J.C. Headwaters, Inc.

TMDL Modeling and Analysis of Diamond Lake, Oregon

Prepared for the
Oregon Department of Environmental Quality
Eugene, OR

By

Joseph M. Eilers[1]
Benn J. Eilers[1]
and
Jake Kann[2]

December 2003

[1]Current Affiliation: MaxDepth Aquatics, Inc. 1972 NE 3rd St. #10, Bend, OR 97701
[2]Aquatic Ecosystem Sciences, 295 East Main St., Ashland, OR 97520

TABLE OF CONTENTS

ABSTRACT…….. .2
INTRODUCTION…….….3
METHODS……..…7
1. Access Database……..7
2. Hydrologic Budget…….…7
3. Hydrodynamic Model……..…9
4. Assessment of Biological Communities……....10
5. Paleolimnology……..12
6. Biological Model……...13
RESULTS……..18
1. Access Database……18
2. Hydrologic Budget and Nutrient Fluxes……18
3. Hydrodynamic Model (CE-QUAL-W2)……27
4. Biological Assessment of Diamond Lake……..28
INTERNAL LOADING AND THE NEED FOR A BIOLOGICAL MODEL….52
DEVELOPMENT AND APPLICATION OF THE FIN-S MODEL……57
TOTAL MAXIMUM DAILY LOAD…….…….……67
1. Fish-Based TMDL……67
2. Uncertainty Analysis……68
CONCLUSIONS……69
REFERENCES…… ..71
ACKNOWLEDGEMENTS……. .75
APPENDICES…… …75

ABSTRACT

Diamond Lake, Oregon is a 1,226 ha lake in the central Cascades located at an elevation of 1580 m. The lake does not meet water quality standards for pH and algae. Since 2001 the lake has been experiencing large blooms of the cyanobacteria, *Anabaena flos-aquae*. In addition, the cyanobacteria have been producing a toxin, *anatoxin-a*, at levels considered potentially injurious to humans. In 2002, the ODEQ issued a contract to JC Headwaters, Inc. (JCH) to analyze the existing conditions and develop a TMDL (**T**otal **M**aximum **D**aily **L**oad) for Diamond Lake. The hydrologic budget and hydrodynamic modeling using CE-QUAL-W2 showed that inputs of nitrogen (N) and phosphorus (P) to Diamond Lake are largely natural. This is in agreement with a previous study of Diamond Lake from 1972-1977 by EPA. However, the changes to the fishery, particularly the recent expansion of tui chub population, have had profound changes on the biology of the lake that have had major consequences on nutrient cycling within the lake.

A STELLA® model of linked differential equations was developed for Diamond Lake to represent the major biological components of Diamond Lake, their interactions, and the nutrient fluxes associated with the biological responses to the fisheries. The model results indicate that the current conditions in the lake can be explained by the changes in the fisheries. The tui chub have eliminated most of the larger, herbivorous zooplankton, resulting in a zooplankton population that is comprised of over 90 percent rotifers. These organisms have a short life-span and have poor grazing efficiency on the phytoplankton. The reduction in grazing pressure reduces the constraints on the phytoplankton. The proliferation of the chub has also eliminated most of the larger zoobenthos such as the amphipods, resulting in a population that is comprised largely of dipterans (Chironomidae) and tubifex worms (Tubificidae). Again, the smaller zoobenthos have shorter life-spans than the pre-fish community and would be expected to increase the rate of cycling nutrients from the sediments.

The modeling scenarios indicate that it is necessary to remove the vast majority of tui chub from Diamond Lake to meet water quality standards. Partial removal of the chub results in compensatory growth of the remaining individuals and the high fecundity of the species results in rapid recovery to levels of maximum biomass supported by the lake. The evidence also indicates that the lake will not meet water quality standards at trout stocking densities employed since 1962. The model was used to estimate trout stocking densities that yielded satisfactory water quality for the majority of climatic conditions. However even under a no-fish scenario, it is likely that water quality standards will be exceeded during climatic conditions consisting of above-average temperature and below-average wind velocities that persist for extended periods. A TMDL was created using incremental rates of fish stocking, assuming that the tui chub are removed.

