productivity at the free-flowing versus impounded reaches. The increased productivity estimates were due to the increased algal biomass at the site. In relating the amount of carbon fixed to the amount of chlorophyll *a* on the substrate (mg C/mg Chl *a*), differences in primary production rates were minimal. The difference between the dissolved oxygen and ^{14}C techniques to estimate productivity were negligible. Although the ^{14}C method is theoretically more sensitive, other comparisons of the two techniques have shown little difference except when algae is exposed to high light intensity (McAllister, 1961 in Vollenweider, 1974).

In comparing the amount of carbon fixed versus the chlorophyll a on the mylar substrate (mg C/mg Chl *a*), there was a decrease in primary productivity rates (*i.e.*, efficiency) in late October at all sites. Rates on mylar substrates ranged from 2.1 to 2.9 mg C/mg Chl *a* in mid-September. By mid-October, rates dropped to 1.0 to 1.3 mg C/mg Chl *a* at the four sites. These rates were similar between sites and did not exhibit a trend between free-flowing versus impounded sites.

Differences were observed in algae accrual in between incubation substrate types. Natural substrate supported the highest algal growth. It has been shown that artificial substrates and natural substrates vary in both algal composition and biomass (Brown and Austin, 1973; Robinson, 1983; Wetzel, 1983; Falter and Olson, 1991; and Kann and Falter, 1989). Chlorophyll *a* values on natural substrate averaged 1.8 times higher than on the tile substrate. However, the variance of the natural rock substrate ABA growth was high and therefore contributed to high within-site variability.

Algal growth on mylar and tile substrates differed. The use of the mylar film was beneficial in determining the algal growth at the site without the interference of excess sedimentation. The AFODW and the AI were lower on the mylar substrate than the tile substrate. These results indicate the importance of using a variety of substrates to adequately sample and depict the attached benthic algal communities.

Conclusions

1. The ABA seasonal growth was highest in August and September coinciding with decreased flows and increased temperature.

2. Through the four impounded reaches on the Snake River, algal growth increased downstream.

3. Algal growth, as determined by both chlorophyll *a* and AFODW below Little Goose, Lower Monumental, and Ice Harbor Reservoirs was less than the algal growth above the reservoir at each site.

4. The free-flowing reaches at Snake RM 148 and Columbia RM 369 had higher water transparency, higher algal growth, and greater primary productivity rates from mid-August to October compared to the impounded reaches.

5. Algal growth was highest at 0.75 m depth except when affected by dropping water levels and wave action. Most algal growth occurred in the shallower depths between 0.75 m and 1.5 m, with decreased growth at 3.0 m.

6. Natural substrates had 1.8 times the chlorophyll *a* compared to tile substrates. Mylar substrates had decreased sedimentation and decreased AI values compared to tile substrates.

7. When comparing surface corrected chlorophyll accumulations, Snake RM 148 (free-flowing) was the most productive site per unit planar area. In all cases, the 0.75 m depth was 1.2 to 3.4 times as productive as the 1.5 m depth. The least productive site was Clearwater RM 11 (free-flowing) closely followed by Snake RM 118.

3.6.6 - Diel Dissolved Oxygen and Temperature

Dissolved oxygen and temperature measurements were taken during a twelve-hour period and twenty-four hour period at Asotin (RM 148) on September 17 and 18 and October 1 and 2, and at Lower Granite Reservoir (RM 118) on September 18 and 19, and October 2 and 3. The diel runs were measured at the peak attached benthic algae chlorophyll *a* values at both sites. Overall, there was very little change in temperature at Lower Granite and a slight decrease in temperature during the night at Asotin (Table 3.6-15a). In October, Asotin had a higher daytime dissolved oxygen peak of 10.8 mg/l compared to Lower Granite's 8.9 mg/l. Lower Granite, however, had a lower night minimum of 6.9 mg/l (Tables 3.6-16 a and b) compared to 7.6 mg/l at Asotin (Table 3.6-15b).

Table 3.6-15a
Dissolved Oxygen and Temperature Measurements at Snake RM 148 (Asotin)
September 17 and 18, 1997

Time	Downstream Dissolved Oxygen (mg/l)	Downstream Temperature (°C)	Upstream Dissolved Oxygen (mg/l)	Upstream Temperature (°C)
2015	8.00	19.8	8.11	19.9
2200	7.85	19.8	8.31	19.8
0025	7.58	19.7	8.26	19.8
0240	7.60	19.7	8.27	19.7
0430	8.05	19.6	8.26	19.7
0635	8.00	19.7	8.28	19.6
0830	8.05	19.5	8.35	19.5

Upstream location at ~RM 152

Table 3.6-15b Dissolved Oxygen and Temperature Measurements at Snake RM 148 (Asotin) October 1 and 2, 1997		
Time	**Dissolved Oxygen (mg/l)**	**Temperature (°C)**
1130	8.20	19.1
1330	8.30	19.0
1530	10.80	19.0
1730	10.80	19.0
1930	8.20	19.0
2130	7.80	19.0
2330	7.80	18.8
0130	7.60	18.7
0330	7.70	18.7
0530	8.00	18.7
0730	8.10	18.8

Table 3.6-16a Dissolved Oxygen and Temperature Measurements at Snake RM 118 (Lower Granite Reservoir), September 18 and 19, 1997		
Time	**Dissolved Oxygen (mg/l)**	**Temperature (°C)**
2030	8.00	20.0
2230	7.60	20.0
0030	7.00	20.0
0230	7.20	20.0
0430	6.80	20.0
0630	6.80	20.0
0830	7.10	20.0
1030	7.00	20.0

Table 3.6-16b Dissolved Oxygen and Temperature Measurements at Snake RM 118 (Lower Granite Reservoir), October 2 and 3, 1997		
Time	**Dissolved Oxygen (mg/l)**	**Temperature (°C)**
1000	8.40	18.1
1200	8.70	18.1
1400	8.85	18.1
1600	8.60	18.0
1800	8.23	17.9
2000	7.50	17.9
2200	7.35	17.9
2400	7.19	17.8
0200	7.00	17.9
0400	6.96	17.9
0600	6.92	18.0
0800	7.10	17.9

3.6.7 - Hydrology

The volume of discharge in the regions rivers was, in many instances, greater during 1997 than it had been in several years. Table 3.6-17 presents a summary of comparisons between long-term average monthly and mean daily discharges, with provisional USGS data from gauging stations on the Snake River, Clearwater River, and Columbia River. The largest discharge values, highlighted by the bold characters, were observed throughout 1997; including the June through October field season.

Table 3.6-17
Comparison of Long-Term Mean Monthly and Maximum Daily Discharges
With 1997 Provisional Data from the USGS Gauging Station
At Anatone, Washington, Spaulding, Idaho, and Priest Rapids Dam, Washington
(All discharge data given in kcfs)

Period	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct
Anatone										
Mean Monthly										
1958 to 1997	28.8	33.1	39.0	48.2	65.7	71.7	30.7	17.8	19.2	21.5
1997	72.3	72.1	77.6	85.0	109.6	117.8	43.7	29.1	31.7	29.3
Maximum Daily										
1958 to 1996	48.2	72.5	90.4	88.7	118.7	134.2	63.9	26.5	27.0	31.5
1997	173.0	85.0	96.2	119.0	152.0	141.0	78.2	33.1	38.1	40.0
Spaulding										
Mean Monthly										
1972 to 1996	10.3	12.0	17.0	21.7	35.5	32.3	12.9	6.2	7.7	4.3
1997	19.2	23.6	32.7	41.2	62.0	46.9	27.7	19.0	4.5	5.2
Maximum Daily										
1972 to 1996	27.3	38.9	35.8	40.7	63.2	73.6	23.9	16.3	11.5	8.3
1997	38.3	35.0	52.1	72.6	81.4	67.3	34.8	26.9	9.3	16.5
Priest Rapids										
Mean Monthly										
1960 to 1996	104.5	108.9	108.7	116.0	162.9	209.8	159.8	110.2	79.6	80.2
1997	150.6	153.6	142.4	173.8	271.7	323.4	197.0	147.3	108.8	118.3
Maximum Daily										
1960 to 1996	168.4	195.0	201.8	189.1	269.8	461.4	294.3	191.0	126.7	118.2
1997	191.0	178.0	211.0	277.0	317.0	410.0	242.0	182.0	139.0	163.0

The pattern of spring runoff, and consequently the relation between the hydrograph and the sampling regime, did differ slightly in each river system (Figure 3.6-48). In the Snake River and Clearwater River, the maximum mean daily discharges of 512 kcfs and 67.3, respectively, occurred in the middle of May. In the Columbia River, however, the peak of the freshet, 410 kcfs below Priest Rapids Dam and 577 kcfs at McNary Dam, occurred between 12 and 13 June. The contrast in timing between the Columbia River and the Lower Snake River system is probably the result of the larger size of the Columbia River watershed and differences in upstream systems operations. In all three rivers, minimum flows during the study period were observed during the first half of September, with discharges of 3.5, 25.4, and 99.9 kcfs at Spaulding, Anatone, and McNary Dam, respectively.

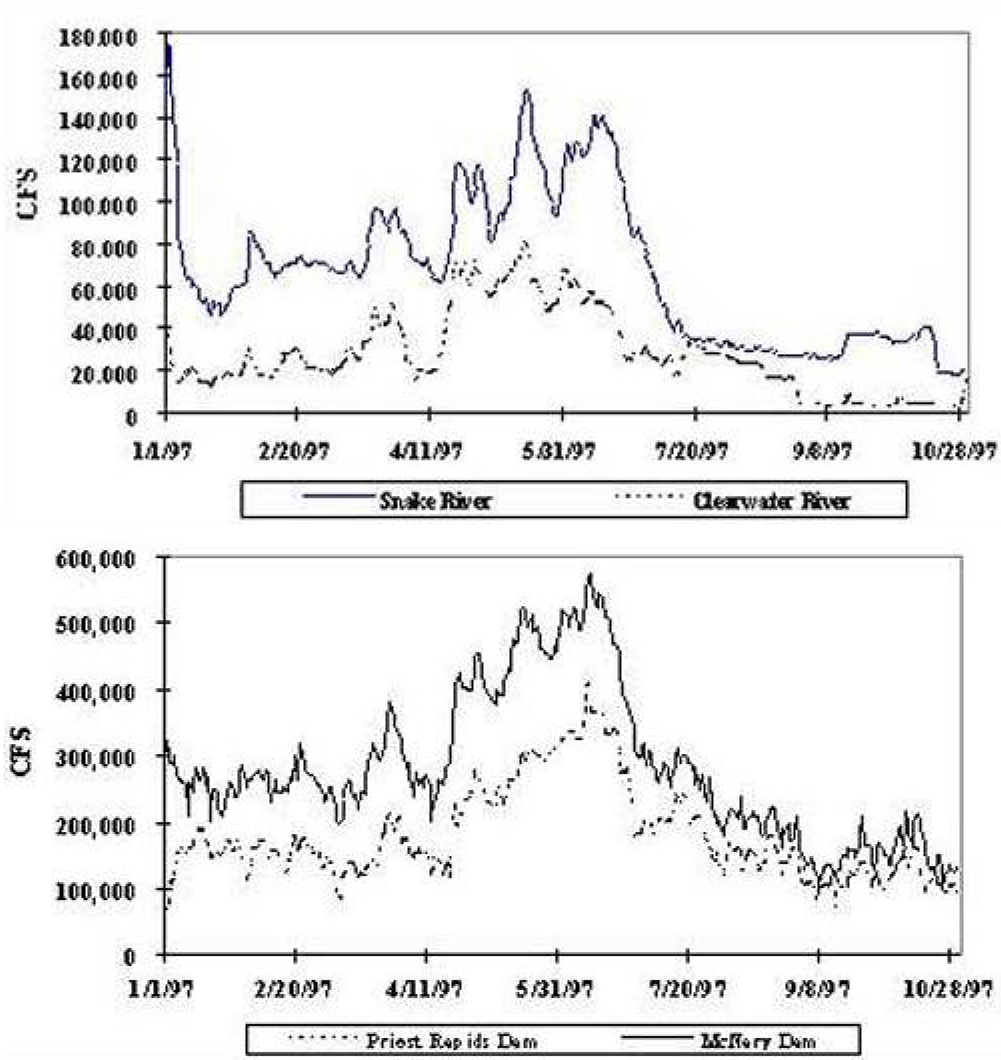

Figure 3.6-48. Hydrographs for the Clearwater River at Spaulding, the Snake River at Anatone, and the Columbia River at Priest Rapids Dam and McNary Dam

In addition to inter-annual variations in the hydrographs of the Lower Snake River that result from changes in snowpack, climate, and other "natural" phenomenon, the system operations of Dworshak Reservoir, Idaho, have also had a local effect. Beginning in 1994, large volumes of water have been released from this water body either for flow augmentation in the Lower Snake River or, as in 1997, to facilitate repairs to the dam. From about mid July to mid August of this year, water releases from the reservoir into the Clearwater River ranged from 20 kcfs to 22.5 kcfs, and were then tapered off to slightly less than 2 kcfs by the end of August. The result of this was a marked increase in the volume of water in the river, to the point where up to 88% of the flow in the Clearwater River during August was attributable to water released from Dworshak

(Figure 3.6-49). The effect on the Lower Snake River was, of course, less, yet during the latter part of July and the first half of August upwards of 52% of the water flowing through Lower Granite Dam originated from the Clearwater River. This difference in the origin of the water may initially appear inconsequential, but due to the dissimilar physical and chemical properties of the Snake and Clearwater Rivers, the end result is apparent in the Lower Snake River, especially in the upper reaches of the system.

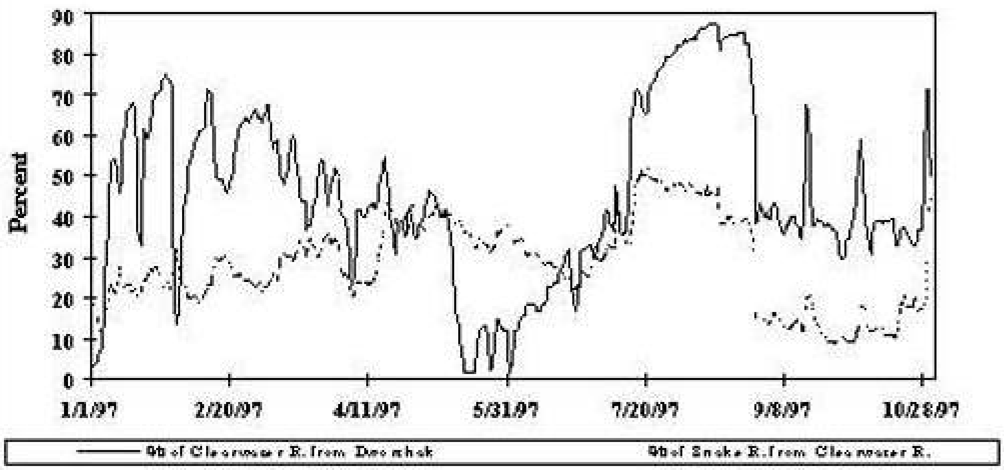

Figure 3.6-49. Percent of the discharge in the Clearwater River that was due to releases from Dworshak Reservoir, and of the discharge in the lower Snake River attributable to the Clearwater River.

3.6.8 - Meteorological Data

Weather stations were installed at the following locations as part of the project: the USACE Clarkston Resource Center (Clarkston), the flume at the fish bypass facility below Little Goose Dam, and on the northern control tower at Ice Harbor Dam. Each weather station was equipped with a Young Model 05103-12 Wind Monitor, a Li-Cor Model LI-190SZ Quantum Sensor, a Texas Electronics Model TRP-525 Rainfall Sensor, and a Young Model 41375VF Relative Humidity/Temperature Probe with multi-plate radiation shield. All of the instruments at a given site were connected to a ZENO-3200 Datalogger that was housed in a waterproof NEMA enclosure.

Data was collected at thirty minute intervals from 11 July to 31 October, 25 July to 31 October, and 13 July to 31 October at Clarkston, Little Goose Dam, and Ice Harbor Dam, respectively. Table 3.6-18 summarizes maximum, minimum, and average statistics for wind speed, wind direction, temperature, precipitation, and solar radiance for each station.

Table 3.6-18
Maximum Daily Average, Minimum Daily Average, and Monthly Average For Wind Speed, Wind Direction, Temperature, Precipitation, and Solar Radiation For Weather Stations Located at Clarkston, Ice Harbor Dam, and Little Goose Dam

Parameter	Month	Clarkston			Ice Harbor Dam			Little Goose Dam		
		Max	Min	Mean	Max	Min	Mean	Max	Min	Mean
Wind Speed (m/s)	July*	2.1	0.9	1.3	2.6	1.2	1.8	4.9	1.1	2.5
	Aug.	3.4	0.7	1.6	4.4	1.2	2.0	5.8	0.8	2.4
	Sept.	4.0	0.6	1.2	7.4	1.1	2.6	8.4	0.8	3.0
	Oct.	2.4	0.5	1.3	7.0	0.8	2.9	8.2	0.9	3.5
Wind Direction (deg)	July*	183.6	82.9	128.1	182.6	138.0	166.3	240.0	82.9	196.1
	Aug.	269.3	88.0	149.3	232.8	127.8	176.7	250.4	107.1	186.1
	Sept.	271.3	94.6	149.9	215.1	111.6	177.6	241.5	129.2	190.9
	Oct.	240.1	95.1	166.4	274.7	86.8	180.1	248.5	101.0	193.7
Temperature °C)	July*	27.1	18.92	24.7	25.0	22.4	23.9	28.5	22.8	25.0
	Aug.	30.6	.3	26.3	29.2	19.6	23.9	31.8	21.1	25.5
	Sept.	27.3	14.8	20.3	23.1	14.4	18.6	25.4	15.6	20.1
	Oct.	21.7	6.6	12.1	18.0	7.1	12.0	17.5	7.7	12.6
Precipitation (mm)	July*	11.4	0.5	3.7	144.5	144.5	144.5	2.3	0.5	1.4
	Aug.	5.1	0.3	2.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sept.	8.1	0.5	2.9	3.6	3.0	3.3	2.8	0.5	1.6
	Oct.	18.8	0.3	3.9	9.1	1.0	4.1	11.4	0.3	4.0
Solar Radiation (W/m^2)	July*	1,009.5	331.4	836.0	1,123.6	303.2	923.7	1,059.9	242.7	951.8
	Aug.	902.6	495.8	739.0	1,033.0	536.2	887.7	1,015.9	579.2	898.0
	Sept.	827.0	239.6	619.3	904.4	265.3	547.5	865.0	228.3	667.6
	Oct.	619.5	109.4	424.2	686.3	127.7	511.4	695.1	142.6	484.3
Solar Radiation (Ly/min)	July*	1.4	0.5	1.2	1.6	0.4	1.3	1.5	0.3	1.4
	Aug.	1.3	0.5	1.1	1.5	0.8	1.3	1.5	0.8	1.3
	Sept.	1.2	0.3	0.9	1.3	0.4	0.9	1.2	0.3	1.0
	Oct.	0.9	0.2	0.6	1.0	0.2	0.7	1.0	0.2	0.7

*Data for July is based on the installation date of the weather station.
Clarkston: July data represents recordings from 11 July to 31 July.
Ice Harbor: July data represents recordings from 25 July to 31 July.
Little Goose Dam: July data represents recordings from 13 July to 31 July.

Solar Radiation

Solar radiation data at the three stations are presented in Figures 3.6-50 to 3.6-52. The highest monthly average solar radiation was recorded at Little Goose Dam during July (951.8 W/m^2), August (898.0 W/m^2), and September (667.6 W/m^2). The greatest monthly average for October was calculated for Ice Harbor Dam (511.4 W/m^2). Lowest monthly averages of solar radiation for July through October were recorded at Clarkston, with values ranging between 836.0 W/m^2 in July to 424.2 W/m^2 in October. The maximum daily value for solar radiation (1,123.6 W/m^2) was recorded during July at Ice Harbor Dam, while the minimum (109.4 W/m^2) was documented in October at Clarkston.

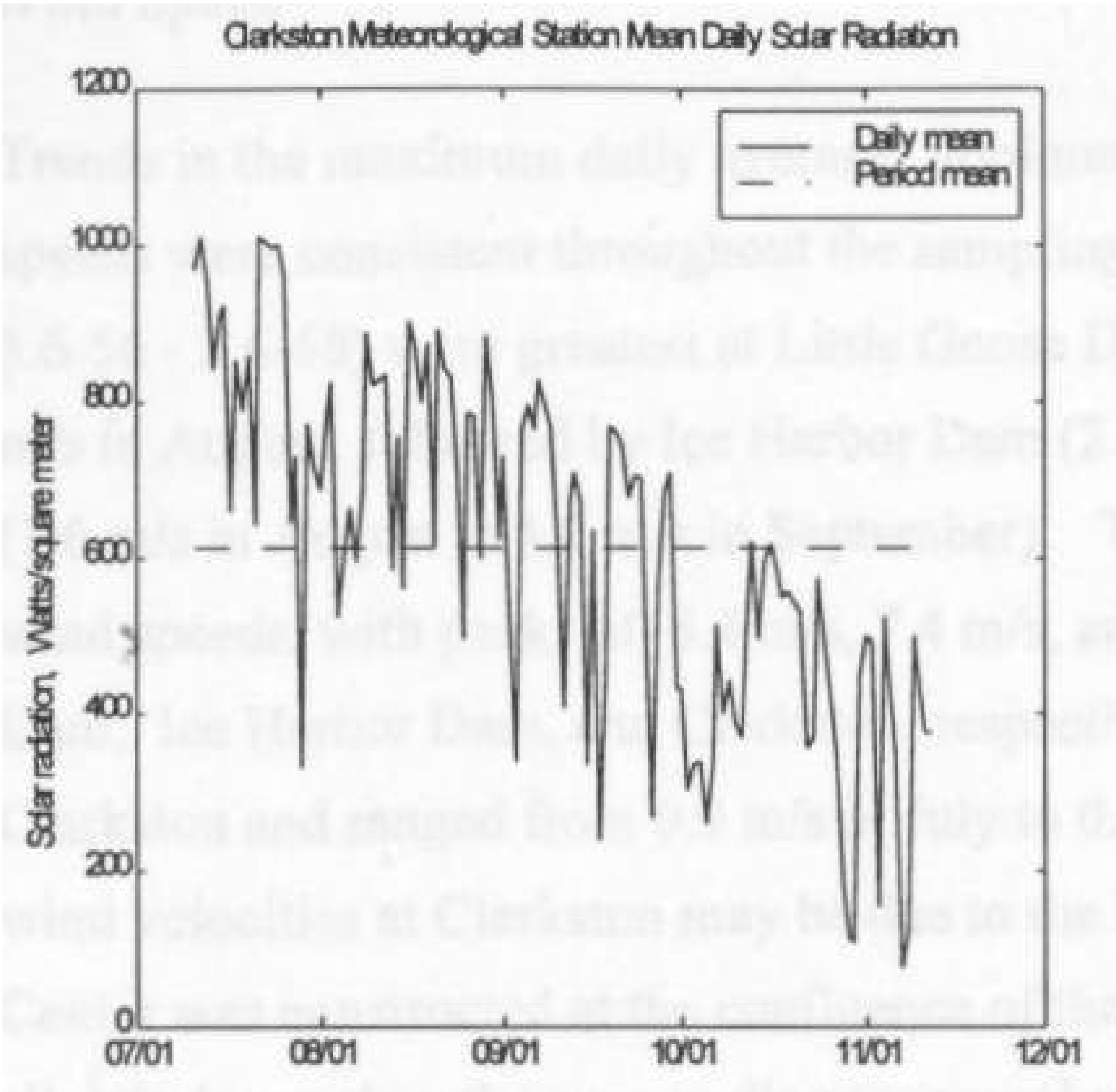

Figure 3.6-50. Daily average solar radiation at Clarkston from 11 July to 31 October

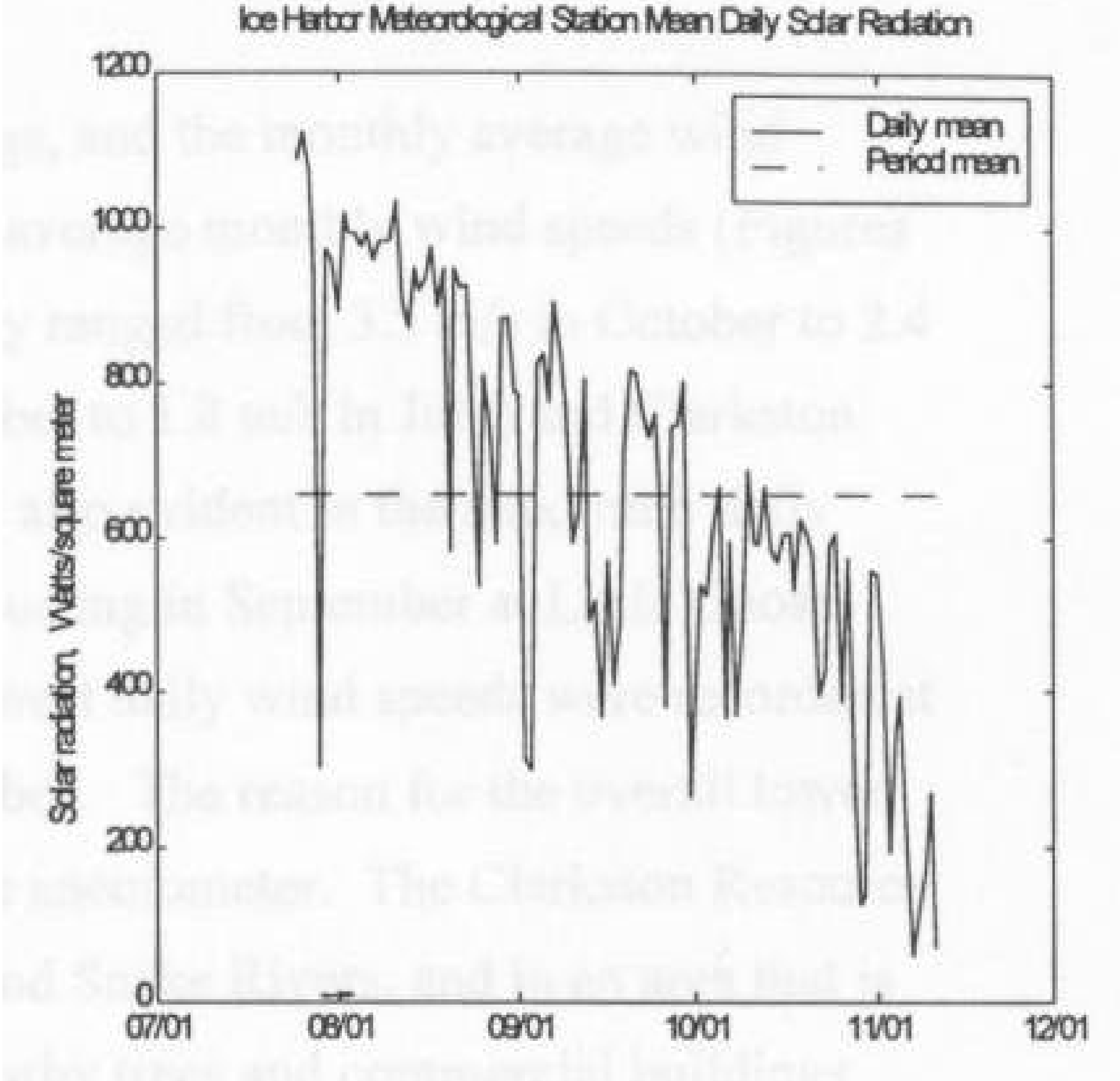

Figure 3.6-51. Daily average solar radiation at Ice Harbor Dam from 25 July to 31 October

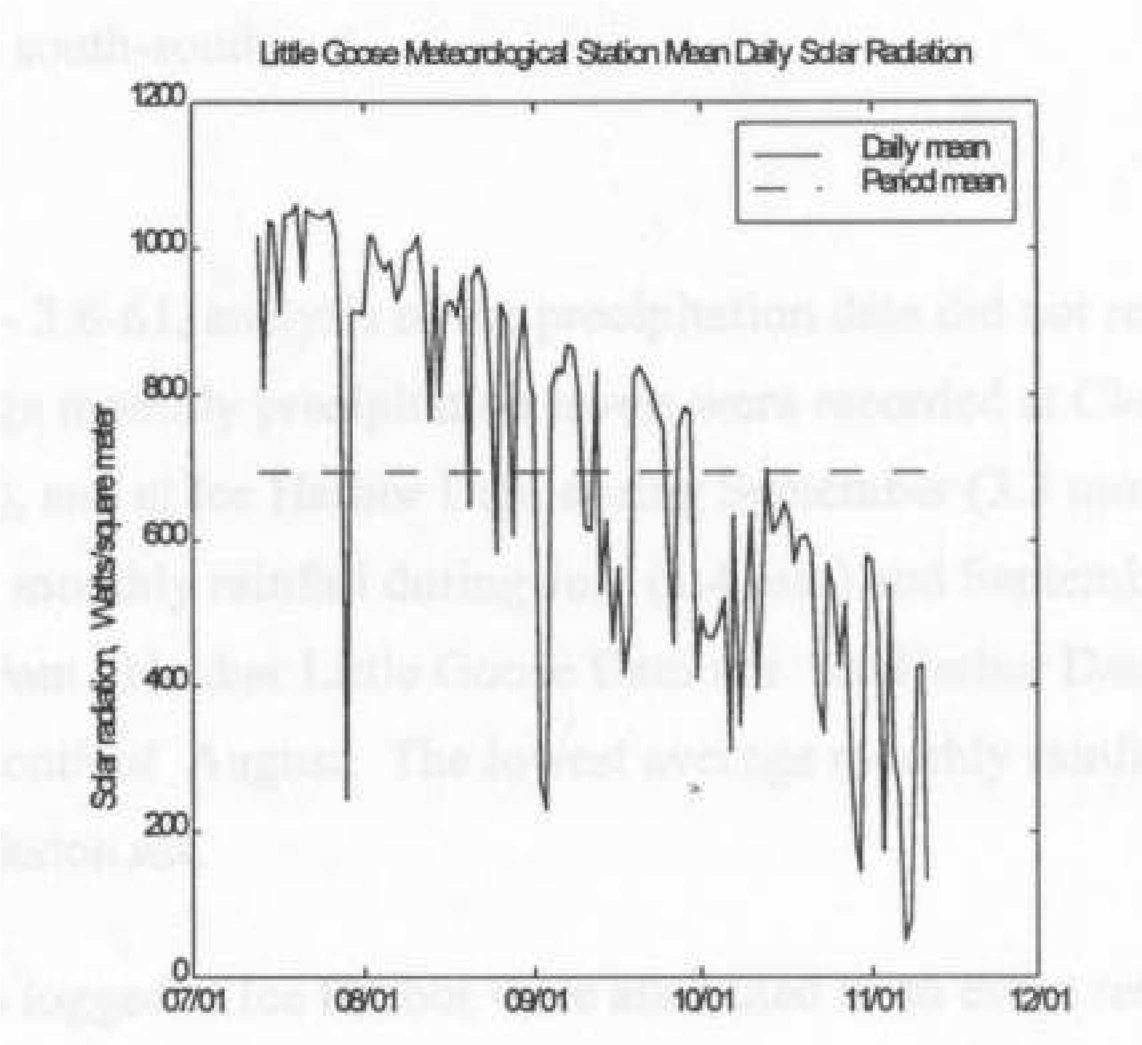

Figure 3.6-52. Daily average solar radiation at Little Goose Dam from 13 July to 31 October

Air Temperature

Average monthly temperatures at Clarkston and Little Goose Dam were similar throughout the sampling season (Figures 3.6-53, 3.6-55). Air temperatures at Clarkston ranged from 26.3 °C in August to 12.1 °C in October, while temperatures at Little Goose Dam ranged from 25.5 °C in August to 12.6 °C in October. Average monthly temperatures at Ice Harbor Dam (Figure 3.6-54) were slightly lower (23.9 °C in July to 12.0 °C in October). The highest average daily temperature (31.8 °C) was recorded during August at Little Goose Dam. The lowest average daily temperature (6.6 °C) was recorded during October at Clarkston.

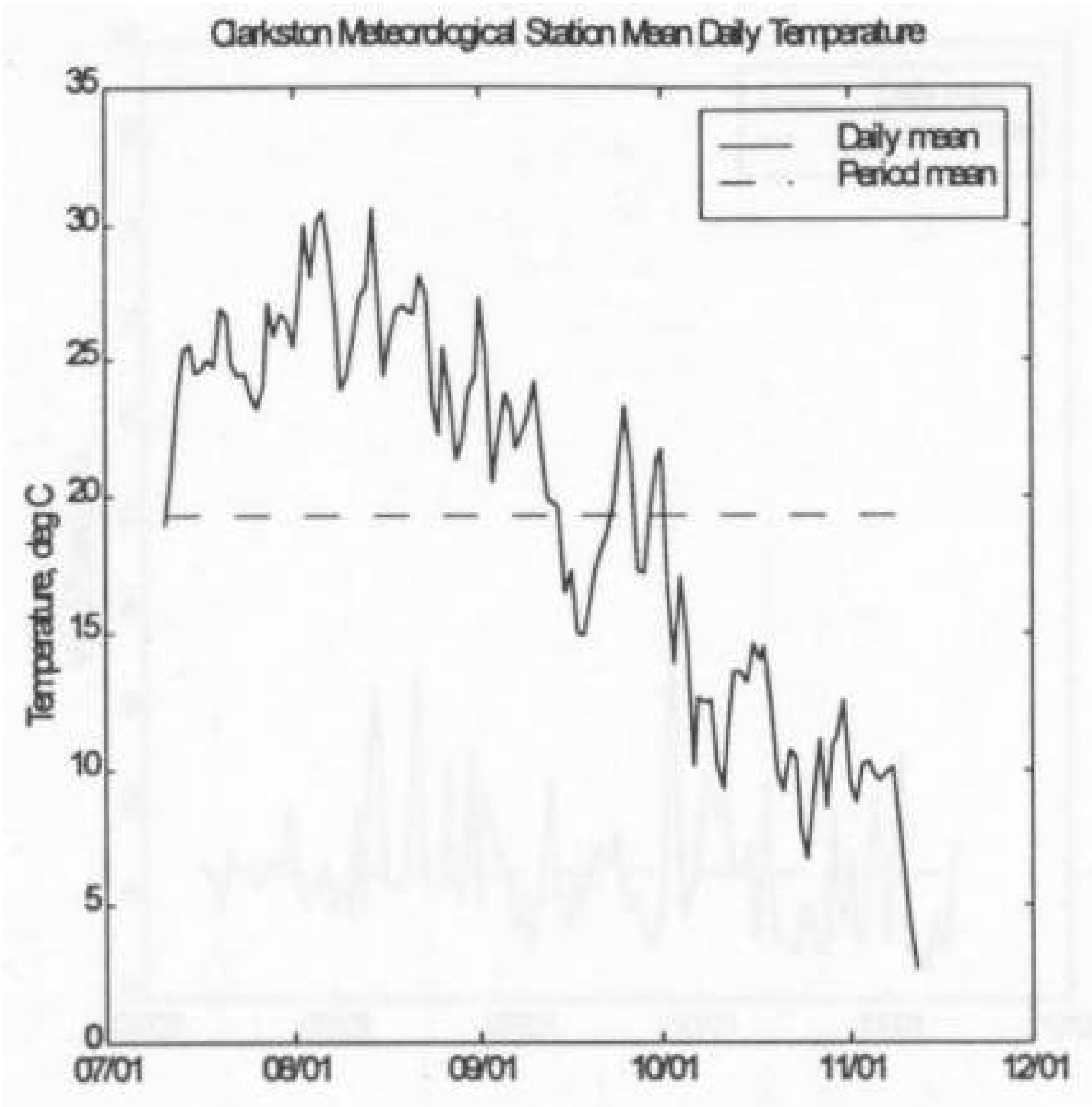

Figure 3.6-53. Average daily ambient air temperature at Clarkston from 11 July to 31 October

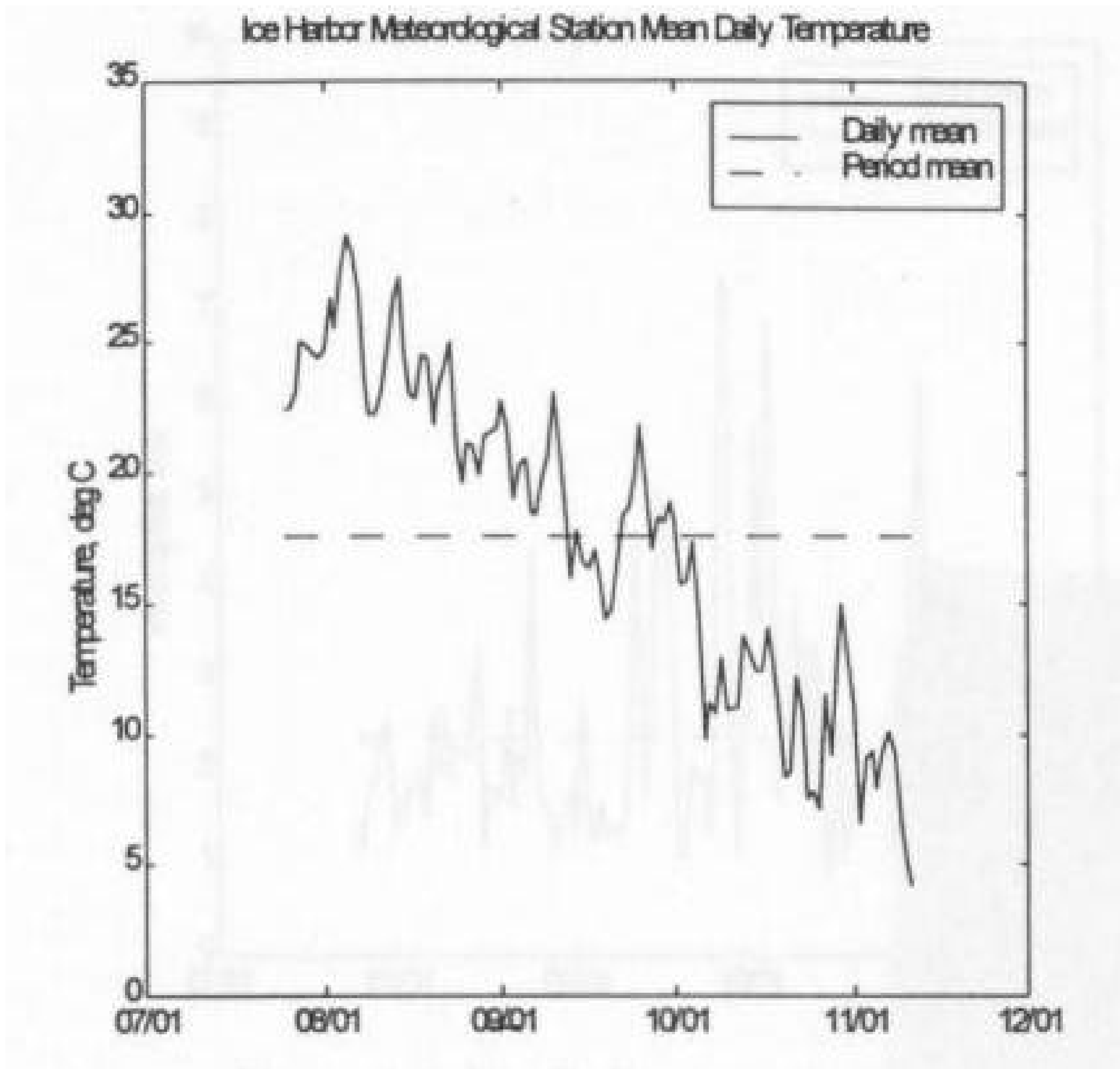

Figure 3.6-54. Average daily ambient air temperature at Ice Harbor Dam from 25 July to 31 October

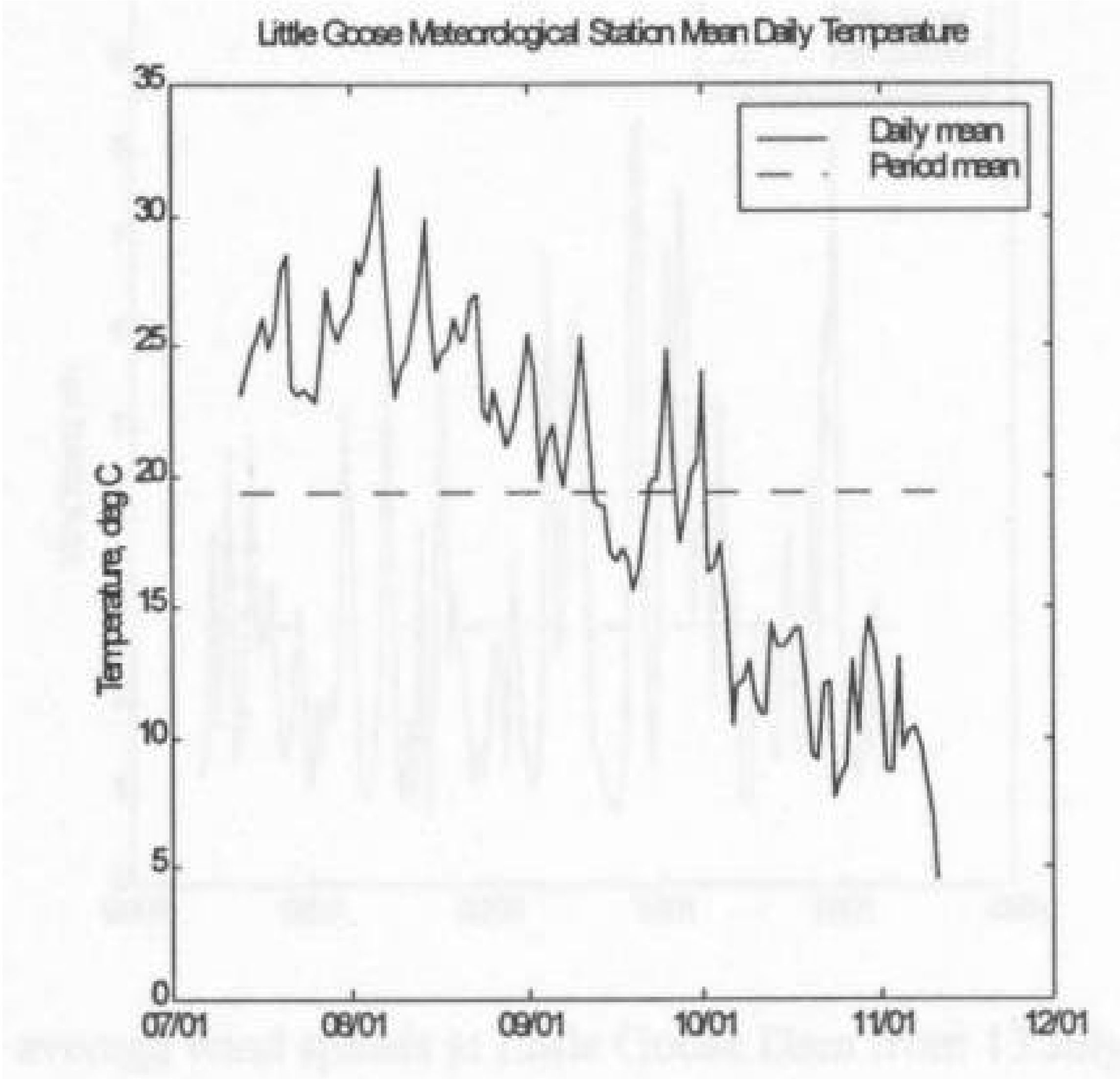

Figure 3.6-55. Average daily ambient air temperatures at Little Goose Dam from 13 July to 31 October

Wind Speed

Trends in the maximum daily average, minimum daily average, and the monthly average wind speeds were consistent throughout the sampling period. The average monthly wind speeds (Figures 3.6-56 to 3.6-58) were greatest at Little Goose Dam where they ranged from 3.5 m/s in October to 2.4 m/s in August, followed by Ice Harbor Dam (2.9 m/s in October to 1.8 m/s in July) and Clarkston (1.6 m/s in August to 1.2 m/s in September). This trend was also evident in the maximum daily wind speeds, with peaks of 8.4 m/s, 7.4 m/s, and 4.0 m/s occurring in September at Little Goose Dam, Ice Harbor Dam, and Clarkston, respectively. The lowest daily wind speeds were recorded at Clarkston and ranged from 0.9 m/s in July to 0.5 m/s in October. The reason for the overall lower wind velocities at Clarkston may be due to the location of the anemometer. The Clarkston Resource Center was constructed at the confluence of the Clearwater and Snake Rivers, and in an area that is slightly lower than the surrounding topography. As such, nearby trees and commercial buildings may be creating a wind shadow, even though the sensor was mounted approximately 20 feet above the ground.

