APPROACH TO DEVELOPING NUTRIENT LOADING CRITERIA FOR FRANKLIN D. ROOSEVELT LAKE

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Water Resources Series
Technical Report No. 133

Report to Washington Department of Ecology
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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

INTRODUCTION

There has been increasing concern recently over the potential degradation of the water quality of Franklin D. Roosevelt (FDR) Lake, a mainstem hydropower storage reservoir located on the Columbia River in northeastern Washington state (see Figure 1). In particular, the problem of nuisance attached algae in FDR Lake, and its relation to eutrophication, has received significant attention in recent years. Much of the work is currently being done by Broch and Loescher (1988a, 1988b,1990), who are investigating the distribution of the benthic alga, *Cladophora*, which is recognized as one of the most important nuisance periphyton species in standing and running water throughout the world (e.g., Clark Fork River, MT; the Baltic Sea, Sweden; the Great Lakes, USA; Manawatu River, New Zealand; Harvey-Peel Estuary, Australia). *Cladophora* is a large-cell, filamentous green alga that is distributed worldwide in fresh, brackish, and salt water. Work in progress by Broch and Loescher includes mapping of *Cladophora* distribution from Grand Coulee Dam to the Canadian border, using qualitative rake samples in shoal areas, and conducting a lake macrophyte survey and sampling for surface water quality.

Relatively high total phosphorus (TP) concentrations at Northport, where the Columbia River begins to merge from river to reservoir, suggest the presence of a large point source load in excess of the normal concentrating effect at low flows (Figure 2). TP concentrations at Northport are well above the Environmental Protection Agency (EPA) recommendation of 25 $\mu g/L$ to prevent development of nuisance conditions and control eutrophication (Figure 2). The cause is suspected to be Cominco Ltd., a lead-zinc smelter and fertilizer plant located in Trail, British Columbia. Currently, if Cominco Ltd. was to discharge at permit limits (11,170 kg P/day and 2,015 kg Zn/day), at the seven-day ten-year low flow of 25,548 cfs

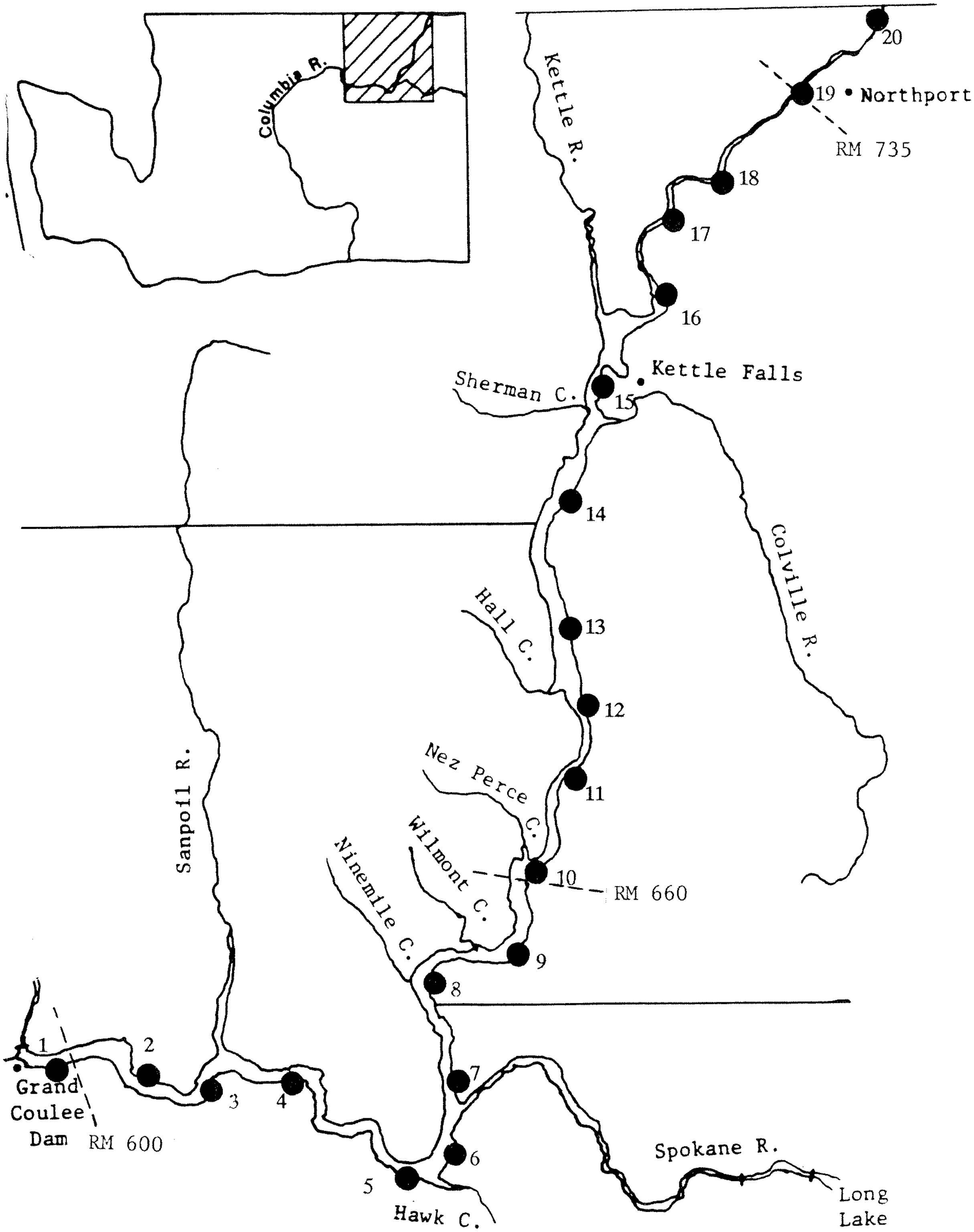

Figure 1. Map of Franklin D. Roosevelt Lake, including major tributaries and sample sites for 1991 longitudinal survey.

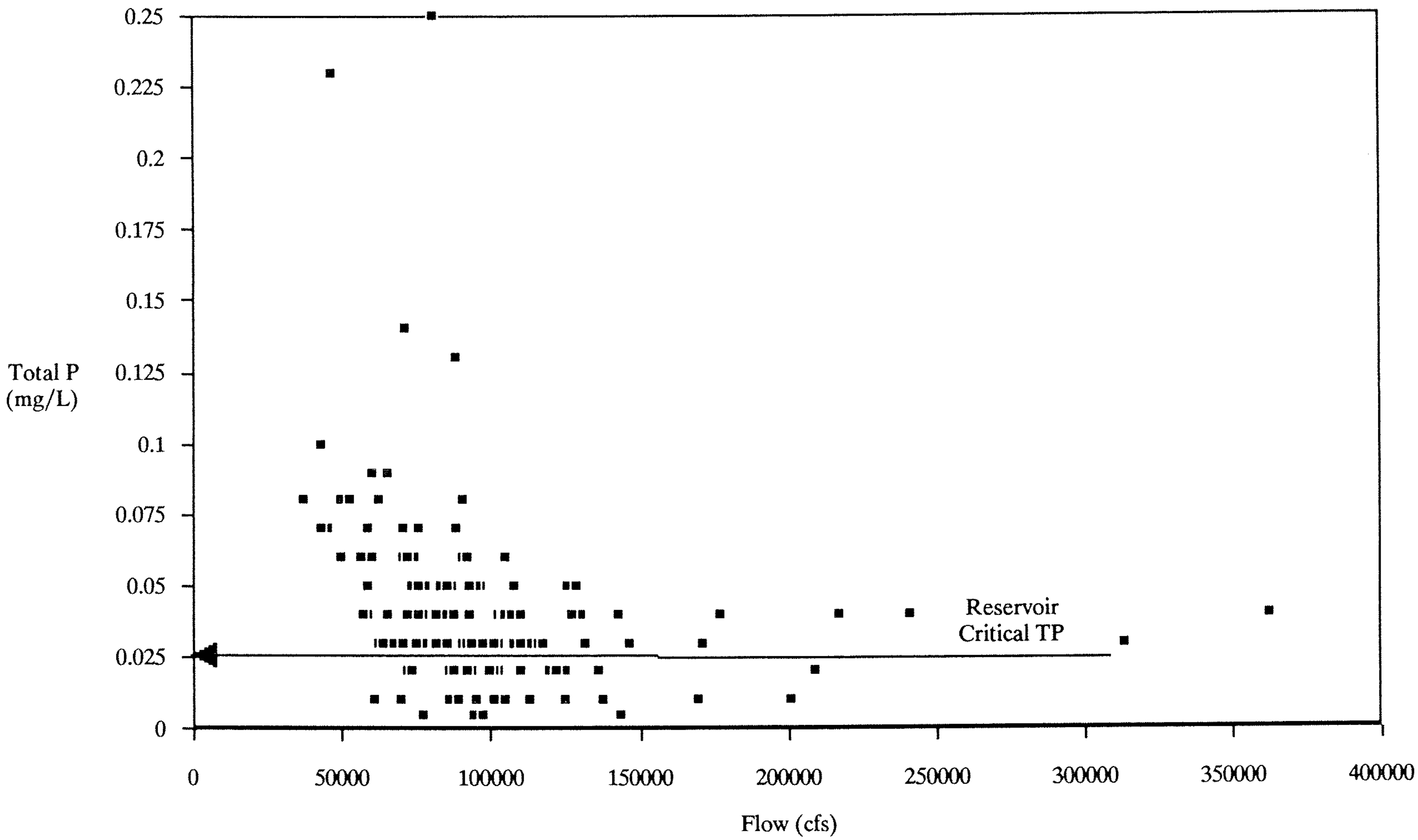

Figure 2. TP concentration vs. flow for Columbia River at Northport
for period 1971 to 1988 (data from USGS).

(Williams and Pearson, 1985), concentrations of TP and zinc in the Columbia River would be allowed to increase by 180 and 32 $\mu g/L$, respectively.

The purpose of this study was to conduct a scoping analysis, using existing information and data collected during the summer of 1991, to evaluate existing and potential problems from eutrophication in the upper Columbia River and Lake Roosevelt. The scoping analysis conducted here had five major goals, as follows:

1) Compile existing data which describe the physical and chemical characteristics of the upper Columbia River and Lake Roosevelt.
2) Collect additional data during the summer of 1991 to further describe the water quality of FDR Lake. This was accomplished by conducting a longitudinal survey involving 20 stations (see Figure 1) from the British Columbia border to Grand Coulee Dam. Also, surface samples from mid-channel were collected monthly at Keller Ferry (station #3) throughout the 1991 summer growing season.
3) Conduct an experiment in artificial channels to determine the critical P concentration for *Cladophora* growth.
4) Evaluate the response of trophic state indicators such as reservoir nutrient concentration, phytoplankton, transparency, and periphyton to a range of assumed foreseeable nutrient loads from all sources within the basin. The analysis includes evaluation of existing data with empirical models from the literature which relate trophic state indicators to nutrient loading.
5) Develop critical nutrient loading criteria for Lake Roosevelt, to the extent possible with existing and proposed sampling data, based on phytoplankton and periphyton response models. If not possible with

existing data, a plan for future studies will be recommended with the aim of ultimately establishing critical nutrient loading criteria for TP and/or TN.

STUDY AREA DESCRIPTION

Franklin D. Roosevelt (FDR) Lake was formed in 1941 following the construction of Grand Coulee Dam (Figure 1). FDR Lake is the largest body of water in Washington state and the sixth largest reservoir in the United States (Johnson *et al.*, 1989; Beckman *et al.*, 1985). The reservoir is located in the rather sparsely populated region of northeast Washington State and extends easterly and then northerly up the Columbia River valley to the Canadian border, a total of 151 miles from Grand Coulee Dam (Wolcott, 1964). The largest communities near FDR Lake are Grand Coulee, population 1195, Kettle Falls, population 1245, and Northport, population 342 (Johnson *et al.*, 1989).

Rather steep canyon walls and significant annual drawdowns preclude significant development of the shoreline of FDR Lake. A large portion of the lower part of the reservoir is bordered by the Spokane and Colville Indian Reservations. The surface of the lake itself is managed by the National Park Service (NPS) for recreational purposes. One of the driving forces for recent investigations of lake water quality is an increase in recreational visitors from 510,000 in 1985 to 1.4 million in 1988 (WALPA News, 1989).

Cominco Ltd. is located approximately 10 miles north of the Canadian border on the bank of the Columbia River in Trail, British Columbia. The plants have been in existence since the turn of the century and include the world's largest lead-zinc smelter and refinery (Johnson *et al.*, 1989). As mentioned previously, a significant amount of metals and nutrients are introduced to the Columbia River by effluents from Cominco's smelter and fertilizer plants. The extent of metals contamination with zinc, copper, lead, arsenic, cadmium, and mercury has previously been reported by Johnson *et al.* (1989). In addition to the metals from the lead-zinc smelter, Cominco also operates a fertilizer plant which discharges nutrients into the Columbia River, elevating the concentrations of phosphate and ammonia, with effluent P being over four times that of N.

The hydrologic input to FDR Lake is dominated by the Columbia, Spokane, and Kettle Rivers, which contribute 89, 7, and 3 percent, respectively, of the annual average inflow to the lake. The Colville, Sanpoil, and numerous small streams contribute the remaining 1 percent of the lake's input (Stober *et al.*, 1981). The outflow from FDR Lake is either through Grand Coulee Dam and down the Columbia River or is pumped to Banks Lake as needed for irrigation storage. At full pool, the reservoir has a length of 243.5 km, an average depth of 33.5 m, a maximum depth of 122.3 m, a storage volume of 11.23 x 10^9 m^3, and a surface area of 334.8 x 10^6 m^2 (Stober *et al.*, 1981; USGS, 1989; Columbia River Water Management Group, 1991). A summary of the morphometric characteristics of FDR Lake, from full pool (1290 ft, 393.2 m) to minimum operating level (1208 ft, 368.2 m), is presented in Table 1.

The U.S. Bureau of Reclamation (USBR) controls the water level of FDR Lake to provide for such uses as flood control, irrigation, recreation, fisheries, navigation, flow regulation, and power generation (Columbia River Water Management Group, 1991). FDR Lake water detention times and surface elevations for the period of 1986 to 1989 are presented in Figures 3 and 4.

Detention times were calculated as average monthly reservoir volume divided

Table 1. Morphometric characteristics of FDR Lake.[a]

Surface Elevation Perimeter (ft)	Area (x10^6 m^2)	Volume (x10^9 m^3)	Length (km)	Max Width (km)	Mean Depth (m)	Max Depth (m)	(km)
1290 (393.2 m)[b]	334.8	11.23	243.5	3.1	33.5	122.3	1,048
1270 (387.1 m)	297.9	9.346	---	---	31.4	116.2	---
1250 (381.0 m)	261.0	7.666	---	---	29.4	110.1	---
1230 (374.9 m)	224.1	6.195	---	---	27.6	104.0	---
1208 (368.2 m)[c]	183.5	4.838	---	2.2	26.4	97.3	---

[a] data from Stober *et al.*, 1981; USGS, 1989; and Columbia River Water Management Group, 1991.