INTRODUCTION

Diamond Lake is currently listed on the Oregon 303d List for non-conformance with water quality standards. The lake routinely exceeds pH values of 8.5 during the summer and experiences nuisance growths of cyanobacteria, primarily *Anabaena flos-aquae* (Eilers 2003b). This has led to closure of the lake to recreational contact during parts of the summers of 2001-2003. As part of the water quality program for the state, water bodies not meeting existing water quality standards are required to undergo an analysis of Total Maximum Daily Loads (TMDL) to assess the nature of the water quality problem, determine the sources, and allocate loads to meet water quality standards. Most of the water quality problems in the state are the product of activities in the watershed that promote the export of increased loads of nutrients (usually phosphorus and nitrogen) or other pollutants, such as biochemical oxygen demand (BOD), bacteria, or toxic substances. These types of problems often lend themselves to an engineering approach that can be simulated with mathematical models of watershed processes. In these cases, a watershed-loading model such as SWAT (Arnold et al. 1995) can be employed to simulate pre-development conditions, current conditions, and a future scenario that presumably provides improved water quality to the receiving waters. Such was the case for Tenmile Lake on the Oregon coast where Eilers et al. (2002) used SWAT to assist in identifying the sources of nutrients and sediment to the lake and simulate a possible solution.

However, in Diamond Lake there are two attributes of the lake and its watershed that preclude this type of approach. First, the watershed is underlain by fractured basalt and andesite (Sherrod 1991), which leads to rapid infiltration rates and a poor correspondence between the topographic watershed and the actual contributing area. Without knowing what portions of the watershed contribute hydrologically to Diamond Lake it is not possible to develop functions to compute watershed loads to the lake with a high degree of confidence. Second, the long hydraulic residence time of the lake provides considerable opportunity for the abundant biological communities in the lake to alter water quality. Most current hydrodynamic models do not allow for extensive differentiation of the various organisms or linkages among higher organisms, although a number of efforts are being made to address this issue (Schindler and Eby 1997; Hakanson et al. 2003; Portielje and Rijsdijk 2003; Simon and Townsend 2003)

Most TMDL analyses for eutrophication-related water quality problems in the United States have focused on total phosphorus (TP) and approaches for limiting TP export to the receiving water. This reflects the fact that the majority of lakes in the United States are P-limited. However, in the Cascades, the groundwater and surface waters come into extensive contact with the bedrock, which is comprised largely of basalt, andesite, and andesitic basalt. These rocks have an abundance of phosphorus, and the weathering reactions yield waters that are naturally high in phosphorus. In contrast to many waters in the United States where P-limitations permit the use of P-based models, many Cascades lakes require an analysis of both P and N to understand the eutrophication process.

Because of these factors, we elected to develop an approach for Diamond Lake that used the hydrology and hydrodynamics to assist in characterizing nutrient fluxes to and from the lake, while using a parallel path to model the lake biota and nutrient fluxes within the lake. In addition to using the recent water quality data to evaluate temporal changes and key processes, we also made extensive use of paleolimnological results to better understand the chemical and biological changes over a longer time span. The conceptual approach is described in Figure 1.