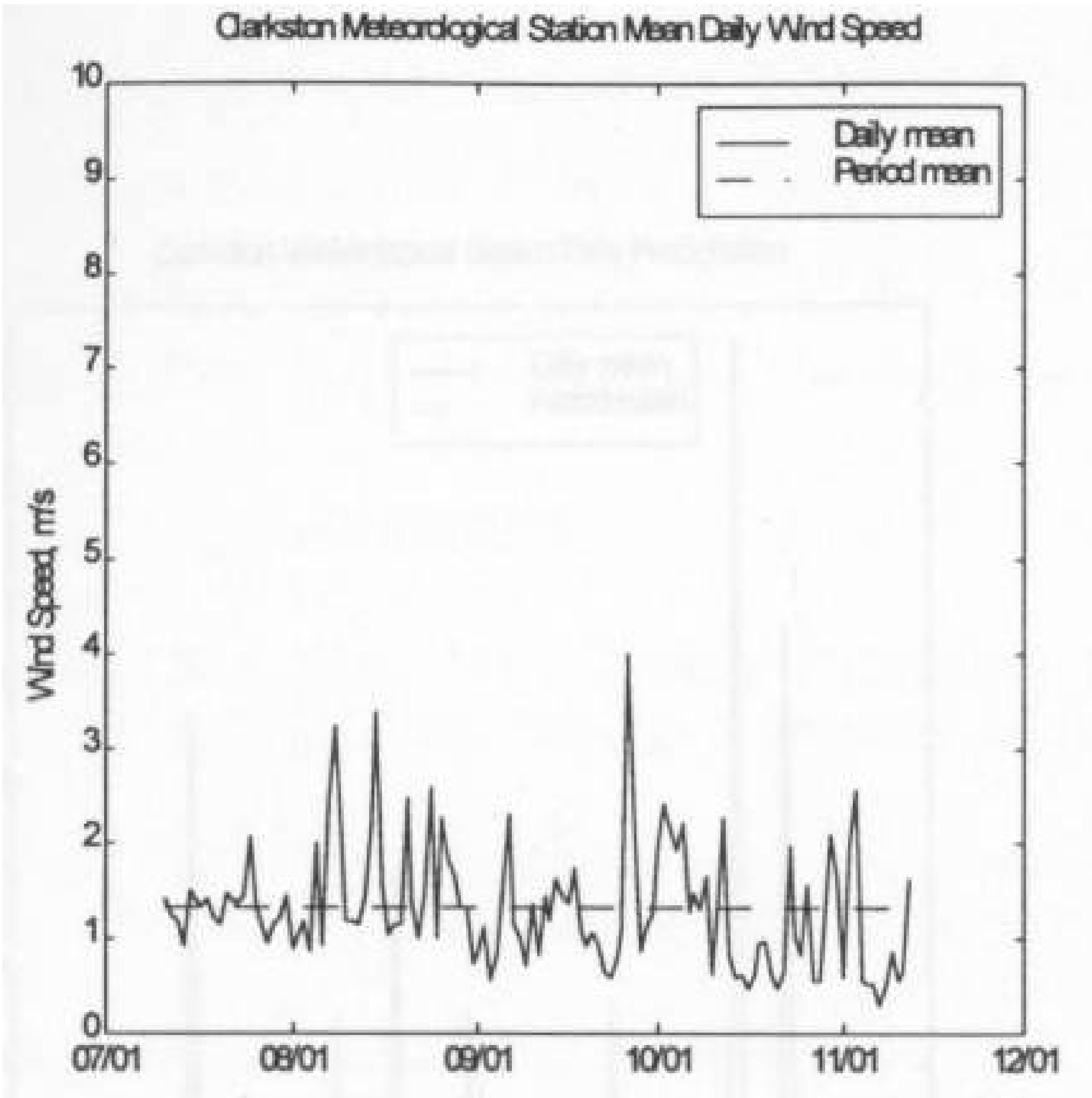

Figure 3.6-56. Daily average wind speeds at Clarkston from 11 July to 31 October

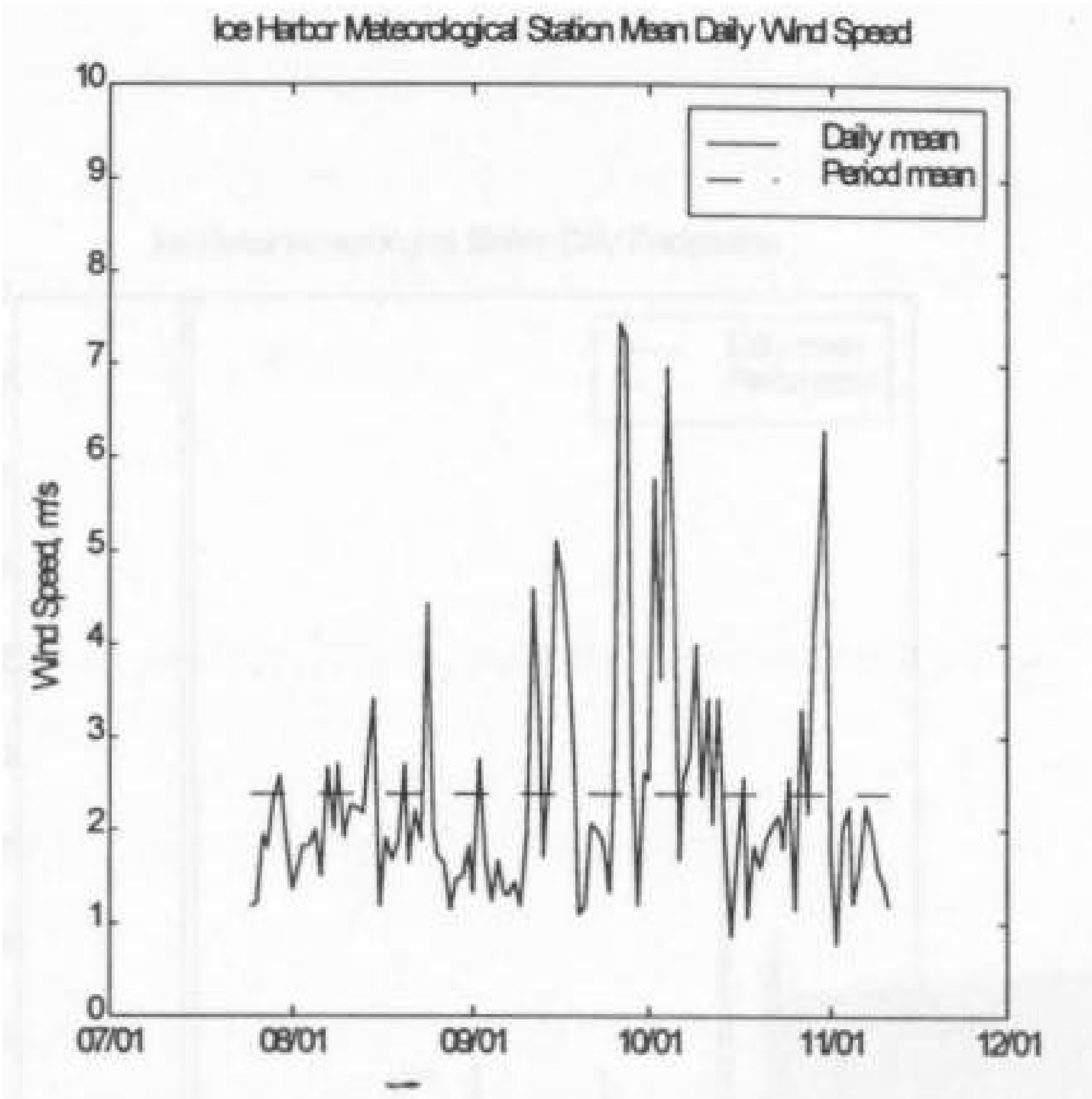

Figure 3.6-57. Daily average wind speeds at Ice Harbor Dam from 25 July to 31 October

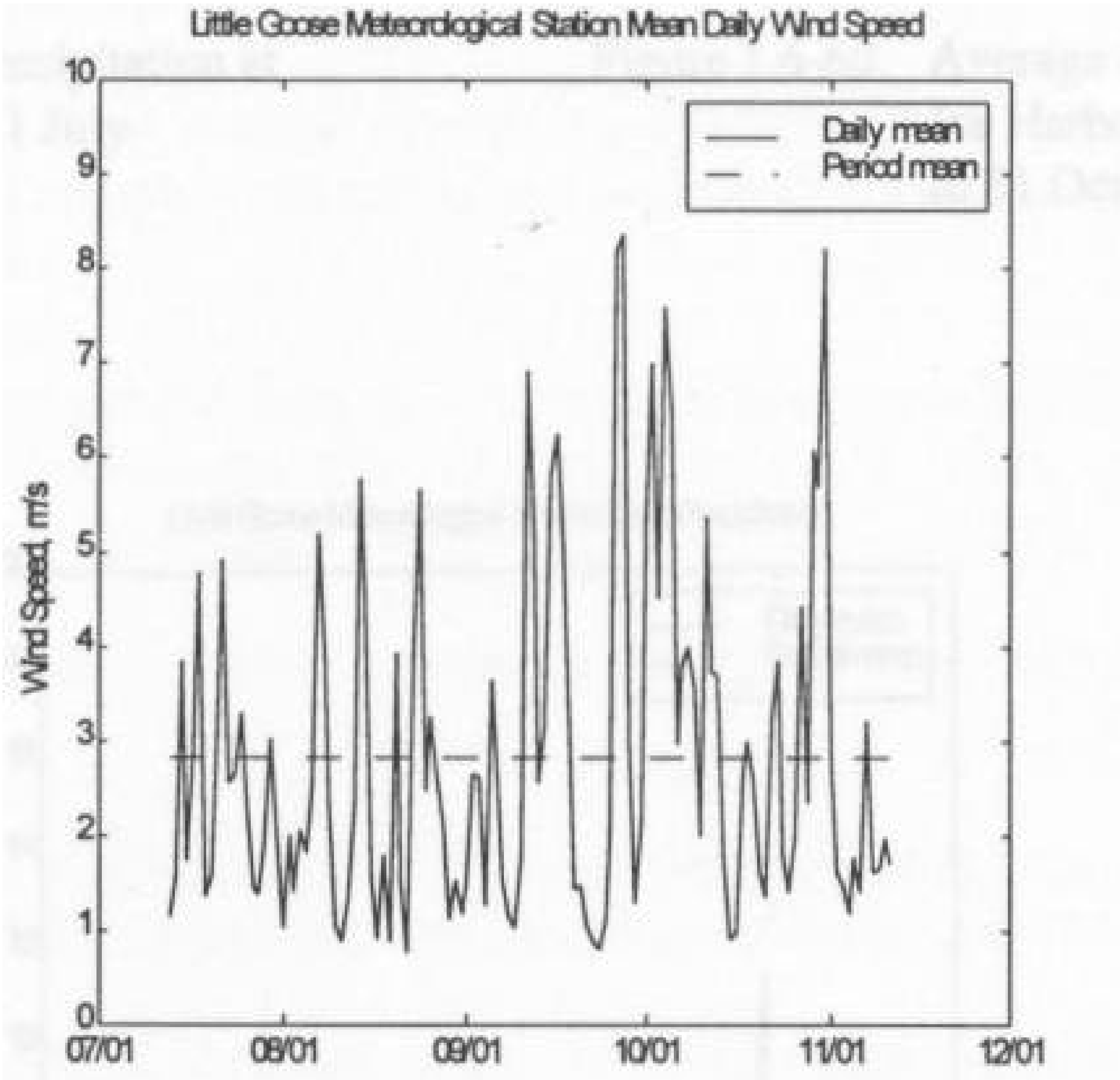

Figure 3.6-58. Daily average wind speeds at Little Goose Dam from 13 July to 31 October

Wind Direction

There were only minor differences with respect to average wind directions at the sites. On average, the winds at Ice Harbor Dam and Little Goose Dam blew from the south, while the average wind direction at Clarkston was south-southeast.

Precipitation

As seen in Figures 3.6-59 to 3.6-61, analysis of the precipitation data did not reveal any consistent trends. The highest average monthly precipitation levels were recorded at Clarkston during July (3.7 mm) and August (2.0 mm), and at Ice Harbor Dam during September (3.3 mm) and October (4.1 mm). The lowest average monthly rainfall during July (1.4 mm) and September (1.6 mm) was recorded at Little Goose Dam. Neither Little Goose Dam nor Ice Harbor Dam had measurable precipitation during the month of August. The lowest average monthly rainfall for October (3.9 mm) was recorded at Clarkston.

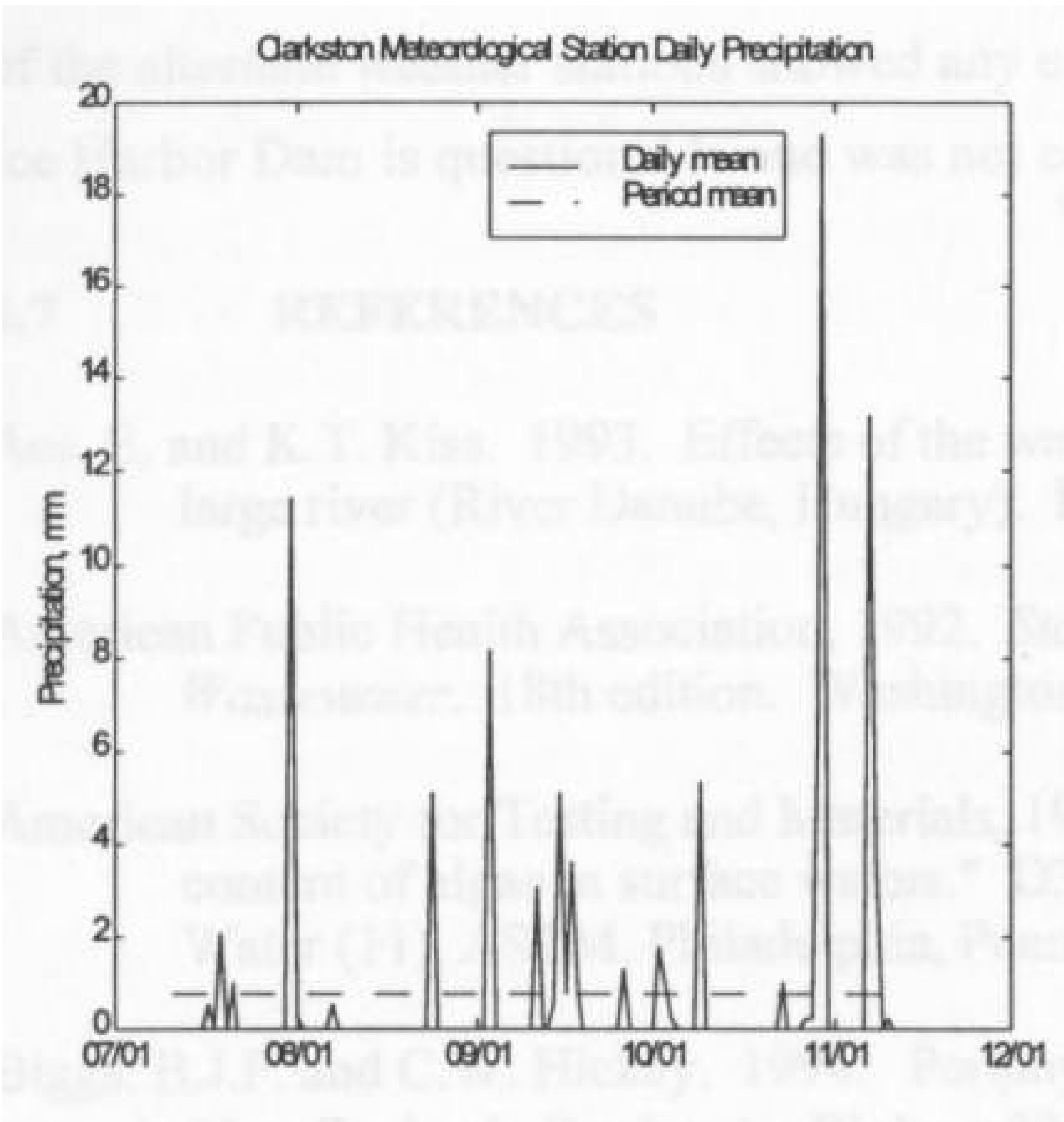

Figure 3.6-59. Average daily precipitation at Clarkston from 11 July to 31 July

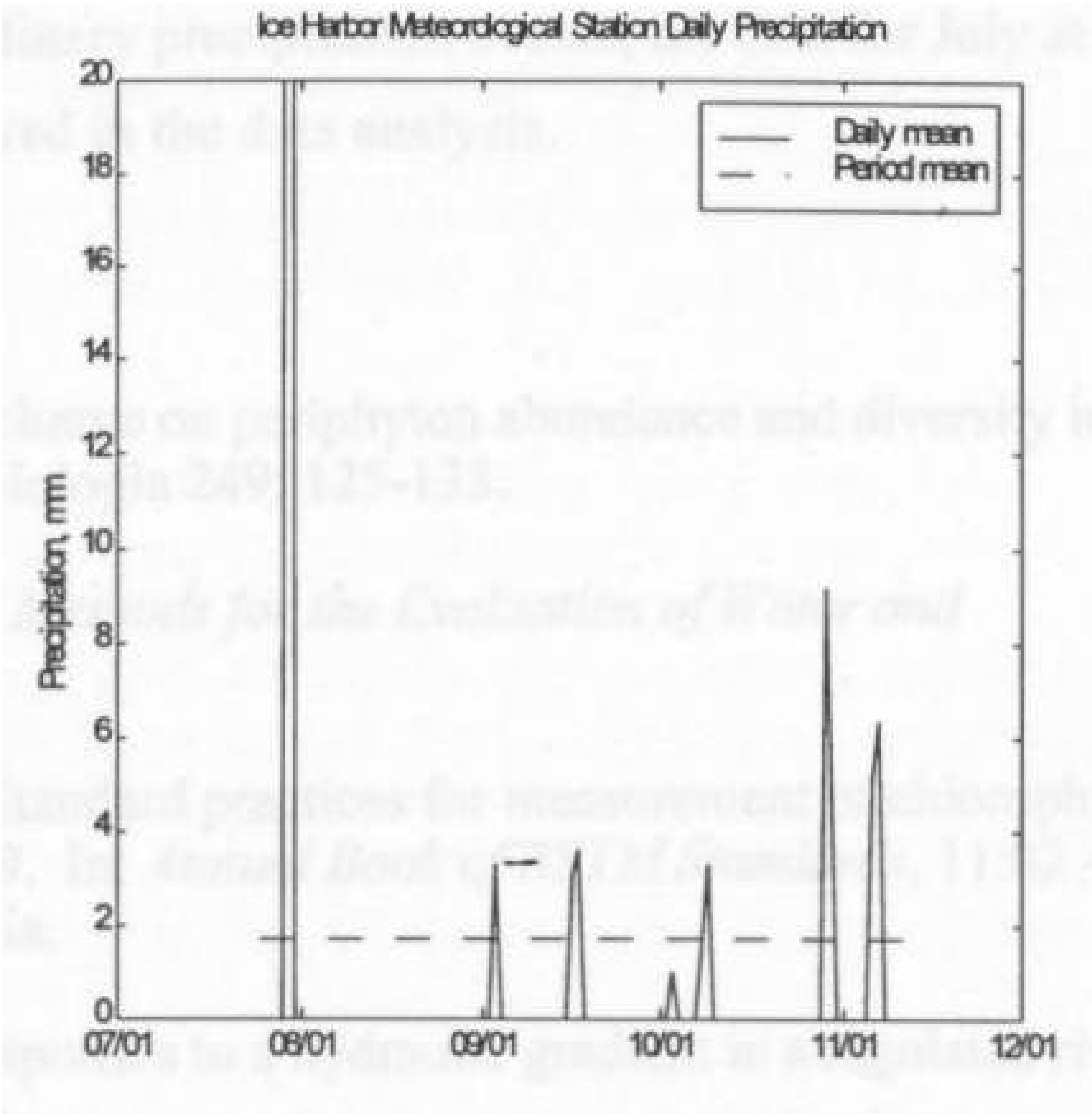

Figure 3.6-60. Average daily precipitation at Ice Harbor Dam from 25 July to 31 October

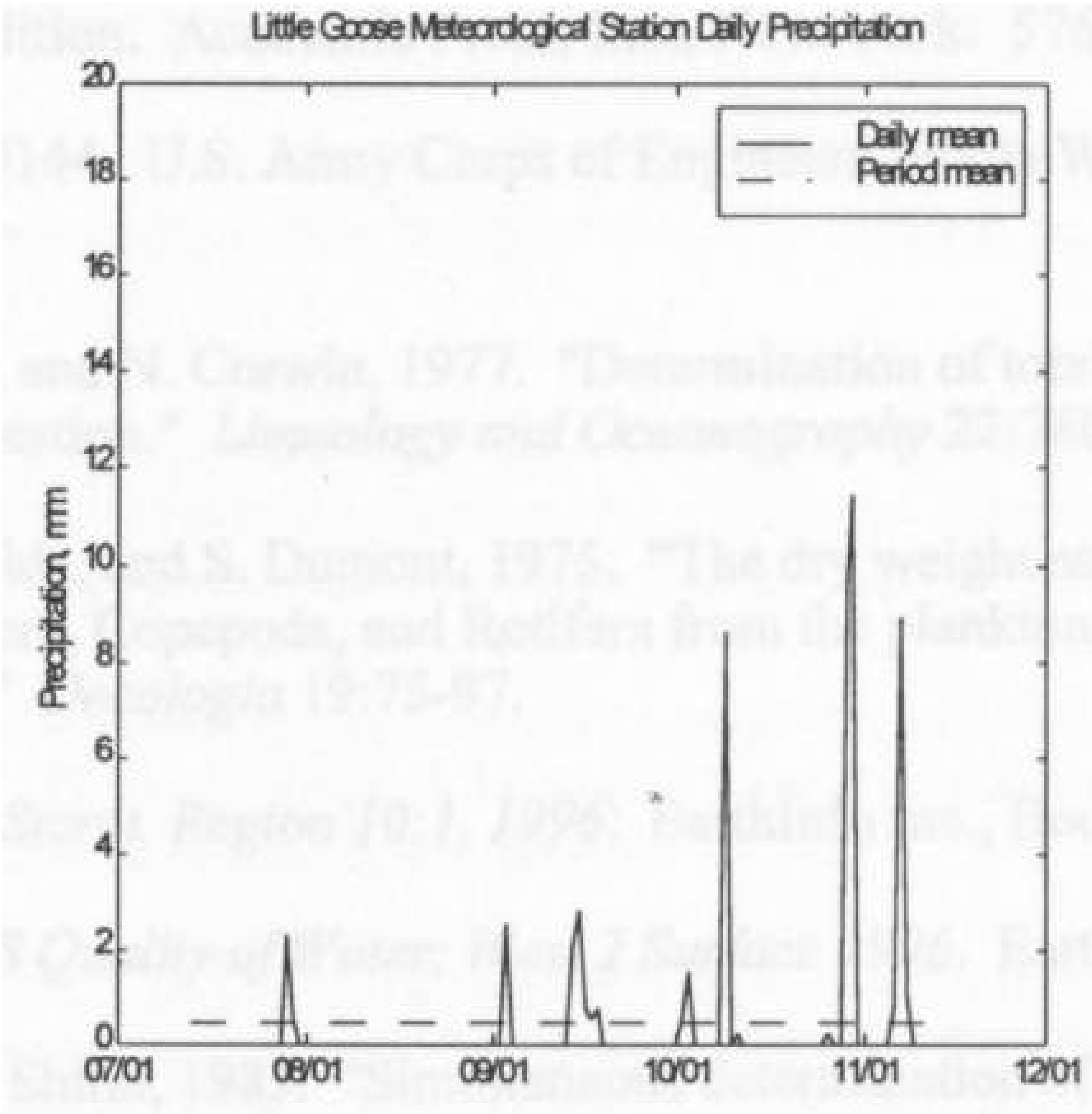

Figure 3.6-61. Average daily precipitation at Little Goose Dam from 13 July to 31 October

The unusually high values logged at Ice Harbor were attributed to an event recorded between 22:00 and 23:00 on 30 July. These precipitation values were compared to data recorded at two Public Agricultural Weather Service (PAWS) weather stations located two miles South of Fishhook Park, Tri-Cities, Washington; and at Washington State University (WSU) at Tri-Cities, Washington. The data was also compared to precipitation levels recorded at a pre-existing weather station located at Ice Harbor Dam and operated by the Corps.

On 30 July, the weather stations at Fishhook, WSU Tri-Cities, and Ice Harbor Dam recorded precipitation events of 0.76 mm, 0.25 mm, and 0.76 mm, respectively. In light of the fact that none of the alternate weather stations showed any extraordinary precipitation events, the data for July at Ice Harbor Dam is questionable and was not considered in the data analysis.

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4.0 - PRIMARY PRODUCTIVITY INDICES

4.1 - INTRODUCTION

The purpose of this task was to develop a multiple regression model to predict planktonic primary productivity levels in the Snake River based on a set of index parameters. One of the objectives of this empirical approach was to select the smallest set of limnological index parameters that would explain the greatest amount of variability in primary productivity.

4.2 - AVAILABLE DATA

The data used for developing an estimation method were from sampling dates during 1994, 1995, and 1997. Those are years for which both primary productivity data (^{14}C uptake rate) and a wide variety of other biological and water quality data were available for several locations in the lower Snake River. The analysis was restricted to data collected in the Snake River as far upstream as river mile 148 and in the Clearwater River up to river mile 11. A total of 172 samples (sampling date/station combinations) were used in the analysis (Table 4.2-1). Occasionally some of the parameters at a station were sampled on a different day than the primary productivity, but as part of the same sampling effort (within the same week). Those were treated as if sampled on the day of primary productivity sampling. Integrated hourly rate (IHR), in mg $^{12}C/m^2/hr$, was used as the dependent variable to represent "true" primary productivity. Data that were compared to the IHR by means of correlation and regression analysis to develop a predictive model included total chlorophyll a, total phytoplankton biovolume, total zooplankton biomass, nutrients (orthophosphate, total phosphate, and total nitrogen), measures of transparency (Secchi depth, photic zone depth, turbidity, suspended solids), "field" data (temperature, dissolved oxygen, conductivity, and pH), concentrations of silica and several ions (calcium, magnesium, sodium, potassium, chloride, and sulfate), and flow data (for both the Snake River and the Clearwater River). The value for the shallowest of the sampling depths was used for chlorophyll *a*, ions, and nutrients. Temperature, dissolved oxygen, conductivity, and pH were averaged over all sampling depths within the photic zone. When a chlorophyll *a*, ion, or nutrient concentration was below the analytical detection limit, one-half of the detection limit was used for the value in the analysis. The 23 potential predictor variables considered in the analysis are summarized in Table 4.2-2.

Table 4.2-1
Sampling Events (Date/Station) Used for Analysis of Relationships
Between Index Parameters and Primary Productivity in the Lower Snake River

Year	Snake River											Clearwater River
	6	18	40	50	68	83	106	108	118	129	148	11
1994		1 Jun 27 Jun 25 Jul 10 Aug 16 Sep 2 Nov 22 Nov				31 May 28 Jun 28 Jul 9 Aug 17 Sep 28 Oct 19 Nov		2 Jun 14 Jun 29 Jun 12 Jul 28 Jul 11 Aug 2 Sep 18 Sep 1 Oct 30 Oct 20 Nov	2 Jun 29 Jun 28 Jul 11 Aug 18 Sep 30 Oct 20 Nov	3 Jun 17 Jun 30 Jun 11 Jul 26 Jul 13 Aug 4 Sep 19 Sep 2 Oct 31 Oct 21 Nov		
1995		18 Mar 9 Apr 13 May 13 Jun 11 Jul 21 Aug 28 Sep				19 Mar 8 Apr 14 May 12 Jun 12 Jul 22 Aug 29 Sep		20 Mar 10 Apr 22 Apr 15 May 30 May 16 Jun 26 Jun 13 Jul 31 Jul 23 Aug 15 Sep 2 Oct 13 Oct	10 Apr 15 May 17 Jun 13 Jul 23 Aug 2 Oct	21 Mar 11 Apr 23 Apr 16 May 31 May 14 Jun 27 Jun 14 Jul 1 Aug 24 Aug 16 Sep 1 Oct 13 Oct		
1997	4 Jun 2 Jul 18 Jul 1 Aug 13 Aug 12 Sep 14 Sep 26 Sep 10 Oct	9 Jun 3 Jul 17 Jul 31 Jul 14 Aug 11 Sep 15 Sep 25 Sep 9 Oct	9 Jun 3 Jul 17 Jul 31 Jul 14 Aug 11 Sep 15 Sep 25 Sep 9 Oct	4 Jun 30 Jun 16 Jul 30 Jul 13 Aug 10 Sep 24 Sep 8 Oct	4 Jun 30 Jun 16 Jul 30 Jul 13 Aug 10 Sep 24 Sep 8 Oct	5 Jun 29 Jun 19 Jul 29 Jul 12 Aug 9 Sep 23 Sep 7 Oct	3 Jun 29 Jun 15 Jul 29 Jul 12 Aug 9 Sep 23 Sep 7 Oct		2 Jun 28 Jun 14 Jul 28 Jul 11 Aug 8 Sep 22 Sep 6 Oct		7 Jun 29 Jun 15 Jul 29 Jul 12 Aug 11 Sep 23 Sep 7 Oct	6 Jun 28 Jun 14 Jul 28 Jul 11 Aug 9 Sep 22 Sep 7 Oct

Station numbers represent river miles upstream of the river mouth.

Table 4.2-2 Variables Evaluated as Potential Predictors of Primary Productivity in the Lower Snake River			
Parameter	**Units**	**Log Transformation?**	**Variable Name**
Total chlorophyll *a*	µg/L	yes	LTOTCHL
Phytoplankton biovolume	mm^3/mL	yes	LPHYBIO
Zooplankton biomass	µg/L	yes	LZOOBIO
Orthophosphate	mg/L	yes	LOPHOS
Total phosphorus	mg/L	yes	LTPHOS
Total nitrogen	mg/L	yes	LTOTN
Secchi depth	m	no	S_DEPTH
Photic zone depth	m	no	PZ_DEPTH
Turbidity	FTU	yes	LTURB
Suspended solids	mg/L	yes	LSS
Temperature	℃	no	TEMP
Dissolved oxygen	mg/L	no	DO
Specific conductance	µS/cm	yes	LSPCOND
pH	pH units	no	PH
Silica	mg/L	no	SIO2
Calcium	mg/L	no	CA
Magnesium	mg/L	yes	LMG
Sodium	mg/L	yes	LMG
Potassium	mg/L	no	K
Chloride	mg/L	yes	LCL
Sulfate	mg/L	yes	LSO4
Snake River flow at Lower Granite	ft^3/s	no	Q_LG
Clearwater River flow	ft^3/s	no	Q_CLW

4.3 - EMPIRICAL MODELING

Each of the potential predictor variables was compared to the IHR by the nonparametric Spearman rank correlation procedure and a scatter plot was examined to evaluate the need for data transformation before performing regression analysis. If the data appeared to be heavily skewed in the plot, which typically occurred when the values spanned a couple of orders of magnitude or more, they were log10 transformed (Reckhow and Chapra, 1983). The variables that were transformed on this basis were IHR, chlorophyll *a*, phytoplankton biovolume, zooplankton biomass, suspended solids, turbidity, orthophosphate, total phosphate, total nitrogen, conductivity, magnesium, sodium, chloride, and sulfate. Because of a few zero values for zooplankton biomass, 0.01 was added to that variable before taking its logarithm.

Stepwise multiple linear regression (MLR) is a technique for reducing the number of predictors while giving up relatively little in ability to predict the parameter of interest (Harris, 1985). MLR models for this analysis were run using the SAS REG procedure (SAS Institute, Inc. , 1989) with log-transformed IHR ("LIHR") as the dependent variable. After an initial run with all 23 potential predictor variables, the data were subset in different ways in an attempt to improve the success of the model. The model selection method STEPWISE was used. The primary criterion of model success was the R-square value, which indicates the proportion of total variation in the dependent variable explained by the regression model.

4.4 - RESULTS

A stepwise MLR run on all samples and all variables produced a model with eight predictor variables (Table 4.4-1) and an R-square of 0.811 (*i.e.*, these variables accounted for 81.1% of the variance in the dependent variable). This model was able to use a total of only 83 of the original 172 observations because of missing values. (A missing value for any of the variables in an observation prevents that observation from being used in the regression model.) The variables contributing the most predictive power to the model were chlorophyll *a* (p=0.0001) and photic zone depth (p=0.0002). Of intermediate importance were phytoplankton biovolume and orthophosphate, both with p=0.0014. Variables of marginal significance were potassium, dissolved oxygen, temperature, and sulfate (ranging from p=0.0164 to p=0.1057). Three of the nine condition index values for the regression were well above 30, an indication of strong collinearity among predictor variables, and hence redundancy in the model.

Table 4.4-1
Multiple Linear Regression Models for Prediction of Primary Productivity from Index Parameters, Calculated from Lower Snake River Data Collected in 1994, 1995, and 1997

	Full Data Set	No Dates Before 15 Jun or After 15 Sep	Only Those Stations Below Palouse River	Only Those Stations Above Palouse River	No Sus Solids	No Ions, Silica, or Turbidity	No Sus Solids, Ions, Silica, or Turbidity	No Sus Solids, Ions, Silica, or Turbidity	No Sus Solids, Ions Silica, Turbidity, or Photic Zone Depth	No Suspended Solids, Ions, Silica, Turbidity, or Phytoplankton Biovolume
Number of objects input	172	90	49	123	172	172	172	172	172	172
Number of objects used	83	42	17	66	96	126	141	141	144	147
Inclusion criterion (p)	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.05	0.05	0.05
R-square	0.811	0.766	0.899	0.745	0.832	0.814	0.829	0.817	0.805	0.741
Value of p for significant variables										
LTOTCHL	0.0001	0.0001	0.0556	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
LPHYBIO	0.0014		0.0020		0.0021	0.0006	0.0001	0.0006	0.0002	X
LZOOBIO		0.0140								
LOPHOS	0.0014				0.0421	0.0004	0.0023	0.0004	0.0016	
LTPHOS										
LTOTN									0.0274	
PZ_DEPTH	0.0002	0.0023	0.0001	0.0875	0.0001	0.0415	0.0286		X	
S_DEPTH										
LSS					X	0.1494	X	X	X	X
LTURB				0.0099		X	X	X	X	X
SIO2			0.0007			X	X	X	X	X
LNA						X	X	X	X	X
LMG						X	X	X	X	X
K	0.0064					X	X	X	X	X
CA						X	X	X	X	X
LCL					0.0022	X	X	X	X	X
LSO4	0.1057				0.0157	X	X	X	X	X
TEMP	0.1021			0.0001	0.0141	0.0001	0.0001	0.0001	0.0001	0.0001
DO	0.0324				0.1199					
LSPCOND						0.0001	0.0001	0.0001	0.0001	
PH		0.0226				0.0575	0.1457			0.0280
Q_LG		0.0998					0.1181	0.0319		0.0001
Q_CLW						0.0622	0.0713			
X = excluded from the analysis										

A reduced data set representing the primary growing season (mid-June through mid-September) was tested to see if the model would be improved by elimination of dates in the fall, winter, and spring, when productivity is generally low. Based on 42 of a possible 90 observations (48 had missing values), this model selected five predictors with an overall model R-square of 0.766 (Table 4.4-1). Again, chlorophyll *a* and photic zone depth were the most important predictor variables. Because of the lower R-square compared to the full data set, however, this approach was not investigated further.

Next, the data were partitioned into two data sets, above and below the inflow of the Palouse River, which is a potentially important influence on the dynamics of the Lower Snake River. Downriver of the confluence, 17 of 49 observations resulted in a model in which the most successful predictor was photic zone depth ($p=0.0001$), and chlorophyll *a* was relatively unimportant ($p=0.0556$) (Table 4.4-1). The overall R-square of this model was 0.899, which was higher than for the whole data set. The remaining data (66 of the 123 observations from upstream of the Palouse) produced a model with $R^2=0.745$, in which chlorophyll *a* and temperature were the most important of four variables selected by the regression procedure (Table 4.4-1). The higher R-square when using only downstream data was thus balanced by a lowering of the R-square when using only upstream data. If the plankton productivity were fundamentally different above and below the Palouse River, the two separate models both are expected to outperform the whole-river model. The fact that the improvement in one was offset by a less successful model in the other suggests that the partitioning of the data set merely resulted in a fortuitous improvement in one at the expense of the other. The small size of the downstream data set (17 observations) does little to inspire confidence in that model, and the upstream model offers no improvement over a whole-river model. On this basis, the approach of seeking different models for subsets of the sampling stations was not pursued further.

As noted earlier, using the full data set of 172 observations and 23 independent variables resulted in a large number of cases in which one or more missing values among the independent variables rendered an observation unusable by the regression procedure (89 of 172 observations, or 51.7% of the data, were ignored by the analysis). Most of these instances were due to one of two reasons: 1) all dates prior to mid-July or early August in 1994, depending on the station, are missing the suspended solids values; or 2) turbidity, silica, and ions data are missing for half of the stations sampled in 1997. Due to these missing values, the information contained in the other independent variables at those stations/dates is unavailable to a regression analysis of the full data set. To avoid this difficulty, regression models were tried 1) without suspended solids data; 2) without ions/silica/turbidity data; and 3) without either. The rationale for excluding these parameters from the analysis was not only that the regressions would have a larger data set available from which to build the model, but also that in earlier regression runs none of these variables played an important role in the final models selected by the analysis. For both suspended solids and turbidity, it can also be argued that by virtue of their high correlation with other measures of transparency (photic zone depth and Secchi depth) most of the information that would be conveyed by suspended solids and turbidity data would still be available for the analysis.

Without suspended solids in the regression, the number of usable observations increased from 83 to 96, and the resulting model had an R-square of 0.832 (Table 4.4-1). Without ions/silica/turbidity (but with suspended solids), 126 observation were usable, producing an R-square of 0.814 (Table 4.4-1). Without either suspended solids or ions/silica/turbidity, the regression used 141 observations and produced a model with $R^2=0.829$ (Table 4.4-1). Clearly, the elimination of these variables from consideration by the analysis enlarged the usable data set substantially without adversely affecting the success of the resultant model. Based on this result, further attempts to refine the model were restricted to the 14 remaining independent variables after eliminating suspended solids, turbidity, silica, sodium, magnesium, potassium, calcium, chloride, and sulfate from consideration. Deleting these parameters from the analysis had the further benefit of reducing some of the redundancy among predictor variables in that there are very strong correlations among the ions, between the ions and the remaining transparency variables, between the ions and the nutrients, and between silica and the nutrients.

The previous model runs used 0.15 as the significance level for entering the model or for staying in the model at each step of the stepwise procedure, which has the effect of including any parameter of even marginal significance in the model. This approach provides the best prediction within the data set at hand, but there is a risk of including variables that do not contribute to the predictive power of the model to the larger population of all dates and locations (beyond those in the 1994-1997 data set). As the purpose of developing this model is to provide a predictive model that can be used in the future to replace direct plankton primary productivity measurements, the final model chosen should be limited to variables that bear a true relationship with productivity, excluding those that just by chance provide a slight improvement in the predictive success within the data set used to develop the model. Based on this reasoning, the 14-variable data set was reanalyzed using 0.05 as the significance level for entering and staying in the model. The resulting six-variable model had only a slightly lower R-square (0.817 compared to 0.829) than the nine-variable model selected using the 0.15 significance level (Table 4.4-1). The most important factors were chlorophyll *a*, temperature, and conductivity (all with p=0.0001). Of secondary importance were orthophosphate (0.0004), phytoplankton biovolume (p=0.0006), and flow at Lower Granite (p=0.0319).

One of the attributes of a useful predictive model is that it is not overly costly to use. Phytoplankton biovolume is a parameter that is more expensive to measure than many of the other variables, and as it was not one of the most important factors in the model, the regression was repeated with 13 independent variables (excluding phytoplankton biovolume) to see if the model produced without it would offer a viable alternative. The resulting model, based on 147 observations, had an R-square of 0.741 and four predictors (Table 4.4-1): chlorophyll *a* (p=0.0001), temperature (p=0.0001), flow at Lower Granite (p=0.0001), and pH (p=0.0280).

Similarly, the regression was run without photic zone depth among the independent variables, to see whether relying on the single transparency measure of Secchi depth would be sufficient for developing the model. Photic zone depth and Secchi depth are highly correlated, and Secchi depth is much simpler to measure. The model derived from regression of 13 independent variables excluding photic zone depth had an R-square of 0.805 and six predictors (Table 4.4-1): chlorophyll *a* (p=0.0001), conductivity (p=0.0001), temperature (p=0.0001), phytoplankton biovolume (p=0.0002), orthophosphate (p=0.0016), and total nitrogen (p=0.0274).

4.5 - PROPOSED INDEX PARAMETERS FOR THE SNAKE RIVER SYSTEM

In nearly all of the various regressions examined, total chlorophyll a was determined to be the most important of the predictor variables selected for the model. This is consistent with the high correlation between chlorophyll a and 14C measurements of primary productivity. Chlorophyll a alone, however, only accounted for about 50% of the variation in productivity. By including other parameters in the multiple linear regression predictive models described above, about 80% of the variance was explained. After eliminating nine variables from the data set that did not contribute much to model success and had restricted the number of useful data observations because of missing values, the best predictive model found, using a significance level of 0.05 for variable inclusion in the model, was this one with an R-square of 0.817 and n=141 observations (Equation 1):

$$\text{LIHR} = 0.5999\ \text{LTOTCHL} + 0.0343\ \text{TEMP} + 0.5461\ \text{LSPCOND} - 0.1977\ \text{LOPHOS} + 0.1990\ \text{LPHYBIO} - 0.00000123\ \text{Q\_LG} - 1.9694$$

where variables are as defined in Table 4.2-2.

The alternate model that was identified by excluding photic zone depth from consideration (as a potential cost saving measure) had essentially the same success, with R2=0.805 and n=144 (Equation 2):

$$\text{LIHR} = 0.5923\ \text{LTOTCHL} + 0.8761\ \text{LSPCOND} + 0.0308\ \text{TEMP} + 0.2275\ \text{LPHYBIO} - 0.1806\ \text{LOPHOS} - 0.3313\ \text{LTOTN} - 2.9710$$

The other alternate model that was identified (by excluding phytoplankton biovolume from consideration, perhaps a more effective cost savings) had a somewhat reduced success (R^2=0.741, n=147), but still much higher than the predictive value of chlorophyll *a* alone (Equation 3):

$$\text{LIHR} = 0.8237\ \text{LTOTCHL} + 0.0394\ \text{TEMP} - 0.00000253\ \text{Q\_LG} + 0.1207\ \text{PH} - 0.3826$$

As another measure of model success besides R-square, the 95% confidence limits for predicted values were examined. The predicted productivity values calculated from Equation 1 were regressed against the actual 14C measurements, and the prediction confidence limits were calculated. For the median of the predicted values, the precision of the model was plus or minus 24.96% of the predicted log transformed productivity (*i.e.*, the true log-transformed productivity should be within 25% of the prediction about 95% of the time).

4.6 - LITERATURE CITED

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5.0 - WATER QUALITY MODEL

5.1 - INTRODUCTION

The lower Snake River between the Idaho/Washington border and immediately upstream of the confluence of the Snake with the Columbia River near Pasco (RM 150-0) was impounded by a series of four hydroelectric dams with navigation locks in the 1960's and 1970's (Figure 3.3-1). Limited 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. 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 normative 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 will not likely approximate water quality to be expected in the normative 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 normative river system. The model of the normative system (with all four dams removed) was built and calibrated using bathymetric and hydraulics data from 1934, and temperature data from the 1950's prior to damming. 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. We used 1994 to represent a dry year, 1995 to represent an average year, and 1997 to represent a wet year. 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. Differences in the thermal and nutrient regimes and the overall productivity of the system, as well as the comparmentalization of the productivity were examined. These differences will be used subsequently to guide the decision-making process in evaluating alternatives for the restoration of anadromous fish stocks in the lower Snake River and stream reaches above. The model was developed to predict system-wide secondary productivity. The model was not designed to predict the biomass of specific species in the aquatic community. A much more detailed modeling effort, using approaches like bioenergetics, would be most appropriate for making species-specific predictions.

We want to stress that the analysis presented herein is best used on a relative basis for comparing flow and hydraulic scenarios rather than as a definitive predictive model for several reasons. First, the data available to calibrate and verify the model were limited. Second, the normative state used as 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. We assume that the stream channel remaining after the redistribution of sediments will be similar to that which was present in the system in 1934, the last pre-dam year extensive bathymetric data were collected on the system.

5.2 - MODEL SELECTION

Selection of an appropriate model to simulate the anticipated water quality changes in the lower Snake River attributed to removal of the dams on the lower Snake River was predicated on finding a model that could answer the questions about biological productivity, could be implemented in a timely manner, contained a full suite of water quality parameters related to productivity, and used available input data for the lower Snake River. Model selection was an iterative process which began with a meeting between the Corps, the project team, and other Corps contractors involved in related efforts on the lower Snake River. The purposes of this meeting were to discuss and define: 1) the purpose and context for modeling biological productivity on the lower Snake River; 2) desired approaches to modeling productivity and available candidate models for this project; and 3) relations between the data requirements of the various candidate models and the field sampling program. This initial meeting and subsequent teleconferences resulted in the development of a profile of the ideal model for this application.

The model selected for use needed to be compatible with the following features and characteristics:

1. Flows and project operations--The selected model needed to be capable of simulating the range of flows and velocities expected in the system, as well as the range of system physical configurations and anticipated flows for the future (normative) lower Snake River. The future system will be free-flowing, but will depend in part on a regulated hydrograph from upstream projects. During times of low flow, the river will have a slightly meandering and minimally braided river channel.

2. Temperature--The impact of altered thermal regimes on salmonids needed to be addressed by the selected water quality model. In order to adequately predict temperature, the model needed to be capable of simulating temperature using the head budget method. Many of the biological and chemical rate processes to be included in the simulations are temperature-dependent. We concluded that accurate temperature modeling would increase the confidence in predictions for the chemical and biological components of the system.

3. Adequate simulation of transport and flow--The selected model needed to be capable of simulating dynamic (unsteady) flows and transport rather than being based on steady state solutions to the equations of motion and continuity.

4. Components and controls of biological productivity--The structure of the model needed to contain compartments or state variables that correspond to the significant components of biological productivity anticipated for the normative river system. These included several functional groups of phytoplankton and benthic algae, zooplankton, benthic invertebrates, and fish. Similarly, the model needed to include those processes or state variables that exert significant controls over primary and secondary biological productivity. For primary productivity, these would include river flows/velocities, soluble nutrients (N and

P), temperature, light, and photic zone depth, and interactions between light penetration and suspended solids (inorganic and organic). For the secondary productivity, these would include flows/velocities, characteristics of bottom substrates (*e.g.*, organic content, embeddedness), temperature, primary productivity, and dynamics of detritus or particulate organic matter.

5. Degree of model evaluation and credibility--We decided that the model selected for this study should have received significant and thorough evaluation through previous applications, so that results from this application would be credible. Ideally, this would include previous applications in the Pacific Northwest.

Numerous models could lend insight into the potential future biological productivity of the lower Snake River. Among those models considered were HEC-5Q, RBM10, 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.