[b] full pool

[c] minimum operating level

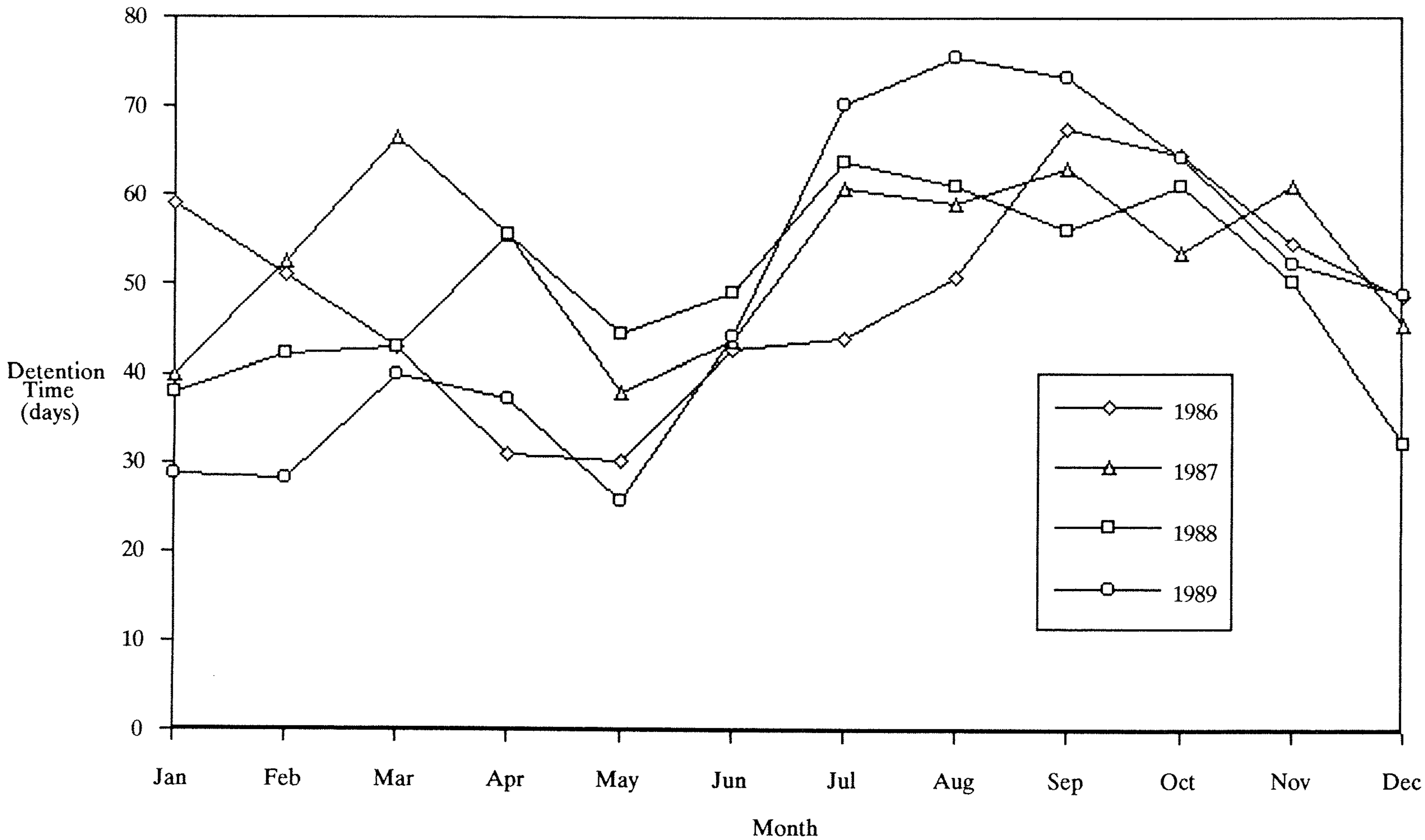

Figure 3. FDR Lake monthly hydraulic detention time for period of 1986-1989.

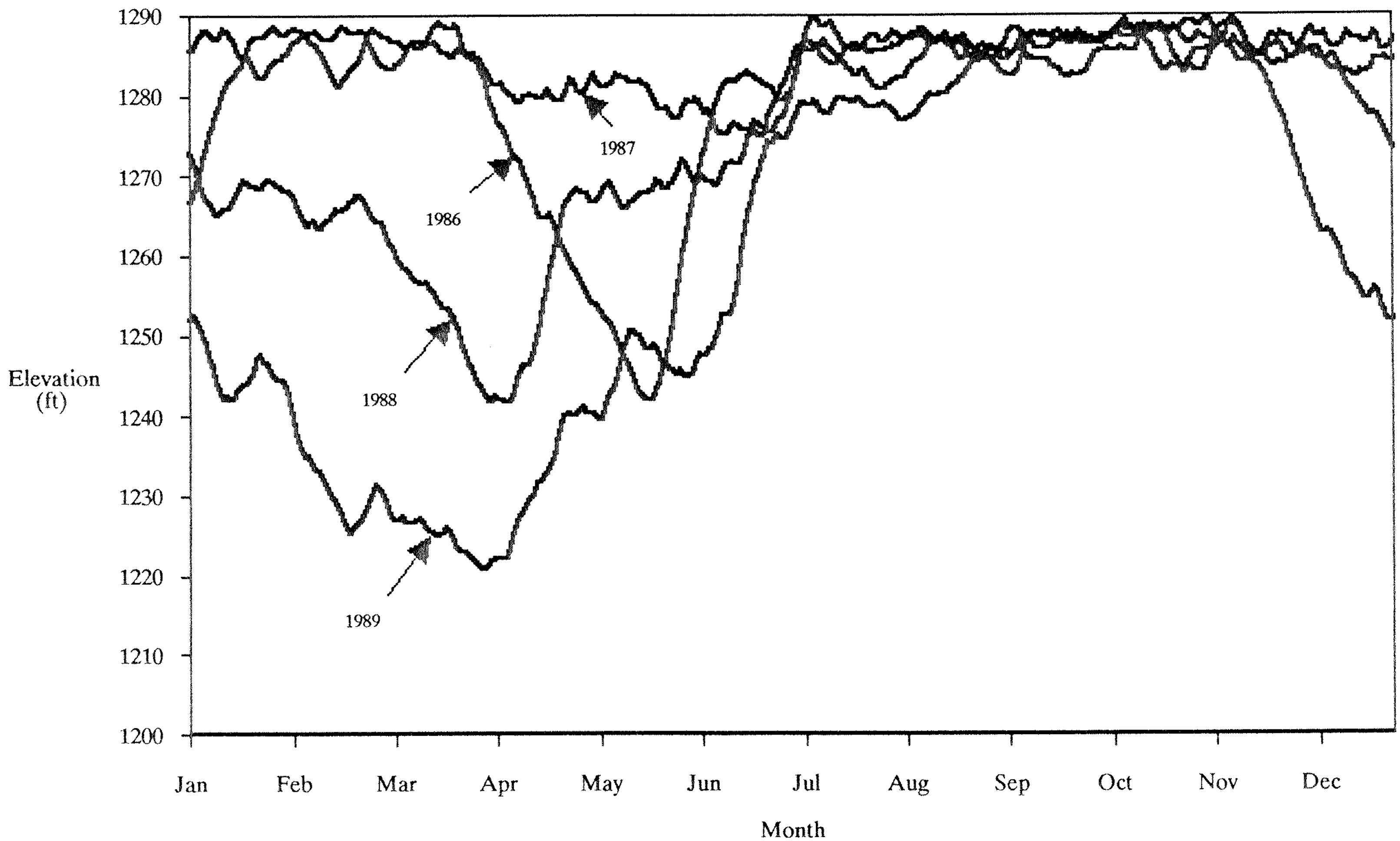

Figure 4. FDR Lake surface elevations for the period of 1986-1989 (data from USGS).

by average monthly outflow. This method assumes uniform outflow from all levels at the dam. Detention times calculated in this manner could underestimate actual detention times in cases where stratification causes significant thermal channeling of flow; however, Jaske and Snyder (1967) found that this effect is minimal for FDR Lake and concluded that the reservoir fills by displacement from upstream and does not develop a density current regime of any significance. Monthly detention times for the period 1986 to 1989 ranged from a minimum of 26 days to a maximum of 76 days and averaged approximately 50 days (Figure 3). For comparison, mean detention time, calculated as the ratio of mean annual minimum storage (in this case for the last 10 years) to the 30-day, 10-year low flow (37,100 cfs), is 74 days. Thus, "lake class" water quality standards apply to FDR Lake as a whole.

The major period of drawdown has been shown to occur from January to June, with the minimum lake elevation usually occurring in April and full pool usually maintained from July through December (Stober *et al.*, 1977; Beckman *et al.*, 1985). This pattern is consistent with the values obtained for the period of 1986 to 1989 and is illustrated in Figure 4.

The minimum annual FDR Lake elevation attained for the period of 1960 to 1990 is shown in Figure 5. Recent drawdowns have been not quite as severe as in past years. This may be due in part to recommendations made by the U.S. Fish and Wildlife Service in a report to the USBR assessing the fisheries of FDR Lake (Beckman, *et al.* 1985). The study concluded that extensive drawdown and corresponding low detention times in the spring flushed nutrients and phytoplankton through the reservoir, reducing nutrient assimilation and, consequently, potential fish food production. Recommendations were made for

Figure 5. Minimum annual FDR Lake elevations for the period 1960-1990 (data from USGS).

the following adjustments in water management of the reservoir in order to increase nutrient assimilation, fish food production, and larval fish habitat: 1) Maintain water levels in April and May as high as possible to increase detention time and 2) Maintain maximum water level at 3-4 meters below full pool every other year to stimulate vegetation to supplement nutrient input and cover for larval fish. As a result of these recommendations, it is possible that future operation of FDR Lake by the USBR will be such that drawdowns are minimized, so that hydraulic detention times are maintained as high as possible. One possible adverse side effect of a reduction in lake surface elevation fluctuations is that it may favor the growth of periphytic algae by allowing longer periods for periphyton development with a stable water surface elevation. However, according to the USBR, the recommendations have not yet significantly affected their control of FDR Lake water level.

METHODS

Several methods were used for collecting and analyzing data for FDR Lake, including the following:

1) Existing data on the physical and chemical characteristics of FDR Lake were compiled from a variety of sources.
2) A longitudinal survey of FDR Lake from the border to the dam was conducted in August, 1991.
3) Monthly surface water quality samples were collected by the Washington State Department of Ecology (Ecology) at the Keller Ferry station from April to September, 1991.
4) Existing data were used to assess potential nutrient and phytoplankton response to loading changes using Walker's model developed for the United States Army Corps of Engineers (USACE) (Walker, 1981, 1982, 1985, 1987).
5) Data obtained from the longitudinal survey and past data were compared with the potential for nuisance levels of periphyton using the Horner-Welch steady-state biomass model for periphyton accrual (Horner, 1978).

COMPILATION OF EXISTING DATA

In an attempt to characterize the physical and chemical characteristics of the reservoir, existing data were compiled from several sources, including the following:

1) Flow data for the major tributaries and elevation data for the lake surface were collected from United States Geological Survey (USGS) monitoring records.
2) Nutrient data for the major tributaries were collected from monitoring records from the USGS and Ecology.
3) Morphometric data were collected from the USBR and from USGS quadrangle maps.
4) Pool water quality data and other limnological data were obtained from reports and a thesis (Stober *et al.*, 1981; Jagielo, 1984; and Beckman *et al.*, 1985).
5) Qualitative assessment of abundance and distribution of various species of periphyton were obtained from progress reports on work by Broch and Loescher (1988a, 1988b, 1990).

Flow data for the major tributaries and elevation data for the lake surface were collected from USGS records for water years 1979 to 1989. The data were retrieved from HYDRODATA database, stored on CD ROM at the University of Washington Natural Sciences Library. Daily flow data were available for the Columbia, Spokane, Kettle, and Colville Rivers, which constitute over 99% of the hydraulic input to FDR Lake (Stober *et al.*, 1981). Daily flow records were also available for the outflow from FDR Lake at Grand Coulee Dam. These were the sum of flow records for the Columbia River downstream of Grand Coulee Dam and flow records for pumped storage to Bank's Lake.

Nutrient data for the major tributaries and outflow were collected from monitoring stations operated by the USGS and Ecology. The data collected consisted of monthly depth-width integrated composite samples for concentration and instantaneous flow data. The data were collected for water year 1980, because that year would allow testing of Walker's reservoir model output against observed pool data collected previously (Stober *et al.*, 1981). The data are used to construct nutrient budgets using Walker's reservoir model. Table 2 summarizes the tributary nutrient data collected for the water year 1980.

Morphological data were collected from the USBR and from USGS quadrangle maps. The USBR provided charts of FDR Lake capacity and surface area as a function of reservoir elevation. The USGS quadrangle maps were used to determine if a significant variation in detention time occurred along the length of the reservoir as a result of morphological variation. The volumes of each 5 mile section along the length of the reservoir were calculated by taking the product of the average depth, derived from soundings of the lake bottom, and the surface area, derived from quadrangle maps. It was determined that a significant volume difference exists between the upper and lower reservoir, implying that splitting the reservoir into two segments may be warranted because of differences in hydraulic detention time. For this reason, FDR Lake was split into two segments at river mile 660 (station #10) for modeling of spatial variation of nutrients and trophic state indicators between the upper and lower reservoir.

The only longitudinal pool water quality data available were obtained from a previous study of the limnology of FDR Lake (Stober *et al.*, 1981). In this study, pool water quality was sampled at eight different sites spread throughout the length of FDR Lake. Water samples were collected monthly from December, 1979, to September, 1980, and were analyzed for TP, soluble reactive phosphorus (SRP),

Table 2. Summary of tributary nutrient data for water year 1980.

Tributary	Parameters Monitored[a]	Frequency of Record	Source
Columbia	TP, SRP, TN, DIN	Monthly	USGS
Spokane	TP, SRP, TN, DIN	Monthly	USGS
Kettle	TP, SRP, DIN	Monthly (Oct-Jun)	USGS
Colville	TP, SRP, DIN	Monthly (Oct-Jun)	USGS
Outflow	TP, SRP, DIN	Monthly	Ecology

[a] DIN = $(NO_2+NO_3\text{-}N) + (NH_4\text{-}N)$

dissolved inorganic nitrogen [DIN = $(NO_2 + NO_3\text{-}N) + (NH_4\text{-}N)$], and chlorophyll a (chl a). Secchi depth measurements were also taken at the same time that water samples were collected. Average, volume-weighted growing season means were calculated from these data and used for comparison with predicted values generated by the Walker reservoir water quality model.

Data on periphyton abundance were collected from previous studies by Broch and Loescher (1988b, 1990). Periphyton were qualitatively sampled at 14 stations located along the lower portion of FDR Lake. The stations selected had significant littoral areas and suitable substrate for *Cladophora* development. Samples were taken during July-August, 1988 and then again in October, 1988. Samples from 0-8 meters were collected qualitatively with a spring tooth rake in an attempt to characterize the distribution of *Cladophora* in FDR Lake. An attempt to estimate relative abundance from these data was unsuccessful.

LONGITUDINAL SURVEY OF FDR LAKE

A longitudinal survey of FDR Lake from Grand Coulee Dam to the Canadian border was conducted during August 21-24, 1991, as part of this contract. Twenty sampling locations were chosen along the length of the reservoir (see Figure 1) to be, as much as possible, evenly distributed and to correspond with stations previously sampled qualitatively by Broch and Loescher (1988b, 1990). Samples for nutrients and periphyton abundance were collected to determine if a downstream relationship exists between periphyton biomass and limiting nutrient content. A decline in the limiting nutrient content downstream (down lake) was expected, as a result of nutrient uptake by phytoplankton and periphyton. A decline to levels which would inhibit accumulation of periphyton biomass would be useful information for determining permissible nutrient loading criteria. Water samples were analyzed for zinc and observations were made on invertebrate grazer abundance to determine if a lack of grazers, as one progressed upstream toward the source of zinc, could be explained in part by zinc toxicity. Such a relation could indicate grazer inhibition and be a partial cause for nuisance *Cladophora* levels.

All sites were sampled for the following constituents: TP, SRP, total nitrogen (TN), DIN, total zinc (TZn), and soluble zinc (SolZn). All water samples were collected as surface grabs in acid-washed polyethylene bottles and immediately stored in coolers on ice. Water samples for SRP and SolZn were filtered in the field through a 0.45 μm Millipore filter. Prior to filtering, the 0.45 μm Millipore filters were soaked overnight in deionized water to remove any trace nutrients. Samples for TN were preserved with concentrated sulfuric acid, which was present in the sample bottles supplied by Ecology. Samples for TP were preserved with two drops of concentrated sulfuric acid, and samples for zinc were preserved with concentrated nitric acid to a $pH<2$ (APHA, 1989).