5.3 - WQRRS

The WQRRS was originally developed by Chen and Orlob of Water Resources Engineers, with subsequent modification and continued maintenance and distribution by the Corps of Engineers Hydrologic Engineering Center (HEC). This model, particularly the hydrodynamics and temperature portions, has been extensively evaluated and applied to reservoir, riverine, estuarine, and coupled river-reservoir systems throughout the U.S. The biological portion of the model, particularly the PC version, has not been as extensively tested. However, the biological relationships in WQRRS are similar to those found in many other biological models. Some of its initial applications were to lakes and reservoir in the Pacific Northwest and in California. The overall model consists of three separate, but integrable modules: a reservoir module (WQRRSR), a stream hydraulic module (SHP), and a stream quality module (WQRRSQ). The three modules may be readily integrated for a complete river water quality/ecological analysis through automatic storage of model results for input to downstream simulations. The stream module treats a stream segment in a 1-D longitudinal manner, while reservoir segments are treated in a 1-D vertical manner.

The WQRRS contains the most complete set of ecological and water quality state variables of the models evaluated. This allowed direct model estimation of significant components of both primary and secondary production in the normative river. Model state variables include: temperature, fish (three functional groups), aquatic insects, benthic animals, zooplankton, phytoplankton (two functional groups), benthic algae (two functional groups), detritus, organic sediment, inorganic suspended solids (up to five types), inorganic sediment, dissolved orthophosphorus, dissolved ammonia, dissolved nitrate, dissolved BOD (*i.e.*, DOC), coliform bacteria, total inorganic carbon, alkalinity, TDS, and pH (not all of the possible state variables were included in our simulations). Temperature simulations were performed based on the head budget method, which evaluates the five major components of the heat budget of a waterbody. The WQRRS, because of its complete state variable structure, also has extensive biological/ecological data requirements. In order to perform the simulations required on the normative system, estimation of many of the components of the biological system and assumptions on the rate processes governing many of the biological state variables had to be made. Where possible, data from the Snake River prior to construction of the dams were used. However, these data were limited, particularly with respect to the biological community. Recent data from the Snake River upstream of its confluence with the Clearwater River were used to estimate some of the biological community components. Other estimations were done from other river systems in the area, such as the middle Snake River, as well as through best professional judgment. Where practical, model sensitivity to errors in estimation of the parameters associated with the biological community of the normative river system was assessed by varying the parameters through the range of values found in the literature. This sensitivity analysis is discussed in detail later in this report.

In numerous previous applications, WQRRS has been shown to be capable of reliable and representative simulation of both temperature and ecological/water quality variables, in both river and reservoir settings, relative to the purpose of model application. Among the candidate models, WQRRS most closely fulfills the diverse requirements and criteria identified for the present project.

5.4 - APPLICATION OF WQRRS TO THE SNAKE RIVER

5.4.1 - Conceptual Model

The water quality model employed in this study had to be based in part on the composition of the aquatic community of the lower Snake River and the relationship between the aquatic community and its physical and chemical environment. The aquatic community in the lower Snake River would change in composition and function if the four dams were removed. The current biological system of the lower Snake River is driven by phytoplankton primary production with small contributions from attached benthic algae. These primary producers are in turn consumed by zooplankton and, to a lesser degree, by aquatic insects and other aquatic animals, which are consumed by planktivorous fish and benthivorous fish, respectively. These fish are eaten by the piscivores. Among the trophic levels represented in the simplified food chain are several cross linkages which make the chain a food web. The generalized food web of the existing impounded lower Snake River is presented in Figure 5.4-1.

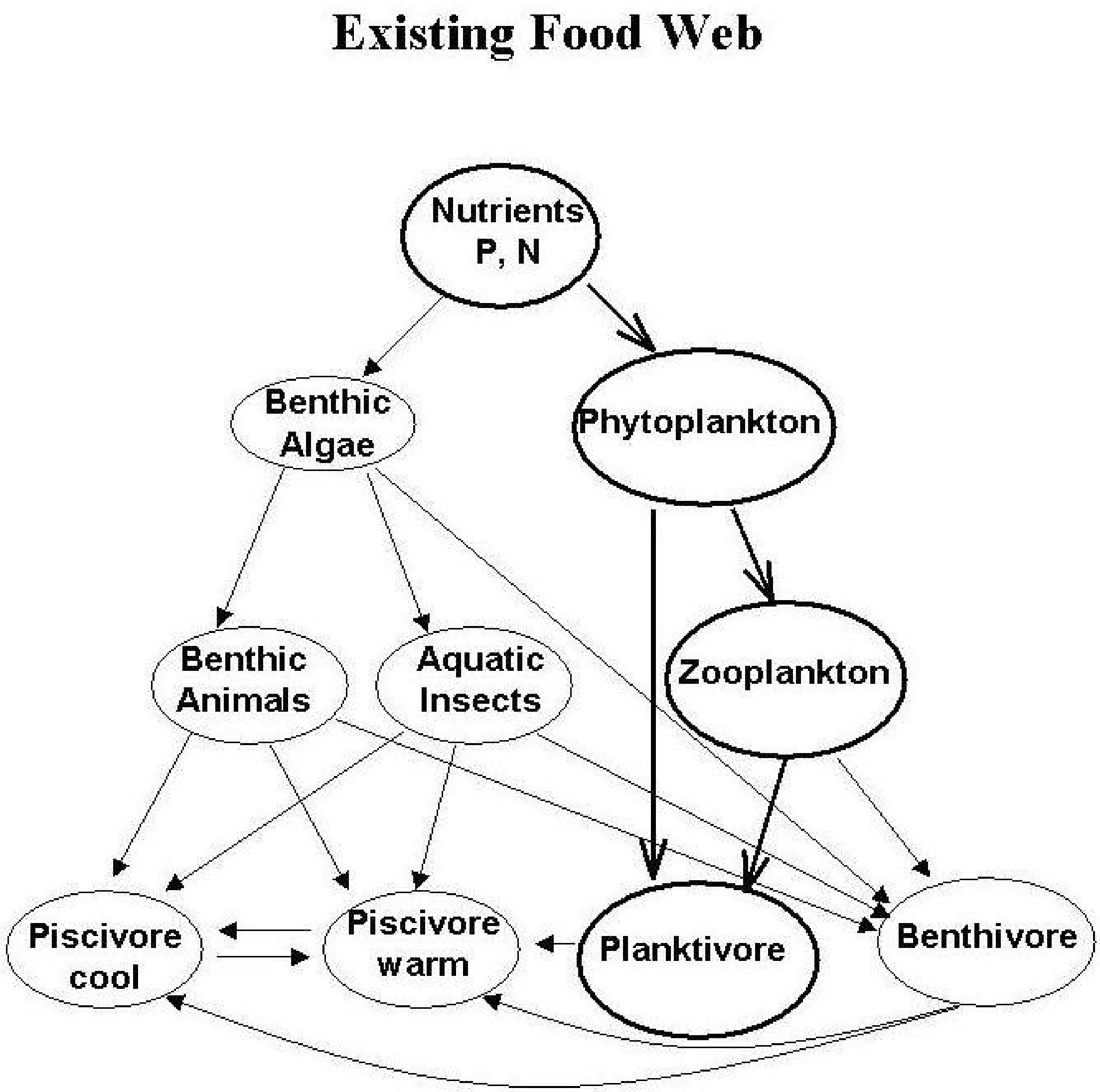

Figure 5.4-1. Generalized Food Web, Impounded Lower Snake River

The food web of the normative river will resemble the existing food web of the lower Snake River in some respects but differ markedly in others. The top piscivores will likely remain northern pikeminnow and smallmouth bass, but the transfer of the bulk of the energy from primary productivity will be through the pathway, that includes attached benthic algae, macrophytes, aquatic insects, benthic animals, and benthivorous fishes. Phytoplankton and zooplankton will become minor components of the food web of the normative river. A generalized food web of the normative river is presented in Figure 5.4-2.

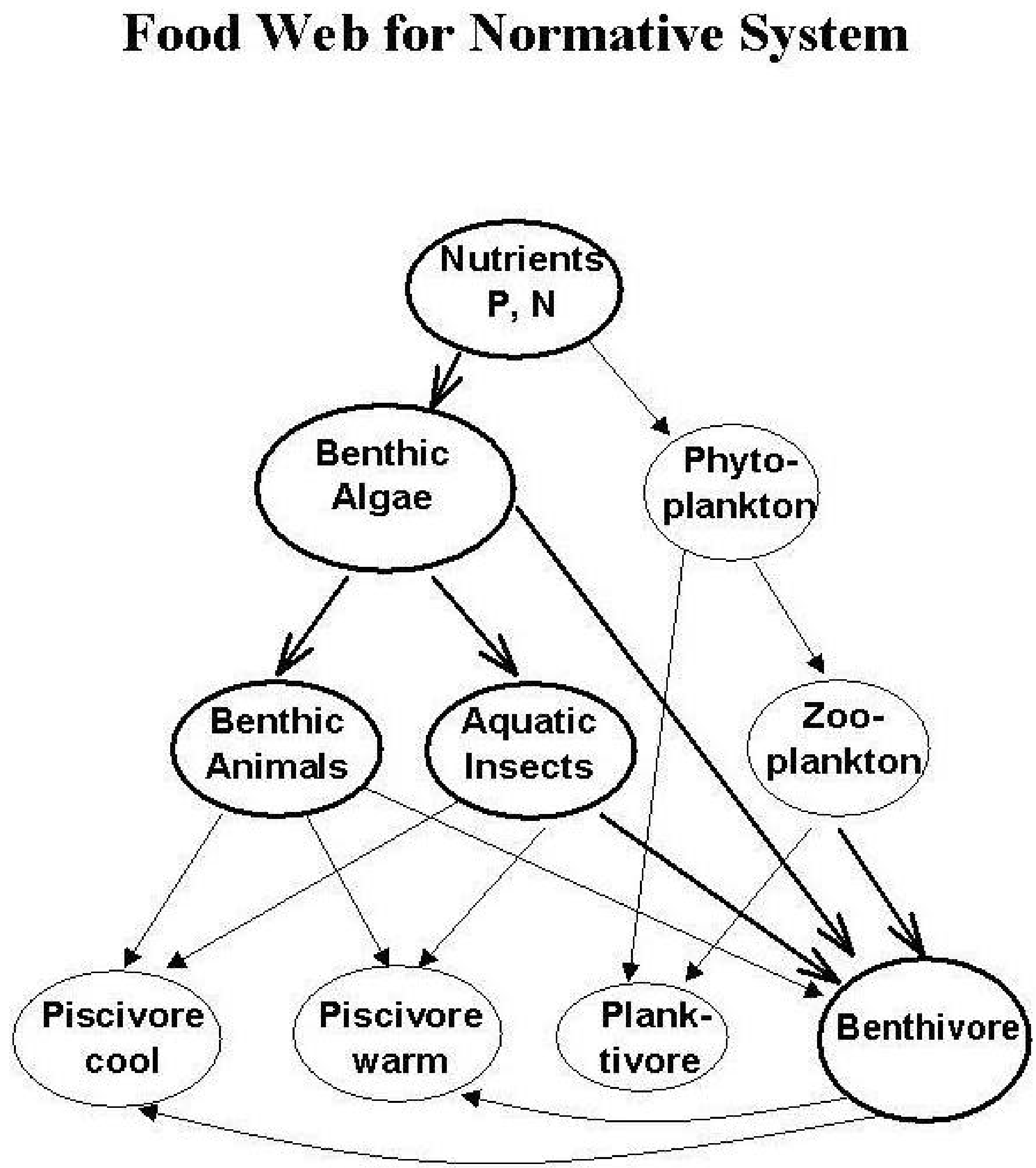

Figure 5.4-2. Generalized Food Web, Normative Lower Snake River

The aquatic community anticipated for the normative river formed the basis for the configuration of WQRRS as it was applied to the system. The available compartments for modeling the aquatic system in WQRRS are presented in Figure 5.4-3. The model limitations, in terms of numbers and types of available compartments for representation of the food web to be simulated, have dictated that elements of the food web and compartments be combined. This simplification can be justified by the absence of information concerning the relationships of many of the specific species in the aquatic community to each other and the feeding preferences of predators on specific prey species within a group. For example, many of the fish species in the lower Snake River community prefer different prey seasonally and as they grow. These fish may start as primarily planktivorous fish, and ultimately switch to benthivorous or piscivorous fish. This food web provides the basis for a modeling effort in that it serves as a guide for the transfer of energy and mass through the system to each of the components. These components make useful compartments for the model.

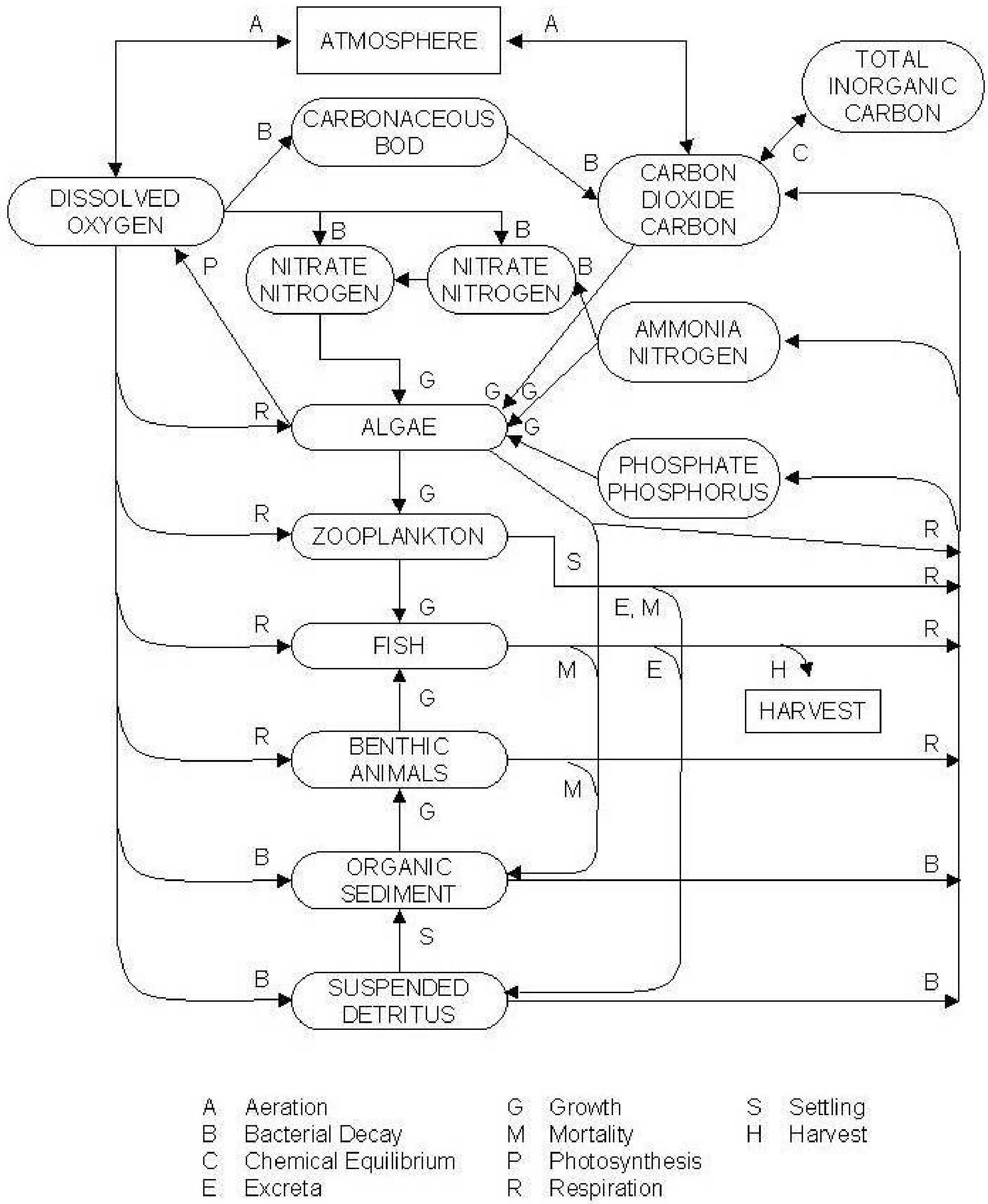

Figure 5.4-3. Quality and Ecologic compartments in WQRRS

The nutrient and physical characteristics that will ultimately determine the primary productivity of the system are represented by the major groups of nutrients essential to plant growth, including both the nitrogen and phosphorus series. Silica, which can, at times, be the limiting nutrient to the growth of diatoms (Wetzel, 1983) was not simulated after a cursory review of historical data suggested that it was not limiting in the existing impounded system. The relative importance of planktonic diatoms to total primary productivity (including benthic productivity) per river mile in the normative river is anticipated to be much lower than in the impounded river, so future silica limitation is not anticipated, and silica was not modeled. Temperature, which forms the basis for many of the rate processes, was modeled using the head budget method. Dissolved oxygen and related oxygen demand was tracked, but was not emphasized in the model due to the extensive reaeration potential expected for the restored system. Biomass and organic sediment was modeled and used as the basis for the flow of mass between the trophic levels in the biota. Detritus was modeled and is incorporated into aquatic insects, macroinvertebrates, benthic animals (crayfish, amphipods, clams, snails, and oligochaetes), and benthivorous fishes.

Nutrients in the model are incorporated directly from the water into the phytoplankton, attached benthic algae, and macrophytes. No compartment was used for the simulation of macrophytes in WQRRS. Macrophytes were included with filamentous algae as one benthic algal compartment (Benthic Algae 1). This can be justified in that WQRRS simulates the growth of attached benthic algae independent of substrate. If light, nutrients, and water velocities are sufficient for benthic algal growth, benthic algae will grow. Some substrate (hard) will be suitable for the growth of attached benthic algae, while other substrate (soft) will be suitable for the growth of macrophytes. The second benthic algae compartment (Benthic Algae 2) is assumed to be unicellular taxa (primarily diatoms) based on community composition data collected in the free-flowing section of the Snake River at Asotin. Phytoplankton are represented in the model for the normative river for comparison to data from the existing systems, but are not expected to be a major component of the primary production. All phytoplankton were combined into one compartment in the model. The phytoplankton compartment is anticipated to be comprised primarily of diatoms. The two attached benthic algae types are preyed upon by macroinvertebrates, benthic animals, and benthivorous fishes, or are lost to scour and decomposition. The scour rate was increased later in the season to simulate losses associated with reduced adherence to substrate by the benthic algae community as light and water temperature become less favorable for growth in the fall.

Zooplankton are expected to be a minor component of the aquatic community in the normative river, due to the anticipated increased water velocities. The contribution of zooplankton to the biomass of secondary producers is anticipated to be minor, so it was not included in the simulations.

The macroinvertebrates were included as one compartment in the model (Aquatic Insects), while the benthic animals - including oligochaetes, crayfish, amphipods, molluscs, and isopods were combined into another group (Benthic Animals). The macroinvertebrate community composition of the lower Snake River under the normative condition cannot be accurately determined at this time. Few data exist on the composition of the 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 Snake River in Hells Canyon. The macroinvertebrate community observed during that study was assumed to be representative of what could be expected in the lower Snake River after removal of the dams. This study indicated that the macroinvertebrate community was composed of Tricoptera (primarily Hydropsyche) and Dipterans (primarily chironomids), and with Ephemiroptera and Lepidoptera also present (Brusven *et al.*, 1973). Macroinvertebrate emergence rates were set based on the assumption that the entire macroinvertebrate community went through one generation during each growing season. Emergence was equally aportioned among months during the growing season. Few quantitative data exist that would suggest the magnitude of the benthic animal community of the lower Snake River under normative conditions. The benthic animal compartment in the model was set such that other compartments in the model that were consumed by or consumed benthic animals were balanced with benthic animals, and predicted biomass of the compartments within literature ranges. Limited data from the Snake River and Clearwater River (Wilson, 1953) in 1950 and 1951 indicate that the biomass values selected for the model are comparable to field values. Macroinvertebrates and benthic animals in the model feed on attached benthic algae and organic sediment. Both groups are, in turn, preyed upon by benthivorous fish and both piscivorous fish groups.

The fish community was divided into three groups: two groups of cool water piscivores (Fish 1 and Fish 2) and planktivores/benthivores (Fish 3). The primary fish species generally represented in the first cool water piscivore group (Fish 1) includes all of the fish-eating life stages of the salmonid species in the lower Snake River. Functionally, this will be a very small group. The second cool water piscivore (Fish 2) includes smallmouth bass, northern pikeminnow, and channel catfish. The benthivorous fish group includes bridgelip and largescale suckers, white sturgeon, and a variety of centrarchids, sculpins, and cyprinids. Planktivory in the system is anticipated to be much lower than benthivory, due tot he virtual absence of zooplankton in the system. This was represented by selecting feeding preference coefficients that favored macroinvertebrates and benthic animals over zooplankton. The mortality of the fish includes death due to disease, angling, piscivory, and migration out of the system. The piscivorous fish groups in the model feed on the benthivorous fish group, macroinvertebrates, and benthic animals. The benthivorous fish group feeds on benthic algae, macroinvertebrates, benthic animals, and detritus.

Snake River System

The WQRRS model was applied to the lower Snake River system from its confluence with the Columbia River to the upstream end of the Lower Granite pool at River Mile 146.5. The model considers inputs from the Snake, Clearwater, Tucannon, and Palouse Rivers. On a mean annual basis, inputs from the Snake and Clearwater Rivers account for over 98% of the total flow in the study reach. The tributary inputs from the Tucannon and Palouse Rivers were volume-weighted into a single tributary so that the model could simulate a period of 200 days. Due to model limitations, keeping the two tributaries separate would have resulted in reducing the simulation period to 167 days.

On the average, the flow in the Palouse River was about 3.7 times larger than the Tucannon River, but together the Palouse and Tucannon Rivers only contribute about 1.5% of the total flow. Although point sources were originally considered, they were not included as input to the final model because their overall contribution to the mass budget was small. Discharges from the treatment plants at Lewiston, Clarkston, and Potlatch were combined in the Clearwater and lower Snake River flows as a part of the upstream inflow to modeled sections of the lower Snake River.

The model was also divided into five separate, but linked, models corresponding to the existing four lower Snake River impoundments and the reach from Ice Harbor Dam downstream to the confluence of the Columbia River. This will allow future investigations to evaluate removing the existing dams separately or in various combinations. Due to model limitations on the number of reaches and segments within a reach, it will also allow more detail in the representation of the habitat of the normative system.

Model simulations for each year (1994, 1995, and 1997) were started on May 1, and extended through October 31 to encompass the majority of the growing season on the lower Snake River.

5.4.1.1 - Model Assumptions

All models are, by definition, simplified representations of the actual system. One of the benefits of this simplified representation is that the model can be manipulated for less cost and in a shorter time than a more complicated model. One of the costs associated with this benefit is that a number of assumptions are used to simplify the real system, and these assumptions impose limitations on the use and interpretation of the model results.

The major assumptions and limitations of the version of WQRRS used in this study include the following:

1. **Normative River is at Steady-State**
   The model constructed for the lower Snake River assumes that all physical, chemical, and biological transitionary processes associated with breaching the dams have ended, and the river is in a normative state with physical characteristics similar to those existing in the river in the mid-1930's. The transitionary processes may take 5 to 10 years or more to establish biological communities that are in a steady state condition with respect to their new normative environment. The model was not developed to investigate the transitional state immediately following the breaching of the dams.

2. **One-Dimensional Assumption**
   The WQRRS assumes a river can be represented by a linear (1-D) system of homogeneous elements (figure 5.4-4). The model limitations associated with this assumption are:

   - Vertical and lateral variations in water quality constituents cannot be predicted.
   - All inflow quantities and constituents entering an element are instantaneously dispersed throughout that element.

3. **Conservation of Mass Assumption**
   The model assumes that the dynamics of each physical, chemical, and biological component can be described by the principle of conservation of mass. The model considers:

   - Mass added by inflows;
   - Mass removed by outflows;
   - Diffusion of mass; and
   - Internal changes due to growth, settling, respiration, grazing, decay, *etc.*

   Limitations are imposed by this assumption because masses cannot be truly balanced. In particular, complex chemical and biological processes are represented by equations incorporating simplified kinetic formulations.

4. **Simplified Ecological Representations**
   Ecological relationships and interactions, by necessity, must be simplified. Several hundred or more biological species exist in nearly all ecosystems and it is virtually impossible to obtain information on all of them. These species must then be aggregated into functional groupings for representation in the model. This aggregation is severe Cone compartment each for all zooplankton, aquatic insects, and non-insect benthic macroinvertebrate species; one compartment for all phytoplankton; two compartments for benthic algae species; and three compartments for selected fish species.
   The model, therefore, cannot:
   Consider competition between individual species within one compartment
   Consider complex biological pathways and formulations; or
   Predict precise numbers of biomass of any individual species.

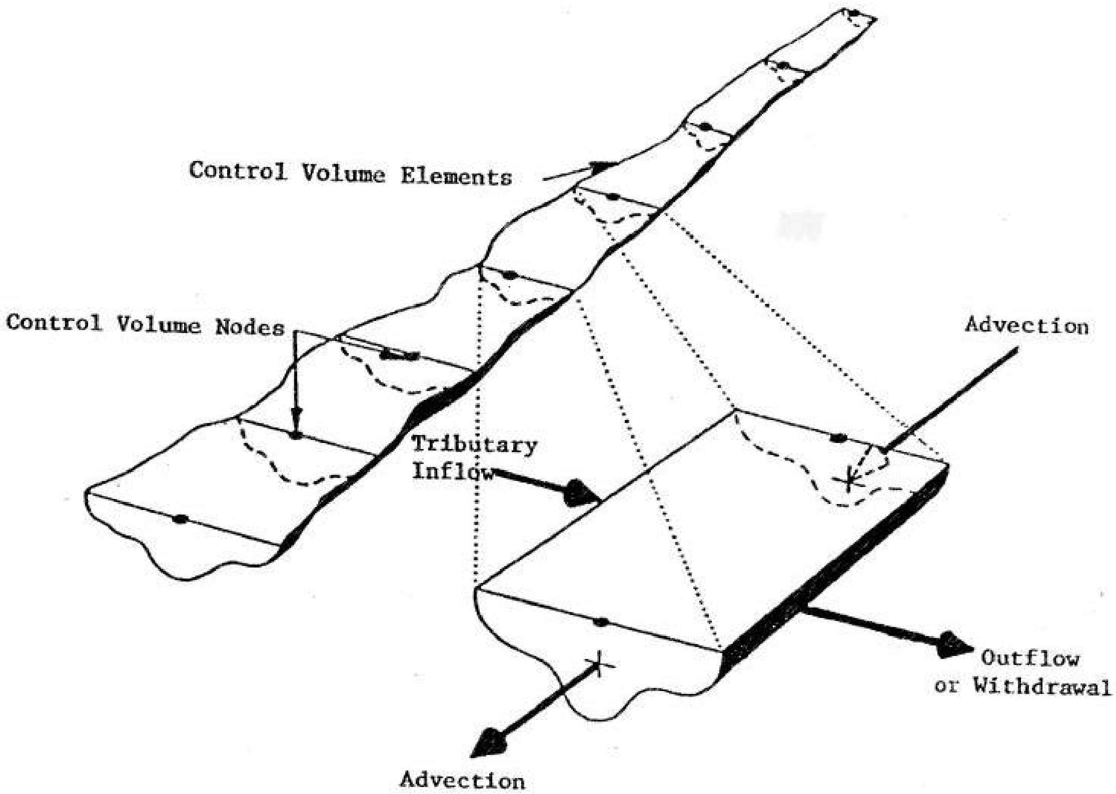

Figure 5.4-4. Geometric Representation of Stream System and Mass Transport Mechanisms

5.4.1.2 - State Variables and Methods of Estimation/Relationships

The project team has compiled a list of the major state variables to be implemented in this project application of WQRRS to the lower Snake River. These variables are presented in Table 5.4-1. Note that this list does not represent the full complement of state variables available for WQRRS; only those variables deemed essential relative to the objectives identified for this project were implemented. Data on nutrient concentrations, primary productivity estimates, light, temperature, dissolved oxygen, detritus, algal enumeration (*e.g.*, chlorophyll *a*), macrophytes, and various faunal groups (*e.g.*, fish, benthos) from previous field sampling efforts on the lower Snake River and elsewhere were used to calibrate the model relative to the chosen list of state variables for this modeling effort. In many instances, data were limited or did not exist. Best professional judgement was used to evaluate the model results in those instances. Existing sampling data were compared to model estimates of the biomass and secondary productivity of higher trophic levels such as benthos and fish. Additional estimates of these state variables, such as fish biomass, were made for comparative purposes using alternative means such as simple empirical indices. In a similar manner, the various index functions to be developed in other tasks of this project for estimating primary productivity were compared with model projections of this process. Reasonable agreement among these various estimates of similar state variables and process rates added further confidence to model predictions. The lower Snake River system poses a unique challenge to model estimates of secondary productivity, in that it is not a biologically-closed system. Several migratory species of fish grow within the system, leave and grow outside the system for a period of time, and then return to the system with additional biomass. These special circumstances were evaluated and eliminated from the modeling effort as an insignificant component of overall system secondary productivity because the biomass of these fish species relative to resident fish species was small.

Table 5.4-1
Major State Variables Simulated in WQRRS for the Lower Snake River

Parameter	Notes
Temperature	Heat Budget Method
Phytoplankton (1 functional group, included in model)	Biomass and Productivity (Gross and Net)
Benthic Algae (2 functional groups, filamentous plus macrophytes and unicellular	Biomass and Productivity (Gross and Net)
Dissolved Nutrients	Orthophosphorus, nitrate, ammonia
Dissolved Oxygen	
Macrophytes (1 functional group, represented by benthic algae type 2)	Biomass and Productivity (Gross and Net)
Benthic Animals	Biomass and Net Productivity
Aquatic Insects	Biomass and Net Productivity
Fish (3 functional groups)	Biomass and Net Productivity
Detritus	
Organic Sediment	
Suspended Solids (inorganic, 1 type)	

The WQRRS state variables that were not implemented in model simulations of the lower Snake River include multiple suspended solids types, coliform bacteria, total inorganic carbon, alkalinity, pH, and toxic constituents. In the case of simulated algal components, different functional groups were simulated based on existing knowledge about the biological dynamics of the Snake River ecosystem (*e.g.*, diatoms and blue-greens for phytoplankton, unicellular, and filamentous for benthic algae). In the case of fish, the model was configured to simulate three distinct functional groups of fish of importance in the lower Snake River system (*e.g.*, two groups of piscivores and one group of benthic feeders).

5.4.1.3 - Model Options

The WQRRS model allows the user to select several different options. Six different methods for the hydraulic computations in the Stream Hydraulics Package (SHP) were used. Four of the six methods are for unsteady flow; these are St. Venant, kinematic wave, Muskingum, and Modified Puls. The Modified Puls method was selected because it provides a stable solution even with higher velocities expected in the normative system, and it is relatively simple compared to the other methods. The St. Venant method was not selected because it would be unstable with the higher velocities expected in the normative river. The kinematic wave method was not selected because it is not included in the PC version of the SHP model.

For simulating water temperature, the user can select the equilibrium temperature approach or the full heat budget. For this project, the full heat budget option was selected because it does not include the linear approximations that are incorporated into the equilibrium temperature approach. Also, selecting the equilibrium temperature approach requires the user to apply a separate model to calculate solar radiation values and surface heat exchange coefficient values.

Model Modifications

In order to get the WQRRS model to simulate the lower Snake River system, a few modifications to the model were required. Several of these were minor code modifications such as checks to insure that constituent concentrations do not drop below zero or cause division by zero errors.

The major modifications were to the fisheries and benthic animal components. Fish 1 was modified so that it could eat Fish 2 and Fish 3. Fish 2 was modified so it could eat Fish 3. Recruitment from Fish 3 to Fish 1 was also included. The benthic animal component was modified so that it would also eat Benthic Algae 1 and 2. The benthic algae scour rate was modified so that it could be varied throughout the growing season.

5.4.1.4 - Study Year Selection

We selected 1994, 1995, and 1997 as study years based on hydrometeorological considerations and the availability of water quality monitoring data to run the model. We wanted to simulate wet, dry, and average hydrologic conditions to band the variability in predictions due to differences in hydrometeorology. Based on mean annual flows measured in the Clearwater River at Spaulding, and the Snake River at Anatone, 1997 ranked as the highest flow year of record, 1994 ranked near the lowest 10% of flows, and 1995 was slightly wetter than normal. Figure 5.4-5 compares the mean monthly flows with the period of record mean for both rivers. Water year 1997 was abnormally wet all year. Water year 1994 was dry during the May to June snowmelt period and during the August to October low-flow period. Water year 1995 followed the historical mean monthly flow pattern well except in August and September when flows were wetter than normal.

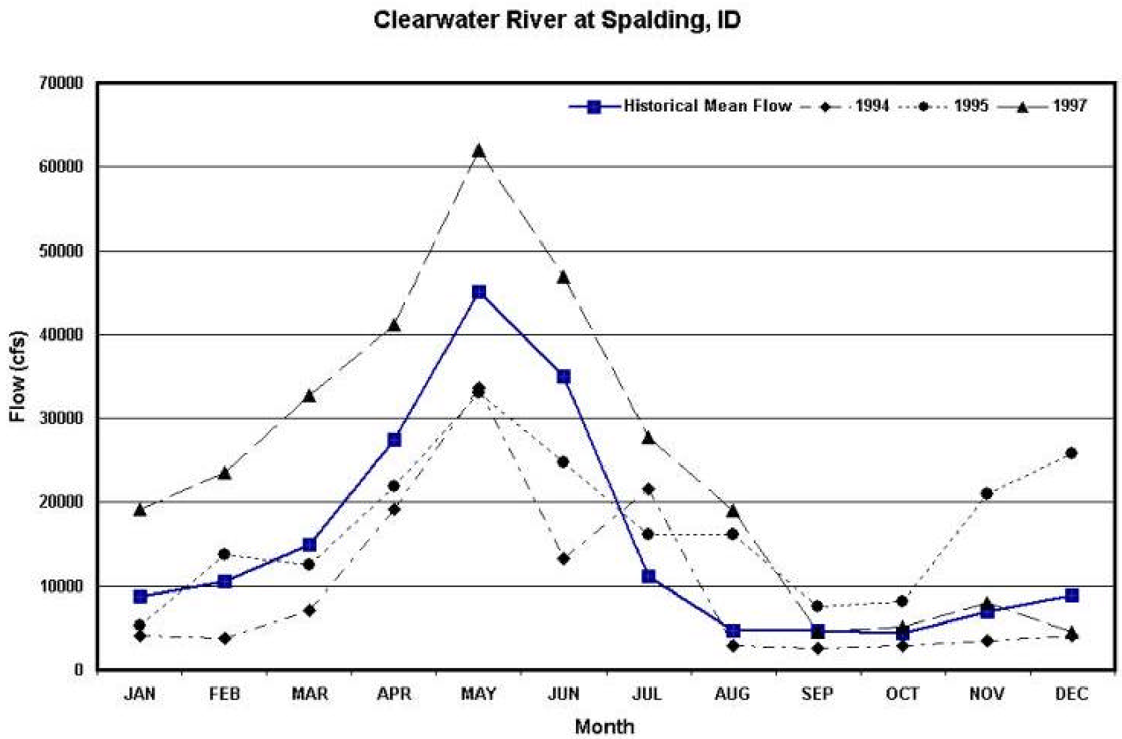

Figure 5.4-5. Combined Snake and Clearwater River flows

Mean monthly air temperatures from Lewiston for the study years and period of record are shown in Figure 5.4-6. The average flow year, 1995, was warmer than normal during April and May, and slightly cooler than normal in August. The dry year, 1994, was warmer than normal during the summer months of July through September. The wet year, 1997, was cooler than normal during June and July, and near normal throughout the rest of the year.

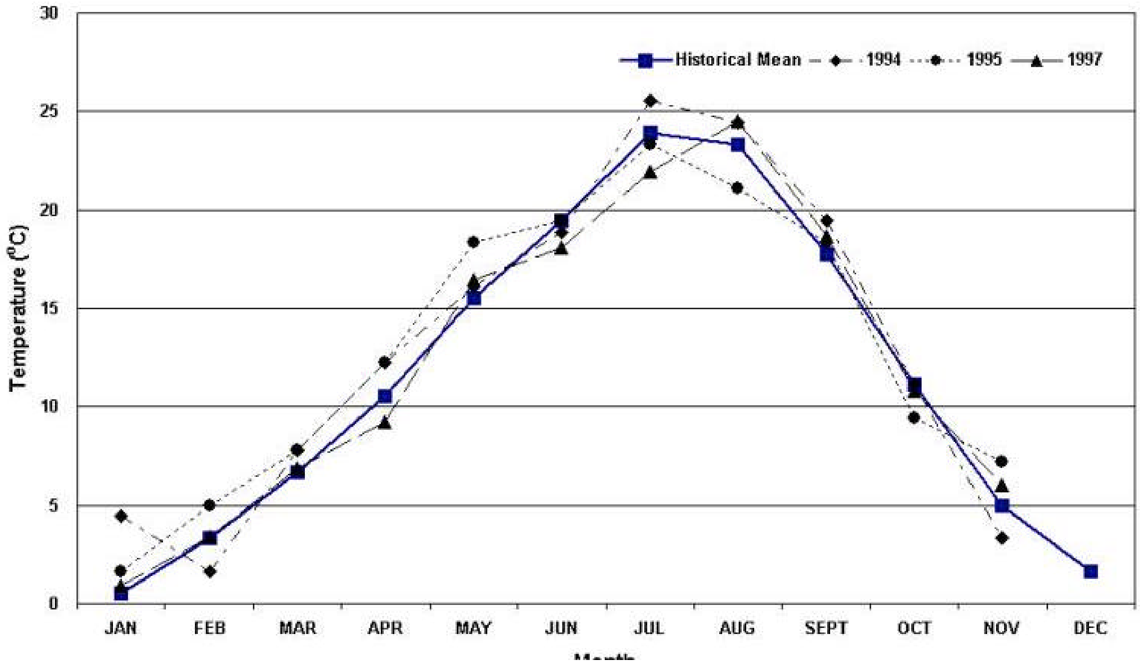

Figure 5.4-6. Lewiston Nez Perce Station monthly mean air temperature

Data from 1956 to 1958 were used to calibrate the temperature model since that period precedes the construction and closure of the four dams on the lower Snake River. The selection of these data is described in more detail in the temperature calibration section. These years tended to be slightly wetter than the average year from the period of record.

5.4.2 - Model Input

Model Configuration

Model inputs consist of geometry, upstream boundary conditions, and model coefficients that can be calibrated. Listing of the calibrated input files are include in Appendix M.

The Stream Hydraulics Package (SHP) model was applied to the lower Snake River from RM 146.5 to the mouth (Table 5.4-2). Because SHP is limited to 41 cross sections and 10 reaches, we decided to simulate the river with multiple SHP models with outflow from one model being used as inflow to the next model downstream. The main advantage of dividing the river into multiple models was to allow a more detailed representation of the river (*i.e.*, more cross sections can be used and the river can be divided into more reaches). The river was simulated with five models representing each of the existing reservoirs, as shown below:

Table 5.4-2 Cross-Sectional Representation of the Lower Snake River			
Model	**River Miles**	**Number of Cross Sections**	**Number of Reaches**
Lower Granite	RM 146.5 to 107.5	40	9
Little Goose	RM 107.5 to 70.3	41	10
Lower Monumental	RM 70.3 to 41.6	33	9
Ice Harbor	RM 41.6 to 9.7	41	9
McNary	RM 9.7 to 0.0	22	1

Each model was divided into multiple reaches to allow pools and rapids to be represented separately in the water quality modeling. Based on the linens showing the 1934 bathymetry, there were 59 different rapids between river mile 146.5 and the mouth. Representation of each individual rapids as a separate reach would require 59 rapids' reaches plus 59 more reaches to represent the pools in between the rapids. This level of detail would require 12 separate SHP models, and was not considered to be consistent with the objectives of the SHP modeling. Therefore, the rapids and pools were aggregated within each model. For example, reach 4 of the Lower Granite model is 1.3 miles long, and represents an aggregation of three individual rapids with lengths of 0.2 miles, 0.7 miles, and 0.4 miles, respectively. The reach lengths were set so that the total lengths of rapids and pools in each SHP model would be the same as from the 1934 bathymetry. The overall bed slope of the river as it appeared on the 1934 mapping was also maintained.

Sources of Bathymetric Data

Several sources of bathymetric data were available for the SHP modeling of the lower Snake River. One source of very detailed bathymetric data were 1934 linens obtained from the USACE, Walla Walla District. These linens consist of maps showing transects of water depths at low flow under pre-impoundment conditions. The transects were spaced approximately 200 to 300 feet apart along the Snake River from the mouth to RM 175.

Limnological Data

Daily minimum, maximum, and average water temperatures were available for the three study years (1994, 1995, and 1997) from the USGS monitoring stations on the Snake River at Anatone, Clearwater River at Spaulding, and Palouse River at Hooper. These average daily water temperatures were used as model input for the simulations. Water temperatures for the Tucannon River were not available for the study years and, therefore, were assumed to be the same as the Palouse River. A comparison of monthly temperature measurements by the State of Washington Department of Ecology (WDE) for both rivers for the study years indicates the temperatures are similar and within about 2EC. The Palouse River temperatures tended to be warmer than the lower Snake River temperatures during the summer months by up to 4EC. Comparison of temperature measurements by the USGS and the WDE on the Palouse River at Hooper for the same days also showed variations of +/- 1 °C.

Water quality measurements for various constituents required in the model were available from Station 148 on the Snake River and Station 11 on the Clearwater River on eight occasions in 1997. These stations were located upstream of any backwater influences. For study years 1994 and 1995, data were only available from Station 140 on the Snake and Station 1 on the Clearwater River on nine occasions each year. Although these stations are located in the backwater area of Lower Granite impoundment, they represent the best data available, and were used. The use of these data is discussed in the following paragraphs. In general, linear interpolation was used between data points to generate daily input values for all parameters. As stated earlier in this section, the inputs from the Palouse and Tucannon Rivers were volume-weighted into a single tributary that enters at mile 59.4. Data for these inputs were obtained form the 1997 database. For 1994 and 1995, data were obtained from the State of Washington Department of Ecology studies through the EPA STORET database.

Dissolved oxygen values were entered as percent saturation based on measured data for the three study years. Daily values were obtained by linearly interpolating between measured values. Measured dissolved oxygen values were consistently above 95% saturation, and often greater than 100% saturation. Since the WQRRS model formulation for reaeration (*i.e.*, O'Connor-Dobbins Churchill, *etc.*) will quickly dissipate oxygen supersaturation and, since the objective of this study was not related to supersaturation, dissolved oxygen input values were assumed not to exceed 100% saturation.

Since dissolved oxygen concentrations are nearly always near saturation in the existing lower Snake River system, although there are periods of time when dissolved oxygen is below saturation, we assumed that oxygen will remain high in the normative system, which will have significantly more reaeration. Therefore, BOD was not simulated. Based on historical USGS values from STORET and the 1997 field program, BOD values were low (*i.e.*, 2 mg/L or less), and should not have a significant effect on model results.

Nutrient inputs of phosphorus and nitrogen were also obtained from the 1994, 1995, and 1997 water quality surveys and linearly interpolated to obtain daily values. Since WQRRS assumes that nutrients are 100% available for uptake by algae, orthophosphorus values were used as phosphorus inputs rather than total phosphorus values. For nitrogen, nitrate and ammonia values were used.

The biological communities in the system were assumed to be in steady state, with inputs equaling outputs and the local populations being controlled by location conditions (including habitat). Therefore, no upstream inputs for any of the biological communities were included in the model except for algae type 1 (phytoplankton as diatoms). These inputs were assumed to be constant at 0.5 mg/L for the Snake and Palouse/Tucannon Rivers and 0.2 mg/L for the Clearwater River, based on chlorophyll *a* measurement sin 1997. Other organic inputs were assumed to be in the form of detritus, which was estimated from TOC measurements at the upstream monitoring stations. Because TOC data were not available for the Snake or Clearwater Rivers for the study years, historical data were used. Historical TOC data on the Snake and Clearwater Rivers show little seasonal variability for the simulation period of May through October. Therefore, a constant value was assumed. The historical data indicates a median TOC value for the simulation period of about 3.9 mg/L for the lower Snake River and 3 mg/L for the Clearwater River. Higher TOC values were observed during the winter months. For the Palouse River, measured TOC values during the study years showed slight seasonal variation, with a median value of approximately 2.0 mg/L. The TOC data were not available for the Tucannon River. Therefore, the median TOC value of 2.0 mg/L was assumed for the combined Palouse/Tucannon tributary. The TOC values were converted to detritus biomass input by assuming the detritus was 40% carbon, the default value used in WQRRS.

Initial biological conditions for each modeled impoundment were required at the beginning of each simulation (May 1). There is little biological data available, particularly at the beginning of the simulations. An initial phytoplankton concentration of 0.33 mg/L was estimated from data collected at Snake River Mile 148 during June 1997. Benthic algae was estimated based on data collected later in the season during 1997 at the same station. An initial biomass of benthic algae of 1,000 mg/m^2 as dry weight was used for both Type 1 and 2. Virtually no data exist for aquatic insects or benthic animals, so best professional judgement was applied along with recommendations from individuals familiar with the system. Aquatic insect data were taken from a study at Hell's Canyon (Brusven *et al.*, 1973) and converted to biomass as dry weight using coefficients from Jorgensen (1979). Initial biomass of insects and benthic animals was estimated to be 1,444 mg/m^2 and 2,888 mg/m^2 as dry weight, respectively. Initial fish estimates were based on data reported by Bennett (1998). Wet weights of fish were converted to dry weight, as required by WQRRS, assuming dry weights are 20% of wet weights (Likens, 1985). Initial estimates for fish 1 are 6 kg/km; fish 2, 67 kg/km; and fish 3, 210 kg/km as dry weight.

One type of inorganic suspended solid was simulated to include the effects of suspended solids on light penetration. Historical data showed direct relationships between suspended solids concentrations and flow on the Clearwater and Snake Rivers. A linear regression was developed for the Clearwater River, and a piecewise linear regression was developed for the Snake River, in both cases, some higher outliers were eliminated from the analysis. The objective of the regression analyses was to develop a relationship that mirrored the general trends of the data. These linear regressions were used to estimate daily values of inorganic suspended solids concentrations for input to the model. For the Palouse/Tucannon Rivers, no obvious relationship existed between flow and suspended solids concentrations. Therefore, linear interpolation between measured data points was used.