Upon returning from the field, samples for TP and SRP were removed from coolers and frozen for later analysis according to Standard Methods (APHA, 1989). Nitrogen and zinc samples were sent on ice to the Ecology Manchester Laboratory for analysis. Filtered samples for nitrate + nitrite (NO_3+NO_2-N) and ammonia were analyzed, respectively, using the cadmium reduction method and the phenate method. Unfiltered samples for TN were digested with persulfate and the digestate analyzed by the same method as for NO_3+NO_2-N. Samples for zinc were analyzed by inductively coupled plasma atomic emissions analysis. Samples for

SRP were analyzed at the University of Washington (UW) using the ascorbic acid-molybdenum-blue method with absorbance measured with a Spectronic 2000 spectrophotometer. Samples for TP analysis were digested with persulfate and the digestate analyzed using the same method as for SRP.

Precision for TN and nitrate was within +/- 10% of duplicates, while ammonia precision exceeded +/- 20%. Blanks showed no detectable levels except for one of the two TN blanks. Blanks for P analysis contained < 1 $\mu g/L$ P and standard deviation of replicates for SRP and TP averaged 0.5 and 0.6 $\mu g/L$, respectively. Blanks for zinc analysis were <4 $\mu g/L$ and precision was within +/- 20% of duplicates.

Periphyton samples were collected by scraping a consistent surface area (25 cm^2) from each of 3 rocks from each sample site. Rocks were chosen in areas of suitable substrate for periphyton growth at a depth of 0-1 meters. The scraped area was gently rinsed with a squeeze bottle of lake water and the rinsate was collected in zip-lock bags, which were immediately stored on ice in the dark.

Upon returning from the field, periphyton samples were analyzed for chl a. Samples were homogenized and then suspended in 250 ml of distilled water. A known volume of the suspension was then filtered directly onto a 25 μm glass fiber filter. Prior to filtering, the glass fiber filters were treated with saturated $MgCO_3$ solution and subsequently rinsed with distilled water. Once filtered, the filters were folded in half, placed in manila coin envelopes, and stored frozen in a darkened glass jar until further analysis.

Following an 18-24 hour extraction in a 90% acetone solution, samples for chl a were analyzed for absorbance in a Perkin-Elmer 3 scanning spectrophotometer (APHA, 1989). The Lorenzen (1967) equation (eqn 1) was used to calculate the μg chl a/L in the suspension volume. The value of μg chl a/L was then converted to mg chl a/ m^2 (eqn 2) to represent surface area coverage.

$$\text{chl a } (\mu g/L) = \frac{11*2.43*[(665o-750o)-(665a-750a)]*E*1000}{V_f*L} \qquad \text{(eqn 1)}$$

$$\text{chl a } (mg/m^2) = \frac{(\mu g \text{ chl a}/L)*Vol*10}{A} \qquad \text{(eqn 2)}$$

where:

E	=	Volume of chl a-acetone extract (ml)
665_o	=	Initial absorption of chl a
665_a	=	Absorption following acidification
750_o	=	Correction for turbidity
750_a	=	Correction for turbidity following acidification
V_f	=	Volume of sample filtered for chl a analysis (ml)
L	=	Length of light path in cell (cm)
Vol	=	Suspension volume of original sample (L)
A	=	Area sampled (cm^2)
10	=	Conversion factor ($mg\text{-}cm^2/\mu g\text{-}m^2$)

Finally, the presence of grazing invertebrate populations was determined qualitatively at each of the sample sites. Samples were picked in the field from the same rocks that were sampled for periphyton using forceps, or, when substrate allowed, a Peterson grab was used. Invertebrates were collected and stored in wide mouth jars in a 30% isopropanol solution.

NUTRIENT AND PHYTOPLANKTON MODELING

Modeling approaches for managing eutrophic lakes and reservoirs can be characterized as either empirical or theoretical. Chapra and Reckhow (1983) define mechanistic (theoretical) models as an attempt to explicitly account for the mechanisms underlying lake dynamics. Therefore, such models attempt to directly simulate the physical, chemical, and biological processes of complex ecosystems and combine these with a simulation of the hydrodynamics of the reservoir. This type of model is particularly useful in situations where a high degree of spatial and temporal resolution is required; however, they can also be used in steady-state form. Nevertheless, large amounts of lake-specific input data are generally required for these models. On the other hand, Chapra and Reckhow (1983) define empirical models as the use of statistical techniques to discern patterns or relationships underlying data for large samples of lakes without necessarily having a theoretical basis. Empirical models are initially calibrated to a large cross section of lakes or reservoirs and are based on the premise of similar trophic behavior among water bodies. The strength of this type of model lies in the fact that application is not water body specific, so models and data are transferable, keeping data collection and analysis requirements low (Reckhow and Chapra, 1983). Empirical models generally deal with spatially and temporally averaged conditions, either annually or over a growing season. Since this particular study is limited to existing data and a high degree of spatial and temporal resolution is not required, an empirical modeling approach was chosen.

Empirical models are generally comprised of nutrient loading models coupled with eutrophication response models. The nutrient loading models utilize an input-output mass balance to predict reservoir pool nutrient concentrations resulting from external nutrient loading, morphometry, and hydrologic conditions. Eutrophication response models then relate the pool nutrient concentration to trophic state indicators (chl a and transparency) using empirical simulations derived from a large cross section of reservoirs. In the input-output nutrient budgets, the reservoir is modeled as a Continuously Stirred Tank Reactor (CSTR), with nutrient inputs from external loadings and outputs via flushing and net sedimentation. Since the rate of mass loss to the sediments is extremely difficult to measure and formulate theoretically, developing empirical simulations for sedimentation losses has been the major focus in nutrient loading models (Chapra, 1982).

The empirical model chosen for this study was developed for the U.S. Army Corps of Engineers by Walker (1981, 1982, 1985, 1987). This model is particularly useful for this application, because it includes a software package which allows for reduction of large amounts of nutrient loading input data and for rapid evaluation of impacts of proposed changes in reservoir management. The model can therefore be used effectively in a predictive mode, where the long-term, steady-state responses of the reservoir are projected in response to changes in controlling variables such as nutrient loading, reservoir morphometry, and reservoir elevation. The software package is composed of three separate programs as follows:

1) FLUX- Program estimates loading of each tributary and outflow from grab sample concentration data and continuous flow records using several possible calculation methods.
2) PROFILE- Program provides graphic display and reduction of observed pool water quality data. Its output is useful for calibrating nutrient

models and eutrophication response models when reservoir-specific data are available.

3) BATHTUB- Program utilizes output from FLUX and PROFILE programs as input and implements water budgets, nutrient loading models, and eutrophication response models.

The BATHTUB portion of Walker's model attempts to increase its generality and applicability by incorporating several improvements on the traditional empirical models used in the past. Walker's model takes into account algal growth limitation by phosphorus, nitrogen, light, and flushing rate. This can be a significant improvement over simple phosphorus-chl a relationships, which are not applicable in situations where factors other than phosphorus are limiting algal growth. This may be a significant point with regard to FDR Lake, because it has previously been shown that portions of the reservoir may be nitrogen limited and that flushing is also a significant factor controlling the growth of algae (Stober *et al.*, 1981; Beckman *et al.*, 1985). Also, Walker's model takes nutrient partitioning into account by requiring both dissolved and particulate nutrient data as input. Nutrient partitioning is important to consider, since reservoirs appear to be more sensitive to dissolved nutrients because of reduced sedimentation rates and increased biological availability (Chapra and Reckhow, 1983). However, the model predicts pool TP only.

While most simple empirical models for lakes simulate nutrient sedimentation as a first-order reaction, Walker found that a second-order reaction was better suited for simulating sedimentation and predicting pool nutrient concentrations in reservoirs. Finally, Walker's model improves on simple empirical models by providing the ability to separate the reservoir into segments, hence allowing for simulation of spatial variability. As indicated earlier, FDR Lake was split into two segments at river mile 660 because of significant difference in hydraulic detention time between the upper (segment #1) and the lower (segment #2) reservoir.

Water Budget

Water budgets were calculated using the BATHTUB portion of Walker's model. The following mass balance equation was used for calculation of the water budget (Walker, 1987):

Total Inflow - Total Outflow = Change in Storage (Δ Storage)

where: Total Inflow = Gauged Inflow + Ungauged Inflow + Precipitation
Total Outflow = Gauged Outflow + Evaporation

The residual in the water budget is attributed to ungauged inflow, which is composed of net groundwater inflow, inflow from ungauged watersheds, and direct runoff into the lake. Ungauged inflow was calculated in the following manner to complete the water balance:

Ungauged Inflow = Gauged Outflow + Δ Storage + Evaporation
- Precipitation - Gauged Inflow

Nutrient Loading Budgets

Nutrient (TP,TN) loading was calculated using the FLUX portion of Walker's model. Nutrient loading was calculated by taking the flow-weighted average concentration multiplied by the mean flow for the averaging period as follows (Walker, 1987):

$$\text{Load} = [(\Sigma\, c_i * q_i)/\text{Mean}(q)] * \text{Mean}(Q)$$

where:

c_i = Measured concentration in sample i.

q_i = Instantaneous flow in sample i.

Mean(q) = Mean of instantaneous flow measurements.

Mean(Q) = Mean of continuous flow measurements.

BATHTUB performs nutrient budget calculations in a manner similar to the water budget, except a term must be included to account for nutrient sedimentation:

Inflows = Outflows + Increase in Storage + Net Sedimentation

Inflow and outflow loading for the major tributaries was calculated using the FLUX program by calculating the flow-weighted average concentration multiplied by the mean flow for the averaging period, as discussed previously. Net sedimentation was modeled as a second-order decay function using the following formulation from Walker (1987):

$$W_s = k_2 * C^2$$

where:

W_s = Nutrient sedimentation rate (mg/m^3-yr)

k_2 = Effective second-order decay rate (m^3/mg-yr)

C = Pool nutrient concentration (mg/m^3)

Using the second-order decay function to model sedimentation, BATHTUB calculates the resulting pool concentration using the following equation:

$$C = \frac{[-1 + (1 + 4*k_2*C_i*T)^{0.5}]}{2*k_2*T}$$

where: T = Hydraulic detention time (y)

C_i = Inflow nutrient concentration (mg/m^3)

The formulation of k_2, the effective second-order decay rate coefficient, is variable depending on whether the budget is being accomplished for TP or TN. Nutrient model #2 was chosen for both TP and TN; in that model, k_2 is formulated as follows (Walker, 1987):

TP: $$k_2 = \frac{0.056*F_{ot}^{-1}*Q_s}{Q_s + 13.3}$$

TN: $$k_2 = \frac{0.0035*F_{in}^{-0.59}*Q_s}{Q_s + 17.3}$$

where:

F_{ot} = (Tributary Ortho-P Load)/(Tributary Total P Load)

F_{in} = (Tributary Inorganic N Load)/(Tributary Total N Load)

Q_s = Surface overflow rate (m/yr)

In addition to surface overflow rate, the effective second-order decay rate coefficient (k_2) is a function of inflow nutrient partitioning (F_{ot}, F_{in}). Since tributary inputs with high particulate-to-soluble ratios would have much greater sedimentation than inputs dominated by soluble nutrients, the F_{ot} and F_{in} terms are incorporated in the formulations to account for these effects of inflow nutrient partitioning.

Trophic State Response Models

The pool nutrient concentration values generated by the nutrient loading models are subsequently used by the BATHTUB trophic state response models to calculate the variables chl a and Secchi depth. The following trophic state response models were used for calculating chl a using BATHTUB (Walker, 1987):

Model 1: Limitation by N, P, Light, Flushing Rate

$$B = B_x/[(1+0.025*Bx*G)*(1 + G*a)]$$

where:

B = Chlorophyll a concentration (mg/m3)
B_x = Nutrient-potential chl a concentration (mg/m^3) = $X_{PN}^{1.33}/4.31$
X_{PN} = Composite nutrient concentration (mg/m^3) = $[P^{-2} + ((N-150/12)^{-2}]^{-0.5}$
G = Kinetic factor = $Z_{mix}*(0.14 + 0.0039*F_s)$
Z_{mix} = Mean depth of mixed layer (m)
F_s = Flushing Rate (1/y) = (Inflow - Evap)/Volume
a = Non algal turbidity (1/m)

Model 2: Limitation by P, Light, Flushing Rate

$$B = B_P/[(1 + 0.025*B_P*G)*(1 + G*a)]$$

where:

B = Chlorophyll a concentration (mg/m^3)
B_P = Phosphorus-potential chl a concentration (mg/m^3) = $P^{1.37}/4.88$
G = Kinetic factor = $Z_{mix}*(0.19 + 0.0042*F_s)$

BATHTUB includes two other chl a models which are much simpler than the two outlined above. These models were used mostly for comparison and only models #1 and #2 were used for analysis.

Secchi depth was calculated by the BATHTUB program using the following formulation (Walker, 1986):

$$S = 1/(a + 0.025*B)$$

where:

S = Secchi depth (m)
a = Non algal turbidity (1/m)
B = Chlorophyll a concentration (mg/m^3)

PERIPHYTON MODELING

Although modeling approaches to manage growth of attached algae are only in early stages of development compared to those for managing phytoplankton, attached algae are apparently a bigger problem in FDR Lake than phytoplankton. Studies of lotic attached algal growth and the conditions creating it are much less extensive than for lentic phytoplankton. Although general agreement exists on the algal biomass levels, average nutrient concentrations, and annual loadings which are associated with the different trophic states in lakes; running water systems (and use of attached algae in standing water) have not received a comparable amount of attention. As with modeling, criteria for nutrient and biomass levels that indicate nuisance levels are only in the early stages of development (Welch *et al.*, 1988, 1989).

In lakes, the biomass of algae per volume is proportional to the original mass of limiting nutrient per volume. Uptake results in additional algal biomass as the limiting nutrient is incorporated into biomass. Therefore, correlations between the concentration of limiting nutrient and algal biomass are expected and have been observed by several researchers (e.g., Dillon and Rigler, 1974; Jones and Bachmann, 1976). In running water systems, on the other hand, the periphyton biomass attained per area at one point does not have a similar relationship with the limiting nutrient concentration of the overlying water, because the nutrients are continuously being supplied from upstream. However, the total biomass developed throughout the affected stream length may be related to the mass of limiting nutrient removed through uptake (Welch *et al.*, 1989). Nevertheless, the uptake rate of nutrients by the periphyton mat is a function of the concentration of limiting nutrient in the overlying water, and hence growth rate and eventual biomass are controlled at least partially by that concentration. This difference in conditions between phytoplankton and periphyton (even in lakes) suggests that biomass-nutrient relations would tend to show saturation at lower levels for periphyton than for phytoplankton. There are several other confounding factors which have been shown to control the growth of periphyton in addition to limiting nutrient content in running water systems; these include current velocity, suspended sediment, shading, grazing, temperature, and substratum type (Horner *et al.*, 1983,1990; Welch *et al.*, in press).

Modeling of periphyton accrual by simple correlation has previously been shown to be unreliable (Jones *et al.*, 1984; Welch *et al.*, 1988). Instead, the modeling process should be dynamic and include growth kinetics as well as several of the confounding factors affecting periphyton growth as described above.

The model used here was developed and calibrated to the growth of periphytic green and blue-green algae in natural streams by Horner (1978). The model was subsequently modified and calibrated in artificial channels to determine the critical P concentration to prevent nuisance biomass development (Horner *et al.*, 1983) and evaluated in natural stream systems (Welch *et al.*, 1989; in press). The model is represented by the following differential equation (Horner, 1978):

$$dB/dt = k_1 * \mu * L * (k_f + k_{fo}) * (B_{max} - B) - k_2 * V^{\theta}$$

With initial condition B=0 at t=0, the integral form of the differential equation is as follows:

$$B = \{B_{max} - (k_2 * V^{\theta})/[k_1 * \mu * L * (k_f + k_{fo})]\} * [1 - e^{-k_1 * \mu * L * (k_f + k_{fo}) * t}]$$

where:

B = Biomass (mg chl $\underline{a}/m^2$)
B_{max} = Maximum sustainable biomass (mg chl $\underline{a}/m^2$)
t = Time (days)
k_f = Mass transfer coefficient for turbulent diffusion (cm/s)
k_{fo} = Mass transfer coefficient for laminar flow (cm/s)
L = Light factor (dimensionless)
μ = P uptake rate (1/day)
V = Velocity at algal cell surface (cm/s)
θ = Exponent to be derived from data (dimensionless)
k_1, k_2 = Constants to be derived from data (units to achieve dimensional consistency)

Table 3 summarizes the values or expressions chosen for the various model variables used in this report. Most were selected by Horner (1978), but some are recent modifications. The growth coefficient, k_1, was developed under running water conditions with a periphyton mat dominated by *Phormidium*; its application to a standing water condition with *Cladophora* may predict excessive growth. However, that tendency should be offset by use of specific values for the very low maximum P uptake rate and much higher half-saturation constant for P uptake developed specifically for *Cladophora* in the Great Lakes. A light factor (L) of 1 was used in the model to best approximate light conditions at 1 m depth.

Table 3. Summary of variables for periphyton accrual model from Horner (1978) with recent modifications indicated*.

Symbol	Definition	Unit	Form or value	Source
B_{max}	Maximum sustainable chl a biomass	mg/m^2	1000*	Walton (1990), Welch *et al.*, (in press)
k_1	Growth coefficient	----	1.46+0.276(P)*	Welch *et al.*, (in press)
k_2	Scour coefficient	----	0.34-0.40	Data
θ	Velocity exponent	----	0.50-0.75	Data
μ	Nutrient uptake rate	1/day	$\frac{\mu max(P)}{K_s+P}$	Michaelis-Menten formulation
μ_{max}	Maximum uptake rate	1/day	$0.22e^{T/10}$ 0.045*	Eppley (1972) Auer & Canale (1982)
K_s	Phosphorus half-saturation constant	μg SRP/L	5* 30*	Welch *et al.* (in press) Auer & Canale (1982)
L	Light factor	---	$(R/R_o)e^{1-R/R_o}$	Steele (1962)
R_o	Optimum light intensity	$\mu E/m^2$-s	8	Data, with literature guidance
k_f	Mass transfer coeff. (turbulent diffusion)	cm/s	$(DV/\pi)^{0.5}$	Canale & Weber (1972)
D	Diffusion coefficient	cm^2/s	$1.5x10^{-5}$	Pasciak & Gavis (1974)
k_{fo}	Mass transfer coeff. (non-turbulent)	cm/s	$0.009(1.018)^{(T-20)}$	Canale & Weber (1972)

RESULTS AND DISCUSSION

LONGITUDINAL SURVEY OF FDR LAKE

The nutrient, metal, and periphyton data from the longitudinal survey conducted August 21-24 are summarized in Table 4. Qualitative observations of periphyton and invertebrate abundance for the survey are summarized in Table 5. TP and SRP values averaged 15.7 (S.D. = 7.6) and 5.5 (S.D. = 3.5) $\mu g/L$, respectively. TP and SRP are also illustrated as a function of distance along the reservoir (Figure 6). According to the Ministry of Environment, the Cominco fertilizer plant was operating normally during the 1991 longitudinal survey, however, actual loading data from the fertilizer plant were unavailable.

One explanation for the downstream trend in TP and SRP is as follows. P in the upper reservoir was mostly in the form of SRP, but proceeding down lake, SRP was taken up and incorporated into phytoplankton and periphyton. Most of the P in the mid-reservoir sections occurred as particulate P and showed up as TP. The SRP concentration was consequently very low. This corresponds with the general spatial trends shown in the 1980 growing season by Stober *et al.* (1981). They observed an increase in chl a concentration downstream from the head of the reservoir to a maximum, followed by declining concentrations down to the dam. SRP also showed a similar magnitude of downstream decrease in 1980 as in this survey, as shown by the values from the Stober *et al.* (1981) data plotted in Figure 6.

TN and NO_3+NO_2-N concentrations were very low in the lower reservoir

Table 4. Data from FDR Lake longitudinal survey (August, 1991).

Station	TP (μg/L)	SRP (μg/L)	TN (mg/L)	NO_{23}-N[a] (mg/L)	NH_4-N (mg/L)	TZn (μg/L)	SolZn (μg/L)	chl a[d] (mg/m^2)	DIN/SRP
1. Spring Canyon	5.7	4.1	<0.1	<0.01	<0.005	4.0	12.0	16 +/- 3	<3.7
2. Swawilla	10.3	5.4	<0.1	<0.01	0.008	4.4	5.9	12 +/- 3	<3.3
3. Keller Ferry	7.9	7.0	0.10	<0.01	0.012	6.4	7.7	48 +/- 9[b]	<3.1
4. Penix Canyon	5.4	4.1	<0.1	<0.01	0.028	4.0	5.3	87 +/- 42	<9.3
5. Lincoln	6.8	1.9	0.10	<0.01	0.023	7.2	8.0	41 +/- 21	<17.4
6. Seven Bays	5.4	2.5	<0.1	<0.01	0.023	5.5	8.9	44 +/- 3[b]	<13.2
7. Abraham	9.3	3.2	<0.1	<0.01	0.020	6.7	13.0	31 +/- 14[b]	<9.4
8. Chimney	13.1	3.5	0.11	<0.01	0.018	5.8	5.9	6 +/- 4	<8.0
9. Enterprise	22.9	3.8	<0.1	<0.01	0.025	6.9	5.9	15 +/- 4	<9.2
10. Hunters	17.0	6.1	<0.1	<0.01	0.040	8.1	11.0	25 +/- 3[b]	<8.2
11. Bissel	30.3	3.2	<0.1	<0.01	0.023	13.0	13.0	31 +/- 10	<10.3
12.Gifford	19.4	3.5	<0.1	<0.01	0.017	13.0	5.0	8 +/- 4	<7.7
13. Daisy	27.1	4.1	<0.1	<0.01	0.011	16.0	6.5	20 +/- 18	<5.1
14. Bradbury	21.2	3.5	<0.1	<0.01	0.040	11.0	21.0	20 +/- 11	<14.3
15. Kettle Falls	18.7	4.9	0.12	0.042	0.046	9.1	10.0	22 +/- 4[b]	18.0
16. Evans	17.0	4.1	0.12	0.052	0.006	7.9	10.0	40 +/- 6[b]	14.1
17. North Gorge	17.7	6.4	0.14	0.057	0.054	7.9	8.6	11 +/- 6	17.3
18. China Bend	24.7	15.4	0.14	0.065	0.033	16.0	11.0	8 +/- 2	6.4
19. Northport	20.0	13.6	0.13	0.058	0.030	14.0	12.0	38 +/- 11	6.5
20. Border	13.5	9.3	0.14	0.051	0.033	11.0	18.0	-----[c]	9.0

[a] NO_{23}-N = NO_2+NO_3-N

[b] *Cladophora* or filamentous green algae abundant

[c] limited access due to swift current

[d] +/- indicates standard deviation of 3 measurements at each station

Table 5. Periphyton and invertebrate observations from FDR Lake longitudinal survey.

Station	Relative Periphyton Coverage	Invertebrates
1. Spring Canyon	100% filamentous greens below 25"	midges, mites, snails- all minute, couple per cm^2
2. Swawilla	80% periphyton on available substrate; <5% filamentous greens	midges, mites, snails- not abundant; One caddis fly
3. Keller Ferry	20% periphyton at 25" on available substrate; 100% filamentous greens; large clumps of *Cladophora* found growing attached to wire mesh	small midges abundant
4. Penix Canyon	100% periphyton at 35" on available substrate; 0% filamentous greens	small snails, midges- only on rocks with periphyton
5. Lincoln	100% periphyton; 5% filamentous greens	low abundance, rocks covered with sediment
6. Seven Bays	100% coverage filamentous greens	midges, snails- quite abundant
7. Abraham	100% periphyton coverage; large clumps of floating *Cladophora* growing attached or lodged to rooted plants	midges, mites
8. Chimney	10% periphyton coverage; 0% filamentous greens	very few found only on limited amount of periphyton
9. Enterprise	100% periphyton; 80% filamentous greens	limited midges, snails, mites- all not abundant
10. Hunters	100% periphyton coverage; 90% filamentous greens	midges, mites, snails- all not abundant
11. Bissel	80% periphyton coverage; <5% filamentous greens	small midges present- not abundant

Table 5. (continued)

Station	Relative Periphyton Coverage	Invertebrates
12. Gifford	Sparse coverage; 50% periphyton coverage; 50% filamentous greens	small midges, snails- not abundant
13. Daisy	100% periphyton coverage below 25"; 90% filamentous greens	midges, snails- very sparse
14. Bradbury Beach	100% periphyton coverage beyond 25" on available substrate; 90% filamentous greens	midges, mites, snails- not abundant
15. Kettle Falls	100% periphyton coverage below 25"; 90% filamentous greens	small midges very abundant
16. Evans	100% periphyton coverage below 12"; 80% filamentous greens	small midges- relatively scarce
17. North Gorge	Very sparse coverage; 25% periphyton on available substrate; 0% filamentous greens	small midges, snails- not abundant
18. China Bend	40% periphyton coverage on available substrate; 90% filamentous greens	small midges and snails quite abundant
19. Northport	60% periphyton coverage; 90% filamentous greens	small midges abundant
20. Border	---------------[a]	-----------[a]

[a] see Table 4

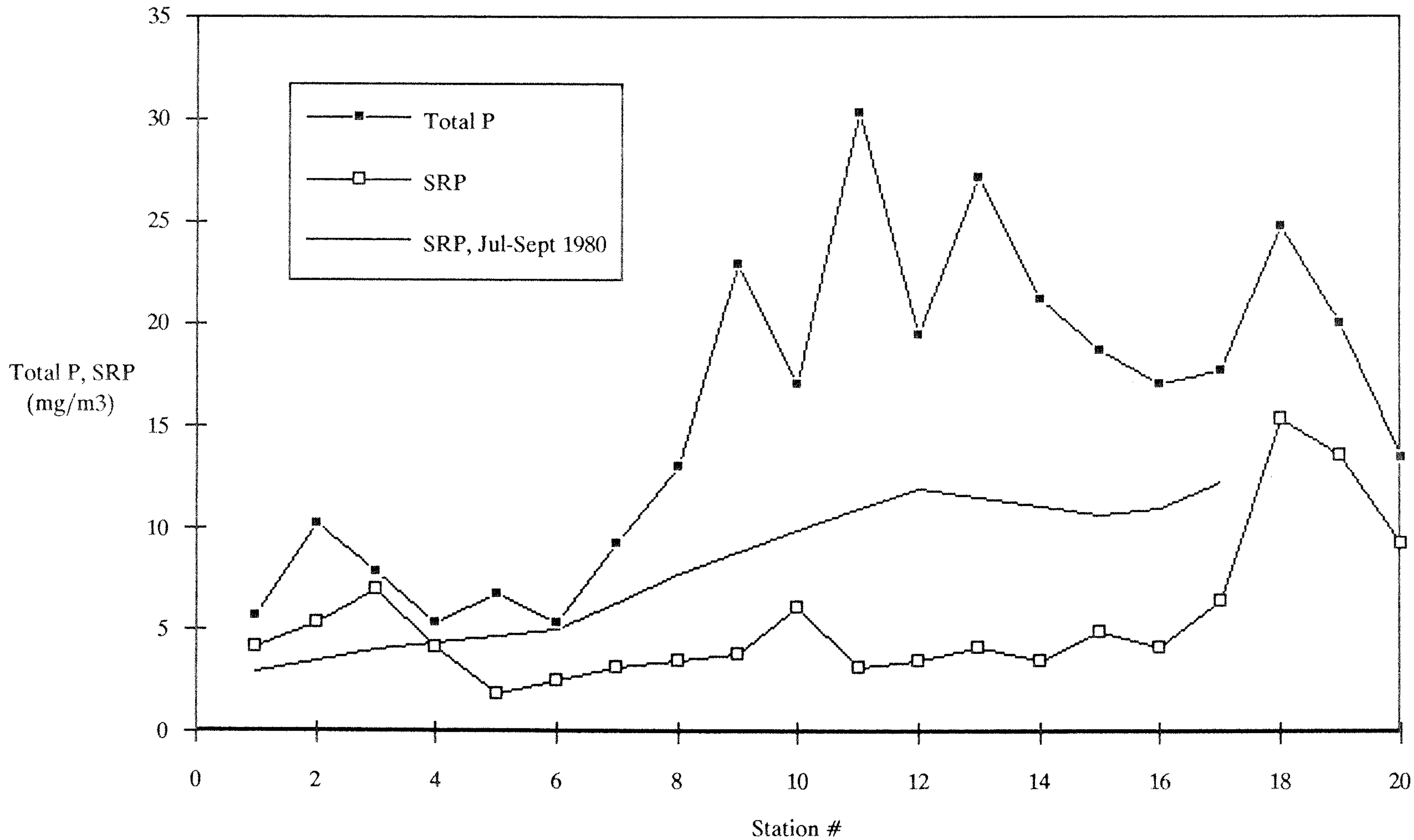

Figure 6. Total P and SRP by station for FDR Lake August, 1991 longitudinal survey. Line without symbols represents average SRP by station for the period July - September 1980 (data from Stober *et al.*, 1981).

(stations 1-14), while they averaged 132 and 54 $\mu g/L$, respectively, from Kettle Falls to the Canadian border (stations 15-20). Low surface nitrogen concentrations in the lower reservoir are probably due to a combination of uptake by phytoplankton and periphyton and decreased mixing between surface and bottom waters, resulting from increased stratification proceeding down lake.

The thermal regime in FDR Lake is, as is the case in most water bodies, dependent primarily on solar insolation and turbulence (Stober *et al.*, 1981). In the upper reservoir, turbulence from the inflow of the Columbia River is high, keeping the water column well mixed with no stratification developing. As one moves down through the reservoir, turbulent mixing decreases, allowing solar insolation to create water column density differences sufficient to restrict vertical mixing, resulting in the development of stratification. Jaske (1969) characterized the typical thermal cycle in FDR Lake as follows:

1) January - June: generally mixed and turbulent due to high flow, developing little or no stratification.
2) July - September: little mixing, with stratification in the lower reservoir at the discharge end, at times extending to Kettle Falls.
3) October - December: cooling and unmixed with oblique orientation of the isotherms.

The DIN concentrations throughout much of the reservoir were relatively low, and NO_3+NO_2-N was below the level of detection of 10 $\mu g/L$ downstream of Kettle Falls (Table 4). Such low DIN concentrations resulted in relatively low DIN/SRP ratios (Table 4). Although the sensitivity limitation for NO_3+NO_2-N presents considerable uncertainty in judging ratios, the mean ratio where

NO_3+NO_2-N is lowest (stations 1-14) is less than 8.7 (S.D. = 4.2). The actual ratio is probably much less than this, considering that NO_3+NO_2-N is <10 μg/L at all of the stations from 1 to 14. On average, ratios below about 7/1 are considered to indicate N limitation (Redfield *et al.*, 1963); that is, if algae take up N and P at their average cell content of 7N/1P, N will be depleted first. The fact that SRP was present at around 5 μg/L through much of the reservoir indicates also that N was limiting; i.e., N was depleted, potentially limiting further biomass increase, with P being left in surplus. In FDR Lake, not only would N probably be exhausted first, but these low levels indicate that uptake by phytoplankton was also probably limited at the time of sampling.