Hydrologic Data

Input values for stream flow for the three study years were taken from mean daily flow data for the Snake River near Anatone (USGS Gage No. 13334300), the Clearwater River at Spaulding (USGS Gage No. 13342500), the Palouse River at Hooper (USGS Gage No. 13351000), and the Tucannon River near Starbuck (USGS Gage No. 13344500). For periods when flow data were missing for the Tucannon River, those data were estimated based on median ratios between monthly flows for the Palouse and Tucannon Rivers. Flow data for the Palouse and Tucannon Rivers were added together and input to the model as one tributary (as discussed earlier).

For the pre-impoundment calibration years (1956B58), input values for streamflow consisted of mean daily values for the same USGS gages except the Snake River near Anatone. Because flow data were not measured at the Anatone gage prior to July 1958, flow data for the Snake River near Clarkston (USGS Gage No. 13343500) were used. This latter gage was located approximately 6 miles downstream of the confluence with the Clearwater River, therefore daily flows for the Snake River upstream of the Clearwater River were estimated by subtracting the Clearwater River flows from the Snake River flows at Clarkston. Because no flow data existed for the Tucannon River during 1956 through 1958, daily flows for the Tucannon River were estimated in the same manner as for the study years (1994, 1995, and 1997). Flow data for the Palouse and Tucannon Rivers were added together and input to the model as one tributary (as discussed earlier).

Meteorologic Data

The WQRRS model requires meteorological inputs of air temperature, dew point temperature, wind speed, cloud cover, and atmospheric pressure to compute the head budget at the air-water interface. These data were available from the National Weather Service monitoring stations at Pasco and Lewiston. Pasco is located near the confluence of the Snake and Columbia Rivers, and Lewiston is located near the confluence of the Snake and Clearwater Rivers at the upper end of the study reach (Figure 3.3-1). A comparison of the mean monthly air temperature and wind speed for the simulation months (May B October) indicates air temperatures are approximately 4EF warmer at Pasco compared to Lewiston. Wind speeds were also about 2 mpg faster at Pasco. To determine the most appropriate data to use, surface (<3 m) water temperatures from the University of Idaho recorders for 1997 for the four existing impoundments were compared (Table 5.4-3). These data suggest that Ice Harbor and Lower Monumental have similar temperatures and, therefore, were exposed to similar meteorological conditions. Little Goose and Lower Granite also had similar but slightly cooler temperatures, indicating they may be more appropriately modeled by using data from Lewiston. However, air temperature data collected by Washington State University at the dam sites of Ice Harbor and Little Goose and the Lewiston Resource Center indicate the micro-climate in the Snake River canyon was better approximated using data from Pasco. Mean daily values were used to investigate seasonal trends.

Table 5.4-3
Comparison of Water Surface Temperatures (1 °F)
From University of Idaho Record Data (Bennett, Unpublished Data)

1997	Lower Granite Site 5	Little Goose Site 8	Lower Monumental Site 11	Ice Harbor Site 14
May	51.6	51.2	52.3	52.4
Jun	57.4	56.8	57.6	57.7
July	64.8	64.7	65.7	65.9
August	67.6	67.5	68.6	69.0
September	69.2	69.1	69.6	68.2

5.4.3 - Calibration

The purpose of model calibration is to refine estimates of certain coefficients and parameters so the model can reproduce observed data over a wide range of environmental conditions. The model is then tested (confirmed) using the same coefficient values with a different set of observed data. The emphasis when calibrating a model is to ensure that temporal and spatial trends in processes such as temperature and nutrient cycling, and algal growth are adequately represented rather than trying to match individual data points. The latter approach can result in the generation of unrealistic coefficient values and parameter estimates, which may not be valid for conditions beyond the calibration simulations.

Because limited water quality data were available from the period prior to impoundment of the lower Snake River, the water quality and biological components of the model were calibrated based on first principles. This means that the model input was refined so that the output reproduced general patterns and long-term averages of observed data or knowledge.

The calibration process followed in this study included several steps. First, the river geometry was specified. The water budget was checked by comparing computed and observed flows over time to ensure conservation of mass. Temperature was calibrated to reflect hydraulics. Temperature calibration was followed by calibration of the water quality parameters. The temperature and water quality parameters were calibrated by comparing observed data with predicted model output for each variable, and then modifying the coefficients to minimize differences. This procedures was not rigorously followed for some water quality constituents, because of data limitations. Finally, the biological processes and components were added and compared qualitatively to available data using first principles. The biological compartments were added one compartment at a time, keeping the higher trophic levels at constant levels. Initially, benthic algae was calibrated to match expected levels and seasonal patterns by keeping aquatic insects and fish constant. Aquatic insects and benthic animals were then calibrated, followed by fish.

Simulations for model verification/confirmation were run for a single year, by comparing the predicted concentrations with actual data from Asotin in the free-flowing reach just upstream of the modeled section of river. Actual biological data exist only for benthic algae. The model was applied to the future free-flowing condition under the assumption that all modeled physical and biological processes have been adequately estimated or calibrated based on available data and information on the existing system, preimpoundment conditions, or from similar systems. Each simulation spans the majority of each calendar year simulated, in order to capture all periods of significant biological productivity in the Snake river.

The WQRRS was parameterized and configured for the Snake river application in a series of stages. Initial literature estimates of rate processes and model coefficient values, together with relevant data derived or calculated from existing information on the Snake River and systems, was coupled with model hydraulics information in order to complete initial simulation predictions of all implemented model state variables (*i.e.*, Table 5.4-1). The input file is presented in Appendix A. These model predictions were compared to available field data to assess the adequacy of the simulated values. Model coefficients and rate processes were then adjusted, as appropriate, to enhance the match between model predictions and field data where field data were available. We employed various approaches for comparing model predictions with field data (*e.g.*, order-of-magnitude comparison, direct comparison with specific measured values, agreement in temporal patterns, and made every effort to compare model projections with data on both state variables and on flux rates.

Hydraulics

The hydraulics (velocity and flow depth) were calibrated by matching velocities and flow depths from the 1934 linens and HEC-2 model simulations. This information was used to develop the storage-volume relationships for the Modified Puls routing. The results were comparable. As an independent check, the flow depth from USDHEW (1964) at Mile 132 was compared with the model output at that same location for various flow rates (Table 5.4-4).

Table 5.4-4 Flow Versus Depth Comparison		
Flow (cfs)	**USDHEW (1964)**	**Model**
10,000	2.5	2.9
31,700	4.1	4.1
67,000	5.6	5.1
191,000	8.6	7.1

As can be seen from the above comparison, the flow depths match quite well at flows less than 67,000, but then diverge with the model predicting the shallower depth. This could result from assuming a constant Manning "n" with flow. These results are valid for that cross section only, and do not represent conditions throughout the entire river system.

A comparison of predicted and observed water surface elevations at flow rates of approximately 24,000 cfs and 100,000 cfs is shown in Table 5.4-5a. The observed water surface elevations were taken from plots of natural water surface profiles from the District's water control manuals for Little Goose and Ice Harbor Dams. These natural water surface profiles were based on field surveys all along the river during different flow regions in 1948 and 1956.

Table 5.4-5a Comparison of Water Surface Elevations		
Section of River	**Approximate Flow (cfs)**	**Average Difference Between Predicted and Observed Water Surface Elevations (ft)**
RM 107.5 to 70.3 (Little Goose)	24,000	-1.3
RM 41.6 to 9.7 (Ice Harbor)	24,000	0.1
RM 107.5 to 70.3 (Little Goose)	100,000	1.3
RM 41.6 to 9.7 (Ice Harbor)	100,000	4.0

This comparison shows that the model tended to overpredict depths at higher flow rates. The overprediction of depths for Ice Harbor at 100,000 cfs was assumed to be partly due to the aggregation of pools and rapids (*i.e.*, the pools and rapids did not align well longitudinally for the comparison).

Velocities predicted by the model were compared with observed velocities from the 1934 bathymetric linens, as shown in Table 5.4-5b. The velocities from the linens correspond to a stream flow rate of approximately 10,600 cfs, and were only available for the rapids sections. This comparison shows that the model was underpredicting velocities in the rapids sections at low flows. A maximum velocity of 24.8 ft/s in 1934 seems unreasonable, probably due to sampling error, but the modeled velocity appears closer to reality. No further comparison fof velocities was possible because there were not any observed velocities at higher flow rates, nor were there any observed velocities in the pools.

Table 5.4-5b
Comparison of Velocities in the Rapids

Section of River	Average Velocity (ft/sec)		Maximum Velocity (ft/sec)	
	1934 Linens	SHP Model	1934 Linens	SHP Model
RM 146.5 to 107.5 (Lower Granite)	7.4	6.8	13.9	12.9
RM 107.5 to 70.3 (Little Goose)	6.8	4.8	12.3	10.6
RM 70.3 to 41.6 (Lower Monumental)	6.9	4.5	24.8	9.9
RM 41.6 to 9.7 (Ice Harbor)	5.9	2.8	12.6	7.0

Temperature

In order to represent free-flowing conditions, data collected prior to the mid-1960's was required. Table 5.4-6 shows a listing of the stations relevant to the study area for which observed temperatures were available prior to the mid-1960's.

Table 5.4-6 Listing of Stations with Observed Temperature Data Prior to Mid-1960's				
Location	**River Mile**	**Period of Record**	**Data Frequency**	**Data Source**
Snake River at Sacajawea	(near mouth)	June through September for each year, 1955 through 1958	6-day averages	Graph of data provided by the Corps
Snake River at Central Ferry	83.2	October 1955 to July 1958	Instantaneous daily measurements	Copies of original USS data sheets provided by the Corps
Snake River near Clarkston (11343500)	132.9	October 1959 to September 1964	Daily min/max	Hydrosphere, Inc., CD (data originally from USGS)
Snake River near Anatone (13334300)	167.2	October 1959 to present	Daily min/max (prior to 1978)	Hydrosphere, Inc., CD (data originally from USGS)
Snake River at Oxbow (13290200)	269.6	October 1957 to September 1973	Daily min/max	Hydrosphere, Inc., CD (data originally from USGS)
Clearwater River at Spaulding (13342500)	11.6	October 1959 to present	Daily min/max	Hydrosphere, Inc., CD (data originally from USGS)
Clearwater River near Peck (13341050)	37.1	October 1964 to present	Daily min/max	Hydrosphere, Inc., CD (data originally from USGS)
Clearwater River at Kamiah (13339000)	68.7	July 1956 to September 1957, October 1958 to July 1959	Daily min/max	Hydrosphere, Inc., CD (data originally from USGS)

As shown in Table 5.4-6, the only pre-impoundment temperature data within the reaches to be simulated were the data collected at Sacajawea, at Central Ferry, and near Clarkston. In order to have observed temperatures to compare with predicted values, the only years that could be simulated were 1956 through 1958. However, during those years there were no data collected near the upstream end of the modeled reach that could be used as upstream boundary conditions for the model.

Several alternatives existed for addressing the absence of observed temperatures during 1956 through 1958 near the upstream end of the modeled area. One alternative was to simulate a different time period, particularly 1960 to 1964, because that is when data were collected near Clarkston (river mile 132.9, which is about 6.4 miles downstream of the Clearwater River. This alternative was not chosen because there were no observed data during 1960 through 1964 for comparing predicted and observed temperatures between the Clarkston station and the mouth of the Snake River.

A second alternative would have been to generate water temperatures for the Clarkston station for 1956 through 1958 using regression analyses of water temperature as a function of air temperatures and streamflow rates. This alternative was investigated by performing regression analyses of water temperatures at the Clarkston station during 1960 through 1964 as a function of air temperatures at Lewiston, Idaho, and streamflow rates for the Clarkston station. This regression analysis yielded a coefficient of determination (r squared) of 0.52. A visual review of predicted and measured data showed this level of accuracy was marginally acceptable.

A third alternative was to use data farther upstream to represent the boundary conditions at the confluence of the Snake and Clearwater Rivers. Observed data for parts of the 1956 through 1958 period were available for the Snake River at Oxbow (approximately 130 miles upstream of the confluence), and for the Clearwater River at Kamiah (approximately 69 miles upstream of the confluence). As shown in Figures 5.4-7 through 5.4-9, temperatures at Kamiah and at Central Ferry during 1956 through 1958 tended to be similar. However, the differences in between temperatures at Oxbow and at Central Ferry during 1956 through 1958 were more variable. Some of the water temperatures at Oxbow were cooler than those at Central Ferry, whereas some were warmer. Due to the lack of a consistent trend between the Oxbow and Central Ferry temperatures and the number of missing values at both Kamiah and Oxbow during the 1956 through 1958 period, these data sets were not selected.

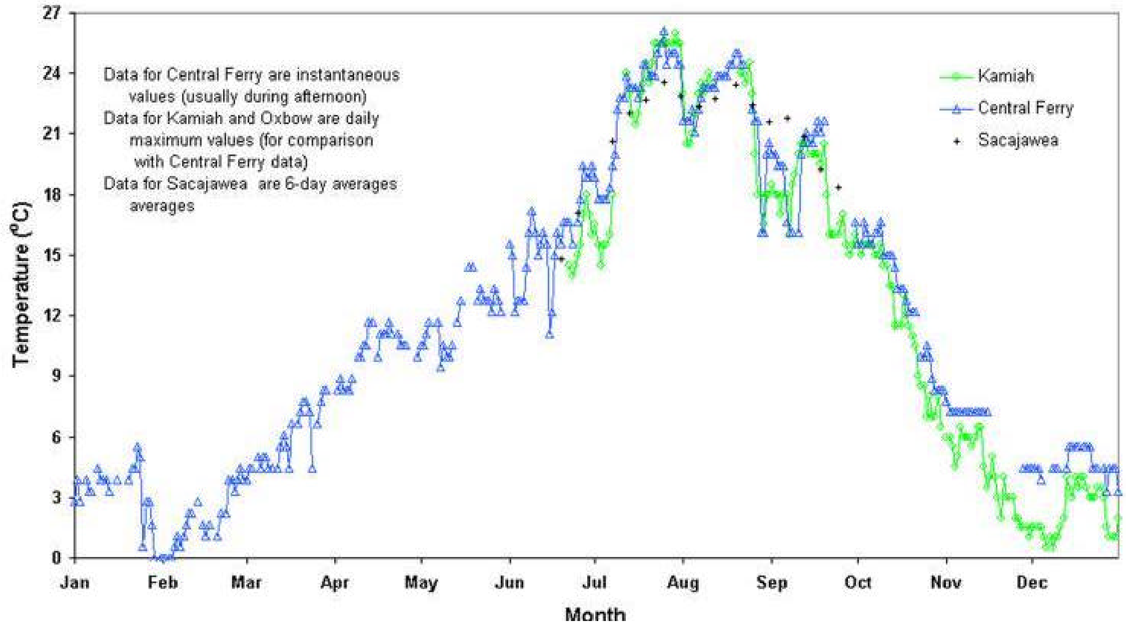

Figure 5.4-7. 1958 Snake River and Clearwater River water temperatures

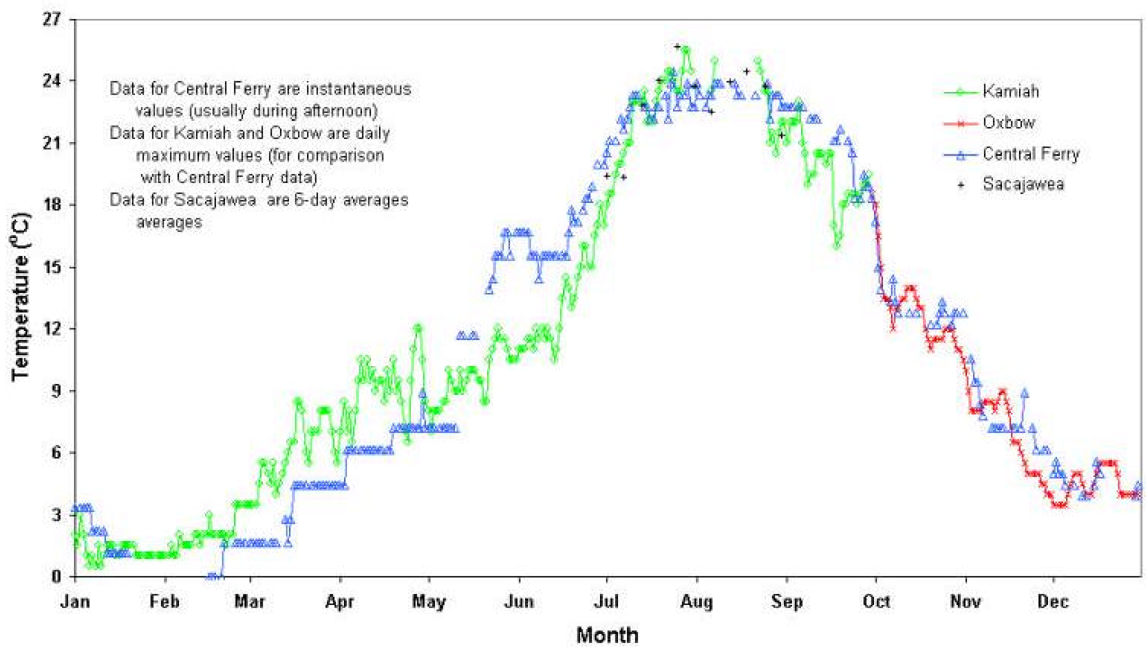

Figure 5.4-8. 1957 Snake River and Clearwater River water temperatures

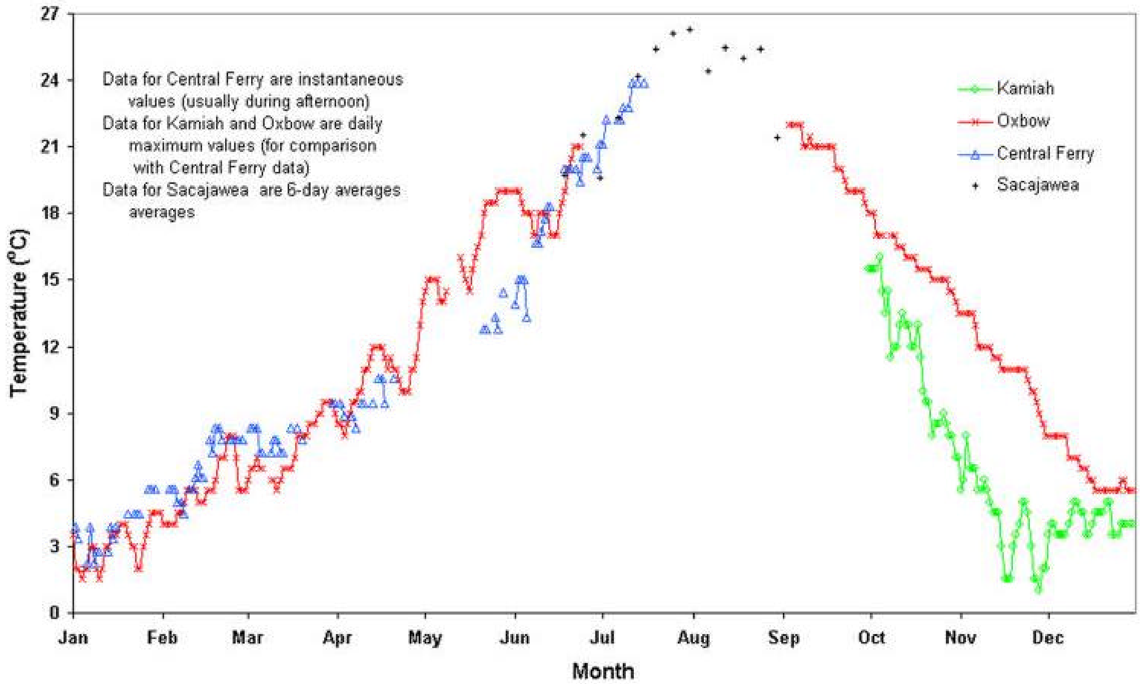

Figure 5.4-9. 1958 Snake River and Clearwater River water temperatures

A comparison of historical water temperature measured at Sacajawea, Central ferry, and upstream stations, indicates no significant longitudinal variation in water temperature within the study reach (Figures 5.4-10 through 5.4-12). Therefore, it was assumed the model should predict relatively constant water temperatures, and the measured temperatures from Central Ferry were used as upstream inputs.

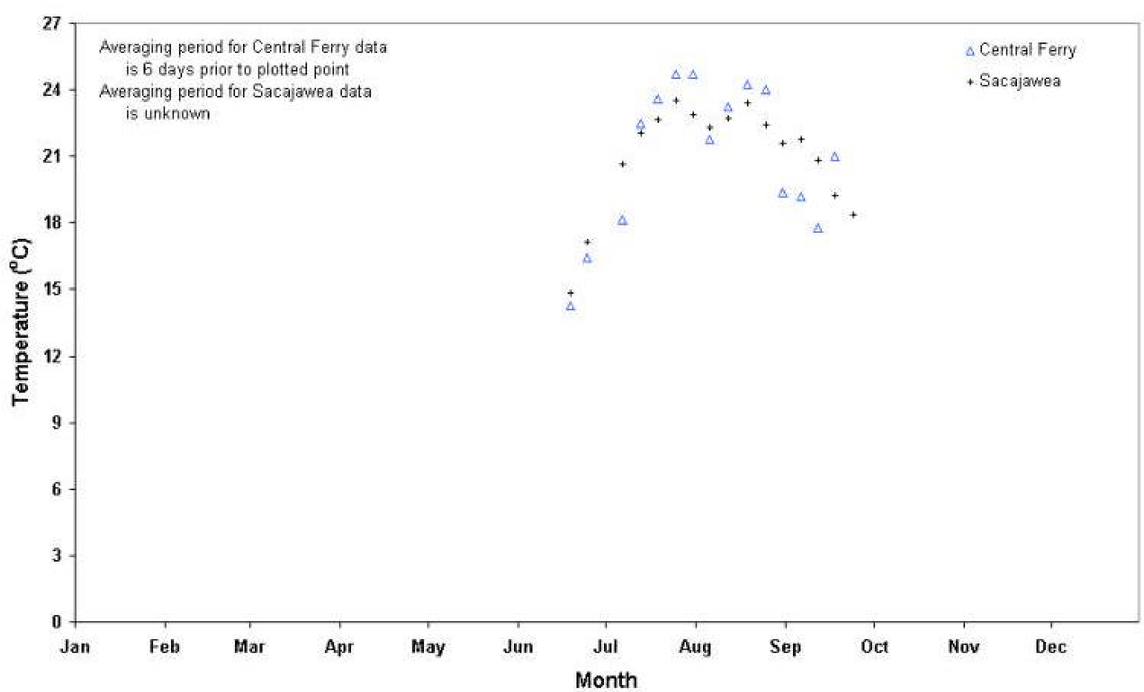

Figure 5.4-10. 1956 Snake River average water temperatures

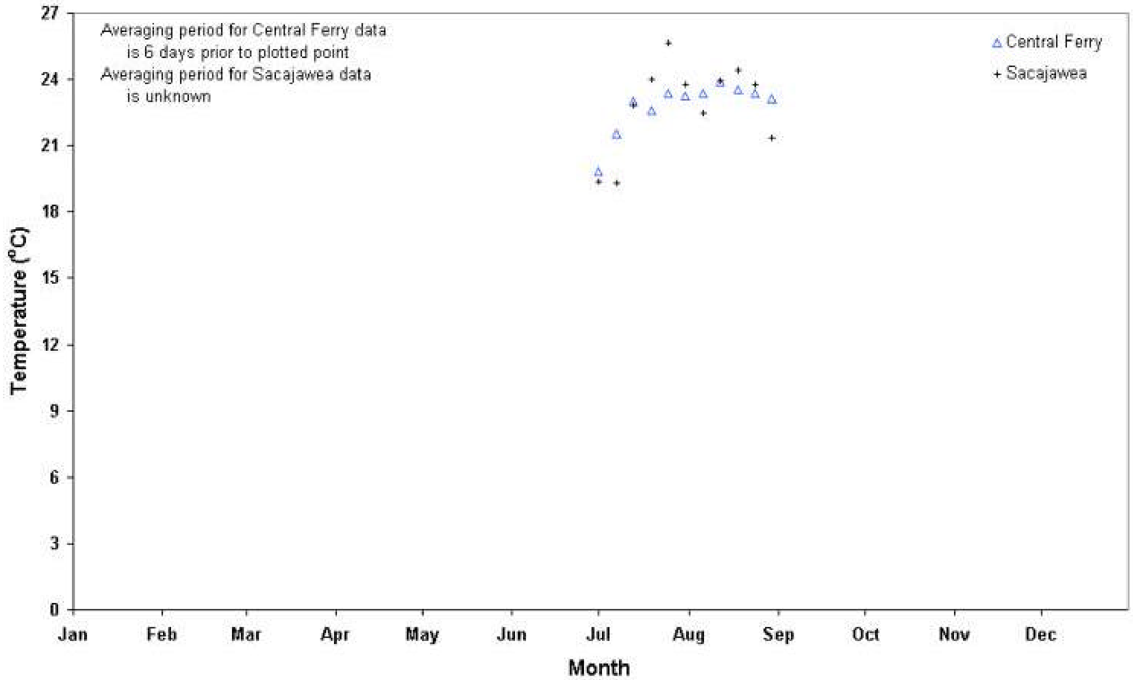

Figure 5.4-11. 1957 Snake River average water temperatures

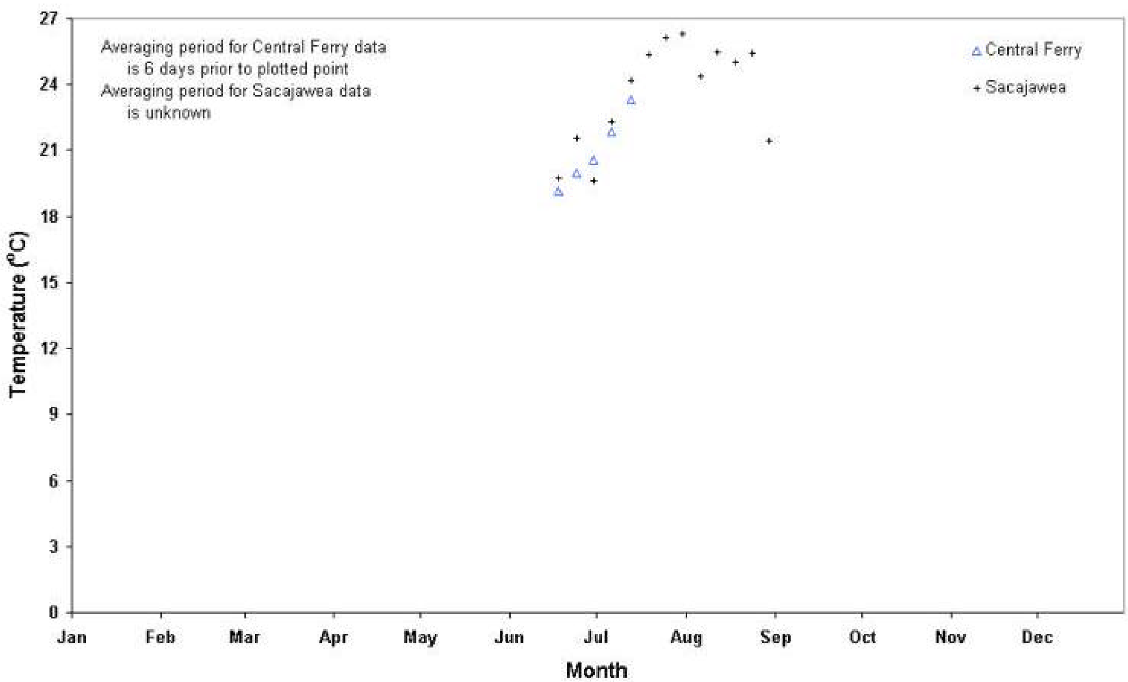

Figure 5.4-12. 1958 Snake River average water temperatures

Data from 1956 to 1958 were first used to calibrate the temperature model. Predicted temperatures were compared with measured temperatures at Central Ferry and Sacajawea. The Sacajawea temperatures were 6-day averages and, therefore, exhibited less variability than the simulated daily numbers. The initial simulations used meteorological data from both Lewiston and Pasco. The Lewiston data were used for the two upstream sections (Lower Granite and Little Goose), and the Pasco data were used for the lower three sections (Lower Monumental, Ice Harbor, and McNary). The initial predictions were good at Central Ferry, but tended to be slightly warm and exhibited too much variability at Sacajawea. In addition, plots of longitudinal temperature variations showed a discontinuity in temperature at the point where the meteorological data changed. Model simulations were then made using only Lewiston meteorological data and only Pasco data. Simulations using the Lewiston data matched the measured temperatures at Central Ferry, but predictions were slightly cool at Sacajawea. In contrast, use of only Pasco meteorological data resulted in temperatures that were too warm at both Central Ferry and Sacajawea. In one instance, temperatures approached 30EC, which was considered unreasonably high. Model simulations were then made using Lewiston data for the upstream two sections, and an average of Lewiston and Pasco data for the lower three sections. This combination of meteorological data provided the best overall simulation, and was used for all remaining simulations. Next, a series of sensitivity simulations was conducted by varying upstream temperature inputs and model coefficients to fine-tune the calibration. The results of the sensitivity analysis are provided in the next section.

The final temperature model calibrations are shown on Figures 5.4-13 through 5.4-18 for 9156, 1957, and 1958. Dates on the figures are presented as Julian dates, where Day 120 corresponds to May 1, Day 151 to June 1, Day 182 to July 2, Day 212 to August 1, Day 243 to September 1, Day 274 to October 2, and Day 305 to November 1. The 1956 data exhibited the largest variability of the three calibration years at Central Ferry and Sacajawea. This variability is in direct response to changes in hydrometeorological conditions, and followed the measured data at Central Ferry. At Sacajawea, the measured data did not show the variability predicted by the model, but it should be noted that measured data represents 6-day averages. Since few specifics are known about the Sacajawea data, we cannot explain differences between the model predictions and measurements, except that more variability in measured values may have been seen had daily measured values been available. The calibration simulations for 1957 and 1958 also matched data better at Central Ferry than Sacajawea. They demonstrated that the model is able to reproduce different temperature patterns.

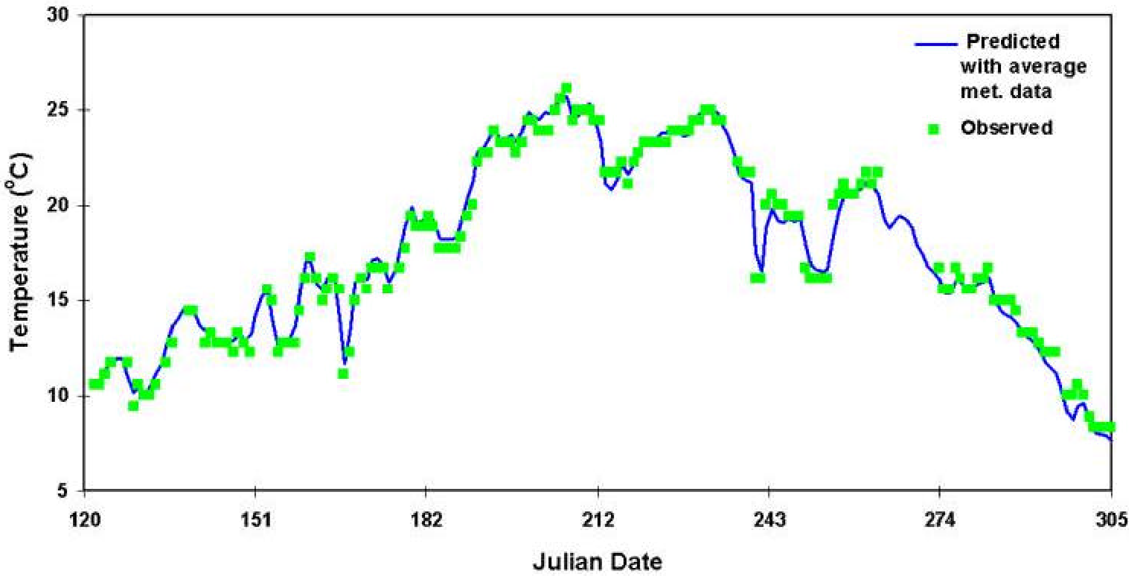

Figure 5.4-13. 1956 water temperatures at Central Ferry

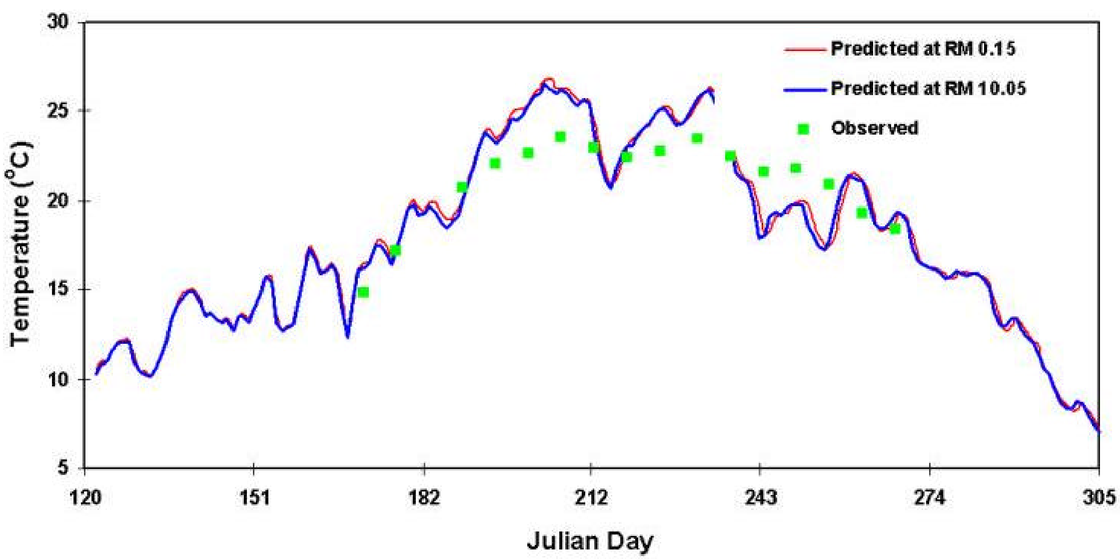

Figure 5.4-14. 1956 water temperatures at Sacajawea

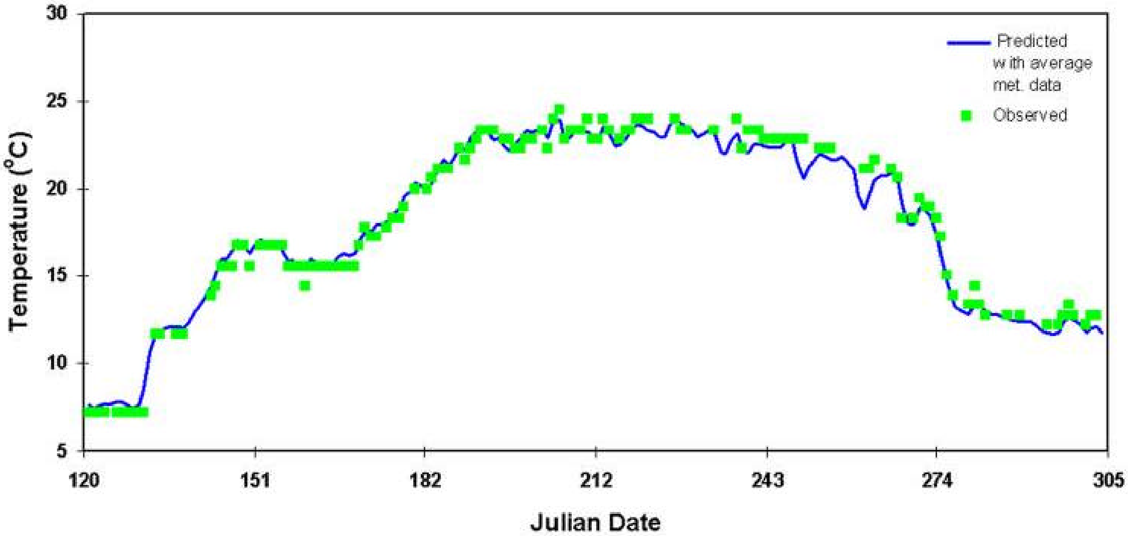

Figure 5.4-15. 1957 water temperatures at Central Ferry

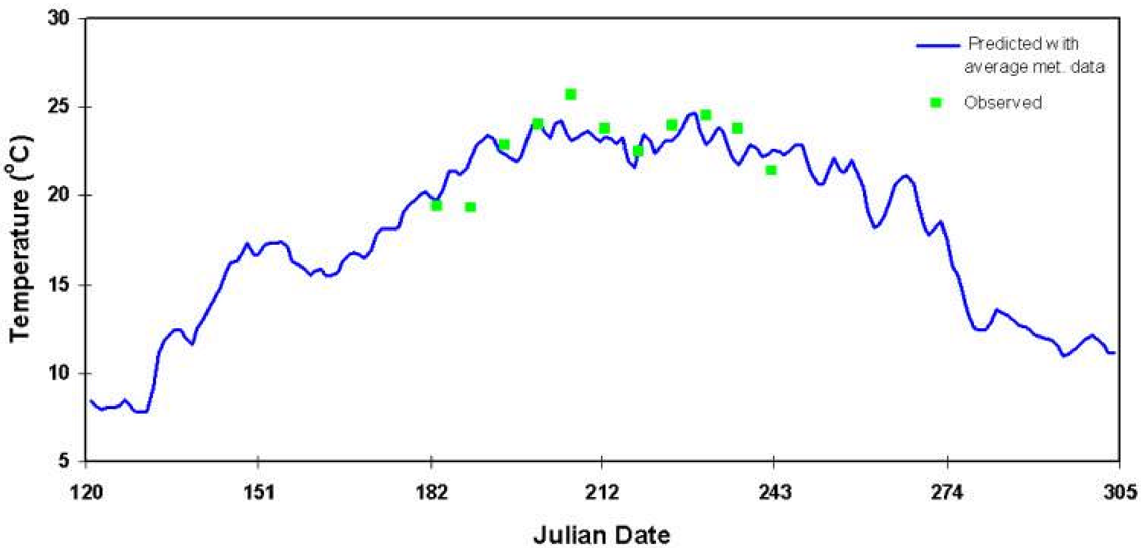

Figure 5.4-16. 1957 water temperatures at Sacajawea

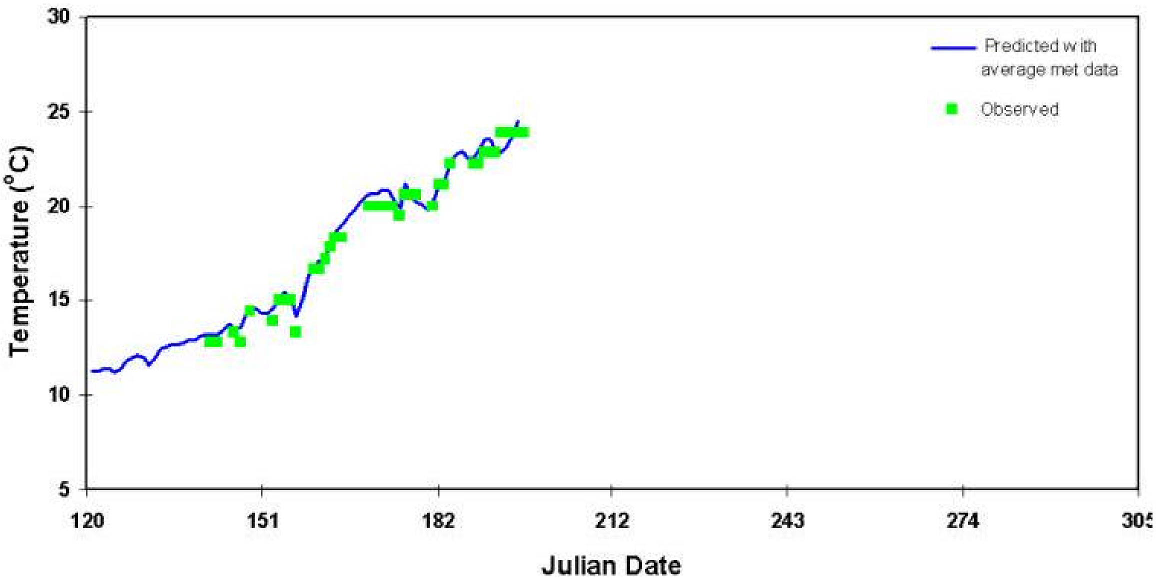

Figure 5.4-17. 1958 water temperatures at Central Ferry

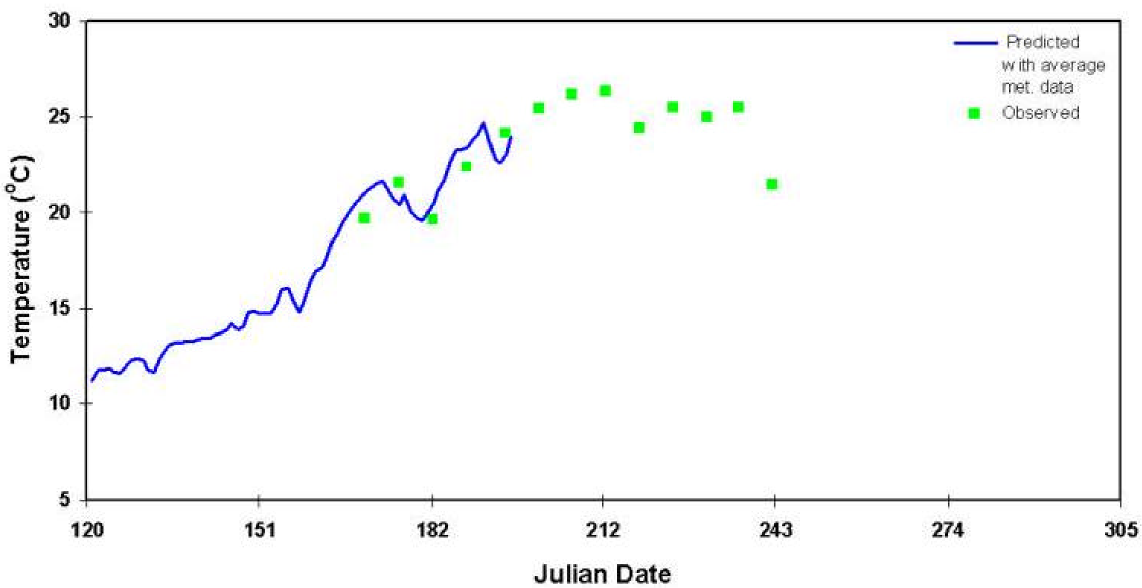

Figure 5.4-18. 1958 water temperatures at Sacajawea

Overall, the calibrated model predicts the correct seasonal warming, maximum temperatures, and fall cooling. As described earlier, minimal longitudinal trends are evident (Figures 5.4-19 through 5.4-21).

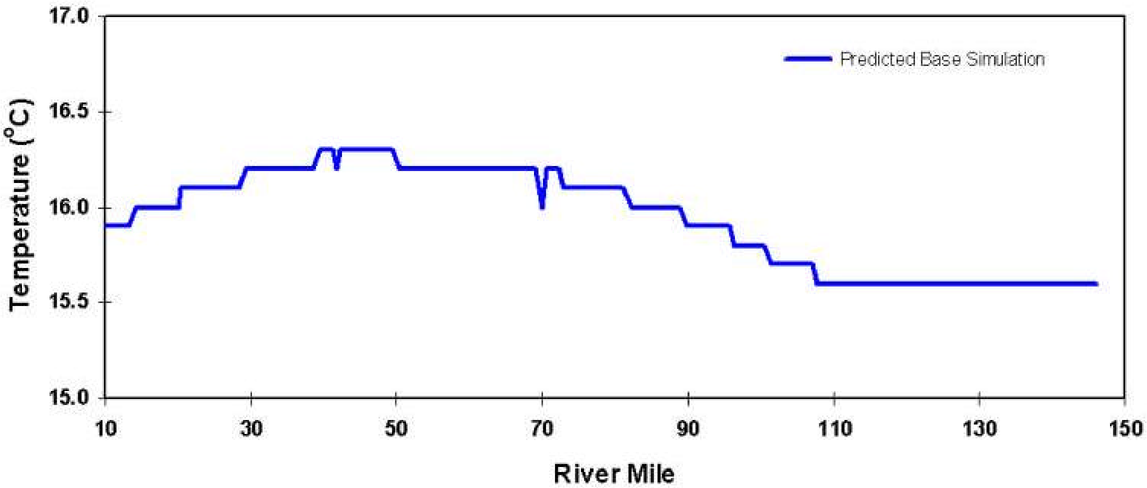

Figure 5.4-19. Predicted 1957 water temperatures for Day 145

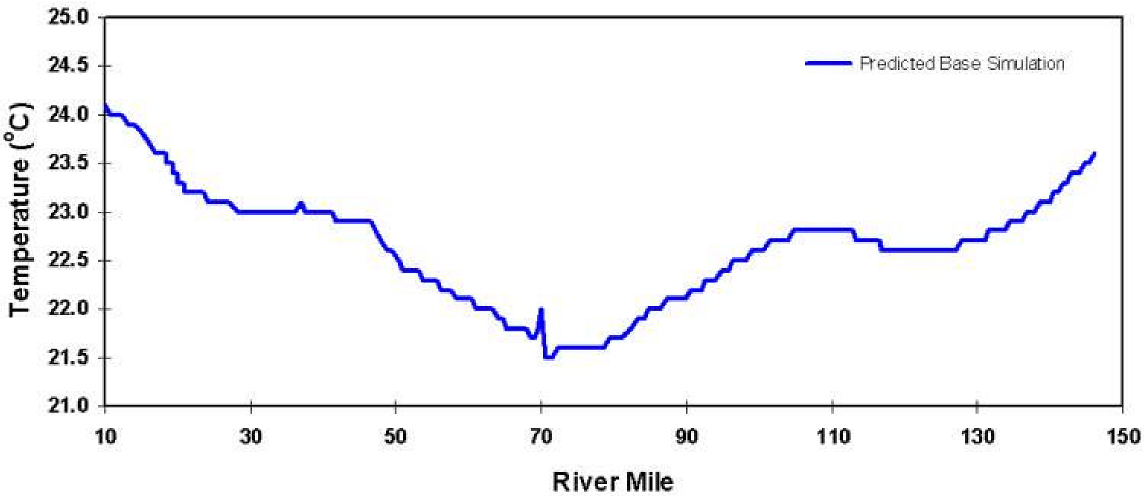

Figure 5.4-20. Predicted 1957 water temperatures for Day 235

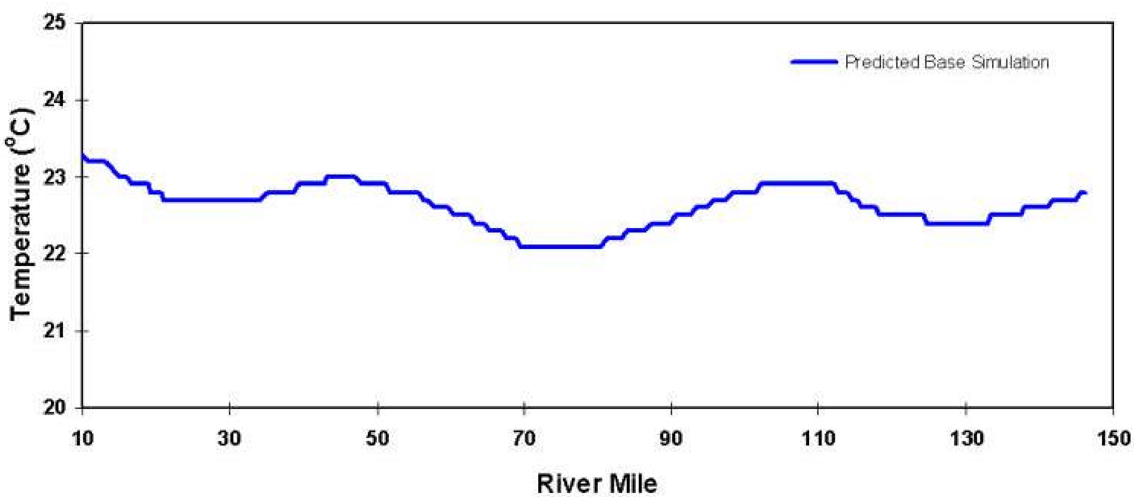

Figure 5.4-21. Predicted 1957 water temperatures for Day 243

5.4.4 - Sensitivity Analysis

Select sensitivity analyses constitute an integral part of the overall model calibration and verification process. Rather than varying individual parameters by set amounts (*e.g.*, +/-10%), we employed a megaparameter sensitivity analysis procedure in this project. Megaparameter sensitivity analysis involves varying several, typically correlated, parameters simultaneously (again, *e.g.*, +/-10%) to determine their joint influence on model projections. This approach provides a more realistic assessment of model sensitivity to sets of variables that are likely to co-vary during the normal function and management of the system of interest.