The actual mean ratios for TN/TP at stations 3, 5 and 8, where TN was at or above detection, was 11.9, which according to analyses by Smith (1982) is probably borderline between N and P limitation. Smith showed from chl a-TP relationships, that lakes with N/P ratios < 10 also had low chl a levels and were probably limited by N. Existing data from FDR Lake (1982-1988) indicate that N was probably limiting in only one third of the samples (Figure 7). The ratios of soluble nutrients are, however, probably a better indication of algal demand-supply at the time of sampling and, from those values, N appears to be the limiting nutrient in FDR Lake. N limitation would appear to be induced by the high P loading. This will be discussed further under model predictions.

TZn and SolZn values were quite low at all of the stations sampled (Table 4). In most cases, SolZn values were equal to or greater than TZn values. The samples for SolZn may have been contaminated with ultra-trace amounts of zinc during the field filtration process, although the filter system was plastic and

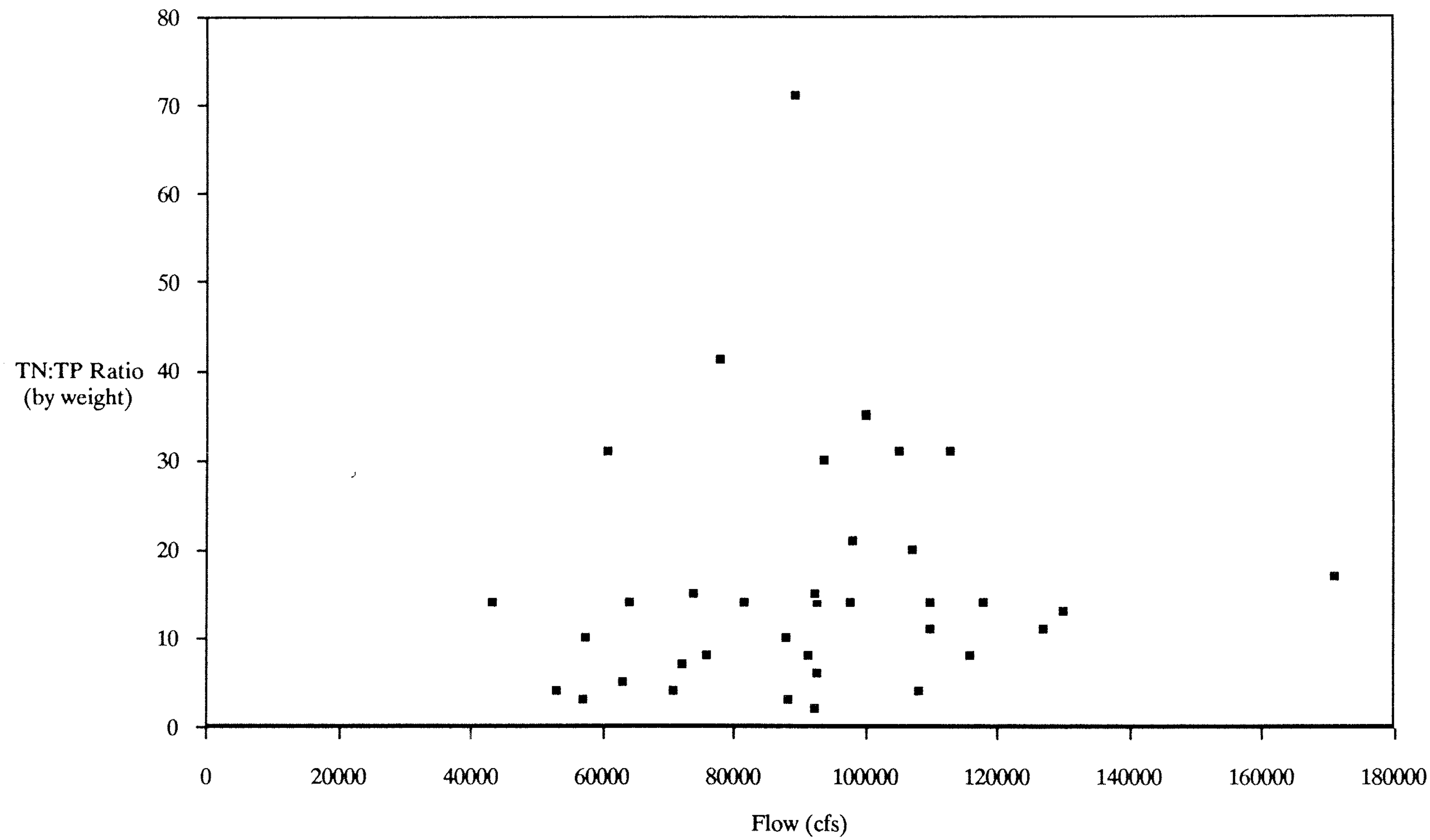

Figure 7. TN:TP ratio vs. flow for Columbia River at Northport for period 1982 to 1988 (data from USGS).

reasonable care was taken to rinse and avoid contamination. The low TZn concentrations observed in the longitudinal survey are consistent with a previous study assessing metals in FDR Lake (Johnson *et al.*, 1989), in which TZn concentrations in the lake and its tributaries were reported to average <1 and from <1 - 52 μg/L, respectively.

Periphyton biomass measured in the longitudinal survey ranged from 6 to 87 mg chl a/m^2 (Table 4). There was no observable trend in periphyton biomass with distance along the reservoir. *Cladophora* was observed in large floating clumps attached to or lodged in wire riprap and rooted plants at the Keller Ferry and Abraham stations. In addition, significant amounts of filamentous green algae were observed at the Seven Bays, Hunters, Kettle Falls, and Evans stations. The low periphyton biomass (less than nuisance levels; <100 mg chl a/m^2, see p. 70) observed in this survey during August may have been a result of the large increase in water level that is typical during the period from April to July, as was shown previously (Figure 4). Broch and Loescher (1988b) experienced similar difficulty with sample site selection during the period of high water level in July - August 1988 and conducted additional sampling during the month of October, 1988. Sites sampled in August, 1988 may not have been inundated long enough for *Cladophora* to develop, because a greater distribution of the alga was found in October, 1988.

SUMMER GROWING SEASON DATA

Seasonal data for the period April - September 1991 at the Keller Ferry Station are summarized in Table 6. Phytoplankton steadily increased until a maximum of 3.8 μg chl a/L was reached in mid-July, and then decreased rapidly

Table 6. 1991 growing season (April-September) data from Keller Ferry station.[a]

Date	TP (μg/L)	SRP (μg/L)	TN (mg/L)	NO_3+NO_2-N (mg/L)	NH_4^+-N (mg/L)	Secchi (m)	chl a (μg/L)
04/10/91	11	10[b]	----	0.13	0.01[b]	3.1	0.69
05/15/91	14	10[b]	0.14	0.01[b]	0.01[b]	2.1	2.00
06/12/91	10[b]	10[b]	0.12	0.02	0.01[b]	2.3	1.48
07/17/91	12	10[b]	0.12	0.01[b]	0.01[b]	3.0	3.82
08/14/91	10[b]	10[b]	0.13	0.01[b]	0.03	7.3	0.80
09/11/91	12	10[b]	0.16[c]	0.03	0.01	11.6	0.01

[a] data from Ecology
[b] detection limit
[c] estimated value

through August and September (see Figure 8). Secchi depth was low from April to July due mainly to increasing turbidity resulting from spring runoff (Stober *et al.*, 1981). Phytoplankton biomass was too low to cause such poor transparency. The transparency increased from July to September due to reduced runoff and the decrease in phytoplankton biomass.

NUTRIENT AND PHYTOPLANKTON MODELING

In order to develop the water budgets, nutrient loading models, and eutrophication response models, a time period for averaging had to be chosen. The period chosen was the 1980 growing season (April - September), because this was the only year in which seasonal, longitudinal pool data exist to assess the accuracy of the model output at predicting pool nutrient content and trophic state response (Secchi depth, chl a). TP loading from the Columbia River in 1980 appears to be fairly representative of the past 10 years. Figure 9 shows the TP load in the Columbia River for the period April - September for 1980-1989. The hydraulic loading over the same period was relatively constant (mean +/- 18%), therefore, the large spike in 1981 was due to elevated TP concentration in the Columbia River and not due to elevated flow. According to the Ministry of Environment, the Cominco fertilizer plant discharged about 1,500 kg/day during 1980 or 0.27 x 10^6 kg over the averaging period used for modeling (April-September, 1980). The plant discharged an average of about 4,700 kg/day or 0.86 x 10^6 kg over the averaging period during 1980-1989. Thus, 1980 was a low P-discharge year. Because flow was relatively constant over the 10-year period, it seems anomalous that P loading was so high in 1980 given the low P discharge from Cominco.

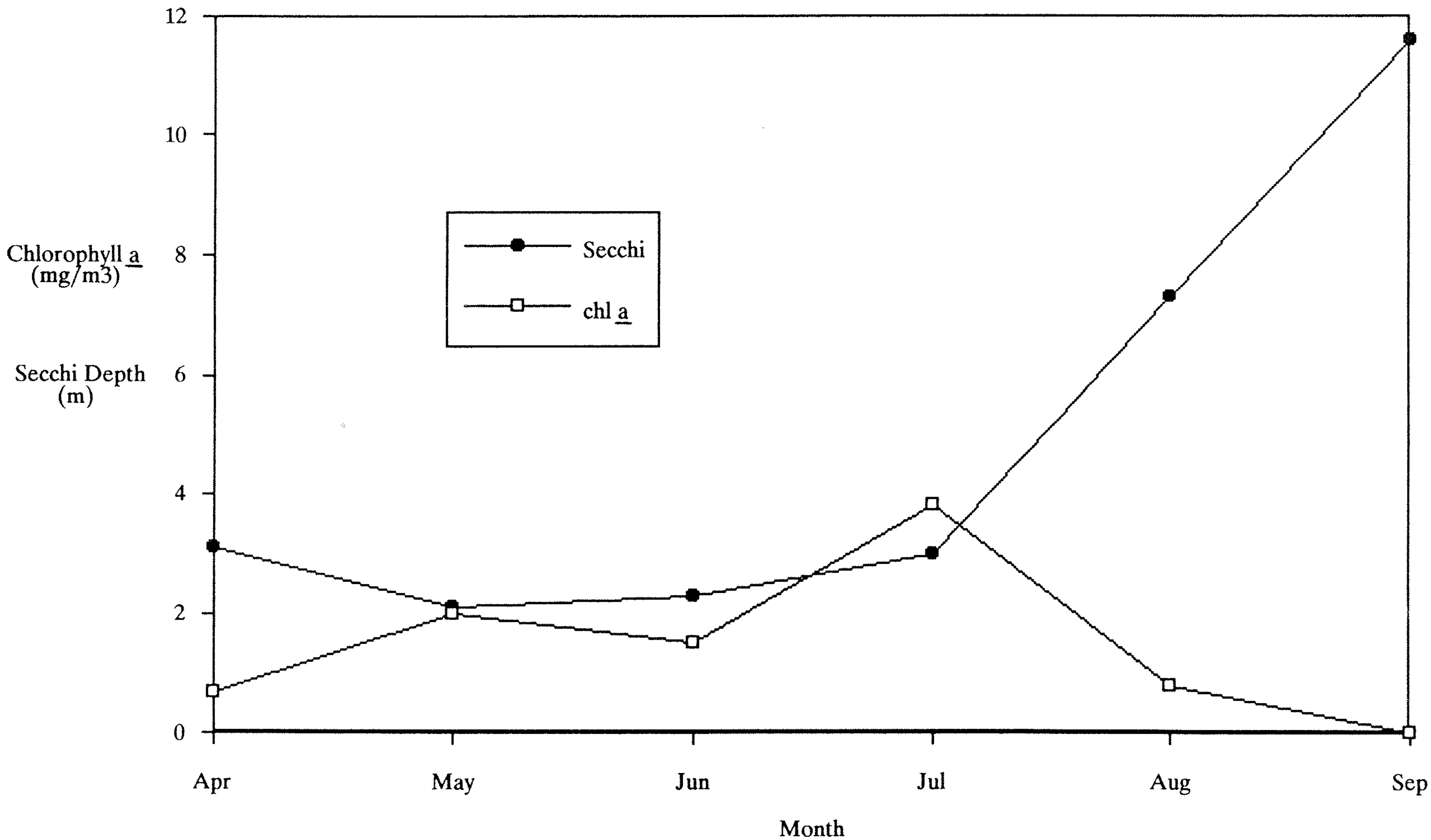

Figure 8. Chlorophyll a and Secchi depth for 1991 growing season (April - September) at Keller Ferry station (data from Ecology).

Figure 9. Columbia River TP load for April-September for period 1980-1989 (data from USGS).

Water Budget

Table 7 provides a summary of the water budget for the averaging period (April-September 1980) used for nutrient and phytoplankton modeling. The hydraulic input to FDR Lake was dominated by the Columbia River, which contributed 84.5 percent of total averaging period input, while the Spokane, Kettle, and Colville Rivers contributed 6.4, 4.4, and 0.3 percent, respectively. Ungauged inflow and precipitation contributed the remaining 4.5 percent total input. The hydraulic residence time for the averaging period was 42 days, with a hydraulic overflow rate of 302 m/y.

Nutrient Loading Budgets

Nutrient loading budgets were calculated for TP and TN using Walker's nutrient loading model, the results of which are summarized in Tables 8 and 9. Similar to the results of the water budget, nutrient input to FDR Lake was dominated by the Columbia River, which contributed 85.5 and 84.9 percent of the total averaging period nutrient input for TP and TN, respectively. The pool concentrations predicted by Walker's nutrient loading model for TP and TN were 32 and 575 mg/m^3, respectively. Predicted pool TP concentration compared very well with that observed by Stober *et al.* (1981), while comparison of predicted and observed TN was not possible because there were no observed TN data from 1980 (see Table 10).

Table 7. Water budget for period of April-September 1980.[a]

Source	Input/Output (x 10^6 m^3)	% of Total Input/Output
Columbia River	42,882.3	84.5
Spokane River	3,238.0	6.4
Kettle River	2,258.0	4.4
Colville River	142.5	0.3
Ungauged Inflow	2,168.8	4.3
Precipitation	77.8[b]	0.2
Total Inflow	50,767.4	100.0
FDR Lake Outflow	45,207.9	89.0
Evaporation	252.2[b]	0.5
ΔS (Change in Storage)	5,307.3	10.5
Total Outflow + ΔS	50,767.4	100.0

[a] data from USGS (1980)

[b] data from National Weather Service (1981)

Table 8. Nutrient (TP) budget for period of April-September 1980.

Source	Input/Output (kg)	% of Total Input/Output
Columbia River	1,989,737[a]	85.5
Spokane River	119,804[a]	5.1
Kettle River	102,738[a]	4.4
Colville River	8,195[a]	0.4
Ungauged Inflow	101,066[b]	4.3
Precipitation	5,611[c]	0.2
Total Input	2,327,151	100.0
FDR Lake Outflow	1,458,707[d]	62.7
ΔS (Change in Storage)	171,247[e]	7.4
Net Retention	697,197[f]	30.0
Outflow + ΔS + Net Retention	2,327,151	100.0

Predicted Pool TP Concentration = 32 mg/m^3

[a] Calculated with FLUX program- product of flow weighted mean concentration and mean flow for the averaging period.

[b] Calculated as product of ungauged inflow from water balance and mean concentration of the major gauged tributaries for the averaging period.

[c] Calculated as product of precipitation from water balance and 80 mg/m^3 (NWS, 1981).

[d] Calculated by BATHTUB program- product of predicted pool concentration and observed outflow from water balance.

[e] Calculated by BATHTUB program- product of predicted pool concentration and observed change in storage from water balance.

[f] Calculated by BATHTUB program- using second-order nutrient sedimentation model.

Table 9. Nutrient (TN) budget for period of April-September 1980.

Source	Input/Output (kg)	% of Total Input/Output
Columbia River	39,580,315	84.9
Spokane River	2,906,384	6.2
Kettle River	2,054,767	4.4
Colville River	129,698	0.3
Ungauged Inflow	1,973,606	4.2
Total Input	46,644,770	100.0
FDR Lake Outflow	26,008,150	55.8
ΔS (Change in Storage)	3,053,270	6.5
Net Retention	17,583,350	37.7
Outflow + ΔS + Net Retention	46,644,770	100.0

Predicted Pool TN Concentration = 575 mg/m^3

Table 10. Comparison of Walker's model predictions with observed data from Stober *et al.* (1981). Calculated for the period April - September 1980.

Parameter	Predicted[a]	Observed[b]
Pool TP (mg/m^3)	32	32
Pool TN (mg/m^3)	575	------
Chlorophyll a (mg/m^3)	7.6[c]	5.4
Secchi Depth (m)	2.4	2.8

[a] Calculated with BATHTUB portion of Walker's model.

[b] Volume weighted mean reservoir concentrations calculated from observed monitoring data (Stober *et al.*, 1981).

[c] Models #2, 3, and 4 predict, respectively, 8.0, 10.5, and 9.0 mg/m^3.

Trophic State Response to Nutrient Loading

Predicted pool nutrient concentrations (TP,TN) and trophic state response variables (chl a, Secchi depth) were calculated for the averaging period (April-September 1980) using the BATHTUB portion of Walker's model. A summary of predicted whole pool nutrient concentrations and trophic state response parameters, compared to observed values from Stober *et al.* (1981), is shown in Table 10. Calculations for chl a were made using Walker model #1, which accounts for algal growth limitation by P, N, light, and flushing. The other three models predict higher chl a concentrations for 1980 (Table 10).

For management purposes, it would be instructive to analyze the response of pool nutrients and trophic state response variables to variations in nutrient loading of both TP and TN from the Columbia River. Figure 10 shows the responses of predicted pool chl a, calculated with Walker's four different chl a models described previously, to variation in TP loading from the Columbia River. TP loading was varied by altering the river TP concentration. That would tend to simulate the effect of Cominco loading, which affects concentration rather than flow. Each model assumes algal growth limitation by different parameters, as indicated in Figure 10. Also included in the figure is the current maximum permit load from the Cominco fertilizer plant. The calculation was based on Cominco operating at maximum permit load for the entire averaging period (6 months @ 11,170 kg P/day). From the chl a models shown in Figure 10, it is apparent that N, light, and flushing all can theoretically contribute to limiting the growth and biomass of phytoplankton in FDR Lake. Inclusion of N reduces biomass whether light and flushing are included or not. Although loss of phytoplankton biomass by flushing

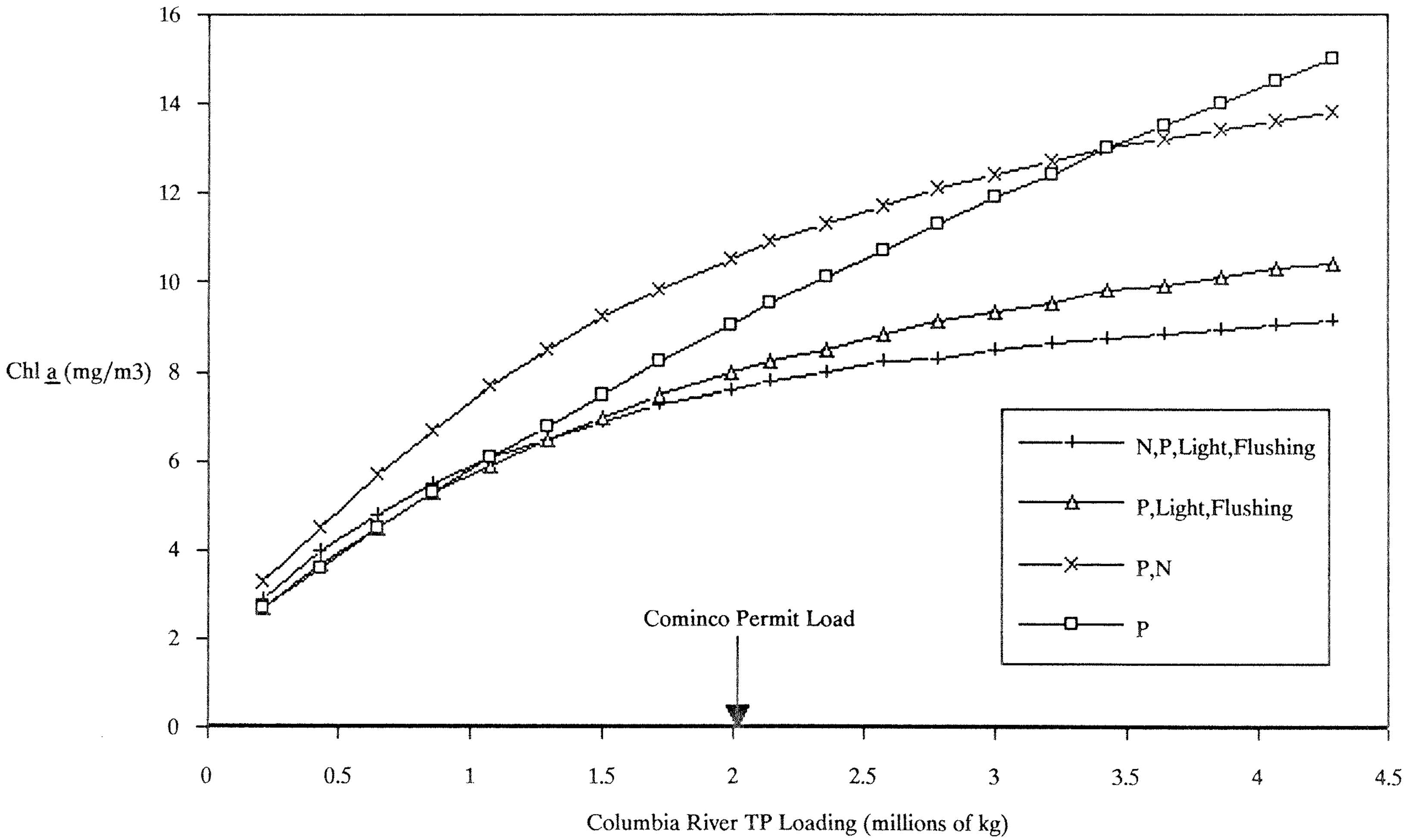

Figure 10. Resulting pool chl a concentration as a function of TP loading from the Columbia River, as predicted by Walker's four different chl a models.

probably has some effect, the relatively low transparencies (see Table 6), due to non-algal turbidity, likely represent a greater effect. Considering the model output, and the earlier discussion indicating possible N-limitation of phytoplankton in FDR Lake, chl a model #1 is considered the most appropriate for FDR Lake, because it accounts for growth limitation by all four factors; N, P, light, and flushing. Output from this model also agreed best with the concentration of 5.4 $\mu g/L$ chl a (calculated as growing-season, volume-weighted average from Stober *et al.*, 1981) observed for the averaging period TP load of 2 million kg from the Columbia River (Figure 10).

FDR Lake was also modeled by splitting the reservoir into two segments, segment #1 being that portion of the reservoir from river mile 660 to the border, and segment #2 being that portion of the reservoir from Grand Coulee Dam to river mile 660. This was done to more realistically project the downstream distribution of SRP, which tends to be higher in segment #1 than in segment #2 (see Figure 6). The mean detention times, calculated as the ratios of mean annual minimum storage (in this case for the last 10 years) to the 30-day, 10-year low flow (37,100 cfs), are 23 days and 51 days for segment #1 and segment #2, respectively. Thus, "lake class" water quality standards apply to both the upper and lower segments of FDR Lake.

Predicted pool concentrations of TP and SRP as a function of inflow Columbia River TP load are displayed in Figure 11. Pool TP values were calculated by Walker's nutrient loading model, and SRP pool concentrations were derived from TP values assuming a TP/SRP ratio of 2.8 (Average pool TP/SRP ratio derived from Stober *et al.*, 1981). Because of less dilution and sedimentation

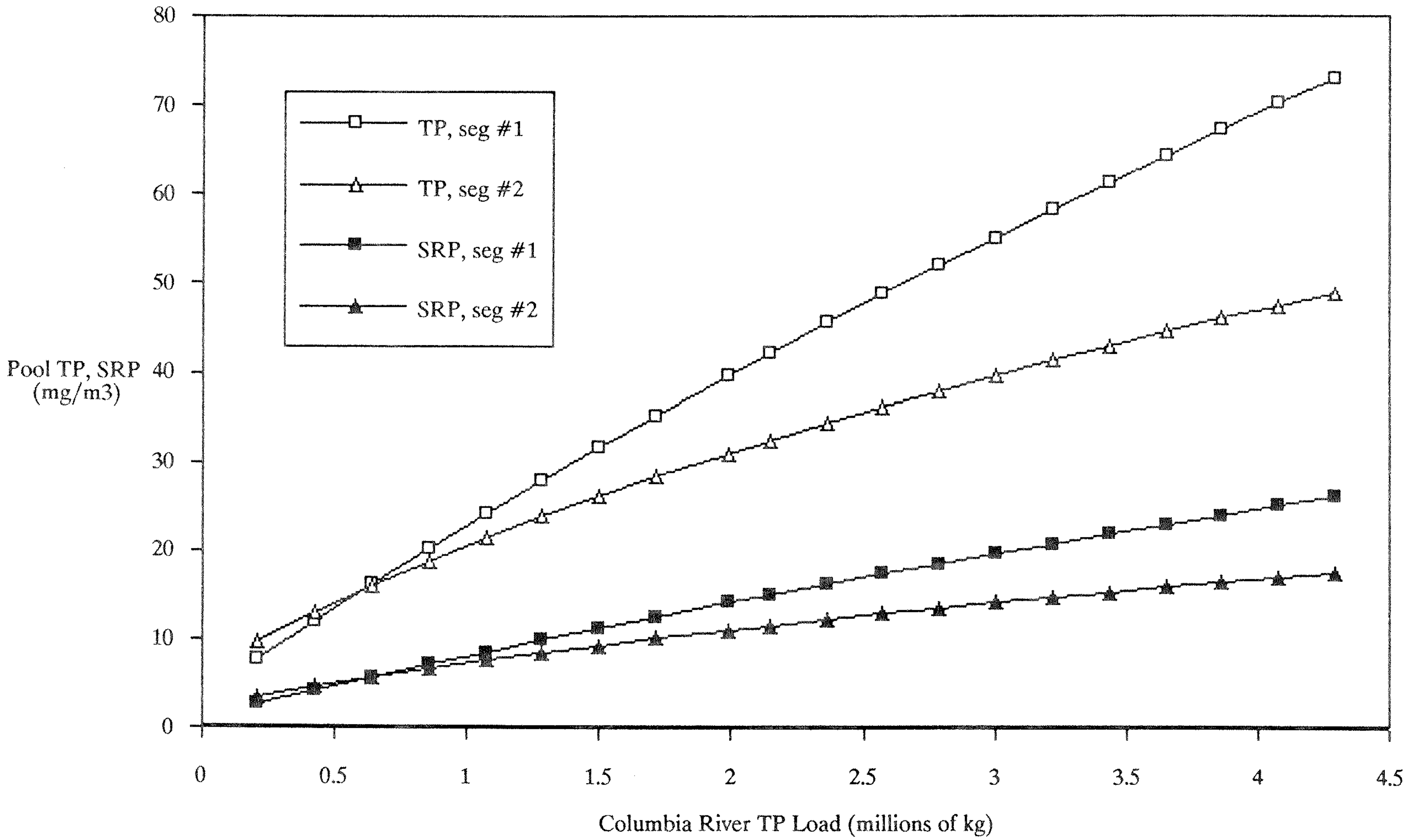

Figure 11. Resulting pool TP and SRP concentrations for upper reservoir (segment #1) and lower reservoir (segment #2) as a function of TP loading from the Columbia River, as predicted by Walker's model.

loss, predicted TP and SRP concentrations in the upper reservoir (segment #1) were larger than for the lower reservoir (segment #2), with the difference increasing with increasing TP loading. Model predictions matched the patterns observed in the 1991 longitudinal survey and the 1980 study, where TP and SRP values for segment #1 (stations 11-20) were generally higher than those for segment #2 (stations 1-10).

The chl a concentrations and Secchi depths corresponding to the predicted pool TP concentrations for segments #1 and #2 are shown in Figure 12. Predicted chl a concentrations in segment #1 were larger than those for segment #2, corresponding to the larger TP concentrations. As a result, predicted Secchi depth is lower in segment #1 than for segment #2. The model suggests that average chl a values in the upper and lower reservoir, respectively, will not increase much above 10 and 9 μg/L even with a doubling of TP loading from the Columbia River. Although mean summer chl a concentrations greater than approximately 9 μg/L indicate an eutrophic state (Porcella, *et al.*, 1980), and trophic criteria are based on means, significant water quality degradation due to phytoplankton, beyond the current potential at about 2 x 10^6 kg TP (1980 loading), is not likely to occur in FDR Lake with increased loading. That is because of biomass limitation by N, light, and flushing. (However, note caveat at the end of this section.) More important to the question at hand, only a slight reduction in phytoplankton biomass (approximately 1-2 μg/L chl a) would be expected if TP loading from the Columbia River were halved, and there is no evidence that the current level of loading is in any way presenting a nuisance in so far as phytoplankton are concerned. On the contrary, recommendations were made to enhance phytoplankton production from the standpoint of fisheries (Beckman *et al.*, 1985).

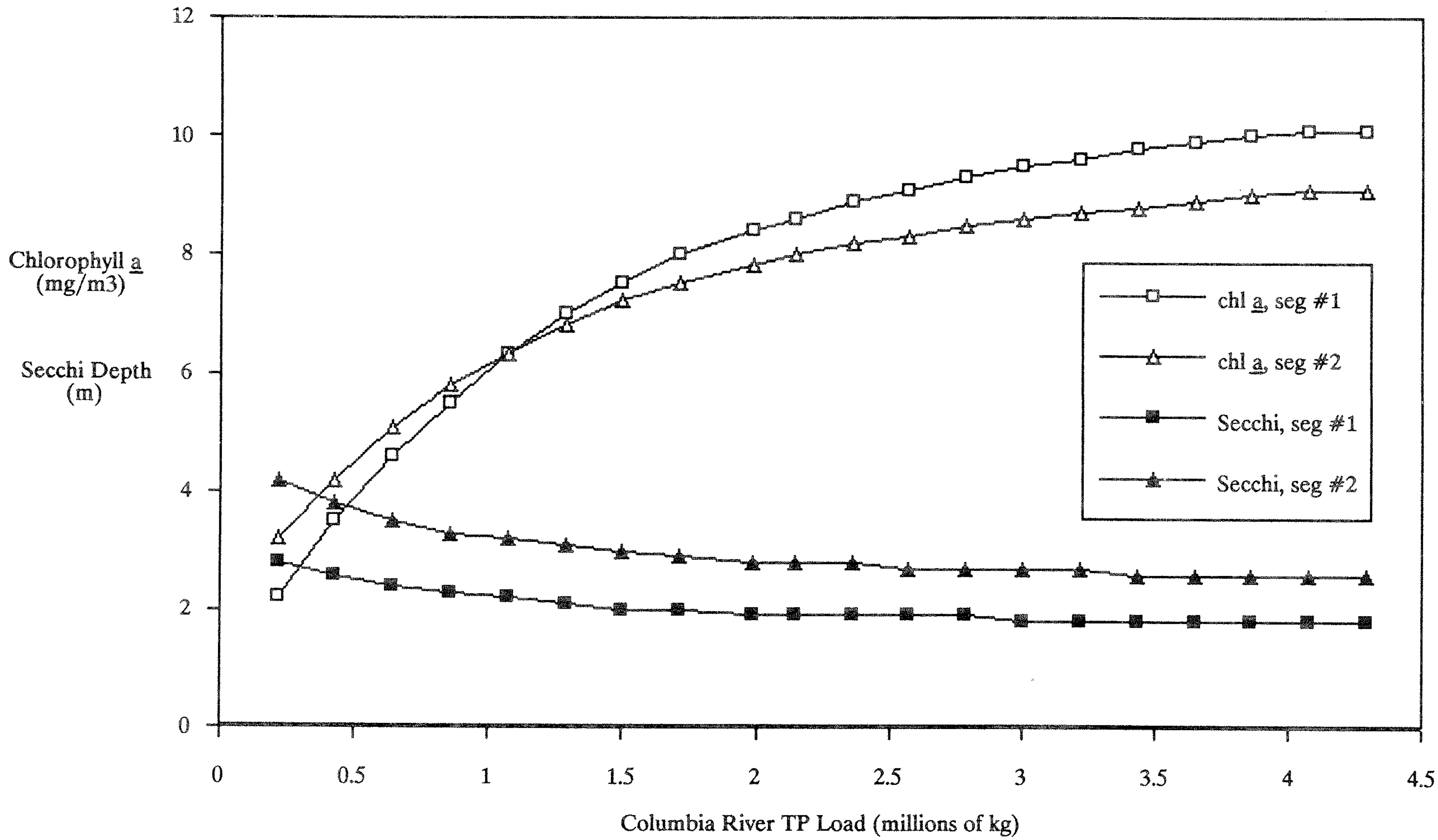

Figure 12. Predicted pool chl a concentrations and Secchi depths for upper reservoir (segment #1) and lower reservoir (segment #2) as a function of TP loading from the Columbia River.