The sensitivity of the temperature predictions to meteorological inputs, upstream boundary conditions, and model coefficients was determined by varying inputs and comparing the results. Although sensitivity simulations were conducted on all years, only results from 1956 are presented. Results from 1957 and 1958 simulations were similar. Sensitivity due to meteorological inputs was determined by comparing simulation results using Lewiston and Pasco meteorological data. The results are shown in Figures 5.4-22 and 5.4-23. In general, use of Pasco data only resulted in temperatures that were too warm, especially at Sacajawea, where temperatures approach 30 °C. Predictions were better when only Lewiston data were used, but they tended to be slightly cool. Variability in model predictions due to meteorological data ranged up to 3 to 4 °C.

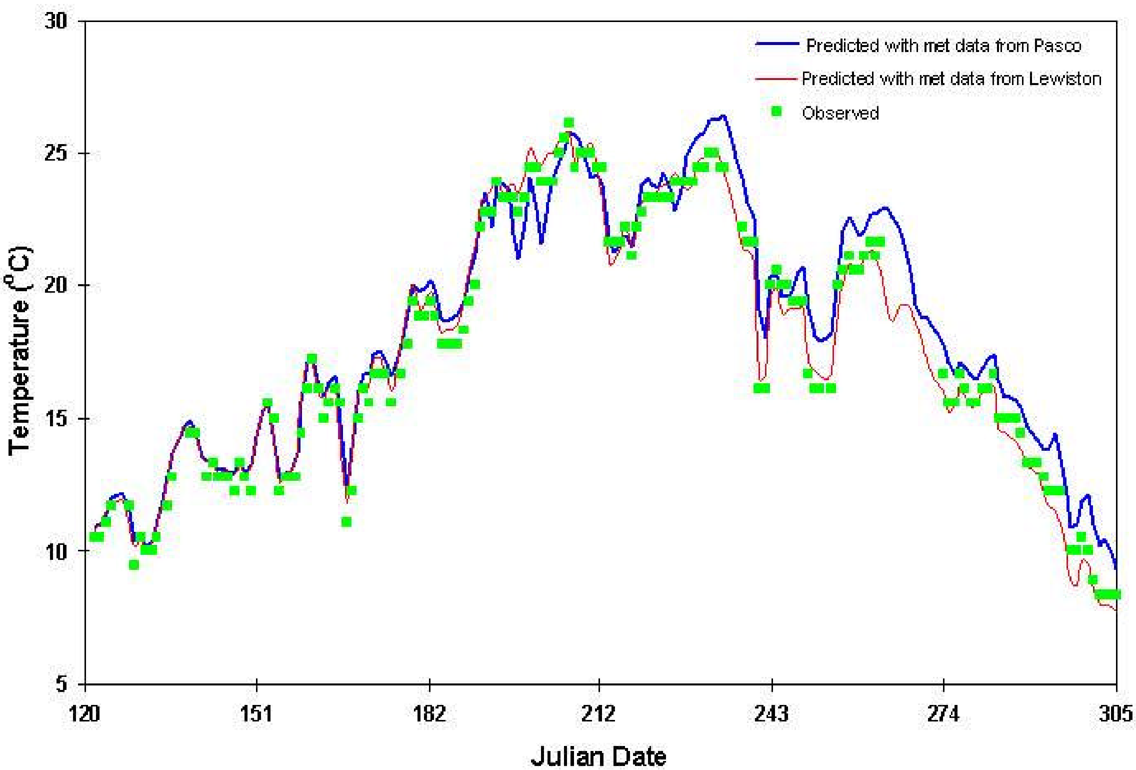

Figure 5.4-22. Observed and predicted 1956 water temperatures at Central Ferry using Pasco and Lewiston meteorological data

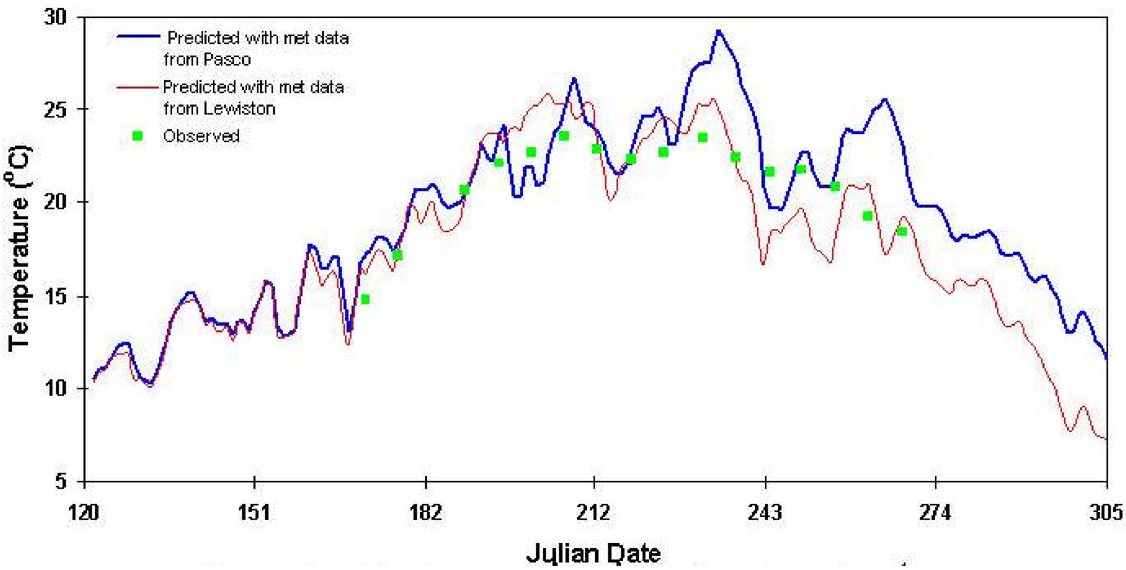

Figure 5.4-23. Observed and predicted 1956 water temperatures at Sacajawea using Pasco and Lewiston meteorological data

Model predictions were most sensitive to inflow temperature boundary conditions, as determined by reducing the inflow temperatures by 2 °C and comparing the results with the base simulation (Figures 5.4-24 and 5.4-25). The perturbed results were consistently about 2 °C less than the base throughout the entire year for all years. This, in part, is probably due to the short travel times for the normative river.

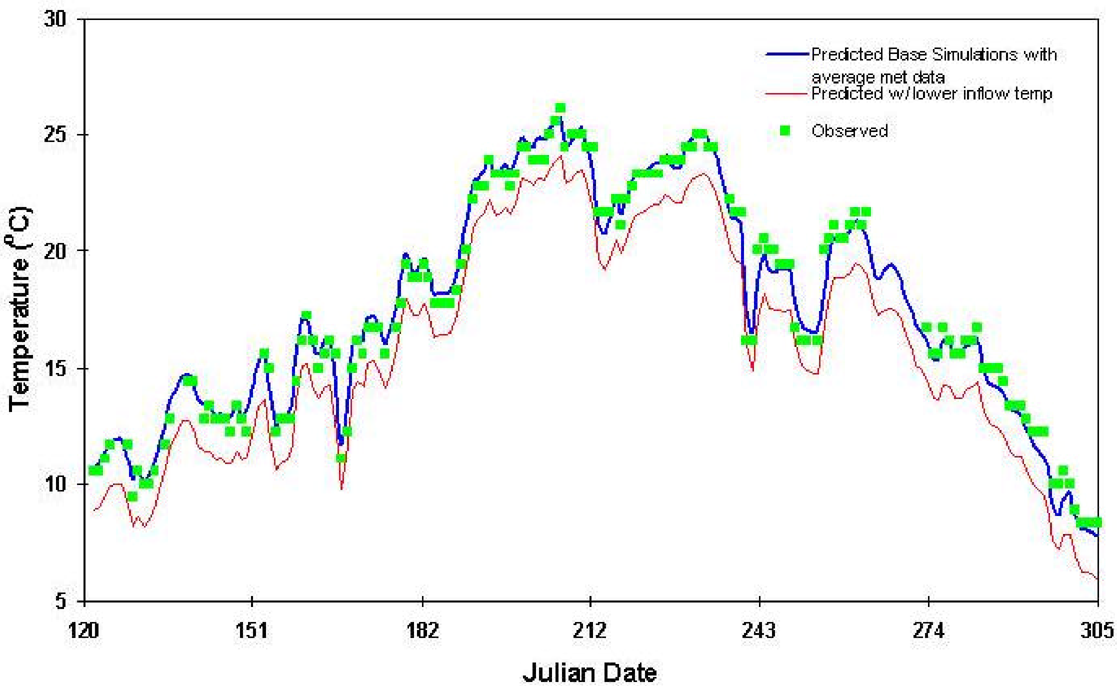

Figure 5.4-24. Predicted 1956 water temperatures at Central Ferry with inflow temperatures 2 degrees C lower than base simulations

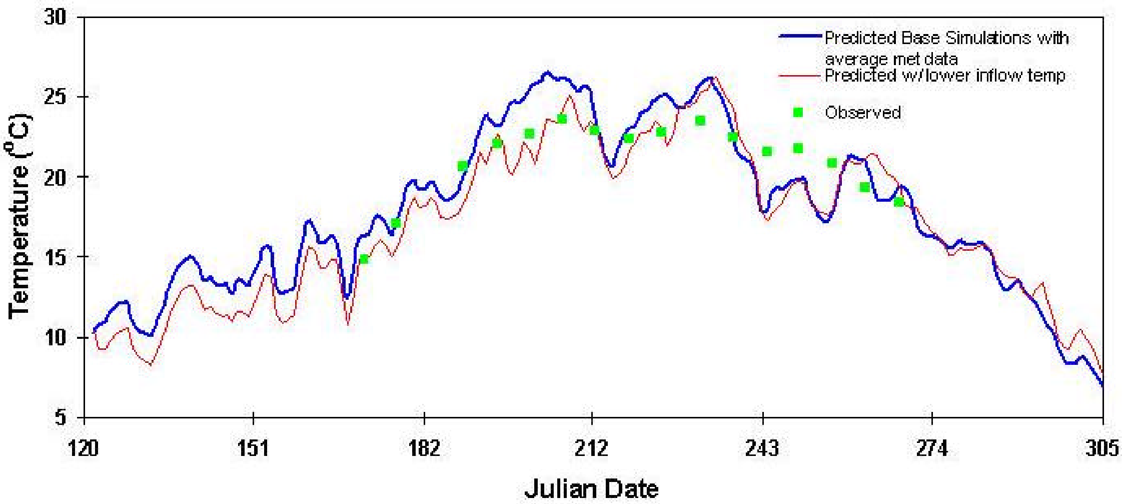

Figure 5.4-25. Predicted 1956 water temperatures at Sacajawea with inflow temperatures 2 degrees C lower than base simulations

The model was not very sensitive to other model coefficients (*i.e.*, less than 1 °C). As expected, model sensitivity was more apparent at Sacajawea because it was farther downstream. Sensitivity to the bed conduction term is shown in Figures 5.4-26 and 5.4-27. Increasing the bed heat capacity smoothed out some of the variability and allowed the river to retain heat longer into the fall. Sensitivity to other model coefficients are shown in Figures 5.4-28 through 5.4-33. Increasing the solar radiation turbidity factor (a measure of atmospheric scattering of solar radiation) from 3 to 5 resulted in slightly higher temperature at Sacajawea. Reducing the solar radiation by 10% was hardly discernable, as was reducing the wind function term BB from 1.5E-09 to 1.1E-09.

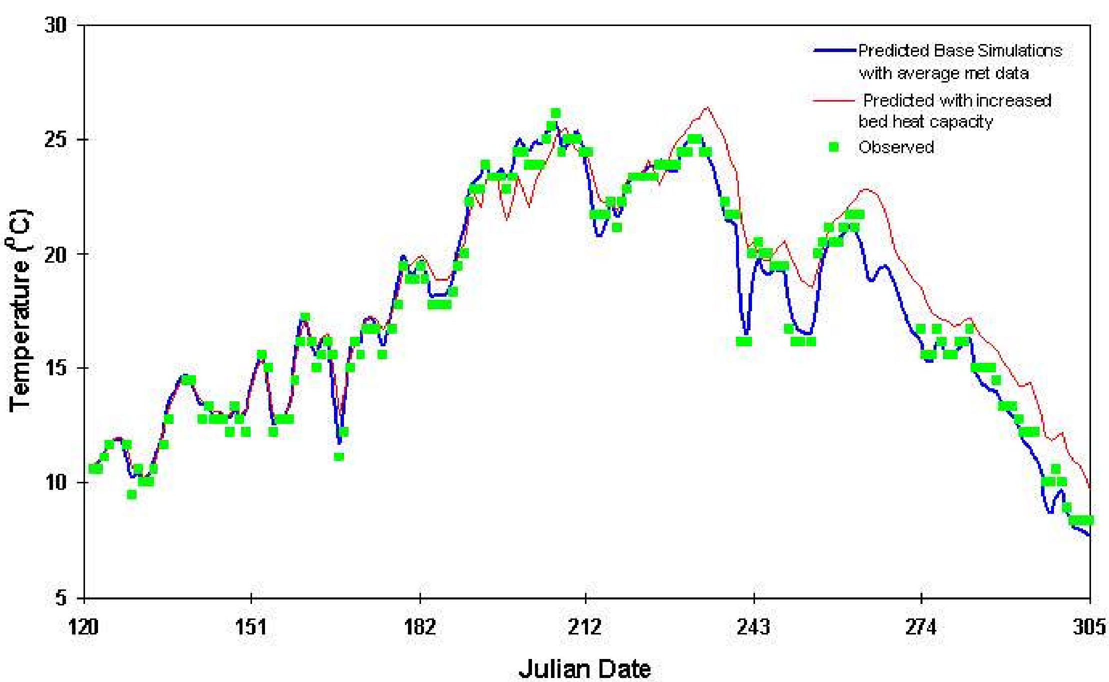

Figure 5.4-26. Predicted 1956 water temperatures at Central Ferry with increased bed heat capacity over base simulations

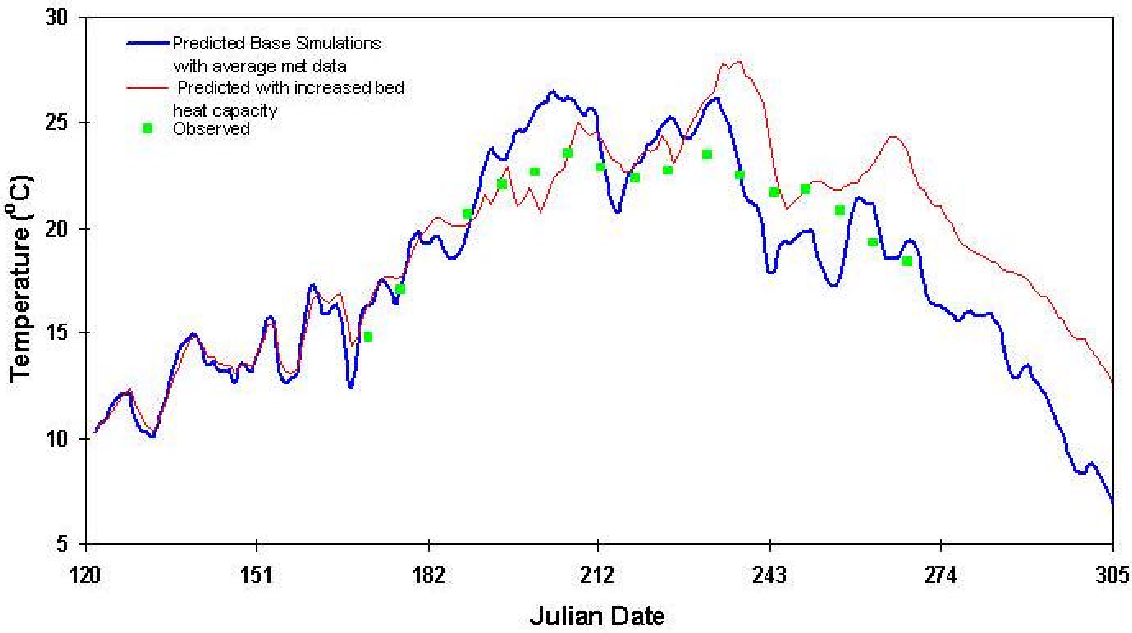

Figure 5.4-27. Predicted 1956 water temperatures at Sacajawea with increased bed heat capacity over base simulations

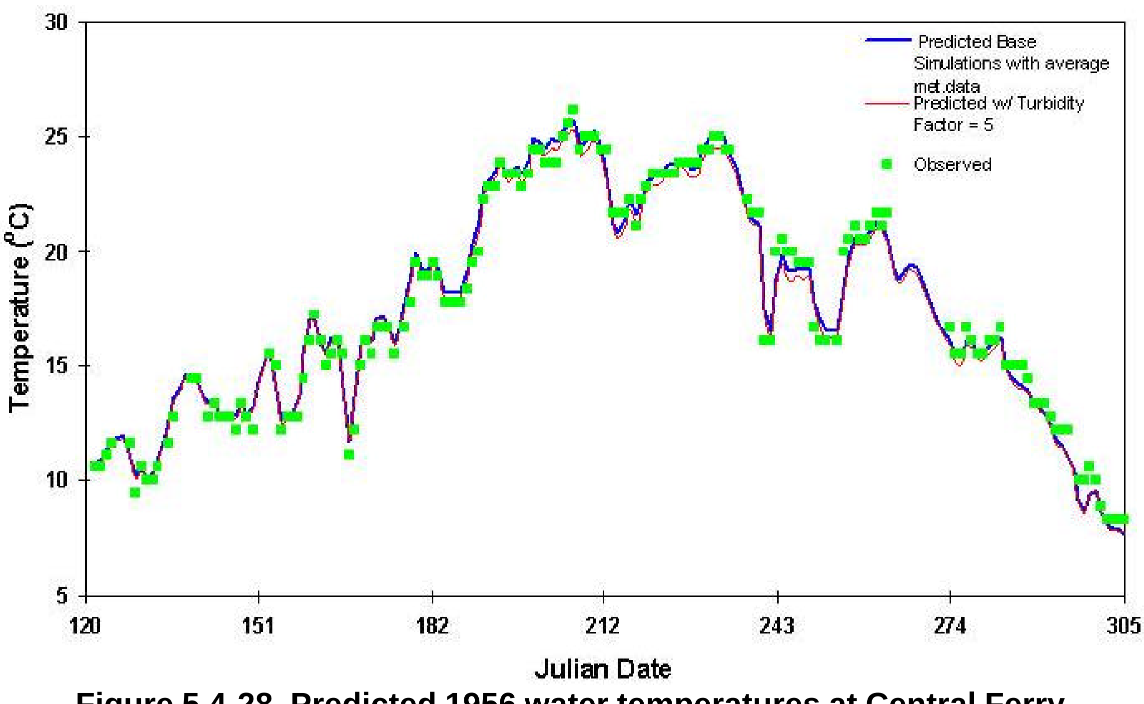

Figure 5.4-28. Predicted 1956 water temperatures at Central Ferry Showing sensitivity to turbidity factor

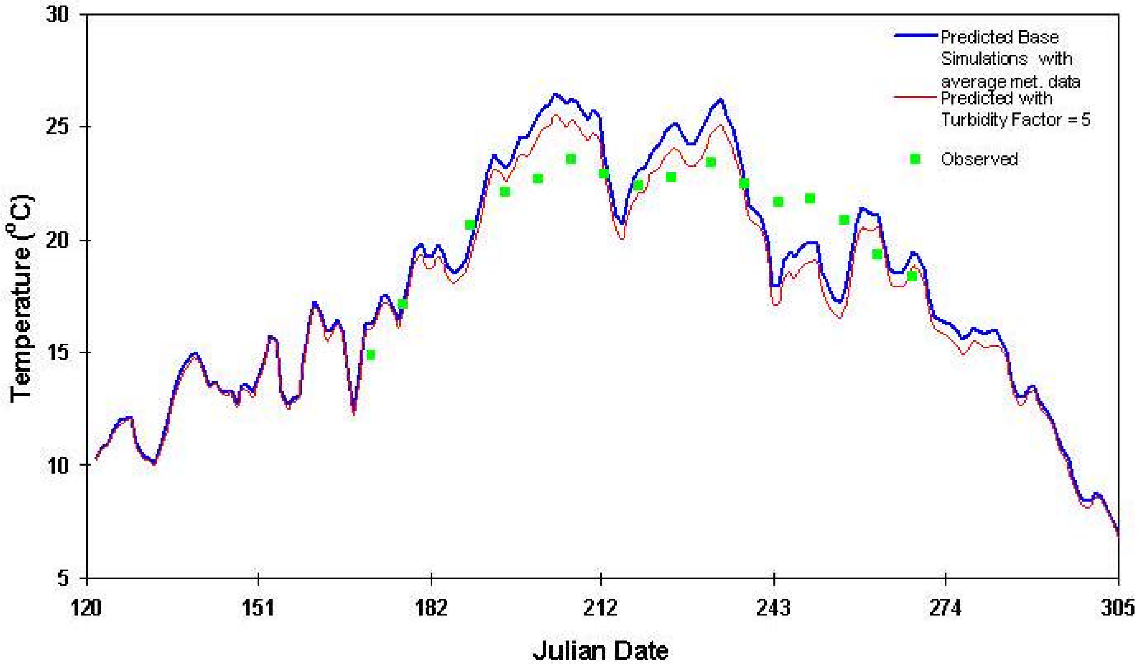

Figure 5.4-29. Predicted 1956 water temperatures at Sacajawea Showing sensitivity to turbidity factor

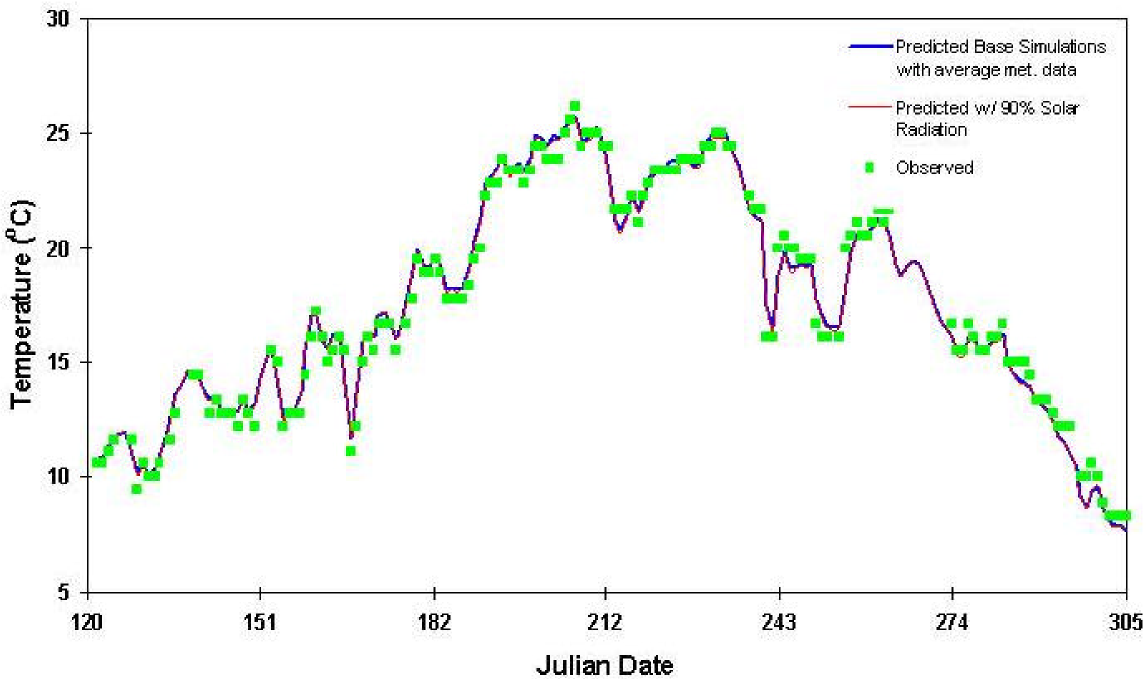

Figure 5.4-30. Predicted 1956 water temperatures at Central Ferry Showing sensitivity to solar radiation

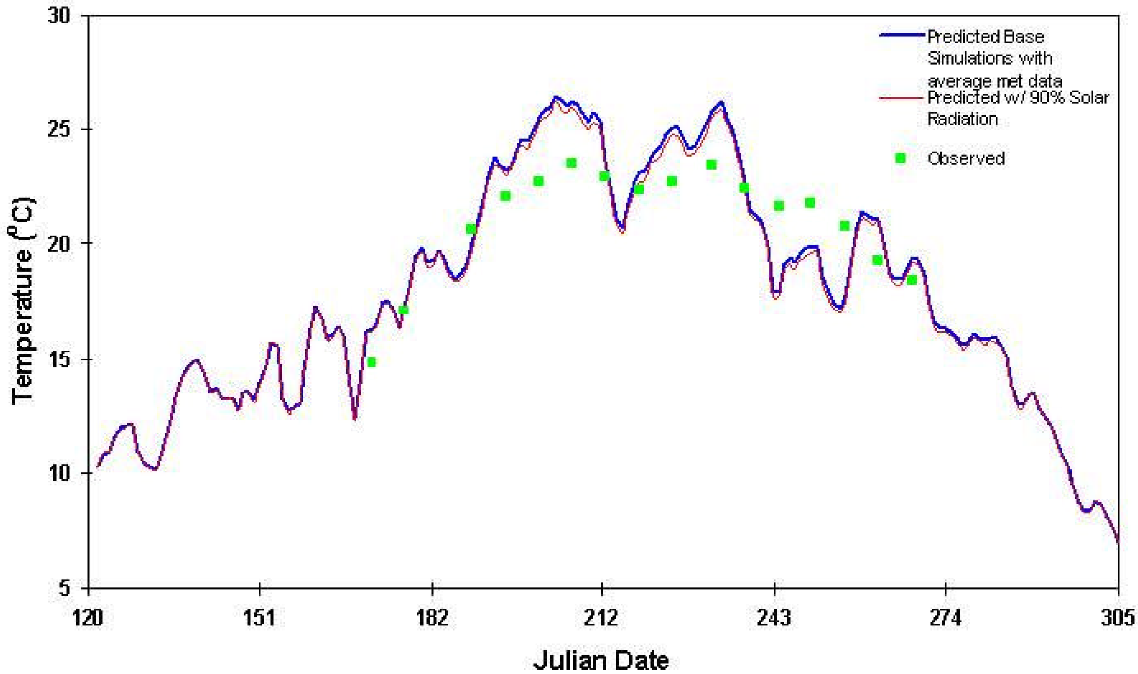

Figure 5.4-31. Predicted 1956 water temperatures at Sacajawea Showing sensitivity to solar radiation

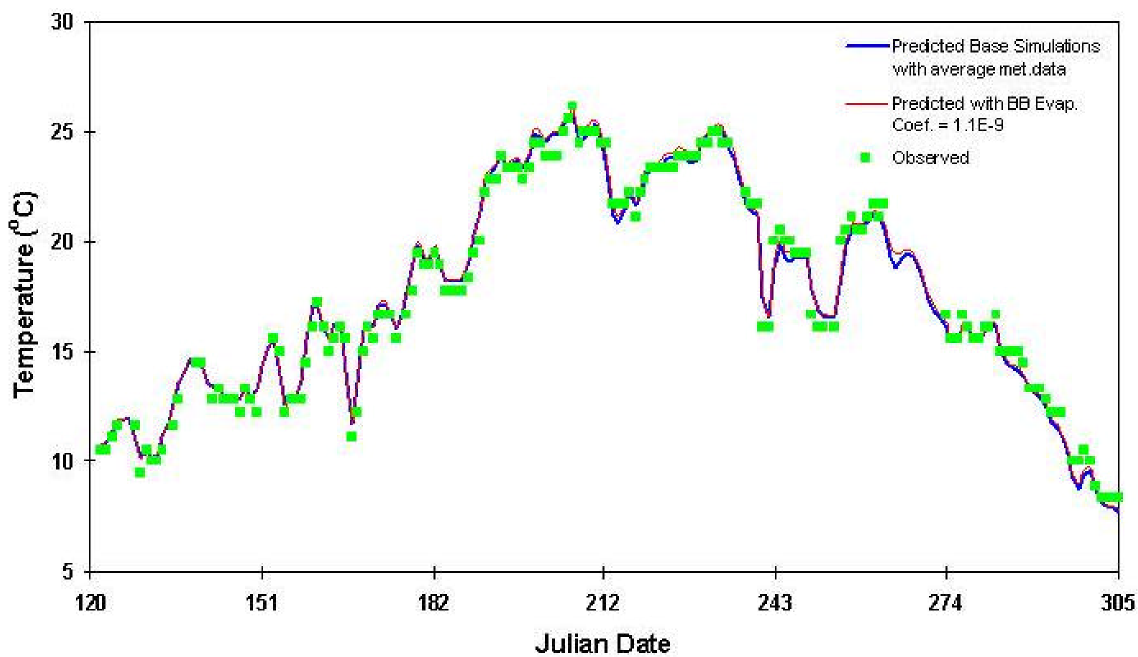

Figure 5.4-32. Predicted 1956 water temperatures at Central Ferry Showing sensitivity to wind function term

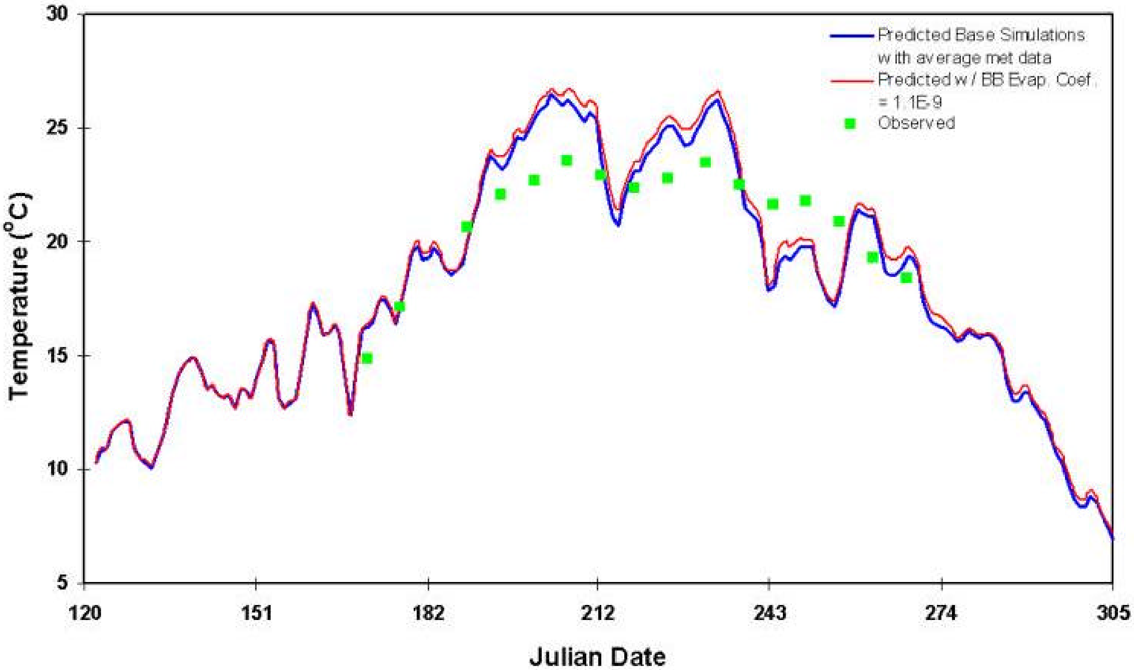

Figure 5.4-33. Predicted 1956 water temperatures at Sacajawea Showing sensitivity to wind function term

The sensitivity analysis clearly shows the importance of upstream boundary conditions. Sensitivity analysis provides greater insight into the response of the system to natural variability and human disturbances or management actions, as well as the "realism" of model projections in comparison with measured data. We selected the parameters to be employed in the sensitivity analysis of the Snake River application based on our initial experience with model calibration and verification.

Biological Sensitivity Analysis

A sensitivity analysis was conducted to determine how the biological components reacted to differences in certain input parameters. Parameters used for the sensitivity analysis were those in which little data was available, either actual data or data from the literature. Other parameters considered for the sensitivity analysis were those which showed the greatest sensitivity during initial modeling simulations. The sensitivity simulations were conducted for Lower Granite pool during 1997.

In general, the primary producers, benthic algae, experienced the greatest sensitivity to most parameters evaluated. Aquatic insects and benthic animals were less sensitive, and fish were the least sensitive.

Results of the biological sensitivity analysis are presented in Table 5.4-7. Decreasing the growth rate of benthic algae 1 and 2 by only 10 percent (from 0.70 to 0.67 1/day), results in 77% and 72% decreases of benthic algae 1 and 2, respectively. The other groups changed by less than 4 percent as a result of changing the benthic algae growth rate.

Table 5.4-7 Sensitivity Analysis of Average Biomass Simulations At Lower Granite Pool During 1997, For Day 213								
Simulation	**Phytoplankton (mg/l)**	**Benthic Algae #1 (mg/m^2)**	**Benthic Algae #2 (mg/m^2)**	**Aquatic Insects (mg/m^2)**	**Benthic Animals (mg/m^2)**	**Fish #1 (kg/km)**	**Fish #2 (kg/km)**	**Fish #3 (kg/km)**
Base Simulation	0.40	5,907	1,898	1,489	2,575	14	132	344
Reduced Benthic Algae growth rate by 10%	0.40	1,373	522	1,422	2,517	14	131	331
Reduce Benthic Algae temperature curve by 1 °C	0.40	22,173	4,909	1,529	2,619	14	132	354
Increase Benthic Algae scour rate from 0.001 to 0.0015 1/day/m^2/sec	0.40	2,439	897	1,446	2,540	14	131	336
Decrease initial Insect and animal biomass by 10%	0.40	6,617	2,051	1,331	2,300	14	130	344
Increase initial fish biomass by 10%	0.40	5,487	1,770	1,446	2,526	16	145	377
Decreased feeding preference of Fish 3 on benthic algae by 10%	0.40	7,974	2,498	1,465	2,517	14	132	328

As with the benthic algae growth rate, the optimal temperature growth curve for benthic algae varied both in the literature and in the WQRRS documentation. The temperature curve was varied slightly by reducing the four inflection points by 1 °C. Lowering the optimal temperature of benthic algae 1 and 2 resulted in increases of nearly four and three times the biomass as that in the base simulation in Lower Granite pool. The secondary producers were again affected minimally by this change.

Benthic algae were also greatly affected by scour. The scour rate parameter was varied from 0.001 1/day/m^2/sec in the base simulation to 0.0015 1/day/m^2/sec. The results were a 59 and 52% decrease in benthic algae 1 and 2, respectively (Table 5.4-7). The secondary producers changed less than 3%.

Initial estimates of aquatic insects and benthic animals were not as well known as the other biological parameters. Initial estimates of insects and animals were decreased by 10 percent for sensitivity analysis purposes. A lower initial biomass of insects and animals resulted in slight increases in benthic algae (12 and 8% for benthic algae 1 and 2). The increases could be attributed to lower predation on the benthic algae. Fish were essentially unaffected by the change. Initial estimates of fish biomass were increased by 10 percent. As a result, benthic algae biomass decreased by 7 percent, and aquatic insects and benthic animals decreased by less than 3 percent.

Changes in grazing preferences could result in changes through the entire food chain. Sensitivity to this parameter was examined through varying the feeding preference of fish 3 on benthic algae. Feeding preferences of fish 3 on benthic algae was decreased by 10 percent. As a result, the feeding preferences of fish 3 on aquatic insects and benthic animals is increased by 7 and 12 percent, respectively. The resulting biomass changes are an increase in benthic algae 1 and 2 of 35 and 32%, respectively, due to a decrease in predation. Aquatic insect and benthic animal biomass decreases by only 2 percent.

Based on this sensitivity analysis, it is apparent that benthic algae, as simulated by WQRRS, is highly sensitive. Small changes in certain input parameters can result in drastically different predicted biomass values. This furthers our contention stated later in this report (Section 5.5.3) that the biological model be used to compare relative differences between alternatives and not be used in a strict predictive sense.

5.5 - MODEL PREDICTIONS AND COMPARISON WITH EXISTING CONDITIONS

This modeling effort encompassed two very specific scenarios, although the capability certainly exists for additional scenario simulations in the future. The model was used initially to simulate the normative river under three hydrologic year classes represent "wet," "dry," and "average" years. Determination of these hydrologic classes was done through statistical examination of the existing flow and water records. While no data existed to verify predictions under the proposed future scenario, historic, pre-impoundment data, particularly from the Lower Granite pool, was used where possible as an order-of-magnitude check on model predictions.

The water quality model was configured in a manner that will allow other scenarios to be investigated in the future.

5.5.1 - Range of Historical Data

Physical characteristics

In order to understand and explain differences in the water quality and biological production of the existing and normative (free-flowing system), some understanding of the difference in physical characteristics between the two systems is necessary. As part of the Columbia River Salmon Mitigation Analysis System Configuration Study Phase I (USACE, 1994), the Corps of Engineers 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 free-flowing system is about one order of magnitude less than for the existing system. At a typical summer low flow of about 25,000 cfs, travel time from the Clearwater River to the Columbia River for the existing system is about 35 days, compared to about 2½ days for the free-flowing system. At a typical May to June flow of 120,000 cfs, travel time would be on the order of 1 day for the normative system and 7 days for the existing system. Therefore, the free-flowing system is flushed very rapidly compared to the existing system.

Water depths and available habitat also vary significantly between the systems. For the existing system, flow depths remain relatively constant throughout the year, and range from about 5 feet in the tailwater areas to over 100 feet at the dams. In contrast, flow depths will vary seasonally with flow in the free-flowing system. During a typical spring runoff period (120,000 cfs), average flow depth over a cross section will be on the order of 25 feet, compared to 15 feet during a typical summer flow condition. Differences in surface area or average width will be similar. The total surface area for the existing system is approximately 35,940 acres compared to 18,370 acres for the normative (free-flowing) system.

Water temperatures in the lower Snake River vary seasonally. Temperatures typically peak between 21 °C and 25 °C during late July and August. Variation in temperatures from upstream to downstream within the study area does occur, but there are no consistent trends. Cooler temperatures downstream have been observed due to colder temperatures entering the lower Snake River from the Clearwater River. Dissolved oxygen concentrations are typically near or above saturation throughout the system. The temperature regime in the normative system is expected to be somewhat different from that which has been observed under the existing impounded system. The dissolved oxygen regime is not expected to change substantially if the lower Snake River is returned to the normative state.

The bottom substrate will differ between the existing impounded system and the normative system. It is assumed that fine-grained sediment deposited behind the dams representing a large proportion of the existing substrate will be flushed from the system, and the free-flowing system will return to a hard substrate as it was prior to impoundment. The substrate will consist primarily of boulders and cobble in fast-moving sections; and sand in deep, grading to cobble and a little sand in shallow slower moving sections.

Existing Water Quality and Biologic Productivity

To put modeling results in perspective, it is necessary to have an understanding of the range of historical water quality and biologic data, as well as a general sense of the anticipated water quality and productivity under the normative system. Because there are few data on the free-flowing system, 1997 data collected at Asotin (river mile 148) are being used as a surrogate for the normative system for this discussion. It should be recognized that the data collected at Asotin are not completely representative of water quality to be expected in the normative river system. Discharges at Lewiston, Clarkston, and Potlatch occur downstream of the station at Asotin, as does the confluence of the lower Snake and Clearwater Rivers. These discharges will all affect the water quality of the normative river system and, as such, have been incorporated into the model of the lower Snake River. During earlier studies (pre-1997), upstream data were collected at river mile 140. This location is not quite representative of free-flowing conditions due to backwater effects.

Historically, extensive water quality data have been collected on the lower Snake River. The most intensive studies were undertaken in 1994, 1995, and 1997. A more limited set of water quality data is available for the pre-impoundment years, especially 1970 to 1972 and 1975 to 1977. Little data on the biology of the system are available for the pre-impoundment years. Likewise, data on the biology of the system are limited for the existing lower Snake River system. The most comprehensive source of biological data for the existing system is the study described previously in this report and in the appendix to this report. The biological data contained in this report are limited to information on the phytoplankton community and production, the zooplankton community, and the attached benthic algae community and production. Extremely limited data exist regarding the macroinvertebrate, fish, and benthic animal communities of the lower Snake River as they exist today. Those databases that do exist for the higher components of the food chain are often of limited utility to the modeling effort, because water quality and physical data were not collected or reported with the data. Data of this sort are only useful for order of magnitude comparisons with model results.

Nutrient concentrations of the free-flowing section were similar to those observed at stations in the impoundments downstream in the lower Snake River at Asotin range from <0.01 to 0.05 mg/L. In the impounded section of the lower Snake river, ammonia concentrations vary over a wider range, from <0.01 to 0.07 mg/L. Nitrite-nitrate nitrogen typically ranges from approximately 0.1 to 1.4 mg/L at Asotin, and 0.04 to 0.95 mg/L in the impounded section of the river. Ortho-phosphorus concentrations ranged from <0.001 to 0.071 mg/L at Asotin, and <0.001 to 0.59 mg/L in the impounded section. Suspended solids concentrations were also similar at upstream and impounded sites. The free-flowing section showed suspended solid concentrations ranging from 1.2 to 65 mg/L, and the impounded section historically showed concentrations from 0.4 to 40 mg/L. Further detail on the 1997 water quality can be found in previous sections of this report. Nutrient concentrations of the magnitude typically observed on the lower Snake River would support a trophic classification of meso-eutrophic to eutrophic. These nutrient concentrations support moderate to high levels of primary productivity in the existing impounded river, and are expected to support similar levels of productivity in the normative river.

Biological data are more limited. Most of the available data is from the 1997 field effort. These data are summarized in great detail in Section 3 of this report. Zooplankton data showed a median biomass at the free-flowing section of 0.247 µg/L versus median values from 0.098 to 11.58 µg/L in the impounded section. Diatoms dominate phytoplankton in the lower Snake River. Phytoplankton concentrations in the free-flowing section were similar to those observed in the impounded section exhibiting a mean biovolume of 578,980 μm^3/mL, which lies between the range of values, 323,697 to 738,062 μm^3/mL, for the impounded section. However, due to the greatly reduced river volume per unit of river length in the normative river relative to the existing impounded river, the overall contribution of phytoplankton to system productivity is anticipated to be small in the normative river. Attached benthic algae will account for the majority of the total primary productivity in the normative system. The most extensive data for attached benthic algae was collected in 1997. The 1997 and 1998 surveys showed the free-flowing site to be more productive than the impounded sites, with respect to chlorophyll accumulations. On a dry weight basis, attached benthic algae at the free-flowing site averaged 24 mg/m^2 at a depth of 0.75 m. The other sites in the impounded section had average dry weight values of 15 to 56 mg/m^2. There should be more substrate available for the growth of attached benthic algae in the normative river than in the existing impounded river, because shallower water depths will allow sunlight to reach more of the river bottom in the normative river, resulting in increased growth of attached benthic algae.

Benthic macroinvertebrates data, which included both aquatic insects and the taxa included in the benthic animal category in the model, were collected in Lower Granite, Little Goose, and Lower Monumental reservoirs from November 1993 to September 1995 (Bennett *et al.*, 1997). Samples were collected on soft and hard substrate. The most common taxa observed in the soft substrate were Oligochaetes, Amphipods (primarily Corophidae), Nematodes, Diptera (primarily chironomids), and Pelecypoda (primarily mussels). In the hard substrate, Diptera (again primarily chironomids), Tricoptera (primarily caddis flies), and Amphipods (both Gammaridae and Corophidae) were the most common taxa.

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 removal of the dams. The Hells Canyon study documented the impact of variable flows on the composition of the aquatic insect community. Results composited over all flows indicated that the aquatic insect community of the lower Snake River, in the vicinity of Hells Canyon, was composed of 7% Ephemeroptera (mayflies) by volume, 45% Diptera (primarily chironomids) by volume, 35% Tricoptera (primarily Hydropsyche), 6% Lepidoptera, and 7% other taxa. The average wet biomass of all samples collected during this study was 8.7 g/m^2. This number was converted to dry biomass to be used as a starting point for the simulation of the aquatic insect compartment in the biologic productivity model. This study did not include any enumeration of benthic animals.