As a result of only slight changes in chl a expected from changes in TP loading, little change in hypolimnetic oxygen depletion rate (HODR) would also be expected from a fractional change in loading from the current level. Walker's model for HODR is a function of mean water column chl a concentration, but its output was not explored here due to the relative insensitivity of phytoplankton chl a to TP. While hypolimnetic DO does deplete at several mid-reservoir stations in FDR Lake during summer (to a minimum of approximately 50 % saturation), Stober *et al.* (1981) concluded that such DO levels, along with generally high oxidation-reduction potentials, indicated the prevalence of oxidative conditions throughout FDR Lake. Moreover, the prediction of HODR is one more level removed from TP-chl a and, therefore, subject to greater uncertainty.

A similar exercise was performed with respect to variations in TN loading from the Columbia River, with TP loading and other parameters held constant. Predicted whole-pool TN and DIN concentrations, as a function of TN loading from the Columbia River, are shown in Figure 13. Whole-pool TN was calculated using Walker's nutrient loading model, and DIN was estimated from TN using a constant TN/DIN ratio of 5.75 (inflow TN/DIN ratio for Columbia River for the averaging period). As was done for TP, chl a concentration using each chl a model was predicted as a function of inflow TN load from the Columbia River and the result is displayed in Figure 14. The maximum permitted Cominco TN load is not shown because it is relatively small (approximately 443,000 kg N for the averaging period). Observed TN loading from the Columbia is about 40 million kg for the period (Table 9). The two curves show that, given the observed TP load, P will quickly become limiting if TN loading increases beyond the observed rate.

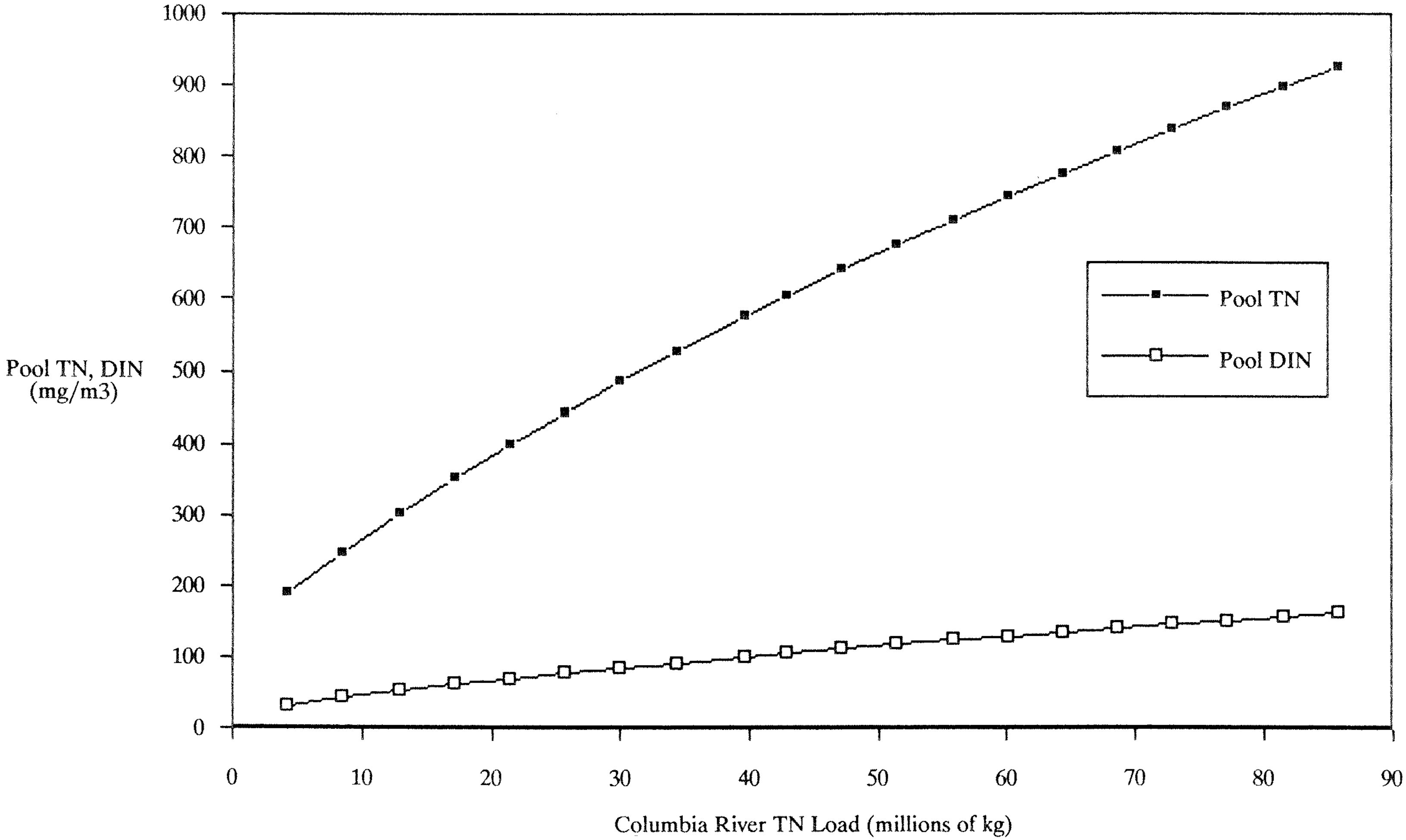

Figure 13. Predicted whole-pool TN and DIN concentrations as a function of TN loading from the Columbia River, as predicted by Walker's model.

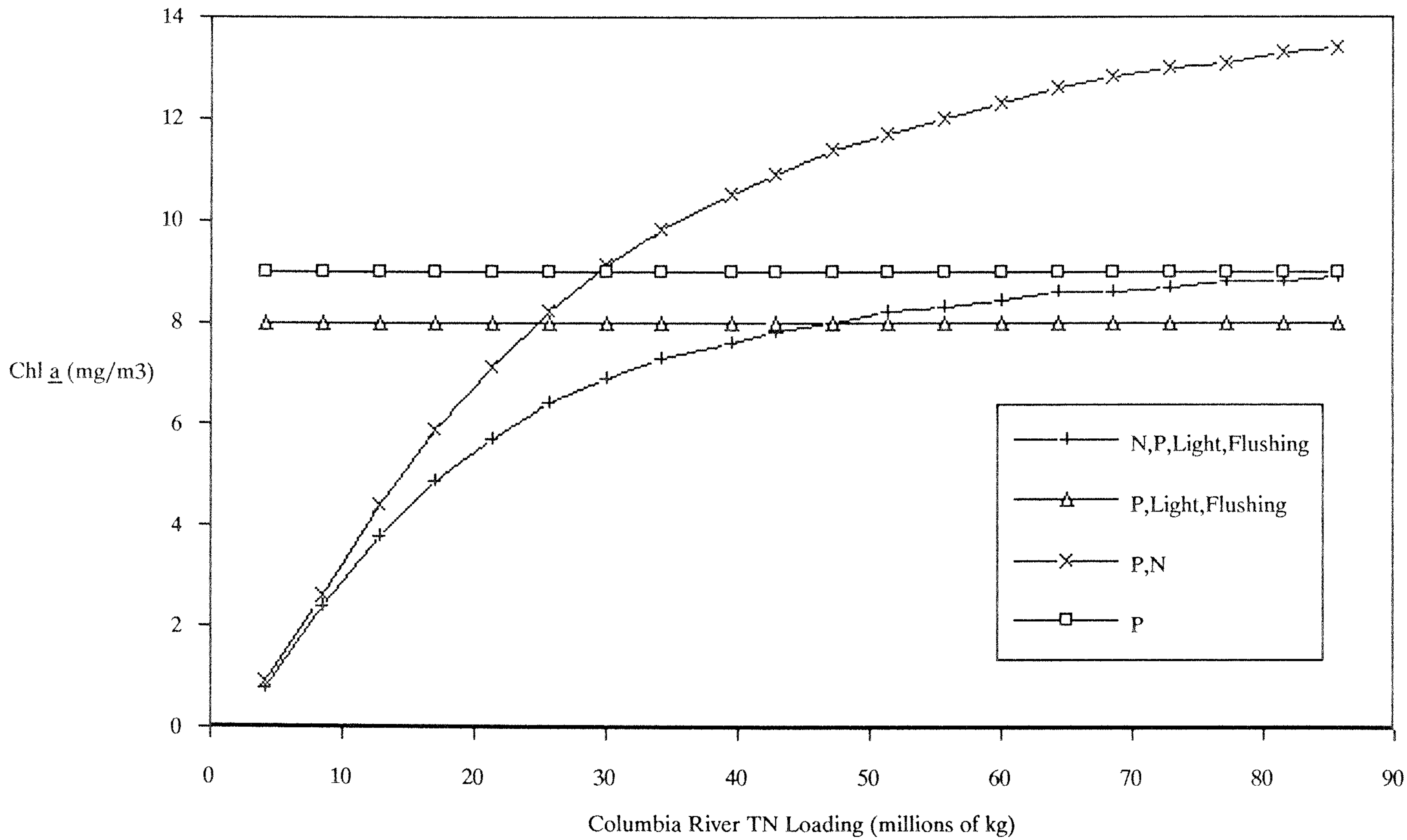

Figure 14. Resulting pool chl a concentration as a function of TN loading from the Columbia River, as predicted by Walker's four different chl a models.

Therefore, the models suggest that phytoplankton is limited about equally by N and P. The flat lines in Figure 14 are due to P only models and TP load is constant.

There should be a caveat added, however, for the expectations depicted in Figure 10 if loading were increased. The model includes nothing about the response of phytoplankton species composition to increased P loading. Selection of N-fixing blue-green algae in response to increased P loading is well known (Smith, 1990). *Anabaena*, an N-fixer, did occur at low levels in July-August of 1980. However, N-fixation is a rather slow, energy-demanding process, with resulting maximum growth rates being very low (0.05/day; Horne and Goldman, 1972). Therefore, the relatively short detention times in the reservoir could prevent significant accumulation of N-fixing blue-greens even if they were favored by increased P loading. Nevertheless, blue-green blooms and mean chl a concentration over 20 μg/L occurred in Long Lake, a reservoir on the Spokane River, in response to increased P loading, and its mean detention time is only 25 days (June-November) (Harper-Owes, 1985). A ten-fold increase from say 2 μg/L chl a should not occur from exponential growth at 0.05/day over 25 days. However, there is short circuiting in the reservoir as the inflow plunges during summer low flow so the surface layer has a longer-than-mean detention.

PERIPHYTON MODELING

Longitudinal Survey

The results of scraping rocks during the August longitudinal survey showed relatively low biomass levels and no definite longitudinal trend (Table 4). If P loading to the river upstream from the lake were responsible for increased periphyton biomass in the lake, one might have expected a trend of decreasing SRP along with decreasing periphyton biomass, especially that of *Cladophora*, downstream toward the dam. Such distributions of *Cladophora* related to P sources have been shown in the Laurentian Great Lakes (Auer *et al.*, 1983). SRP is more pertinent than TP, because that form determines the uptake rate of P by the periphyton mat, and for other reasons due to the differences between the response of periphyton and phytoplankton to nutrient concentration increase as explained earlier. Furthermore, SRP is the nutrient form used to model *Cladophora* biomass in the Great Lakes (Auer *et al.*, 1982). However, while the August survey showed SRP to decrease longitudinally toward the dam, periphyton biomass showed no longitudinal trend (Table 4). Moreover, *Cladophora* was observable at only a few stations and then it was either floating free or lodged in wire riprap along the shore or on submerged branches. As mentioned before, the shoreline sampled in August may not have been inundated long enough for *Cladophora* to have become established.

Cladophora nonetheless occurs in FDR Lake and presents an obvious nuisance condition. The suspended, settled and caught strands of *Cladophora* at the Abraham station (#7) in August, for example, definitely presented a nuisance condition to recreationists. The question is, has nutrient enrichment, especially by P, favored the development of nuisance or near nuisance biomass levels of *Cladophora* in FDR Lake and what loading and/or concentration of SRP would have prevented its biomass from reaching nuisance proportions? To provide answers to that two-part question, some experimental results of *Cladophora* response to environmental variables and two modeling approaches will be discussed. The literature on growth response to enrichment will be relied upon, because *Cladophora* could not be established in artificial channels in the time allotted for this study (see recommendations).

Growth Requirements for *Cladophora*

Cladophora is probably the most important nuisance periphytic alga and becomes dominant in moderate to highly enriched streams and lakes. Its occurrence in nuisance proportions has usually been linked with high P loading in the Laurentian Great Lakes (Auer and Canale, 1982; Lin, 1971; and Neil and Owen, 1964) and other fresh and brackish waters throughout the world. The Great Lakes experience is probably most pertinent to the case of FDR Lake.

A thorough analysis of the *Cladophora glomerata* (typically the problem species) problem at Harbor Beach in Lake Huron, south of Saginaw Bay, Michigan, offers the basic physiological and ecological understanding of this nuisance species that may be most pertinent to the FDR Lake situation. Auer and his co-workers found that *Cladophora* reached a maximum biomass of 200-300 g dry wt/m^2 (about 450-675 mg chl a/m^2) near a source of P at Harbor Beach. Internal cell P content varied from about 0.4% where SRP was high to 0.1% at 0.5 km from the site, where SRP had declined to relatively low levels. Half-saturation kinetic constants for SRP determined experimentally varied with cell P content and ranged from 30-250 μg/L (Auer and Canale, 1982). SRP uptake was greater at low cellular P levels. They concluded that *Cladophora's* half-saturation constants were not much different than for phytoplankton in general (15-480 μg/L). Other literature reviews have found a lower range for phytoplankton (1-50 μg/L; EPA, 1987). However, Rosemarin (1982) found a lower range for half-saturation for *Cladophora* than Auer and Canale (6-31 μg/L). The Great Lakes P kinetic work was conducted in 2-L beakers equipped with stirring bars. While that probably is appropriate for standing water, or wave-washed beach areas, it would probably not represent the turbulence that a periphytic mat experiences in running water, where P saturation occurs at much lower concentrations (see below).

The maximum specific uptake rate (μ_{max}) for the Harbor Beach algae was 0.045/day. Such a slow rate is due to the large size of that alga; in contrast, small phytoplankton have maximum uptake rates of 0.25-1.5/day (Auer and Canale, 1982). Uptake rate and the half-saturation constant varied inversely as internal cell P varied from 0.1-1.0%. The optimum range for temperature and light intensity for net photosynthesis was 13-17 °C and 300-600 μE/m^2-s (Graham *et al.*, 1982). As a result of the interaction of light and temperature, the optimum depth for growth of *Cladophora* was one meter; net photosynthesis decreased greatly at higher temperature and light.