The aquatic insect community anticipated for the normative river system will likely be similar in composition to that described in the 1973 Hells Canyon study. The aquatic insect portion of this community will likely still have a high proportion of chironomids, as exhibited in the existing impounded system. Tricoptera will remain moderately abundant, while Ephemeroptera (mayflies) and Lepidoptera will represent the balance of the community.

The planktivorous/benthivorous fish community in the existing river system is dominated by largescale sucker, bridgelip sucker, and a variety of centrachids and cyprinids. This group also includes the young-of-the-year and juveniles of members of the piscivore community. The piscivore community in the existing impounded lower Snake River system is dominated by smallmouth bass and northern pikeminnow, with less numbers of salmonids and catfishes. In this system, both plantivory and benthivory play important roles in the transfer of energy through the food chain.

Piscivorous fish species in the normative river system will include smallmouth bass, northern pikeminnow, and channel catfish. A smaller group of piscivores in the normative system will include all of the fish eating life stages of the salmonid species in the lower Snake River, while the benthivorous fish group will include bridgelip and largescale suckers, white sturgeon, and a variety of centrarchids, sculpins, and cyprinids. Planktivory in the normative system is anticipated to be much lower than benthivory, due to the virtual absence of zooplankton in the system. Benthivory will play the dominant role in the transfer of energy through the food chain.

5.5.2 - Temperature

Temperature differences between the existing and normative 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 (river mile 83.2) and Sacajawea (near river mile 0). Although the study years represent the spectrum, dry through wet conditions, and the period 1956 through 1958 represents more average conditions, the comparison clearly shows some distinct differences. As previously described by Bennett *et al.* (1997), maximum temperatures during the summer months of July through August are anticipated to be approximately 2 to 5 °C higher under the normative system - approaching 26 to 27 °C. 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 free-flowing system.

Model simulations for the normative system were also compared to the measured data for the study years in Figures 5.5-1 through 5.5-12, at river miles 110.5, 80.5, 44.0, and 15.5. The model versus measured data compares periods of similar hydrometeorological conditions. At river mile 110.5, measured temperatures in the existing system are similar to predicted temperatures for the normative system, except the existing system lags the normative system. The time lag increases as the year progresses and river flows decrease. The response appears to be directly related to increased travel times and volumes associated with the existing system. As one moves downstream, differences between the normative and existing system become greater. The predicted temperature regime in the normative system changes little downstream, due to the short travel times. In contrast, temperatures in the existing system become more diffused and do not illustrate the synoptic variations found upstream. Maximum temperatures in the existing system become less than the normative system, and heat is retained longer into the fall. Differences between the study years appear to be related more to the magnitude of synoptic variations and the overall magnitude of maximum temperatures than seasonal patterns. The model predicts a higher peak summer temperature for 1994. All three study years exhibit similar seasonal trends with respect to temperature. Temperatures generally rise until mid summer, and then decline through the fall.

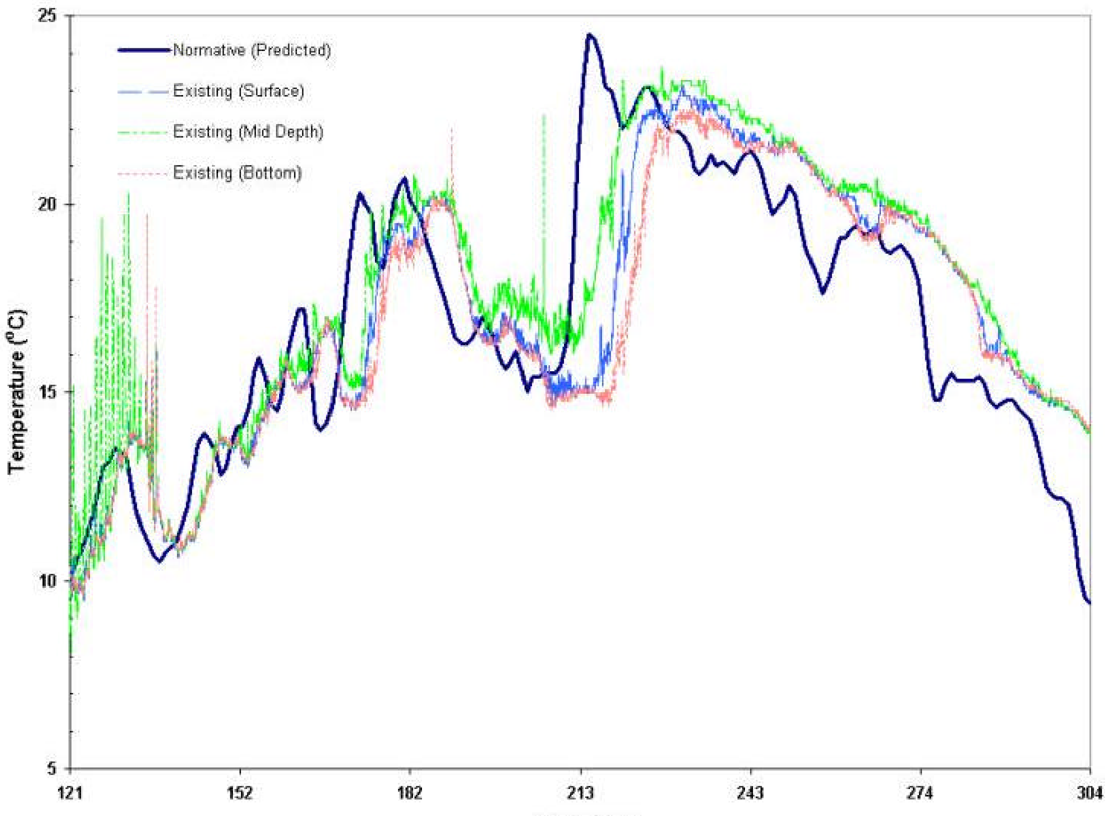

Figure 5.5-1. 1994 water temperatures at Snake River mile 110.5

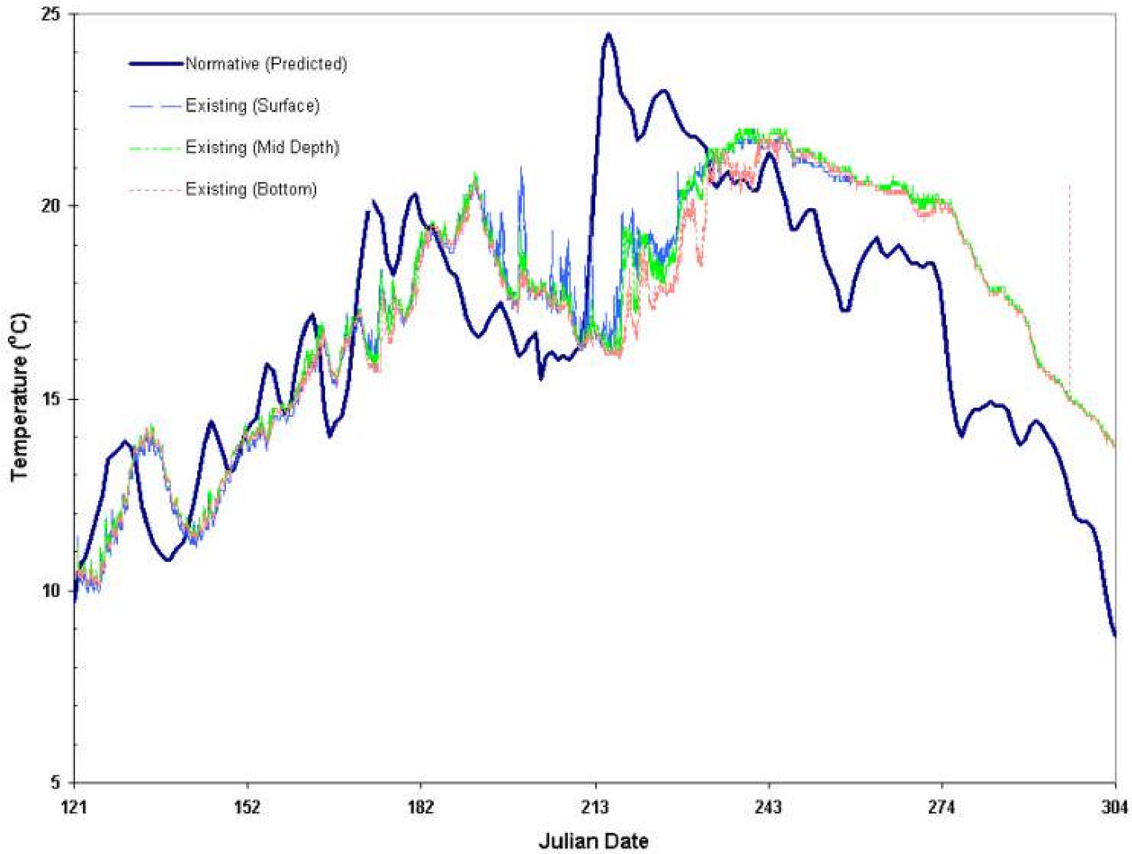

Figure 5.5-2. 1994 water temperatures at Snake River mile 80.42

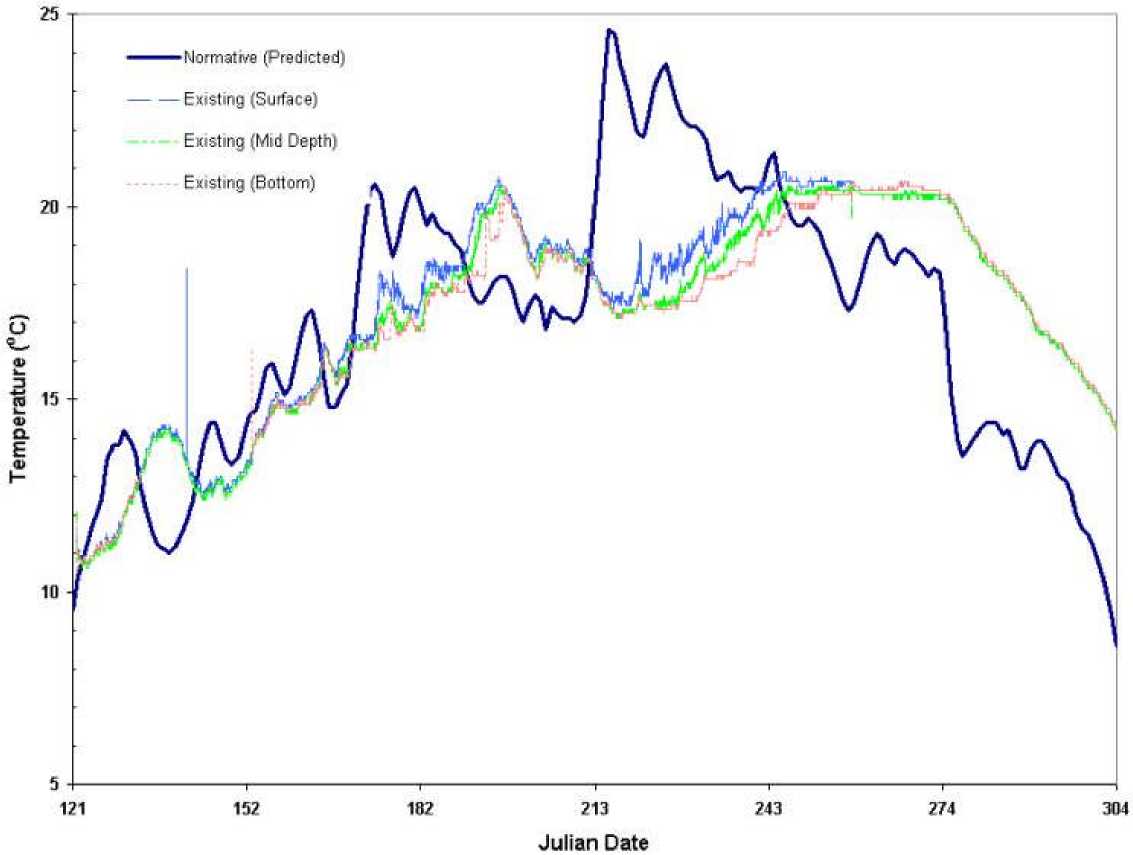

Figure 5.5-3. 1994 water temperatures at Snake River mile 44.15

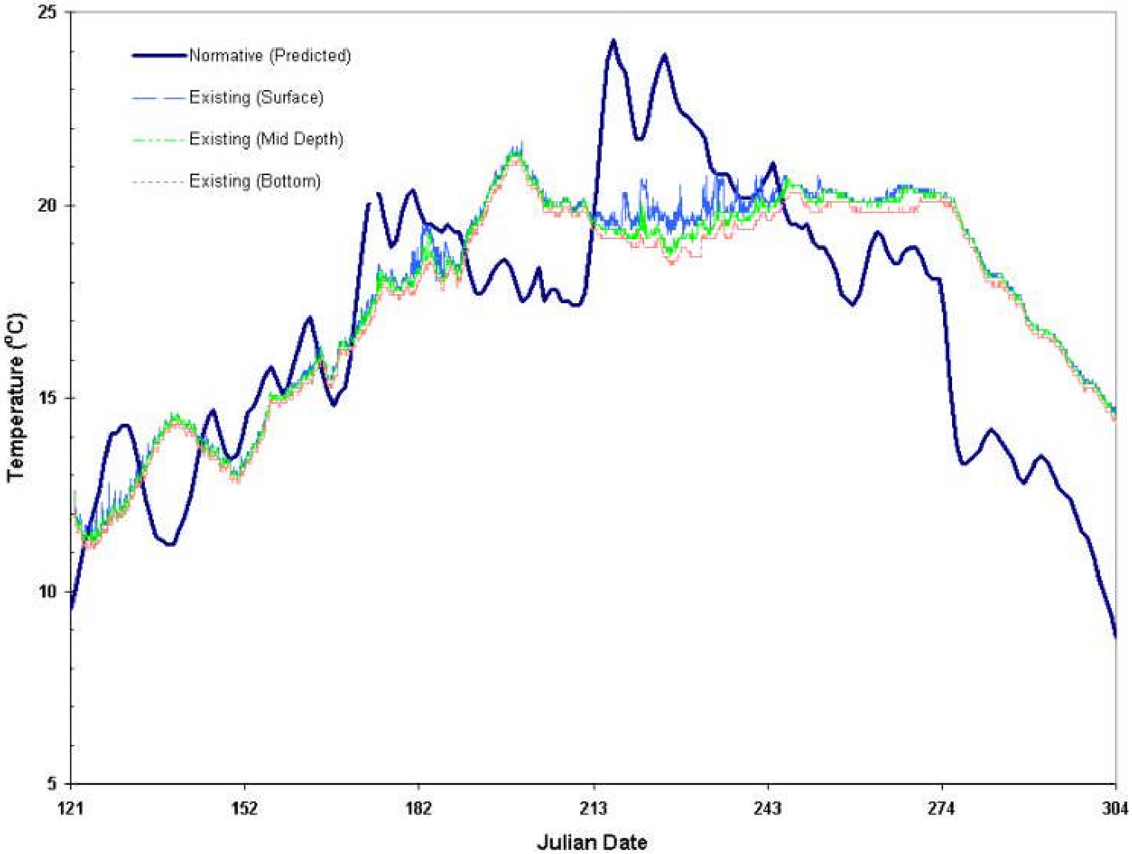

Figure 5.5-4. 1994 water temperatures at Snake River mile 15.94

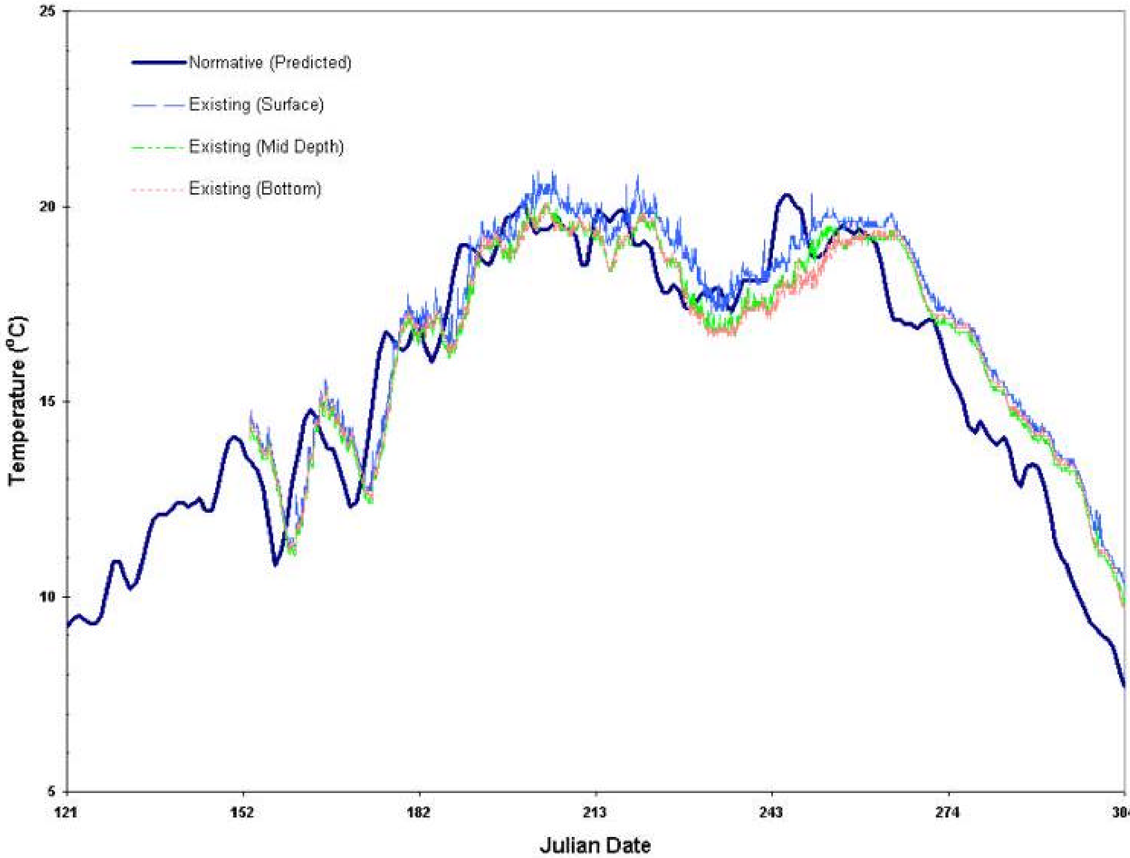

Figure 5.5-5. 1995 water temperatures at Snake River mile 110.5

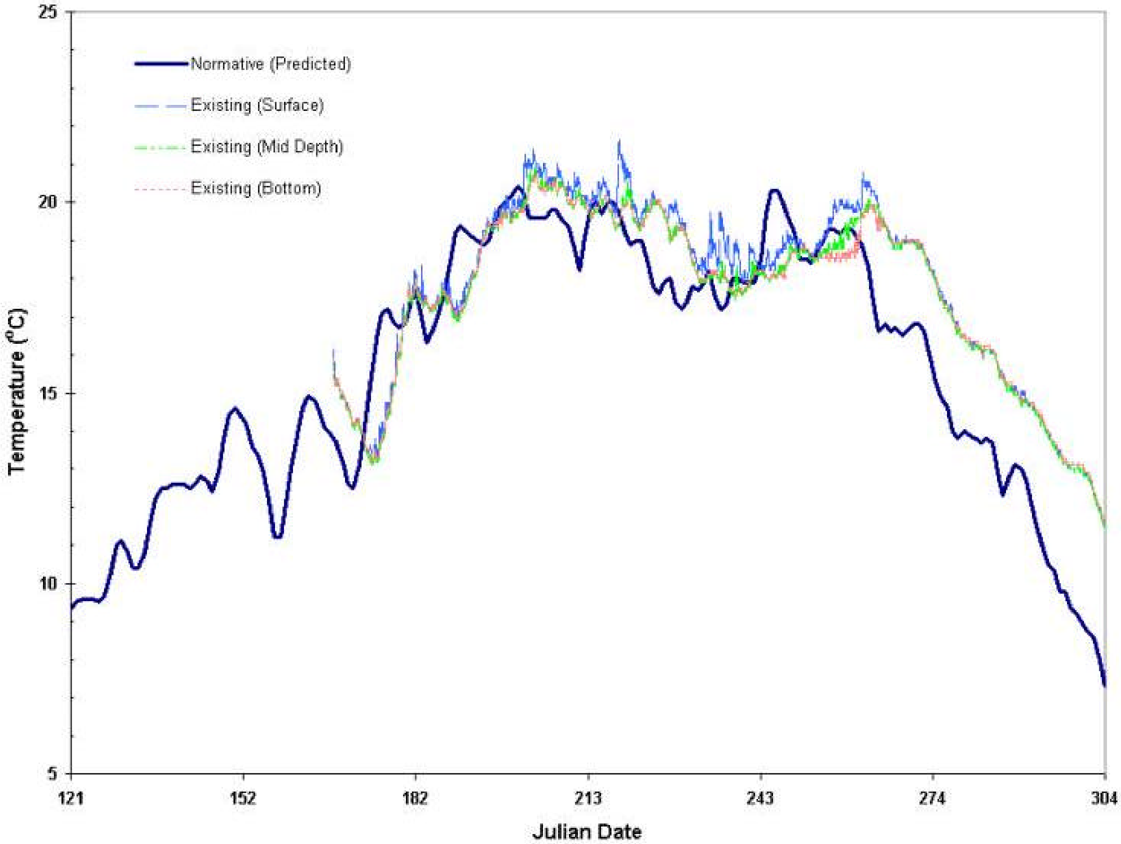

Figure 5.5-6. 1995 water temperatures at Snake River mile 80.5

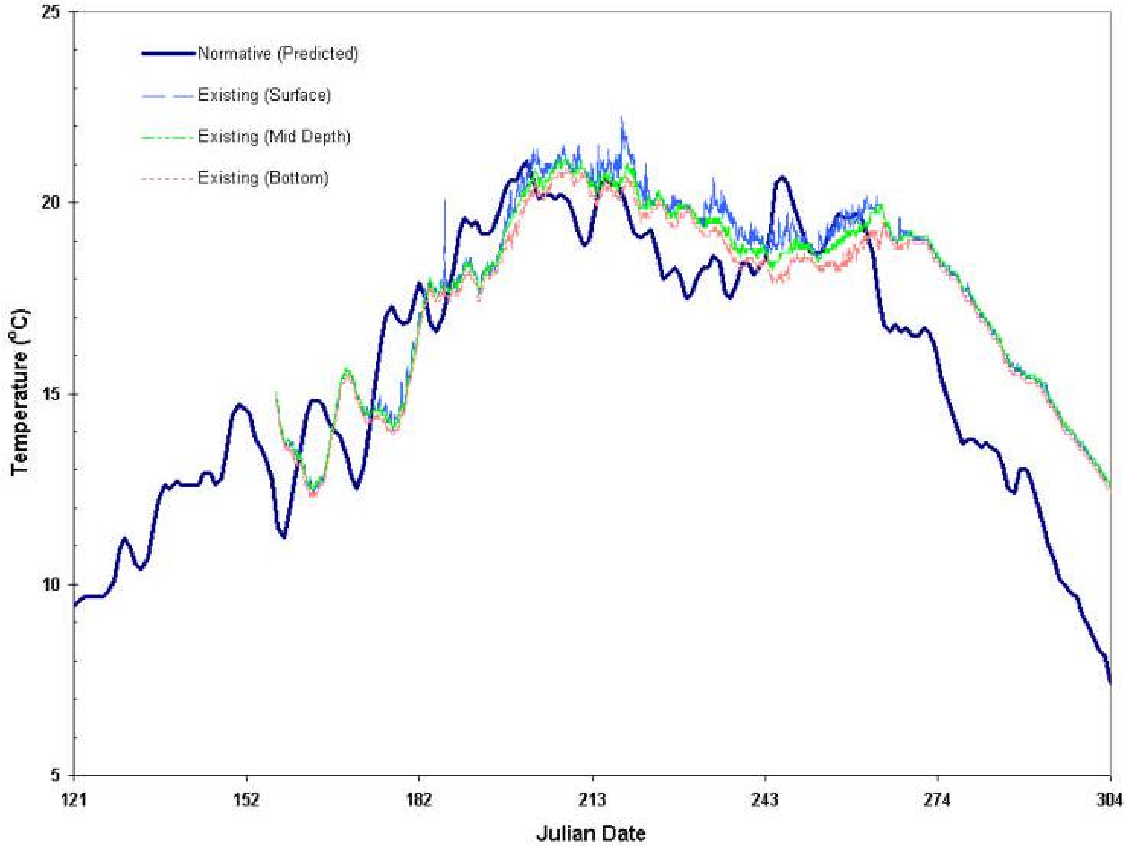

Figure 5.5-7. 1995 water temperatures at Snake River mile 44.0

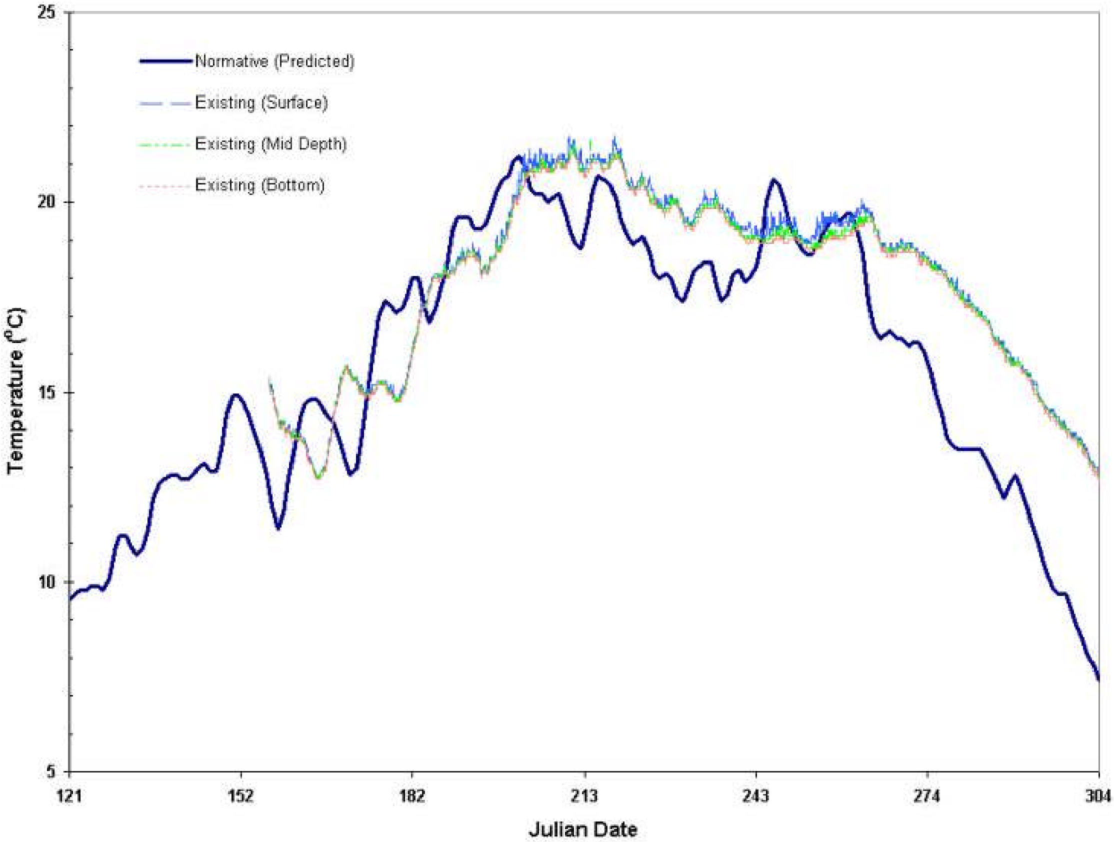

Figure 5.5-8. 1995 water temperatures at Snake River mile 15.5

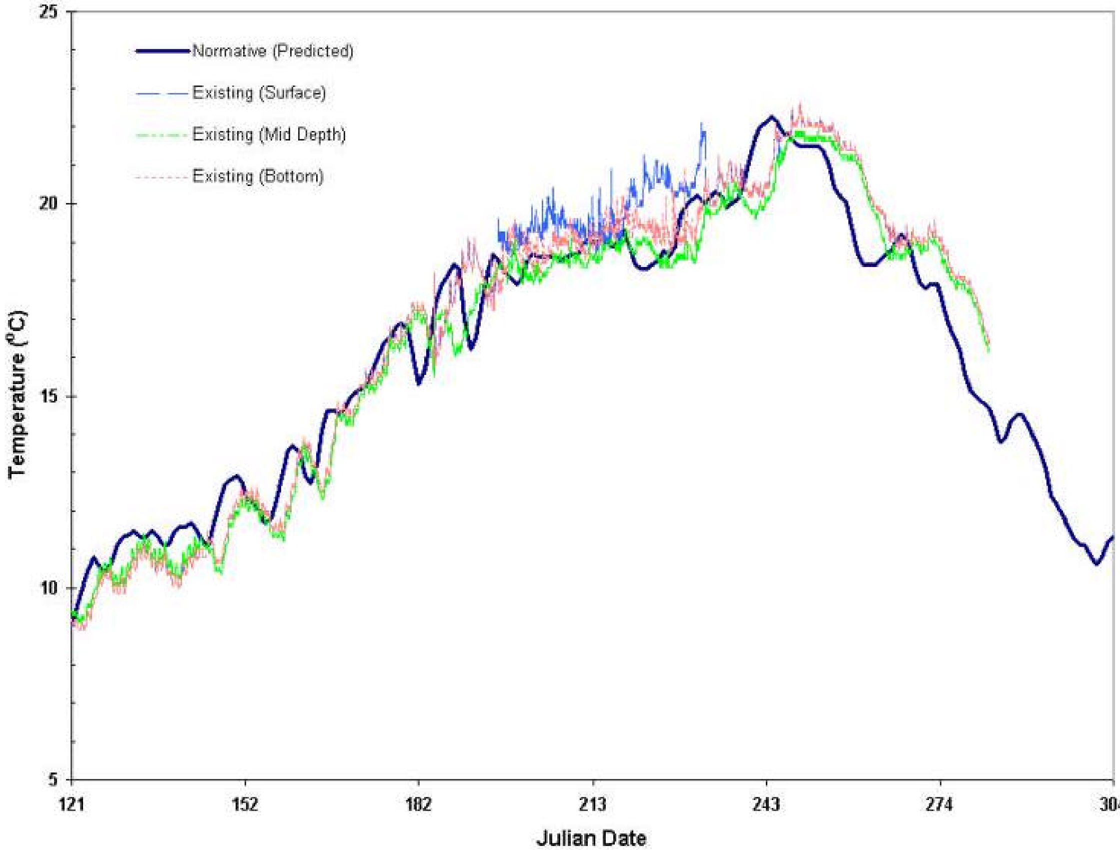

Figure 5.5-9. 1997 water temperatures at Snake River mile 110.5

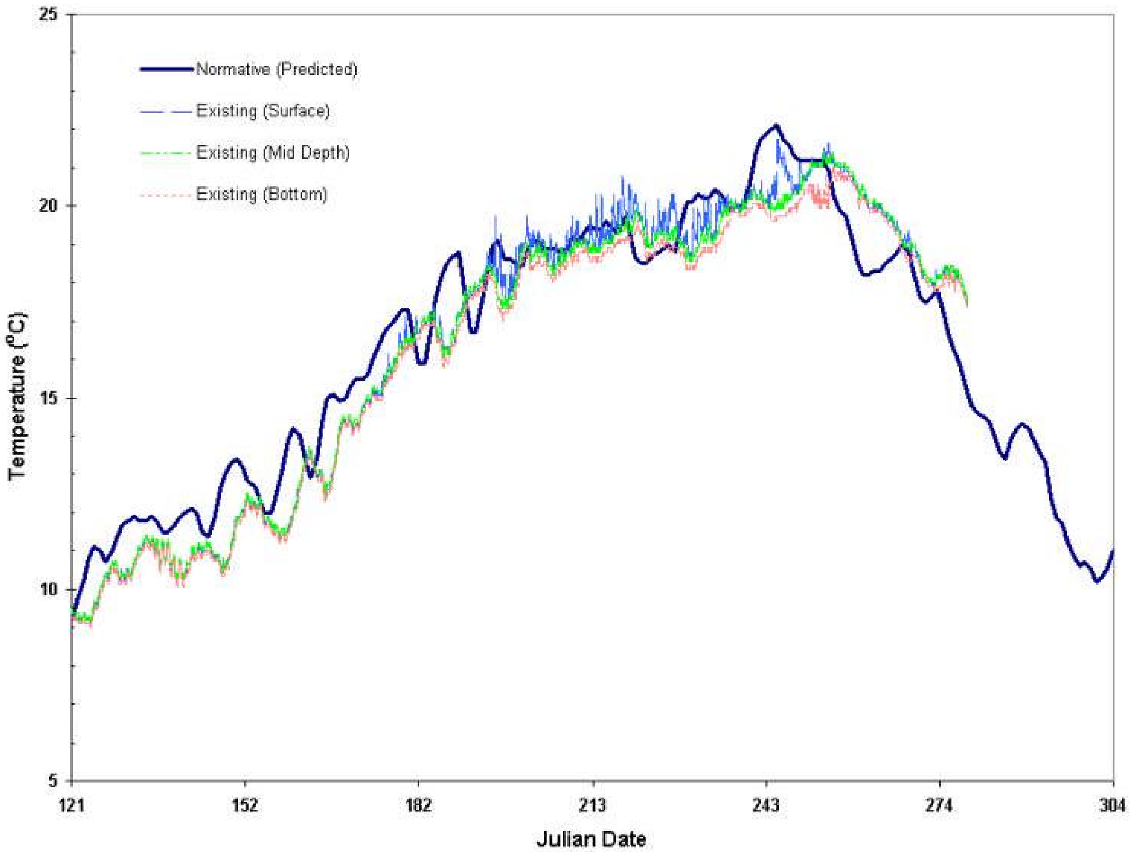

Figure 5.5-10. 1997 water temperatures at Snake River mile 80.5

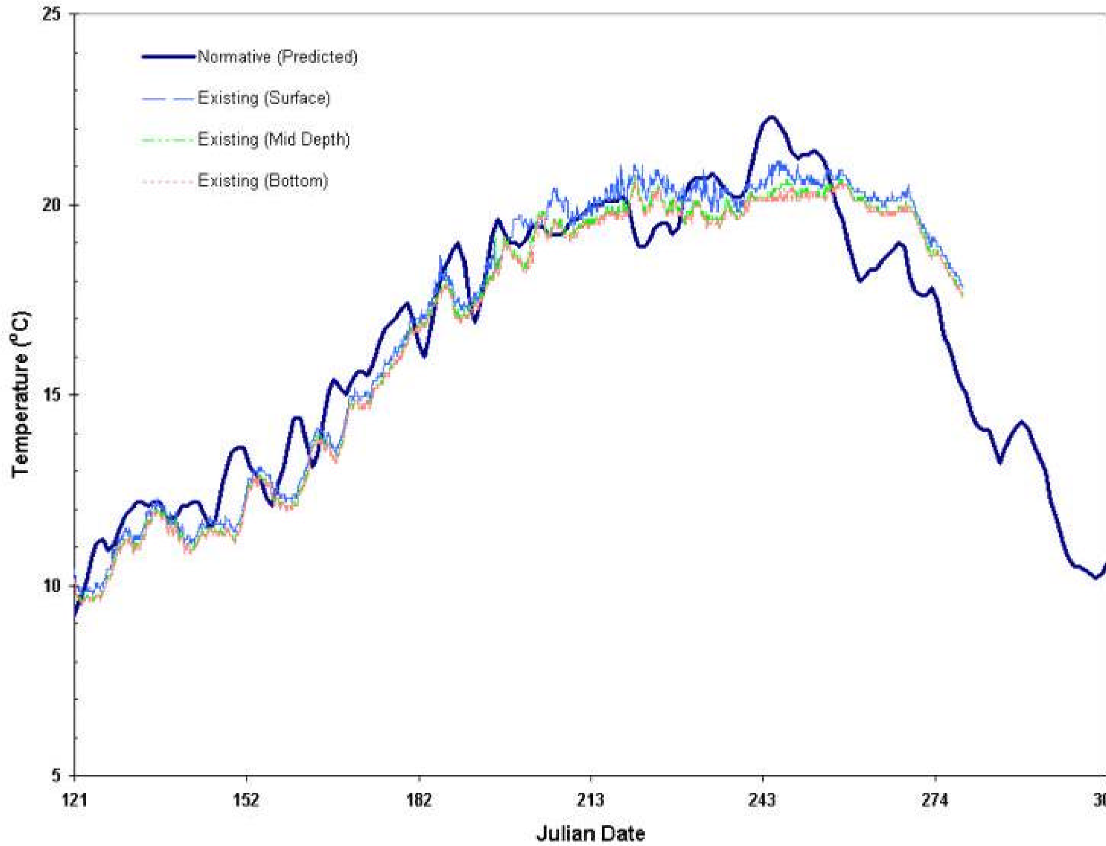

Figure 5.5-11. 1997 water temperatures at Snake River mile 44.15

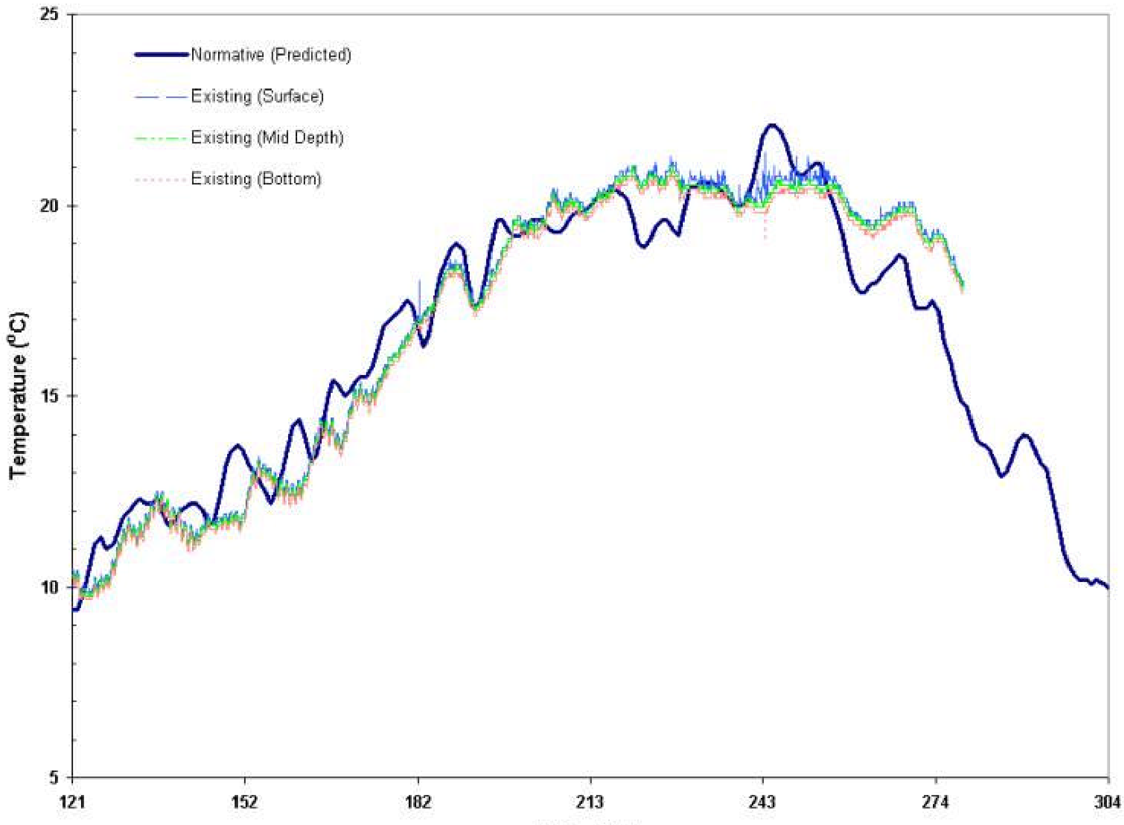

Figure 5.5-12. 1997 water temperatures at Snake River mile 15.94

5.5.3 - Biological Productivity Modeling Results

The risk of misinterpreting the biologic productivity model results is great. The model cannot be used in a strict predictive application due to the lack of adequate biological calibration data (see Section 5.4.3). However, the model is useful to examine differences in biologic productivity under several different hydrometeorologic scenarios. A relative comparison of biologic productivity during wet, dry, and average years can be made with the model, and may represent an important decision-making tool among others for evaluating the impact of dam removal on the ecology of the lower Snake River. The model may also be useful in the future for evaluating the impact of alterations in the flow regime of the river upstream of the project area on the biologic productivity in the study area. Manipulation of the timing, temperature, and magnitude of flows to the project area is possible given the highly regulated nature of the Snake and Clearwater Rivers upstream of the study area.

An understanding of the physio-chemical configuration of the lower Snake River system is necessary to interpret the results of the primary and secondary productivity simulations. The majority of the nutrients entering the lower Snake River are present at the upstream end of the study area. The Palouse and Tucannon Rivers enter the lower Snake River at roughly river mile 59.4, and account for a slight increase in the ambient nutrient concentrations of the lower Snake River. Other than those inputs and internal recycling of nutrient through respiration and decay processes, there are no other significant inputs of nutrients to the 140-mile study area. The nutrient concentrations of the lower Snake River in 1994, 1995, and 1997 support a trophic classification for the lower Snake River as meso-eutrophic to eutrophic. Primary productivity in a system with a nutrient regime such as this would be expected to be high. Nutrient concentration predictions in the normative river system, while not the focus of this study, generally remained high enough to keep growth of primary producers from being limited by nutrients. The normative system was expected to be highly productive at both the primary and secondary levels.

Interactions between the compartments included in the biologic productivity portion of the model make presentation of results challenging. The model allows fish to move between elements in the model, as would be expected in a natural river system. This movement is reflected in the model results by wide fluctuations of fish abundances in a given computational element over the simulated growing season in response to the availability of preferred food. These fluctuations ripple downward through the food chain, resulting in local extinction of some trophic levels and extensive growth in others. As a result, viewing data over time in a specific computational element provides little insight into river productivity. This element to element variability is best illustrated by viewing snapshots of the entire lower Snake River system on selected days in the simulation. These days were selected to coincide with days presented in figures for the temperature simulations. Figures 5.5-13 to 5.5-44 include snapshots of days 152, 183, 213, and 274 for each of the components of biologic productivity in the lower Snake River. Variability between computational elements for each of the components of biologic productivity are apparent in these figures. In order to quantify differences between productivity between pools and among years, the average standing crop of each biologic component for each pool was calculated on selected days in the simulation. These results are presented in Tables 5.5-1 through 5.5-4. The McNary pool referenced in those tables only refers tot he portion of McNary pool between the base of Ice Harbor Dam and the confluence of the Snake River and the Columbia River.

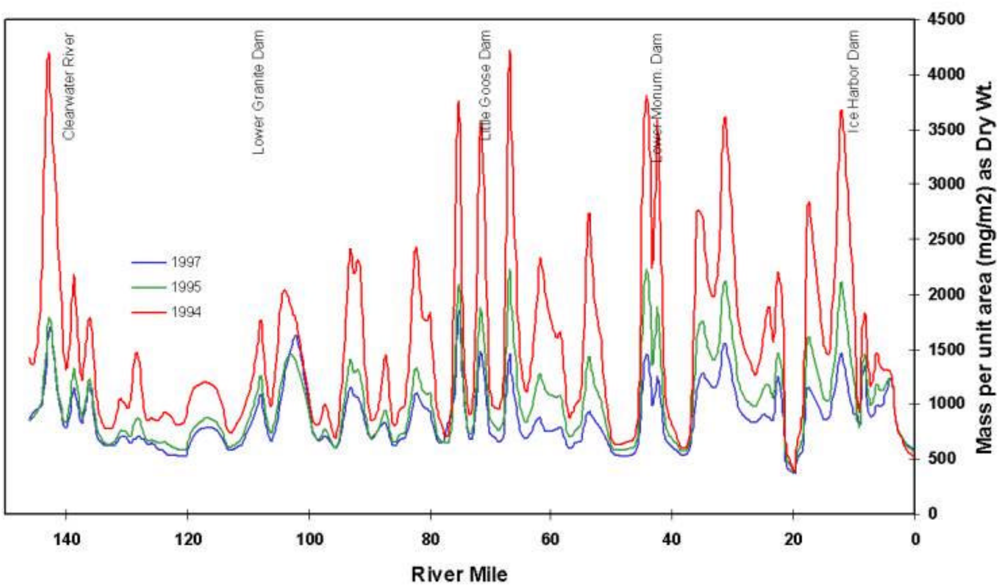

Figure 5.5-13. Predicted Benthic Algae #1 for Day 152

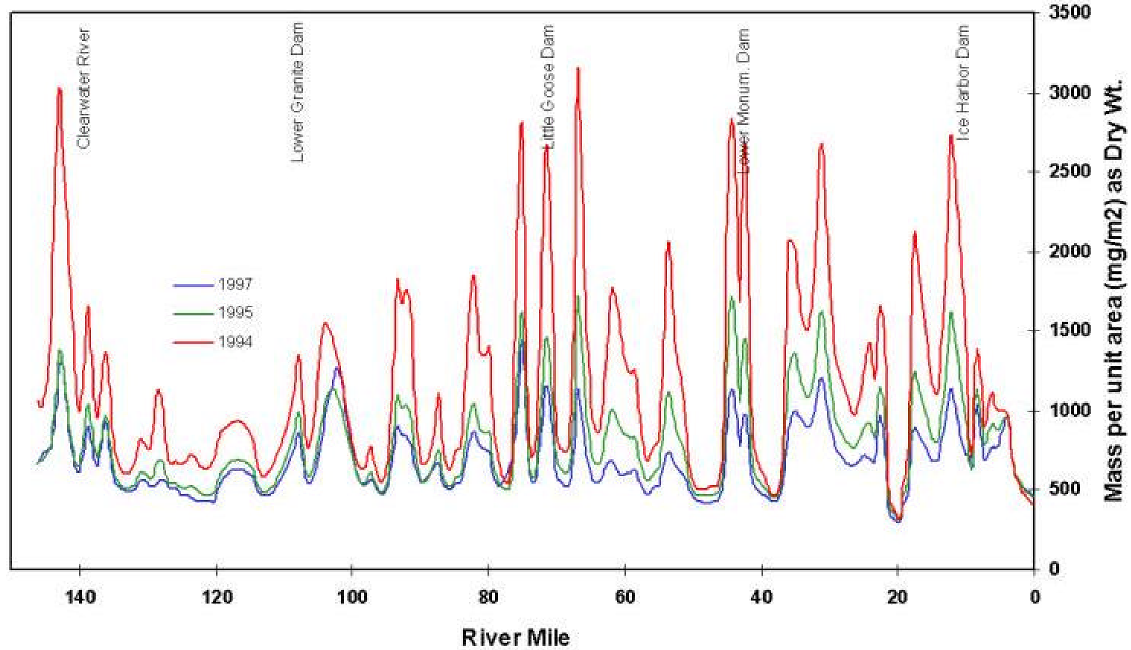

Figure 5.5-14. Predicted Benthic Algae #2 for Day 152

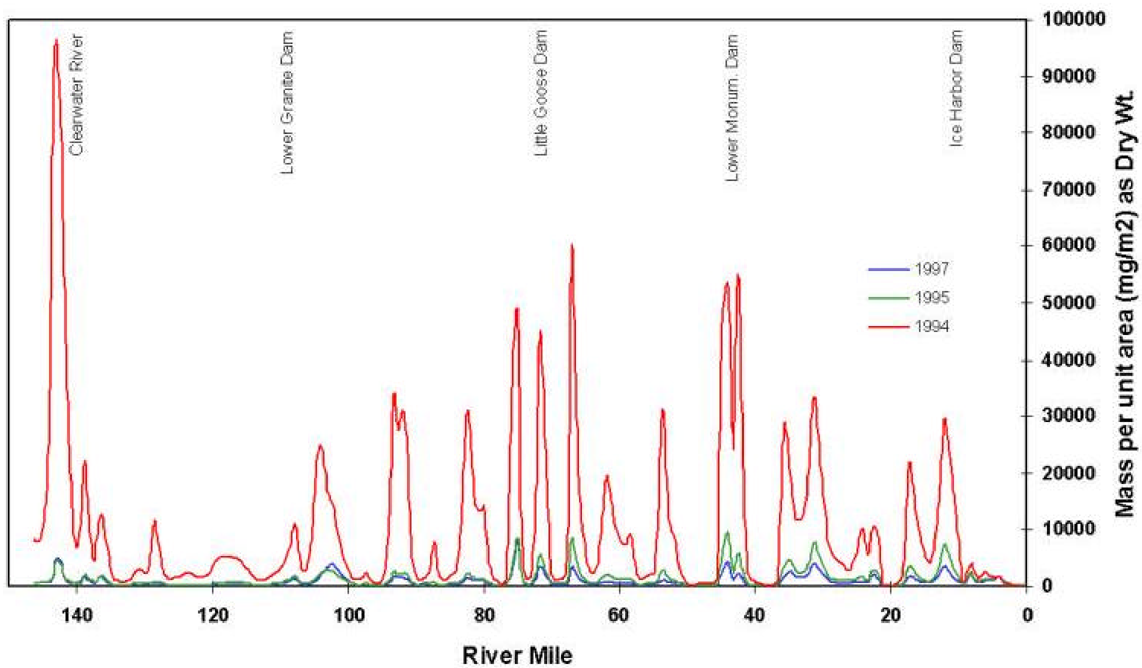

Figure 5.5-15. Predicted Benthic Algae #1 for Day 183

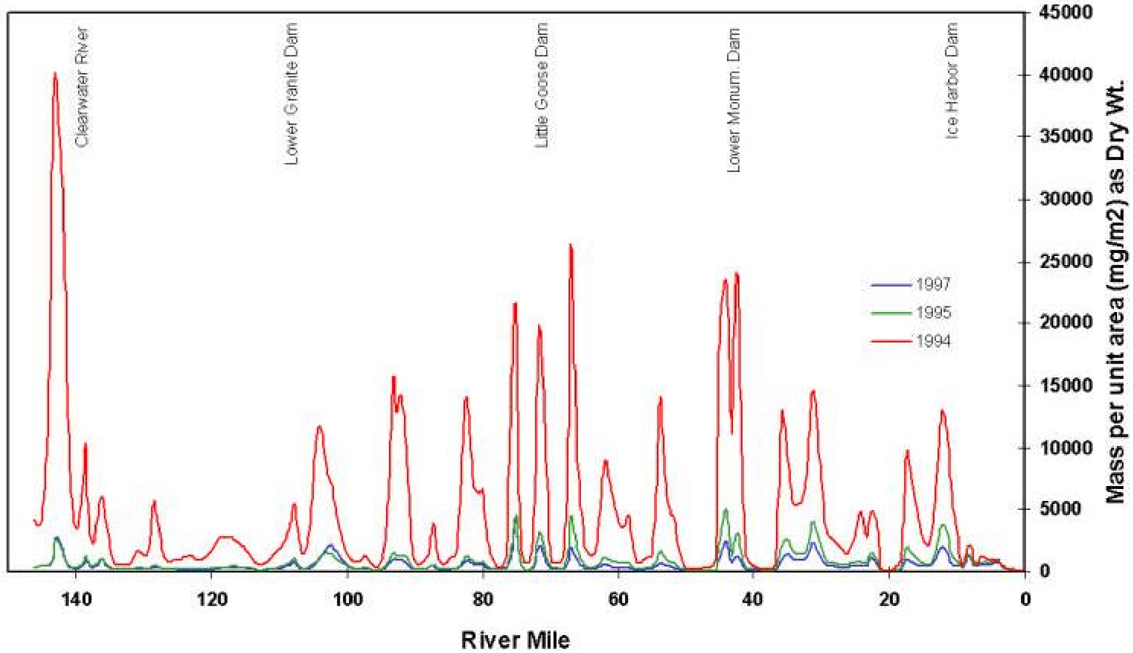

Figure 5.5-16. Predicted Benthic Algae #2 for Day 183

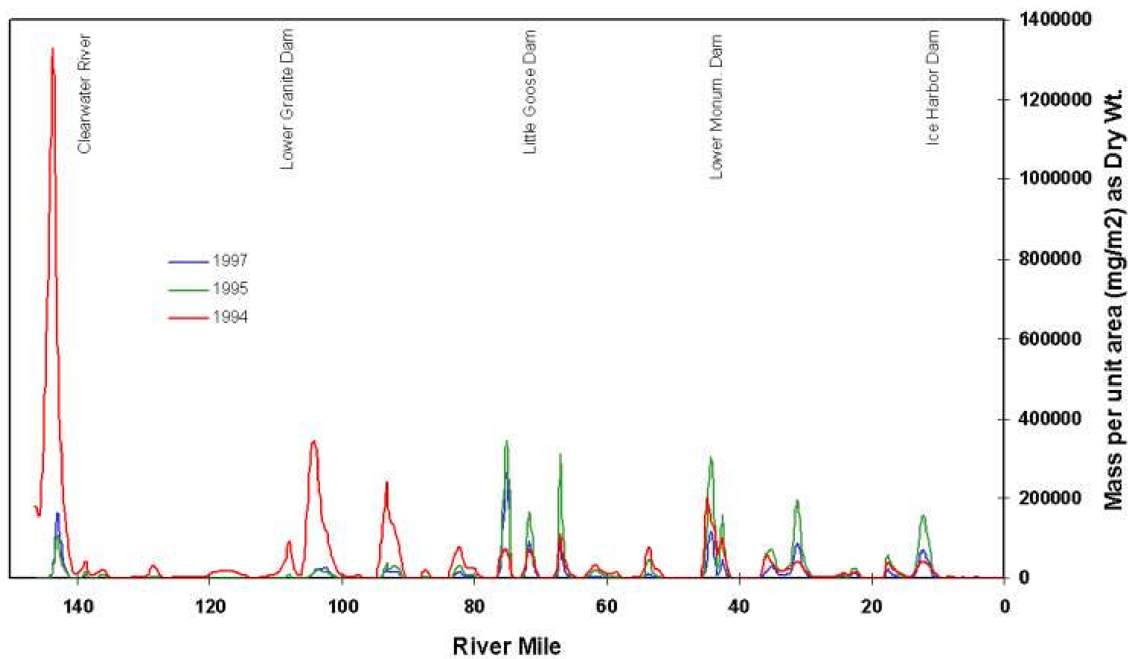

Figure 5.5-17. Predicted Benthic Algae #1 for Day 213

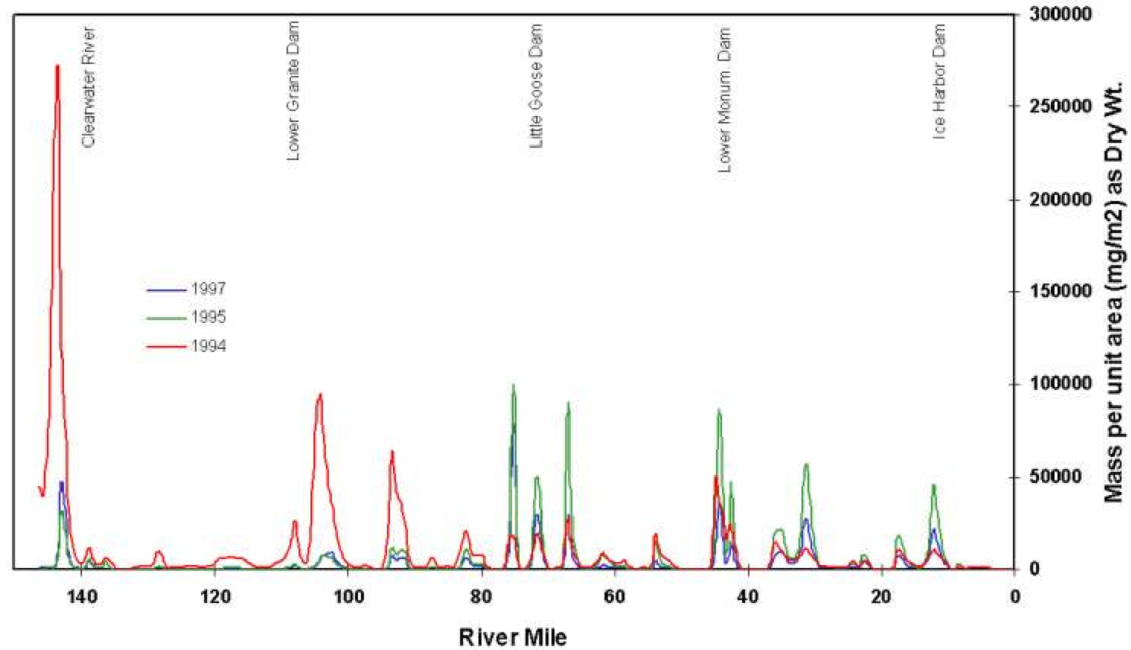

Figure 5.5-18. Predicted Benthic Algae #2 for Day 213

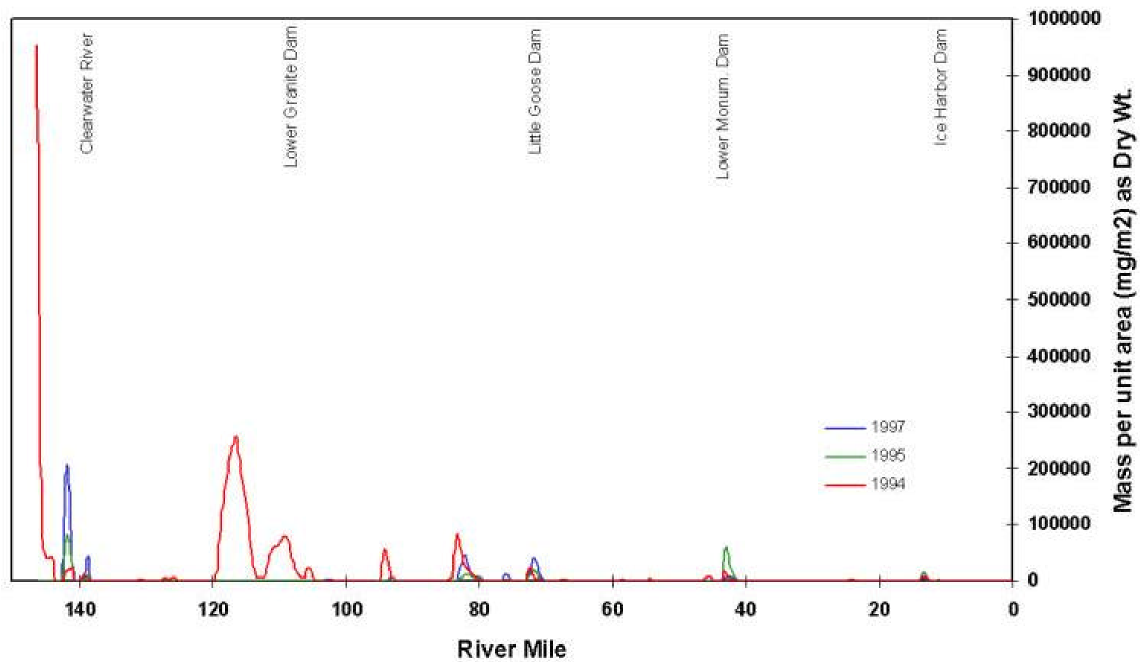

Figure 5.5-19. Predicted Benthic Algae #1 for Day 274

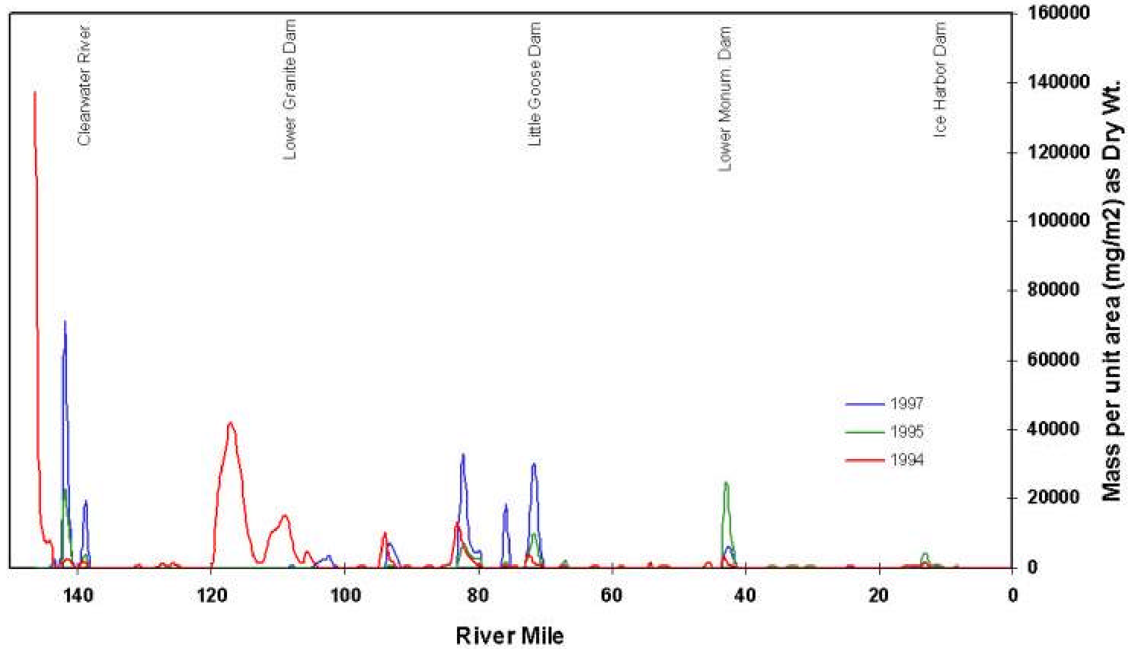

Figure 5.5-20. Predicted Benthic Algae #12 for Day 274

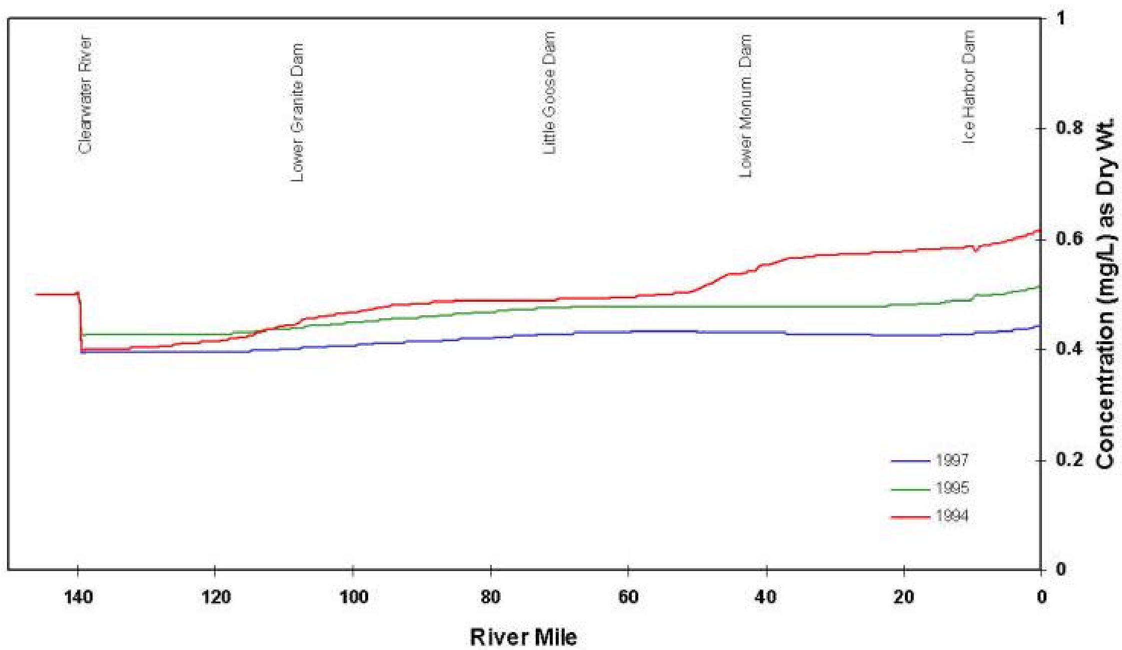

Figure 5.5-21. Predicted Phytoplankton #1 for Day 152

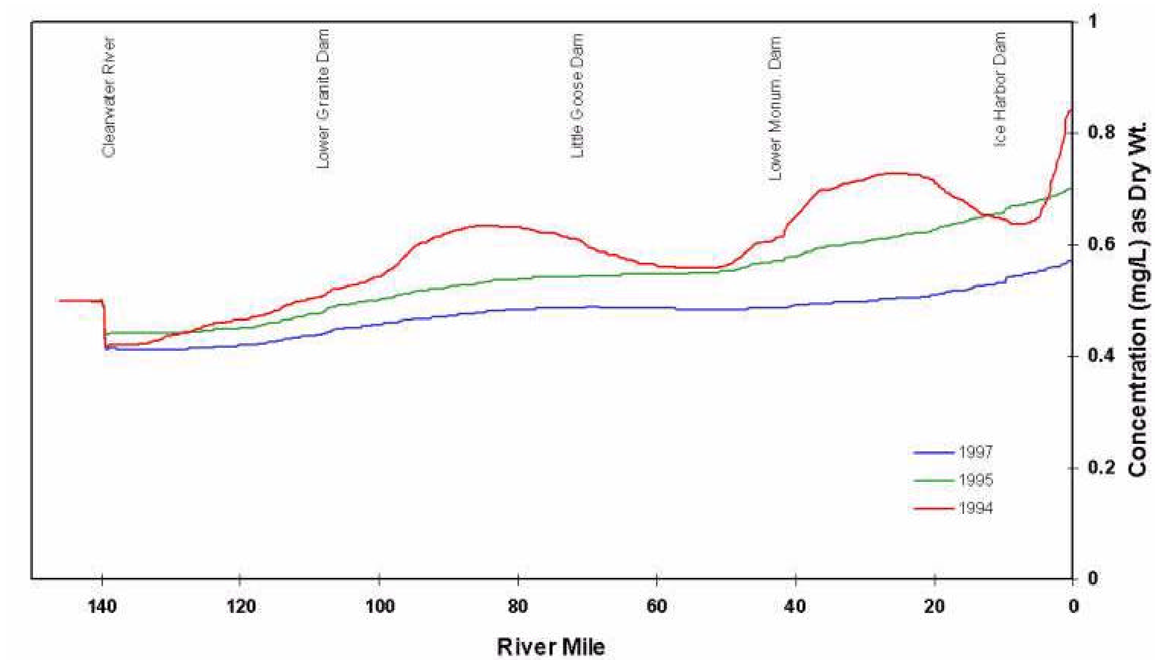

Figure 5.5-22. Predicted Phytoplankton #1 for Day 183

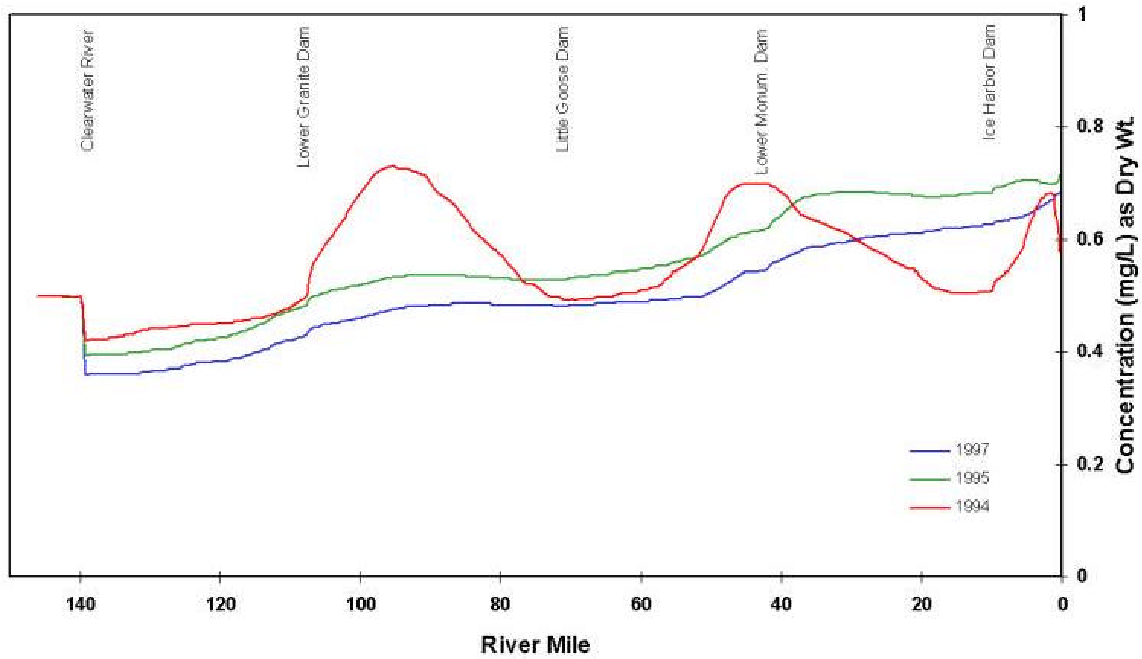

Figure 5.5-23. Predicted Phytoplankton #1 for Day 213

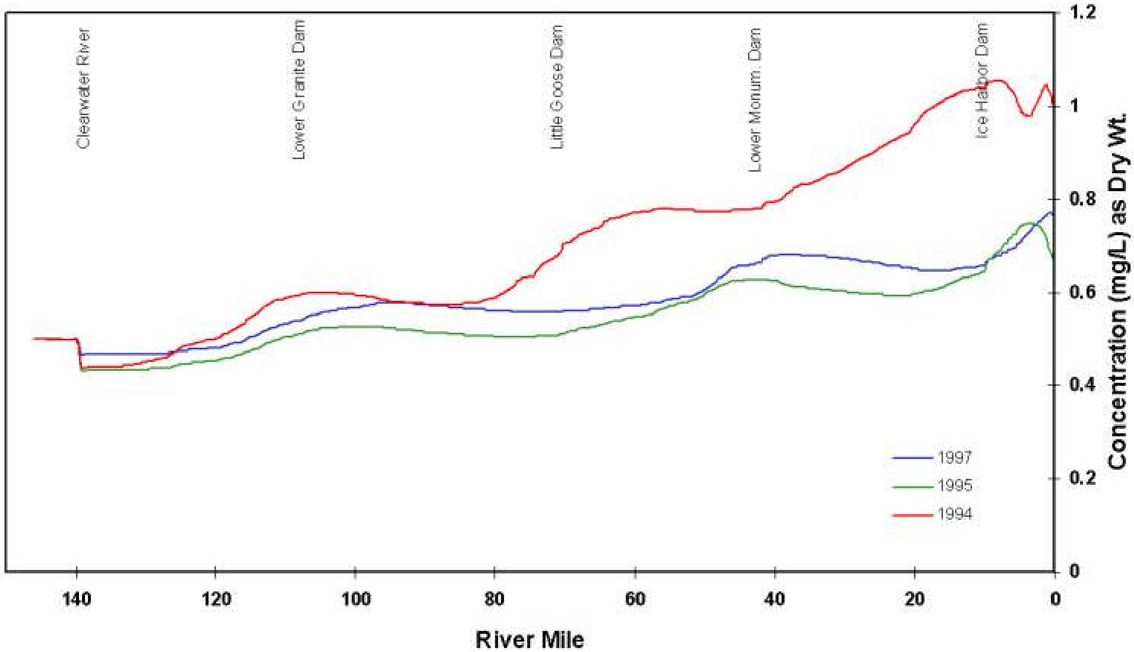

Figure 5.5-24. Predicted Phytoplankton #1 for Day 274

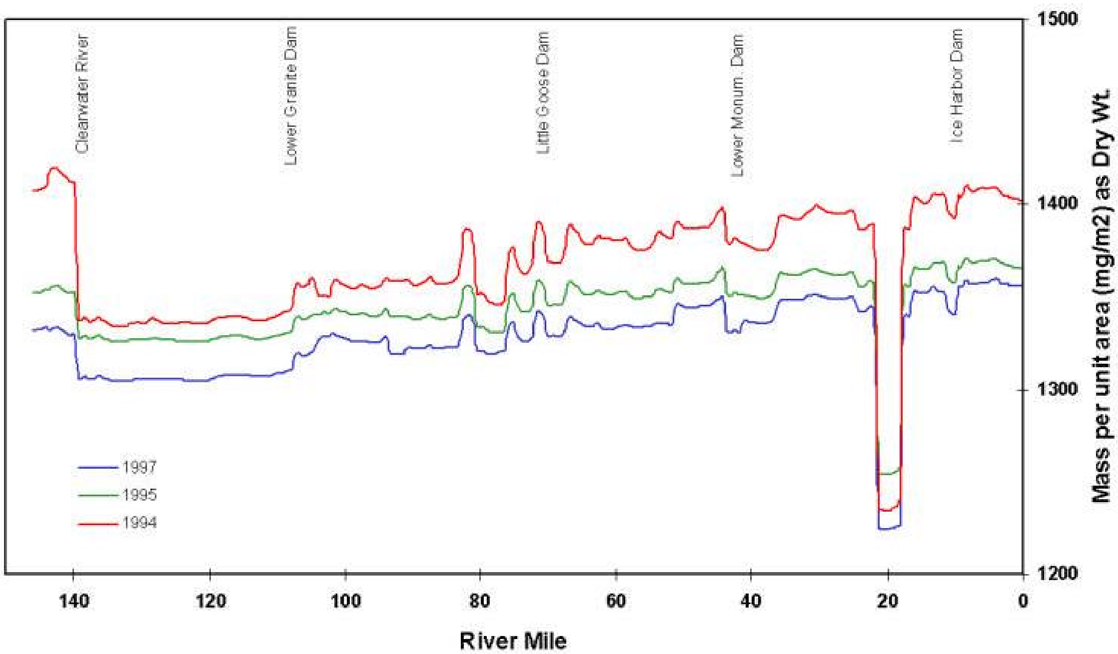

Figure 5.5-25. Predicted Aquatic Insects for Day 152

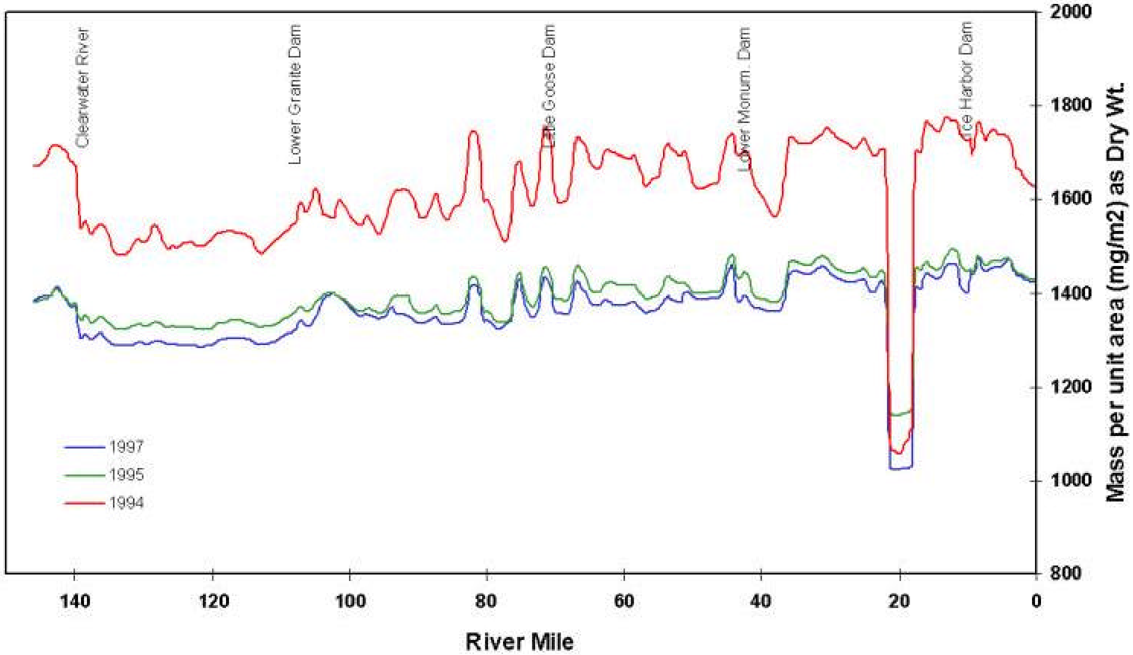

Figure 5.5-26. Predicted Aquatic Insects for Day 183

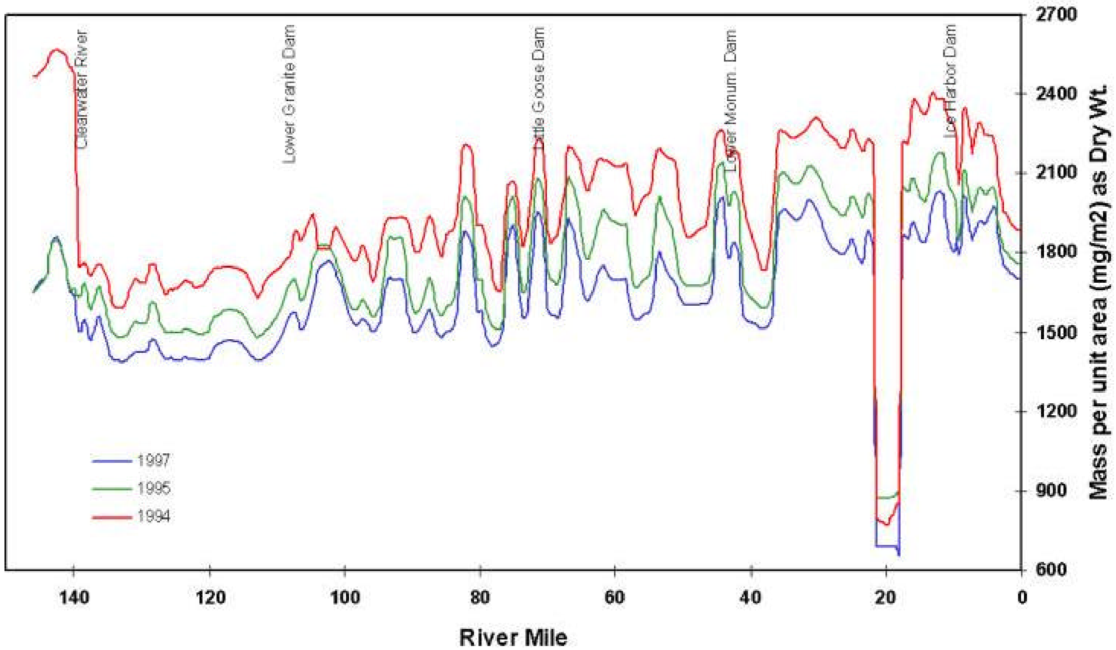

Figure 5.5-27. Predicted Aquatic Insects for Day 213

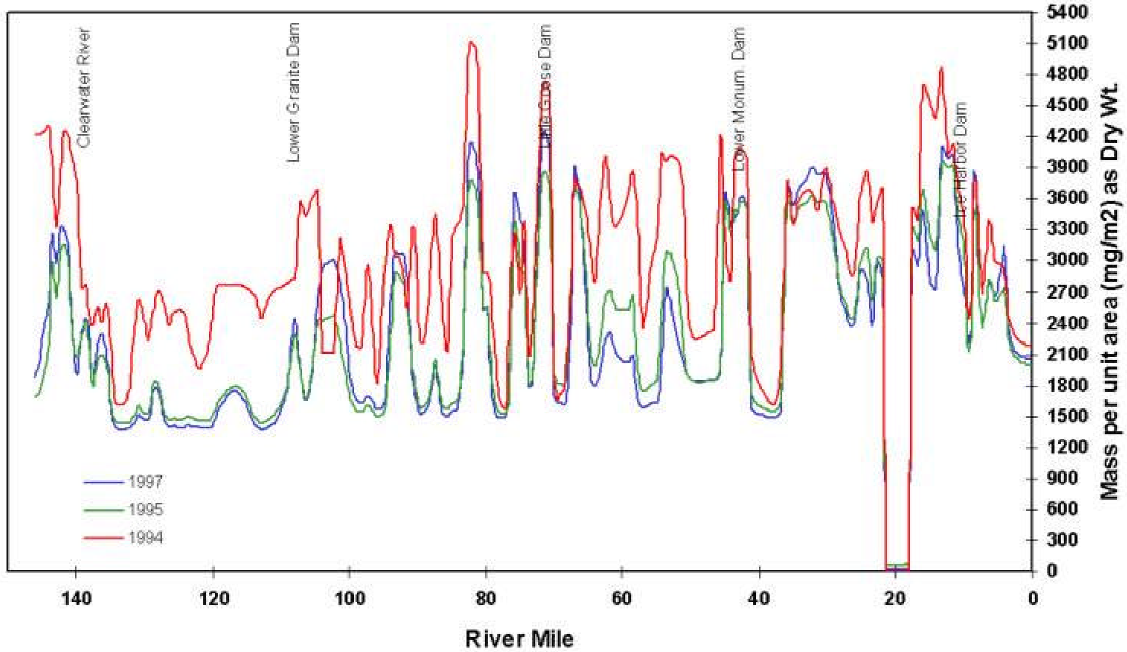

Figure 5.5-28. Predicted Aquatic Insects for Day 274

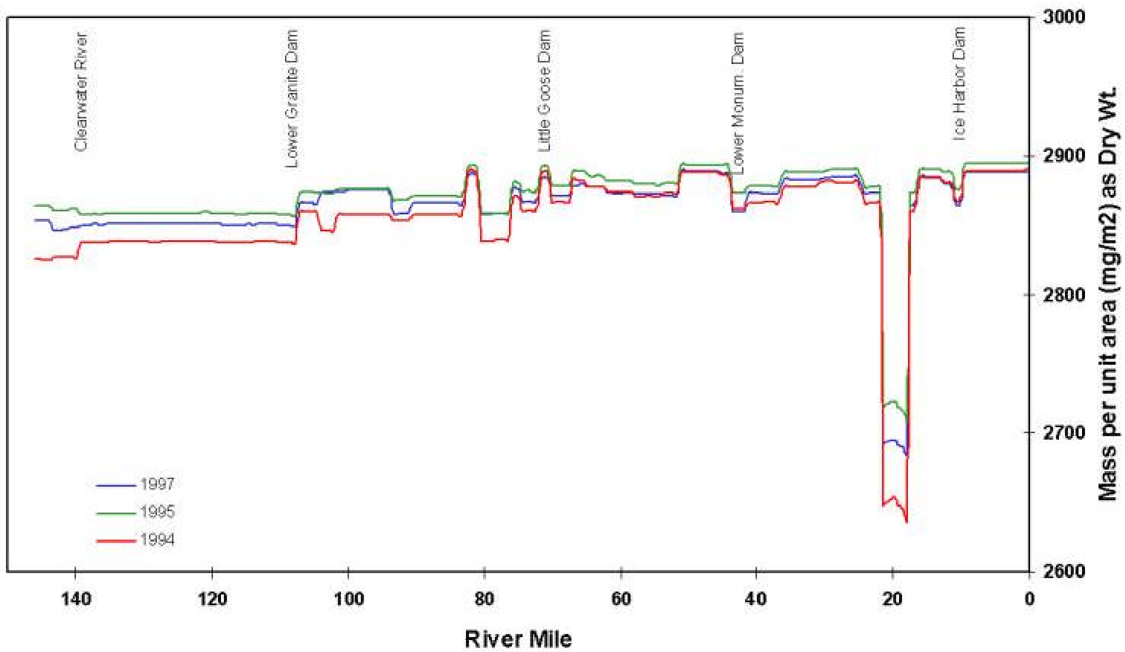

Figure 5.5-29. Predicted Benthic Animals for Day 152

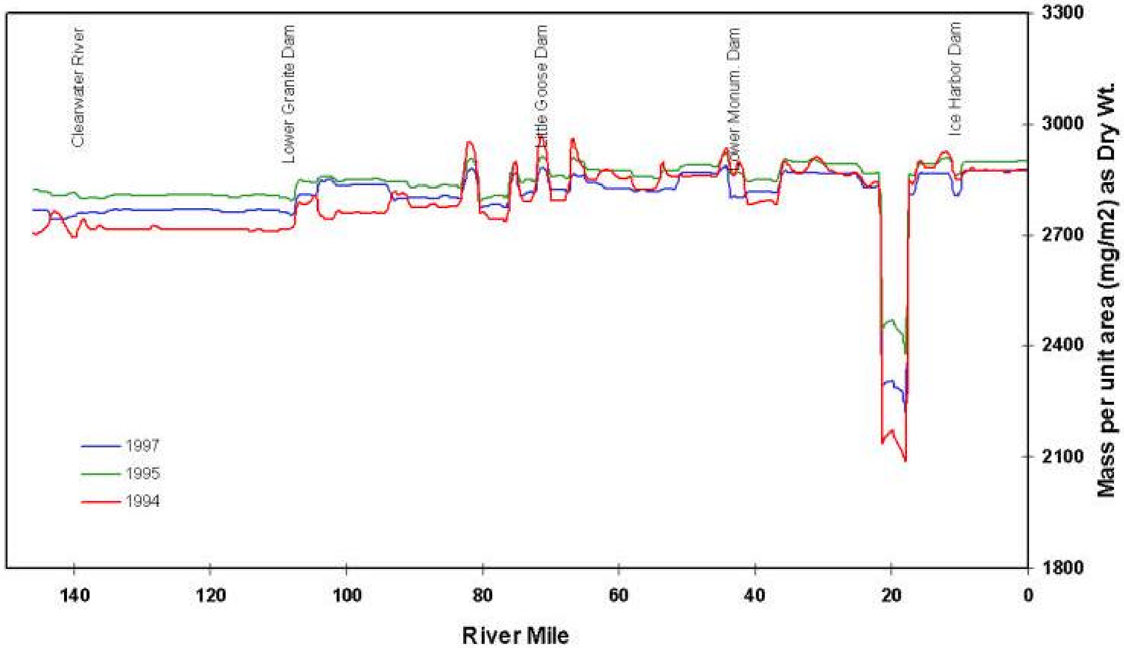

Figure 5.5-30. Predicted Benthic Animals for Day 183

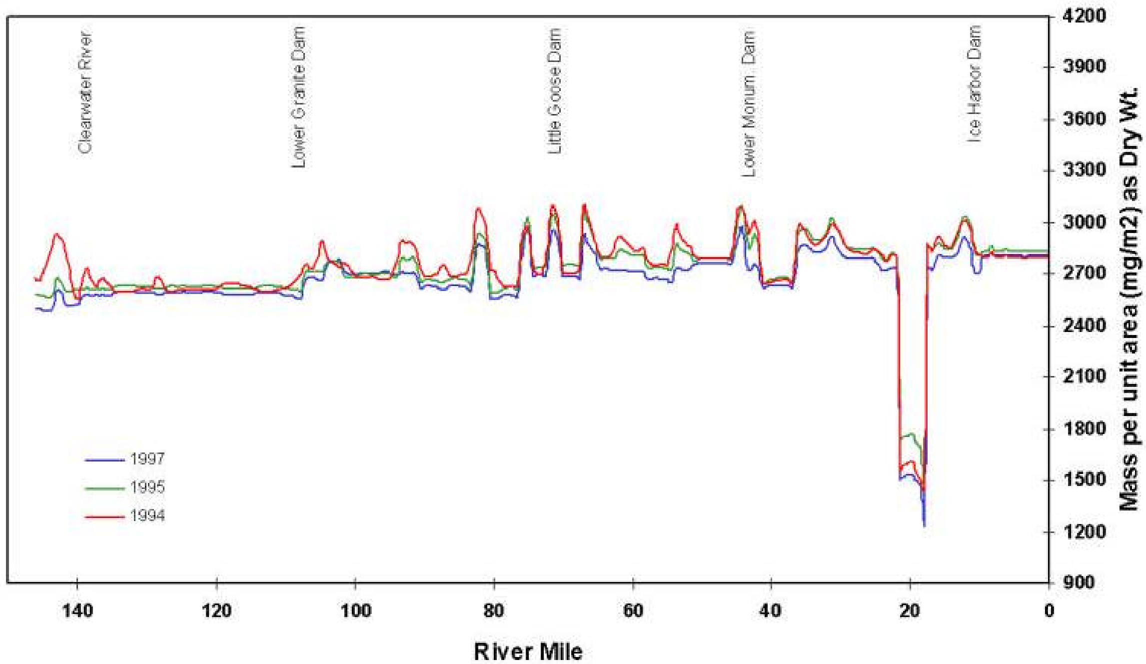

Figure 5.5-31. Predicted Benthic Animals for Day 213

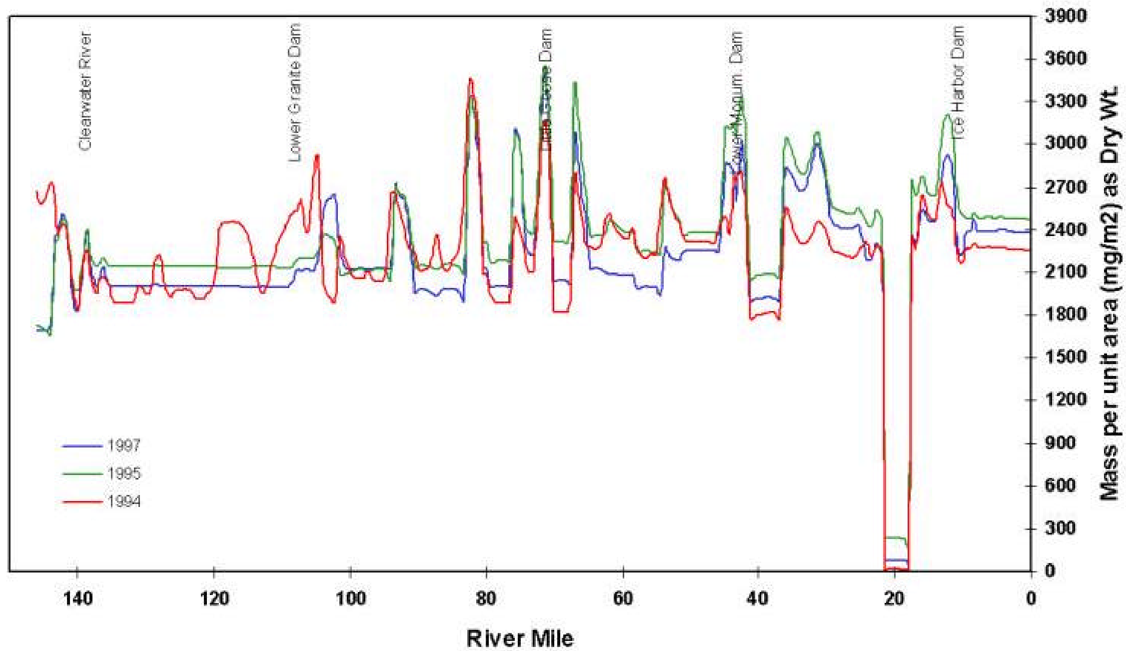

Figure 5.5-32. Predicted Benthic Animals for Day 274

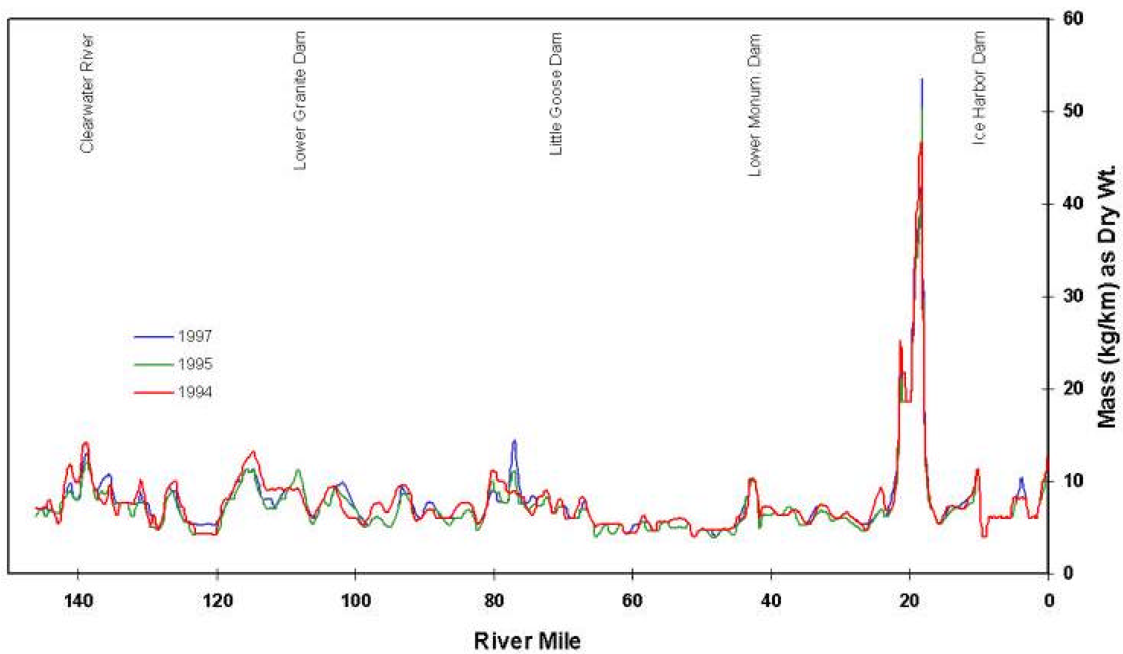

Figure 5.5-33. Predicted Fish #1 for Day 152

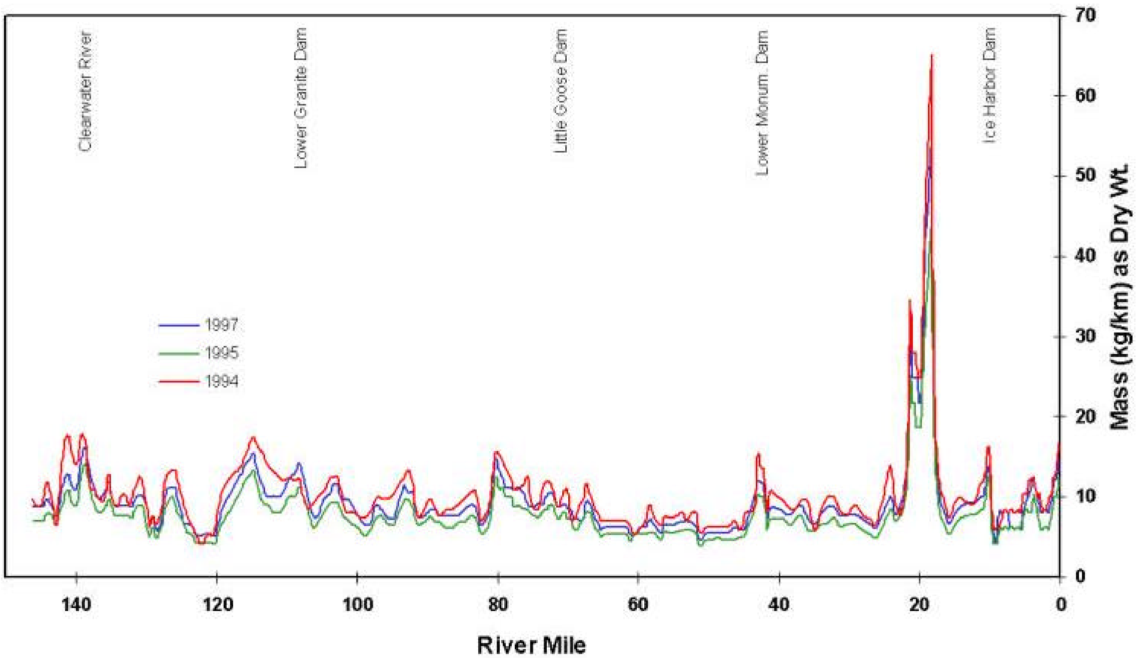

Figure 5.5-34. Predicted Fish #1 for Day 183

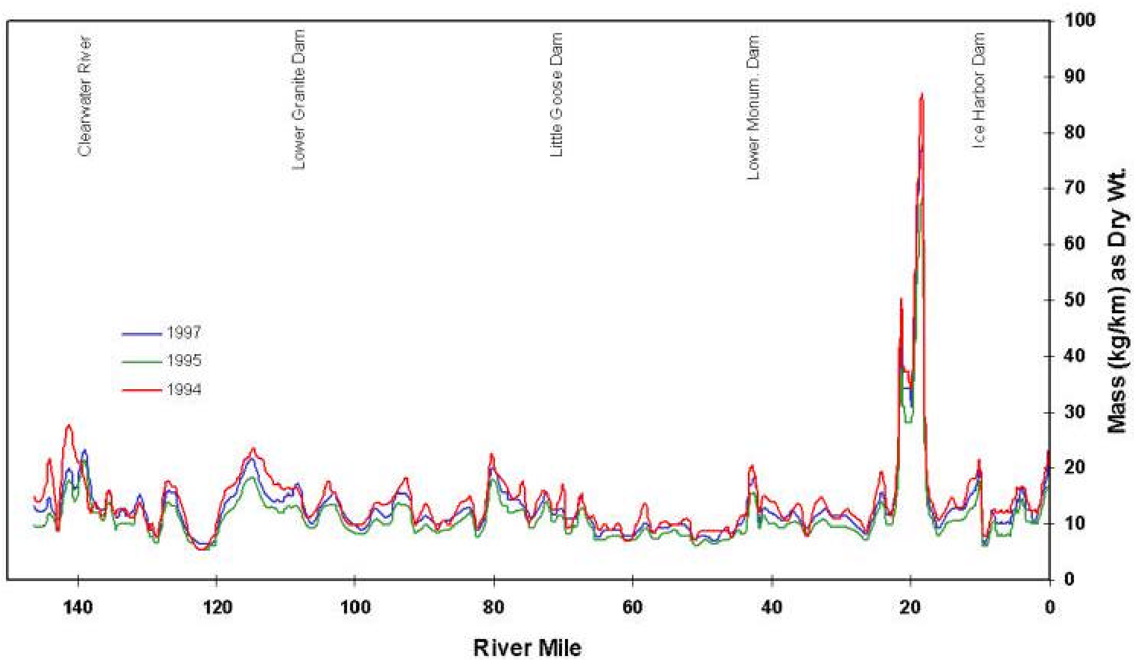

Figure 5.5-35. Predicted Fish #1 for Day 213

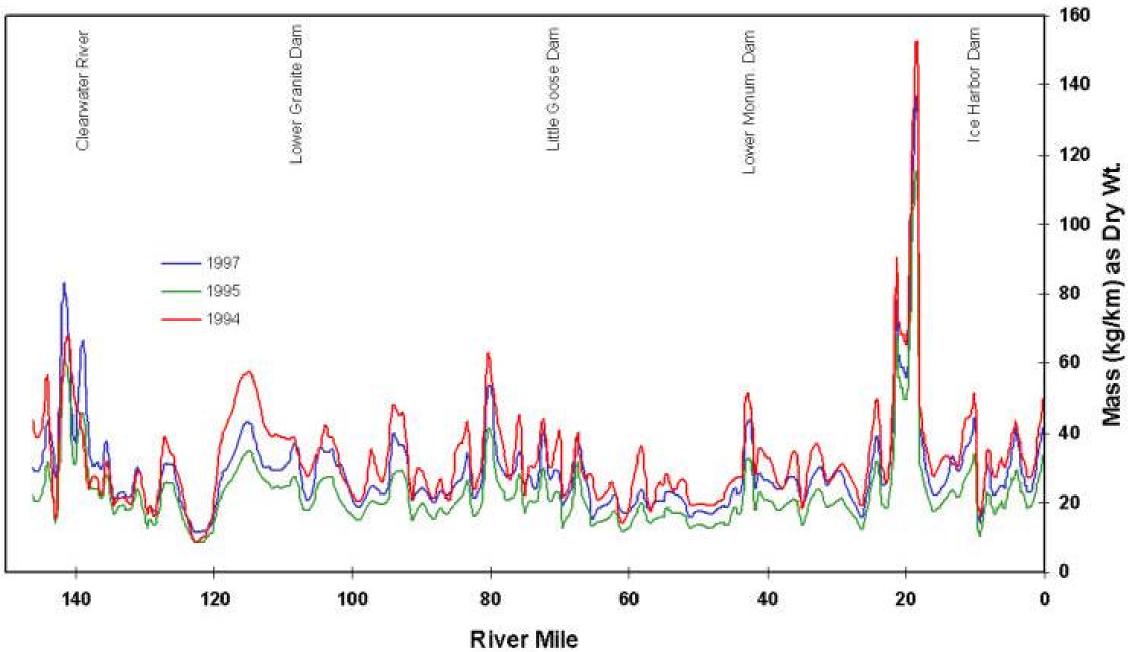

Figure 5.5-36. Predicted Fish #1 for Day 274

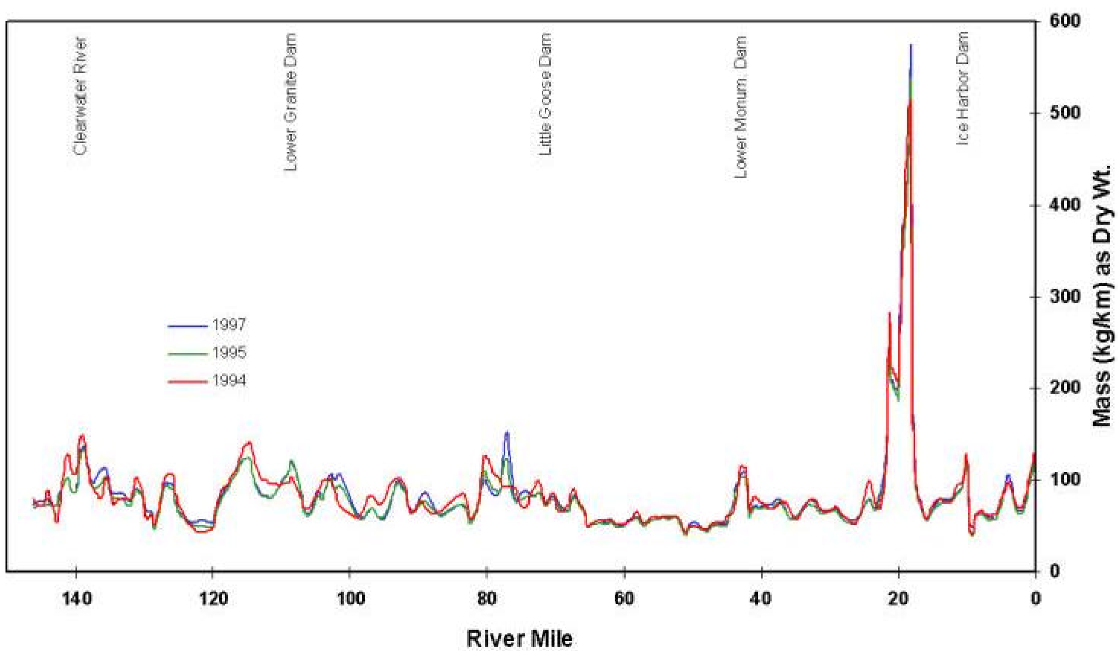

Figure 5.5-37. Predicted Fish #2 for Day 152

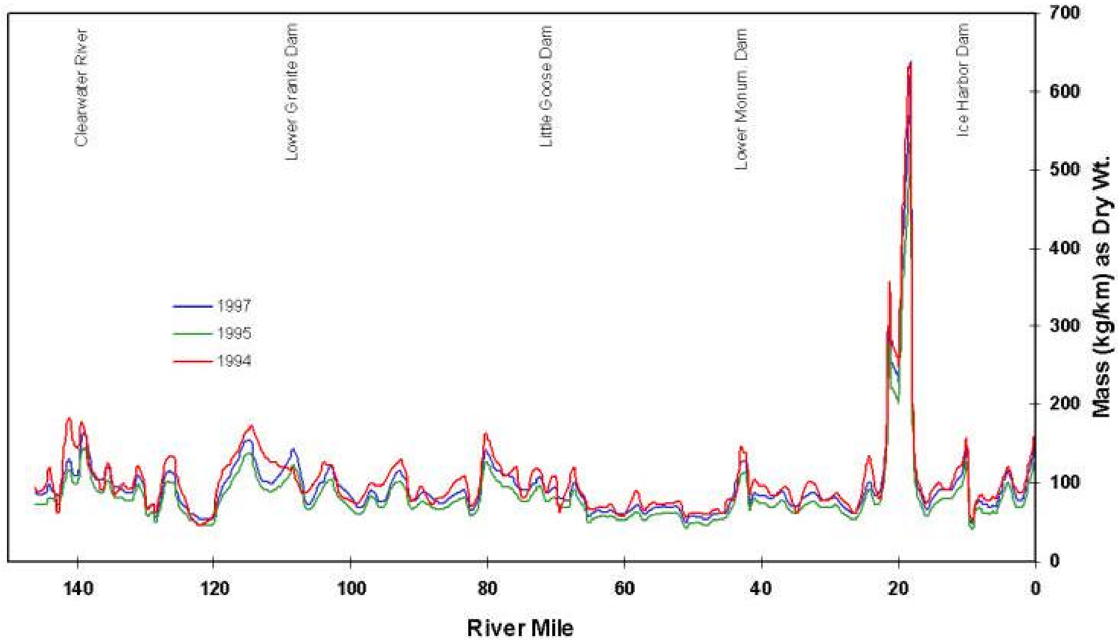

Figure 5.5-38. Predicted Fish #2 for Day 183

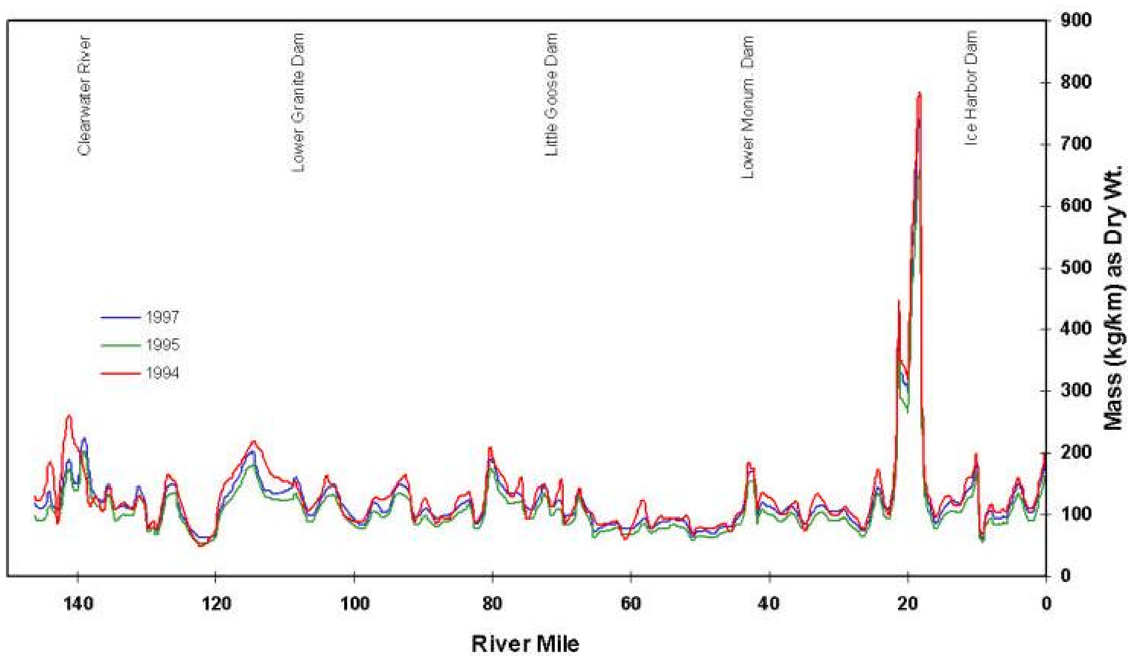

Figure 5.5-39. Predicted Fish #2 for Day 213

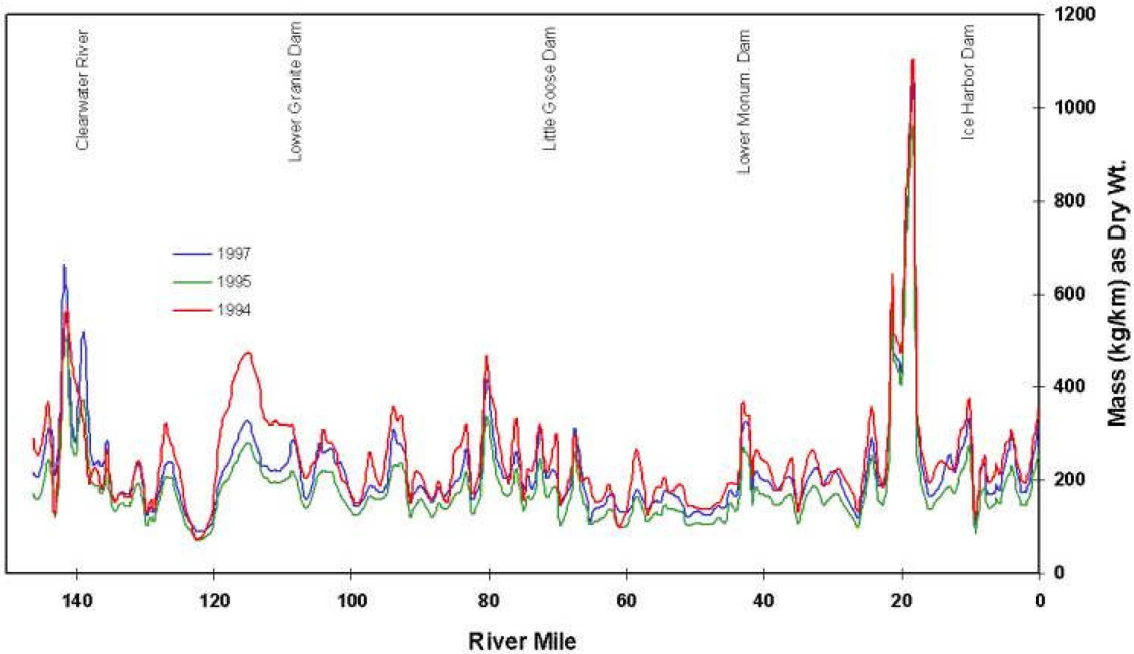

Figure 5.5-40. Predicted Fish #2 for Day 274

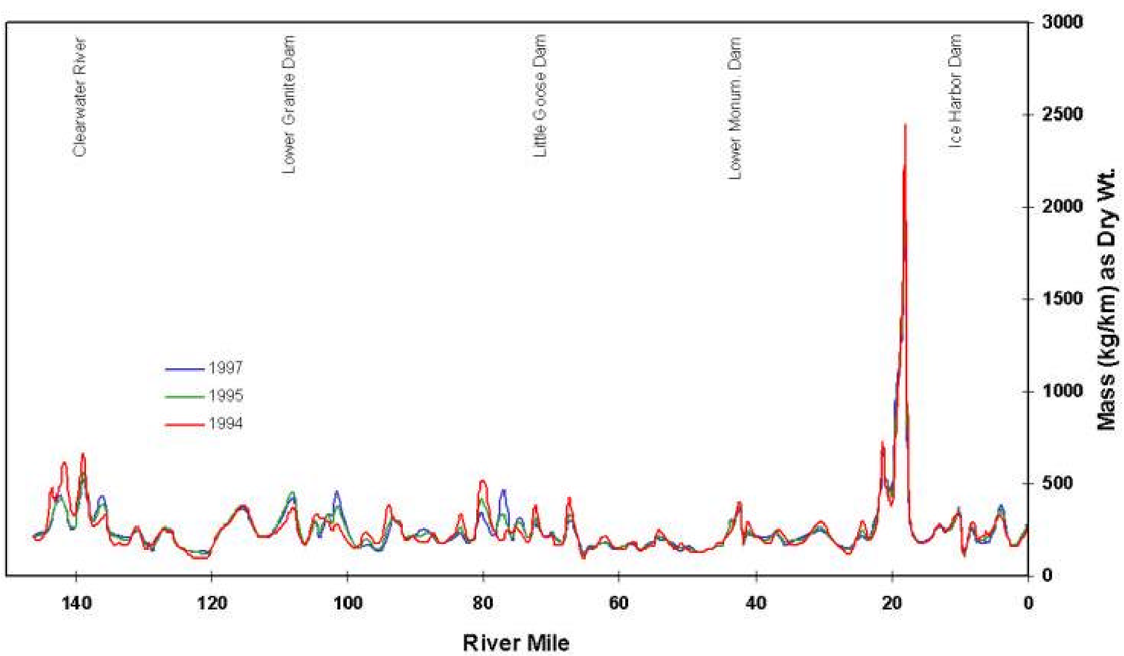

Figure 5.5-41. Predicted Fish #3 for Day 152

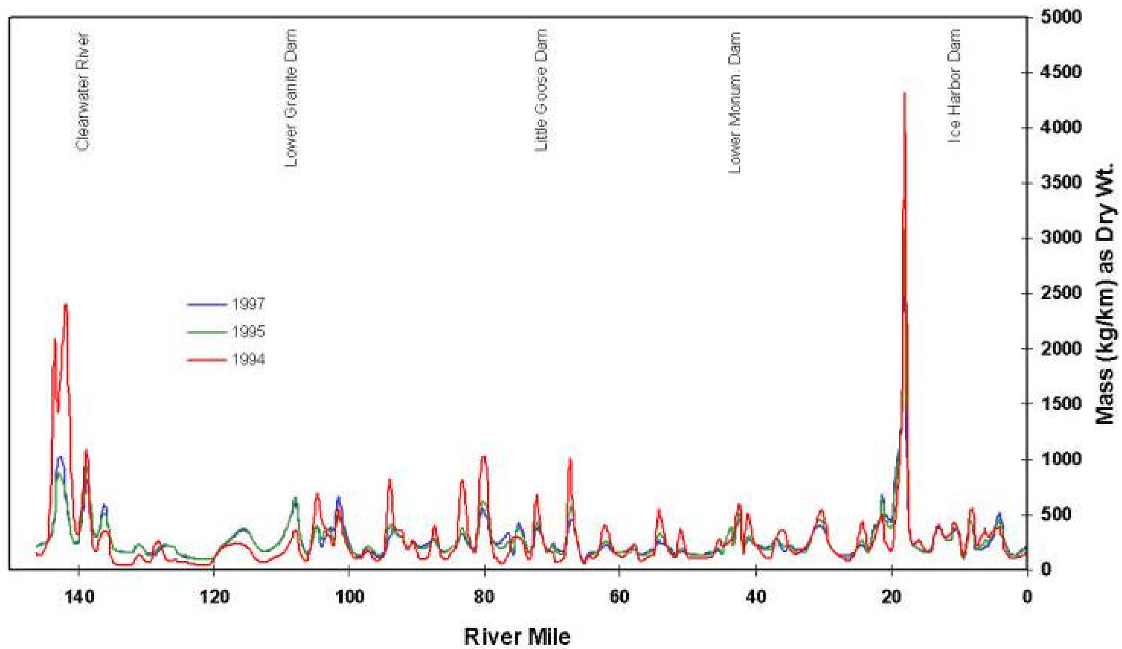

Figure 5.5-42. Predicted Fish #3 for Day 183

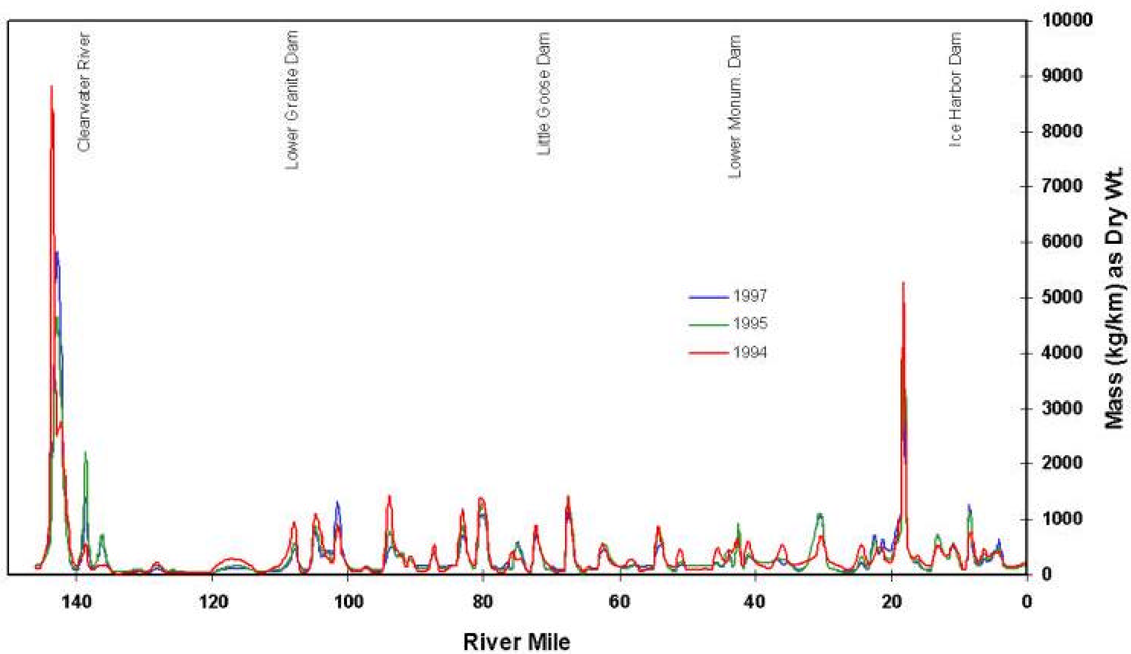

Figure 5.5-43. Predicted Fish #3 for Day 213

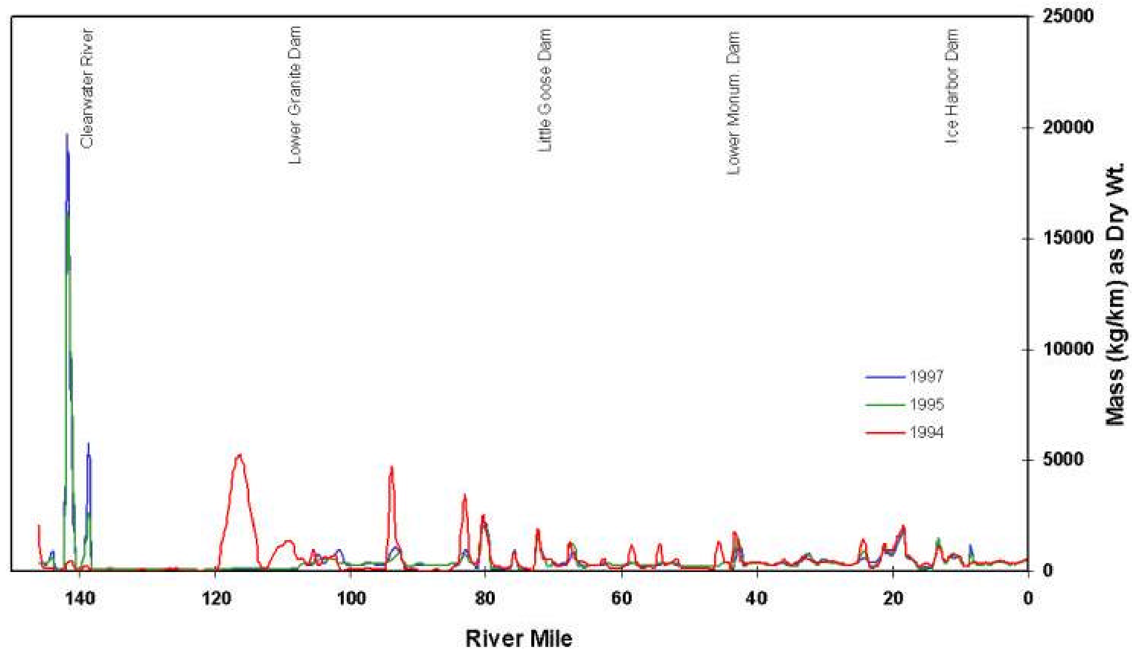

Figure 5.5-44. Predicted Fish #3 for Day 274

Table 5.5-1
Average Biomass Simulations by Lower Snake River Pool for Day 152

Pool	Year	Phytoplankton (mg/l)	Benthic Algae #1 (mg/m^2)	Benthic Algae #2 (mg/m^2)	Aquatic Insects (mg/m^2)	Benthic Animals (mg/m^2)	Fish #1 (kg/km)	Fish #2 (kg/km)	Fish #3 (kg/km)
Lower Granite	1994	0.43	1,295	988	1,350	2,836	8	90	272
	1995	0.44	847	670	1,332	2,859	8	85	269
	1997	0.41	782	620	1,310	2,850	8	89	269
Little Goose	1994	0.48	1,489	1,141	1,359	2,857	8	83	250
	1995	0.46	1,011	797	1,341	2,873	7	79	248
	1997	0.41	940	742	1,325	2,869	7	82	247
Lower Monumental	1994	0.51	1,768	1,343	1,381	2,875	6	63	190
	1995	0.48	1,117	875	1,353	2,883	5	60	188
	1997	0.43	833	657	1,336	2,875	6	62	187
Ice Harbor	1994	0.57	1,537	1,164	1,336	2,793	10	105	314
	1995	0.48	1,039	812	1,322	2,824	9	99	311
	1997	0.43	831	652	1,302	2,811	10	103	310
McNary	1994	0.60	1,103	846	1,406	2,889	7	78	233
	1995	0.50	978	767	1,368	2,894	7	73	231
	1997	0.43	918	719	1,357	2,888	7	76	230

Table 5.5-2 Average Biomass Simulations by Lower Snake River Pool for Day 183									
Pool	**Year**	**Phytoplankton (mg/l)**	**Benthic Algae #1 (mg/m^2)**	**Benthic Algae #2 (mg/m^2)**	**Aquatic Insects (mg/m^2)**	**Benthic Animals (mg/m^2)**	**Fish #1 (kg/km)**	**Fish #2 (kg/km)**	**Fish #3 (kg/km)**
Lower Granite	1994	0.46	10,018	4,625	1,544	2,717	11	112	318
	1995	0.46	784	472	1,342	2,807	8	90	292
	1997	0.43	697	417	1,312	2,763	10	103	292
Little Goose	1994	0.60	11,146	5,274	1,597	2,790	10	103	290
	1995	0.52	1,392	807	1,379	2,843	8	83	269
	1997	0.47	1,228	711	1,360	2,817	9	95	269
Lower Monumental	1994	0.58	14,218	6,494	1,675	2,858	8	78	221
	1995	0.55	1,918	1,079	1,417	2,875	6	63	205
	1997	0.49	911	540	1,385	2,835	7	72	203
Ice Harbor	1994	0.70	7,693	3,538	1,479	2,600	13	131	366
	1995	0.62	1,656	925	1,336	2,723	10	105	340
	1997	0.51	946	547	1,277	2,643	12	120	338
McNary	1994	0.69	1,400	757	1,707	2,872	10	98	266
	1995	0.68	994	588	1,456	2,898	7	78	252
	1997	0.55	873	519	1,448	2,874	9	90	253

Table 5.5-3
Average Biomass Simulations by Lower Snake River Pool for Day 213

Pool	Year	Phytoplankton (mg/l)	Benthic Algae #1 (mg/m^2)	Benthic Algae #2 (mg/m^2)	Aquatic Insects (mg/m^2)	Benthic Animals (mg/m^2)	Fish #1 (kg/km)	Fish #2 (kg/km)	Fish #3 (kg/km)
Lower Granite	1994	0.46	74,646	17,119	1,837	2,648	15	142	407
	1995	0.44	4,394	1,510	1,581	2,618	12	117	344
	1997	0.40	4,907	1,898	1,489	2,575	14	132	344
Little Goose	1994	0.62	50,632	14,192	1,889	2,762	14	127	341
	1995	0.53	17,811	5,683	1,720	2,723	11	108	316
	1997	0.47	12,814	4,219	1,624	2,690	13	121	312
Lower Monumental	1994	0.57	29,549	7,714	2,075	2,855	11	99	261
	1995	0.57	32,615	9,944	1,862	2,824	9	83	243
	1997	0.50	10,396	3,551	1,703	2,739	10	93	234
Ice Harbor	1994	0.56	10,725	2,838	1,695	2,374	18	166	433
	1995	0.68	19,371	5,951	1,576	2,433	14	138	402
	1997	0.60	8,324	2,813	1,414	2,303	17	155	388
McNary	1994	0.60	887	364	2,134	2,798	14	126	308
	1995	0.70	1,252	526	1,939	2,837	11	104	297
	1997	0.65	1,214	522	1,856	2,805	13	117	293

Table 5.5-4 Average Biomass Simulations by Lower Snake River Pool for Day 274									
Pool	**Year**	**Phytoplankton (mg/l)**	**Benthic Algae #1 (mg/m^2)**	**Benthic Algae #2 (mg/m^2)**	**Aquatic Insects (mg/m^2)**	**Benthic Animals (mg/m^2)**	**Fish #1 (kg/km)**	**Fish #2 (kg/km)**	**Fish #3 (kg/km)**
Lower Granite	1994	0.49	46,728	7,742	2,706	2,159	34	276	685
	1995	0.46	1,990	561	1,784	2,132	24	198	538
	1997	0.49	4,771	1,762	1,786	2,023	31	240	577
Little Goose	1994	0.60	6,030	1,238	2,889	2,287	33	244	556
	1995	0.51	1,364	682	2,197	2,356	22	180	440
	1997	0.56	3,606	2,756	2,335	2,298	29	219	467
Lower Monumental	1994	0.76	1,248	292	3,212	2,367	26	190	407
	1995	0.57	3,553	1,648	2,538	2,566	17	139	339
	1997	0.60	743	459	2,346	2,294	22	170	320
Ice Harbor	1994	0.93	273	72	2,159	1,453	40	288	580
	1995	0.61	473	179	1,924	1,750	27	222	519
	1997	0.66	122	67	1,871	1,572	35	262	510
McNary	1994	1.02	0	1	2,754	2,271	34	240	397
	1995	0.71	54	17	2,455	2,484	22	176	376
	1997	0.72	73	20	2,565	2,392	29	216	394

Attached benthic algae plots showing the predicted biomass of attached benthic algae throughout the lower Snake river on selected dates are presented in Figures 5.5-13 through 5.5-20. The biomass of attached benthic algae presented along with the biomass of other biologic components of the model are presented as average biomass per unit area over the entire wetted area of each computational element. Attached benthic algae biomass is extremely variable as one moves through the system. This is primarily a function of differences in velocities and water depths between computational elements. Low biomass values are consistently found around river mile 20. The river is a canyon, and is narrow and deep at this location. In addition, later in the simulation year, benthivorous fish redistribute and exert localized predation impacts on the attached benthic algae community.

The biomass results for the compartment for Benthic Algae 1 mirror the biomass results for Benthic Algae 2, but are slightly higher than the biomass predicted for Benthic Algae 2. This is primarily because Benthic Algae 1 (filamentous algae) and Benthic Algae 2 (unicellular and colonial algae) are preferred differently as a food source by several taxa. Day 152 peak biomasses (ca. 4,000 mg/m^2 in 1994, ca. 1,500 to 2,000 mg/m^2 in 1995 and 1997) of both Benthic Algae 1 and Benthic Algae 2 were predicted to occur near river mile 70 (Figures 5.5-13 and 5.5-14). Benthic algae biomasses predicted on this date are low relative to other dates, because of high river flows, high suspended solids concentrations, high water depth, and cold water temperatures. On day 183 (Figures 5.5-15 and 5.5-16), the predicted biomass of both Benthic Algae 1 (ca 90,000 mg/m^2) and Benthic Algae 2 (ca. 40,000 mg/m^2) peaked above the confluence of the lower Snake and Clearwater Rivers. This peak only occurred in 1994. Other peaks were predicted to occur near river mile 70 and again near river mile 45. Again, these peak biomasses were highest during 1994, the low flow year. On day 213 (Figures 5.5-17 and 5.5-18), predicted biomasses were much higher than on the prior two simulation dates discussed above (ca. 130,000 to 280,000 mg/m^2). The locations of the biomass peaks, however, were similar to those observed on day 183. The highest peaks in the upper portion of the simulated section of the lower Snake River were predicted for the 1994 year simulation, while the highest peaks lower in the river were associated with 1995 (an average flow year). On day 247 (Figures 5.5-19 and 5.5-20), again the highest peaks were observed above the confluence of the lower Snake and Clearwater Rivers for the 1994 simulation year. The magnitude of the peaks on this date were lower than those observed on day 213, reflecting reduced algal growth in response to a reduction in solar radiation and lower water temperatures, and increased losses of benthic algae through mortality and subsequent scour.

Average biomass of attached benthic algae by pool and simulation year is presented in Tables 5.5-1 through 5.5-4. In general, the highest biomasses of attached benthic algae are predicted to occur in the reach that is currently occupied by the Lower Monumental pool. Early in the growing season (Days 152 and 183), predicted attached benthic algae biomass is highest for 1994, a simulated dry year, while later in the season (days 213 and 274), the highest predicted biomasses are associated with 1994 in Lower Granite and Little Goose pools, and 1995 in the downstream pools. In the river reach now occupied by the Lower Granite pool, benthic algal productivity on all dates was predicted to be highest during 1994, a dry year. The highest predicted biomass of 1994 is corroborated by sampling data from 1998, a low flow year (Appendix O). This is not always the case in the lower reaches, where the maximum biomass is often predicted to occur during an average year (1994), and occasionally in a wet year (1997).