The significance of the above environmental requirements of *Cladophora* is that in a lake with little turbulence its growth will apparently be enhanced at ambient SRP concentrations as high as 50 μg/L, or even higher. Moreover, it is very responsive to P; Whitton (1970) reviewed the literature on *Cladophora* and found no case where N limited the development of the species (review by Broch and Loescher, 1988). *Cladophora* takes advantage of nearshore sources of P in the Great Lakes where ambient SRP away from shore is only about 1 μg/L. In FDR Lake, on the other hand, the source is the river upstream from the reservoir, resulting in the observed longitudinal decrease in SRP, rather than an onshore-tooffshore gradient as in the Great Lakes. However, attached algae should have an advantage over phytoplankton in FDR Lake, if SRP is high enough, because their biomass accumulation is not limited by flushing and they may be limited less by N than phytoplankton. Also, referring back to the chl a prediction from Walker's model (Figure 10), it appears that N and P are about equally limiting at the current P loading and, therefore, reduction in P loading should limit phytoplankton growth and probably periphyton as well.

Research with a smaller filamentous green alga (*Mougeotia*) and a blue-green alga (*Phormidium*) in channels with current velocities from 5-60 cm/s has shown that much lower in-channel concentrations of SRP saturate growth rate and biomass (Horner *et al.*, 1983; Welch *et al.*, 1989; Walton, 1990). A model developed from that work fit the channel data best if the half-saturation constant was set at 5 μg/L SRP (Welch *et al.*, in press). Consequently, the critical level of SRP limiting biomass development (for a 30-day period) was around 10 μg/L with current velocity at 20 cm/s (A, Figure 15). Growth saturation at low SRP concentrations has been shown by other workers as well. Freeman (1985) found that *Cladophora glomerata* growth in the Manawatu River, New Zealand, was saturated at around 5 μg/L SRP. And, of course, Bothwell's (1985, 1989) work has shown that running water diatoms are saturated at SRP concentrations of 1-3 μg/L, although some additional biomass accrual will occur up to 25 μg/L. Thus, there appears to be a major difference in P uptake kinetics of periphytic algae (even *Cladophora* and other filamentous forms) between running and standing waters that probably has important management implications.

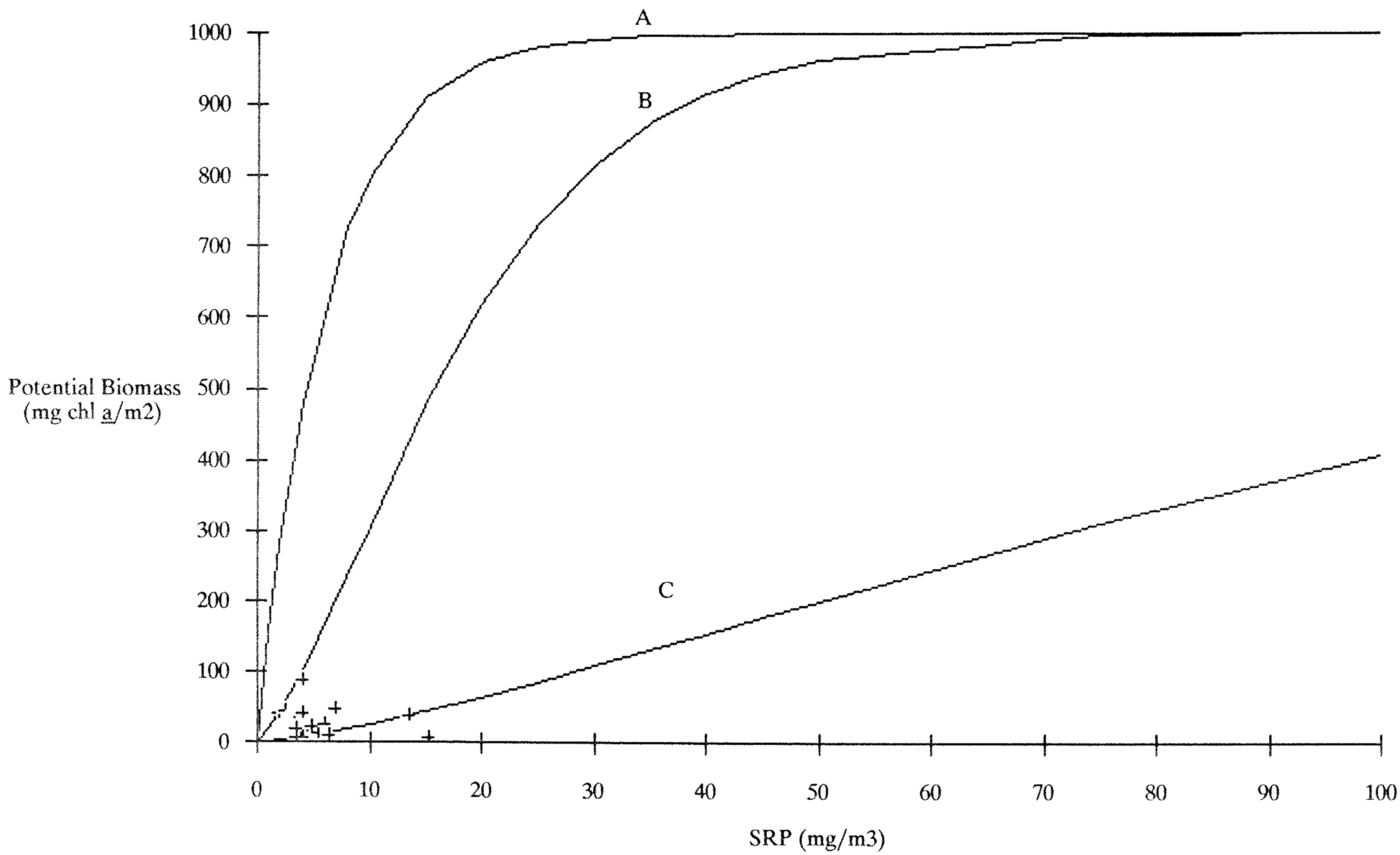

Figure 15. Potential periphyton biomass as a function of SRP, as predicted by Horner-Welch steady-state biomass model for periphyton accrual. Comparison with observed biomass from 1991 longitudinal survey (+). Curve A: $K_s = 5$ $\mu g/L$, $V = 20$ cm/s, $\mu_{max} = 1$/day, t=30 days; Curve B: $K_s = 30$ $\mu g/L$, $V = 1$ cm/s, $\mu_{max} = 1$/day, t=30 days; Curve C: $K_s = 30$ $\mu g/L$, $V = 1$ cm/s, $\mu_{max} = 0.045$/day, t=45 days

Therefore, growth kinetic constants developed from the work with *Cladophora* in the Great Lakes were used in the Horner-Welch model to predict *Cladophora* response to increased (and decreased) P loading in FDR Lake. Both curves B and C in Figure 15 were computed with low velocity (1 cm/s) and the same B_{max} and K_s, but curve C was computed with the low μ_{max} observed for Great Lakes *Cladophora* (0.045/day) and 45 days growth, although there was really little difference in curves for 30 and 45 days. Selection of that μ_{max} accounts for the slow response (C, Figure 15). The responses indicated by both of these curves show a much greater sensitivity of *Cladophora*, over the range of SRP observed longitudinally in FDR Lake, than does phytoplankton to the observed range in P loading (and, hence, pool TP concentration). The values used for the various parameters in the Horner-Welch periphyton model for curves A, B, and C (Figure 15) are summarized in Table 11.

The periphytic chl a levels, observed in the August 1991 survey, are also shown in Figure 15. Obviously, the potential for *Cladophora* biomass is much greater. None of the bottom substrate available for sampling had any *Cladophora* growth, so these observations do not represent in any way a test for this model. Additional littoral sampling should be completed to obtain representative biomass levels of *Cladophora* for model calibration. Considering the success attained with the Canale and Auer (1982) model in the Great Lakes, data collection should probably be oriented to satisfying that model's requirements as well.

Table 11. Summary of parameter values used in Horner-Welch periphyton accrual model (curves A, B, and C of Figure 15).

Symbol	Unit	Curve A	Curve B	Curve C
B_{max}	mg/m^2	1000	1000	1000
k_1	---	1.46+0.276(P)	1.46+0.276(P)	1.46+0.276(P)
k_2	---	0.3	0.3	0.3
θ	---	0.45	0.45	0.45
μ_{max}	1/day	1.0	1.0	0.045
K_s	mgSRP/L	5	30	30
μ	1/day	$\frac{\mu max(P)}{K_s+P}$	$\frac{\mu max(P)}{K_s+P}$	$\frac{\mu max(P)}{K_s+P}$
D	cm^2/s	$1.5x10^{-5}$	$1.5x10^{-5}$	$1.5x10^{-5}$
V	cm/s	20	1	1
k_f	cm/s	$(DV/\pi)^{0.5}$	$(DV/\pi)^{0.5}$	$(DV/\pi)^{0.5}$
k_{fo}	cm/s	$.009(1.018)^{(T-20)}$	$.009(1.018)^{(T-20)}$	$.009(1.018)^{(T-20)}$
T	^oC	22	22	22
L	---	1	1	1
t	days	30	30	45

Critical Limit for P Loading in FDR Lake

The most logical basis for recommending interim P loading criteria for FDR Lake is the result obtained with the Horner-Welch model using *Cladophora* and kinetics constants from the Great Lakes experience. With that approach, loading criteria should be set on SRP as well as TP. The mean inflow SRP concentration during the periphyton growing season should be considerably lower than the inflow concentration observed in 1991 of approximately 15 μg/L (see Figure 6), because that elevated concentration is probably responsible for the current level of *Cladophora* in the lake. According to Ecology (1988), the concentration of TP from Cominco maximum permitted discharge, diluted by low-flow in the Columbia River (25,548 cfs), is 180 μg/L. According to Ministry of Environment (1979), TP = 1.67 SRP in the fertilizer plant's effluent. Therefore, the diluted SRP concentration due to Cominco maximum permitted discharge is approximately 108 μg/L; with the background river SRP concentration being <3-8 μg/L (Ministry of Environment, 1979), the sum is around 114 μg/L. Actual loading during 1980-1989 averaged 4,730 kg/day (42% of their permitted load), which means the actual low-flow concentration entering the reservoir should have been about 50 μg/L. Nevertheless, nearly all of the SRP entering FDR Lake from the Columbia River comes from the fertilizer plant.

With such a high inflow concentration of SRP, it is not surprising that an algal response has occurred. The non-algal turbidity reducing light, especially in the upper reservoir, as well as N limitation and loss through flushing, probably account for the relatively low levels of phytoplankton observed, as suggested by Walker's model (Figure 10). Perhaps, if phytoplankton were less inhibited by light, *Cladophora* would not be a problem, because phytoplankton would have outcompeted them for SRP. Consequently, the existence of a *Cladophora* problem, instead of a phytoplankton problem, may be indirectly due to non-algal turbidity. Detention time should be adequate for phytoplankton biomass development.

An approach to developing P-loading limits would involve the use of model predictions such as Figures 11 and 15. Some reduction in loading may be called for, considering the existence of a *Cladophora* problem, the high inflow SRP concentrations, and the precedents in the Great Lakes and elsewhere for *Cladophora* response to high P. TP and SRP in segment #1, at the 1980 TP loading of about 2 million kg for the averaging period (approximately 340,000 kg/month), are expected to be about 40 and 12 $\mu g/L$, respectively (Figure 11). At that level of SRP, *Cladophora* biomass could potentially reach between 50 and 400 mg chl $\underline{a}/m^2$, depending on what maximum uptake rate was chosen (Figure 15). A nuisance biomass level is considered to fall somewhere between 100 and 200 mg chl $\underline{a}/m^2$ (Welch *et al.*, 1988). If the TP load were reduced by one half, the expected TP and SRP in segment #1 would be about 20 and 8 $\mu g/L$, respectively, which would lower expected *Cladophora* biomass to below 200 mg chl $\underline{a}/m^2$ using the highest uptake rate (Figure 15). Before an absolute recommendation for P-load reduction can be made, however, actual *Cladophora* biomass levels should be measured in the lake and a predictive model calibrated for conditions that exist there. Furthermore, the actual current contribution by Cominco should be included in the modeling process as a source separate to the Columbia River. That was not done here due to a limitation of in-reservoir data for calibration. Also, 1980 appears to be a low P-discharge year for Cominco.

SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

1. Phytoplankton growth and resulting biomass in FDR Lake are apparently limited by N, light, and to some extent, flushing, as indicated by an empirical model developed for reservoirs. This contention is supported by the lower-than-predicted chl a concentration observed in 1980. Thus, current levels of P loading do not produce as much biomass as they might in other less turbid and flushed lakes. In addition to the model results, evidence for N limitation is that nitrate is undetectable in the lake while SRP is low but still detectable. There is the possibility of N-fixing blue-green algae, which were not considered in the model, responding to further increase in P loading. Nevertheless, blue-green algae are not now important and reduction in P loading would not be expected to result in water quality benefits in so far as phytoplankton are concerned.

2. *Cladophora*, a notorious nuisance periphytic green alga, does occur in nuisance proportions in FDR Lake. Nuisance levels were observed, but could not be measured on originating substrate, possibly because the littoral existing in August, when surveyed, had not been inundated for a sufficient time.

3. A periphyton biomass model calibrated for running water, best used with kinetic growth constants peculiar to shoreline growth of *Cladophora* in the Great Lakes, showed that reductions in P loading should reduce *Cladophora* biomass in relation to SRP concentration decrease, especially in the upper reservoir. An approach for recommending critical P loading from the Columbia River indicates that with a one-half reduction in load, *Cladophora* biomass should be reduced below a perceived nuisance-causing level

(determined for running water). However, a recommendation is not possible at this time without extensive measurements of biomass in the reservoir and verification with a predictive biomass model. Up-river and Cominco sources of P to the reservoir should be separated in the modeling process to more clearly observe the response to load reduction.

4. A thorough monitoring of *Cladophora* biomass at about 10 stations throughout the length of the reservoir is recommended. Sampling should be frequent enough during the growing season (April-October) for the sampled littoral to have experienced a sufficient period of inundation for colonization to occur. There is reason to believe that littoral biomass is greatest in the fall due to longer inundation. Ambient and cellular P concentrations should be determined concurrently. Light extinction coefficients should be determined to evaluate the depth-available light relationship in the littoral for modeling purposes (optimum light occurs at about 1 m in the Great Lakes work).

5. Data should be collected that is appropriate to calibrate the Canale-Auer model, which was used successfully in the Great Lakes, to establish P loading criteria for the control of *Cladophora* growth in FDR Lake. Alternatively, the Horner-Welch model could be used, as illustrated here, but not without confirming the beneficial effect of high SRP (up to 50 μg/L) concentrations on *Cladophora* P uptake and growth at low current velocities, which appears to be the case from the Great Lakes work. The channel experiments as proposed should be conducted with *Cladophora* for that confirmation. Establishment of *Cladophora* in experimental channels will be attempted during January-February, 1992, as part of this study. Such confirmation would be beneficial

even if the Auer-Canale model were used. Current velocity was not specified in the Auer-Canale experimental and/or modeling work and some evidence for greatly reduced uptake rates at low current velocity would be valuable to the modeling exercise. Actual measurement of current velocity over the littoral substrate in the reservoir would be helpful in linking the modeling effort with experimental results.

6. More complete seasonal and longitudinal data on nutrients and phytoplankton are needed to verify Walker's model. Although these seem less critical than periphyton data, there is really only one years data available for the reservoir proper and that was during a year of low P discharge from Cominco (1980). Physical conditions (turbulence and short detention time) in FDR Lake are apparently inhospitable to blue-green algae; *Anabaena* occurred but in low abundance in 1980. Blue-green blooms and chl a above 20 μg/L occurred in Long Lake (lower Spokane River) in response to increased P loading and its average detention time was shorter than in FDR Lake.

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