Phytoplankton plots showing the predicted biomass of phytoplankton throughout the lower Snake River on selected dates are presented in Figures 5.5-21 through 5.5-24. Predicted phytoplankton biomass is relatively stable as one moves through the system, particularly when flows are high. This is primarily due to the short time of travel through the system and the turbulent, well-mixed nature of the predicted normative system. In general, predicted biomass of phytoplankton varies between 0.4 mg/l and 0.8 mg/l, with the greatest longitudinal variability and some slightly higher biomasses being predicted for 1994 (a dry year) in the downstream reaches. This may be attributable to the relatively longer time of travel through the systems during dry years, allowing some modest phytoplankton growth in slower moving sections. There is no clear pattern of differences in phytoplankton biomass among pools (Tables 5.5-1 through 5.5-4), although predicted phytoplankton biomass are somewhat higher in the McNary pool.

Figures 5.5-25 through 5.5-28 show the predicted biomass of aquatic insects throughout the lower Snake River on selected dates (note differing scales). Aquatic insect biomass is somewhat variable as one moves downstream through the system although less variable than biomasses predicted for attached benthic algae. Slower growth rates of aquatic insects and longer cycling times are likely responsible, in part, for the increase in population stability. Aquatic insects are also not as closely tied to light intensity at the river channel bottom as a determinate of growth and biomass (they are only connected to the extent that they rely on attached benthic algae as a food source). Substrate throughout the normative river system should be suitable for the growth of aquatic insects. In general, the highest biomasses of aquatic insects were predicted for 1994 (a dry year), whereas the lowest were predicted for 1997 (a wet year). Predicted biomass of aquatic insects increases over the course of each growing season by roughly a factor of two despite emergence losses throughout the growing season. Biomass was not simulated in the winter. In general, aquatic insects were predicted to be somewhat more abundant in the downstream reaches. A reduced biomass of aquatic insects was predicted near river mile 20 throughout the growing season in all three sampling years. This is likely in response to low predicted biomasses of attached benthic algae in this section of the river, which are an important part of the diet of aquatic insects. This section of the river is a deep canyon. The model uses an average water depth across the computational element. Average water depths in this section are at or below the lower limit for photosynthesis.

The predicted biomass of benthic animals throughout the lower Snake River on selected dates is presented in Figures 5.5-29 through 5.5-32 (note differing scales). Benthic animal biomass exhibits a variability similar to that of aquatic insects. As with aquatic insects, slower growth rates of benthic animals and longer cycling times are likely responsible, in part, for the increase in population stability. Benthic animals are also not as closely tied to light intensity at the river channel bottom as a determinate of growth and biomass (they are only connected to the extent that they rely on attached benthic algae as a food source). Substrate throughout the normative river system should be suitable for the growth of benthic animals. In general, slightly higher biomasses for benthic animals are predicted during 1995, although the biomasses predicted for all three years are very similar. On average, predicted biomasses are lowest during 1994 early in the season (days 152 and 183), and during 1997 later in the season (days 213 and 274). In the early part of the simulated season (days 152 and 183), benthic animals remain fairly consistent throughout the lower Snake River. As the season progresses (days 213 and 274), there is more variability in benthic animal biomass throughout the river. Fluctuations from simulation element to element are due to food availability and predation within the particular element. Biomass, for the most part, fluctuates around 2900 mg/m^2. A similar depression in biomass is predicted around river mile 20, as was predicted for aquatic insects. Potential reasons for this depression are the same as was stated for aquatic insects.

The predicted biomasses of Fish 1 throughout the lower Snake River on selected dates are presented in Figures 5.5-33 through 5.5-36. Fish 1, cool water piscivores (salmonid species, fish eating lifestages), shows very little year-to-year variability. The long cycling times of fish, where the standing crop or biomass is the integration of several to many years of individual and population growth, may mask year-to-year variability at the time scale modeled. The biomasses are generally predicted to be slightly higher for Fish 1 during 1994. The variability between computational elements is sometimes quite large.

This variability is driven by the availability of the food source for Fish 1 (zooplankton, aquatic insects, benthic animals, fish 2, and fish 3). Within each pool, the fish are free to move in and out of computational elements based on food availability. On average, the highest biomass of Fish 1 are predicted to occur within the Ice Harbor pool, while the lowest biomass is predicted for the Lower Monumental pool. Fish 1 biomasses increase through the season, and distribution becomes more variable within each pool (Tables 5.5-1 through 5.5-4). Predicted average biomasses for all three years range from 5 to 40 kg/km. A predicted drop in Fish 1 biomass in Lower Monumental and McNary pools is currently unexplained.

Fish 2, also cold water piscivores (smallmouth bass, northern pikeminnow, catfish), exhibits the same pattern for predicted biomasses as Fish 1 throughout the lower Snake River on selected dates (Figures 5.5-37 through 5.5-40). Food sources for Fish 2 are similar to those for Fish 1, which explains why the patterns of predicted biomasses are similar. Fish 2, which is more plentiful than Fish 1, averages from 60 to 288 kg/km (Tables 5.5-1 through 5.5-4).

The predicted biomasses of Fish 3 throughout the lower Snake River on selected dates are presented in Figures 5.5-41 through 5.5-44. Predicted biomass of Fish 3 (benthivores) was generally higher during 1994, like Fish 1 and 2. However, year-to-year variability was still quite low. Again, variability between computational elements can be quite high. Food sources for Fish 3 consist of aquatic insects, benthic animals and, unlike Fish 1 and 2, benthic algae and organic sediment. Fish 3 biomass is predicted to be much higher than that of fish 1 and 2. Average biomass of fish 3 ranges from 187 to 685 kg/km (Tables 5.5-1 through 5.5-4). The highest average biomasses are predicted to occur in the Ice Harbor and Lower Granite pools.

5.5.4 - Comparison of Existing and Normative System

Comparison of the total biomass of primary producers predicted by the model with measured biomass of primary producers from 1997 is presented in Table 5.5-5. Reliable estimates of biomass of attached benthic algae are only available for Lower Granite Pool in 1997. These results indicate that, on a per unit length basis, primary productivity in the lower Snake River will be substantially higher under the future normative system than under the existing impounded system. In addition, the bulk of the primary productivity will be shifted from the phytoplankton to the attached benthic algae component of the food chain. The modeling results suggest that this increase in primary productivity will result in increased biomass of aquatic insects and benthic animals. Ultimately, fish species that can utilize attached benthic algae, aquatic insects, and benthic animals will be favored over species that utilize zooplankton.

Table 5.5-5 Comparison of Existing Data in Lower Granite Pool Day 213 to Predicted Values for the Normative System			
Existing/Impounded		**Normative**	
Phytoplankton			
Concentration (mg/L)	**Total Biomass (kg/km)**	**Concentration (mg/L)**	**Total Biomass (kg/km)**
0.376	718	0.401	380
Attached Benthic Algae			
Biomass at Sampling Point (mg/m^2)[a]	**Total Biomass (kg/km)**	**Biomass at Sampling Point (mg/m^2)[b]**	**Total Biomass (kg/km)**
40,369	208	7,805	1,997

[a]Biomass estimates are for one sampling location in the littoral zone.
[b]Biomass estimates from WQRRS are an average across the stream channel.

The comparison of the model predictions to data on the existing system for the biological components of the system that represent secondary production is complicated by the lack of information on the existing system. Some data exists describing the macroinvertebrate and resident fish community of the Lower Granite impoundment, but not for the years that were simulated.

5.5.5 - Summary

The results of the modeling exercise undertaken as a part of the assessment of biological productivity in the lower Snake River are summarized below:

1. Water temperatures in the lower Snake River are predicted to reach higher summer peaks during dry years under the normative river condition than under the existing impounded river condition. Under wet and average hydrometeorologic conditions, peak summer temperatures are projected to be similar to those observed for the existing system.

2. Water in the lower snake River is predicted to cool faster in the fall in dry, average, and wet years under the normative river condition than under the existing impounded river condition.

3. Benthic algae will dominate the primary productivity of the normative system. Phytoplankton productivity will be of lesser importance.

4. Total primary productivity is predicted to be higher in the normative river than in the existing impounded river.

5. The fish community biomass of the normative system will consist primarily of benthivores rather than the mix of planktivores and benthivores in the existing system.

6. 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, and decreased water velocities and associated scour. The elevated aquatic insect and benthivorous fish production is a function of increased benthic algae production. Elevated aquatic insect production in dry years should result in higher year class strength for fish species such as salmonids with life stages (smolts) that have a high percentage of aquatic insects in their diet.

7. The greatest changes in secondary productivity can be expected to occur in those trophic levels that feed directly on the attached benthic algae with lesser effects seen as trophic levels are more removed from the primary producers.

8. The biologic productivity model cannot be used in a strict predictive application due to the lack of biologic calibration data for the normative system. Relative comparisons of scenarios consisting of different hydrometeorologic or nutrient loading conditions is an appropriate use for the biologic productivity model.

5.6 - REFERENCES

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