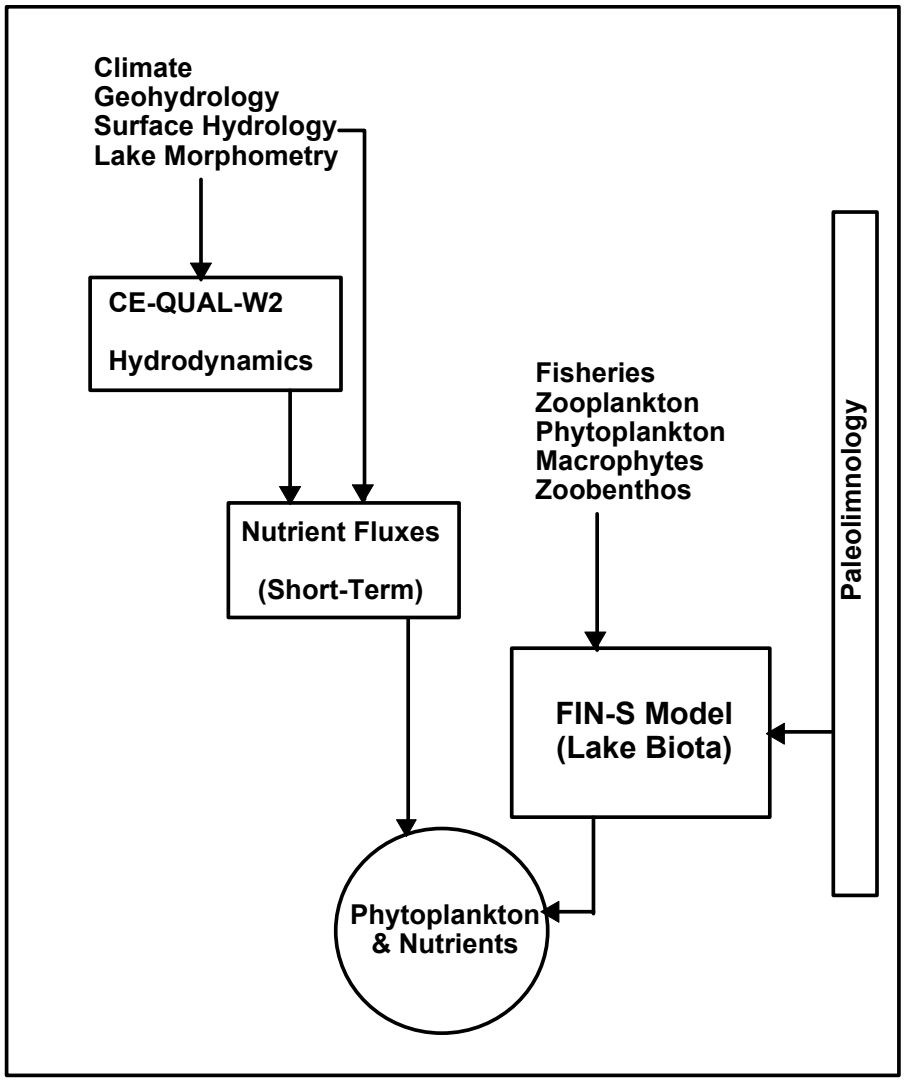


Figure 1. Conceptual structure for the approach used to develop the TMDL modeling of Diamond Lake.

METHODS

The approach to developing the TMDL for Diamond Lake consisted of:

- Organizing the available data into an Access database,
- Developing an hydrologic budget using the most recent data,
- Using the recent bathymetric data and hydrologic budget to calibrate a hydrodynamic model of flows and temperature regimes,
- Evaluating the recent changes in biological communities,
- Evaluating the long term changes in biota using paleolimnology, and
- Developing a STELLA-based model of biota and nutrient cycling.

The specific methodologies associated with each of these activities are described below.

1. Access Database:

The data available for assessing conditions in Diamond Lake includes several major data sets and a number of miscellaneous activities that needed to be organized in a systematic fashion. We chose to use Microsoft Access® to serve as the structure for this task because this software is widely available. The data were organized into logical units based on investigator and core elements within each investigation, such as water chemistry, phytoplankton, and biology (Appendix A). The database is structured to receive input in future years.

2. Hydrologic Budget:

The hydrologic budget was derived from direct measurements of stream flow for the major two inlets, Silent Creek, Short Creek, the outlet, Lake Creek, measurements of precipitation, climate data, and recent groundwater investigations. The data for Silent Creek consist of continuous measurements of flow for June-October, 2003, whereas flow measurements on Short Creek were measured only occasionally. Precipitation has been measure since 1981 at the SNOTEL site at Diamond Lake and an additional weather station that was installed at the lake in May 2003. Evaporation was calculated using a modified Penman equation using the data collected at the weather station and relative humidity collected at nearby Cinnamon Butte. Groundwater inflow was calculated using Darcy's Law for parallel flow 'tubes' configured around the lake in conjunction with the network of groundwater observation wells installed in July and August 2003. The information available for computing the components of the hydrologic budget is summarized in Table 1.

Table 1. Sources of data available for calculating a hydrologic budget for Diamond Lake.

Component	Source	Period of Data	Comments
Precipitation	Crater Lake National Park	1991-present	Two sites; rim and lake level
	NRCS	1981 - present	SNOTEL site #442; Station ID = 22f18s
	RAWS		Cinnamon Butte
	Diamond Lake	May 15, 2003 - present	Tipping bucket gage
Evaporation	Odell Lake, NCDC historical listing	1948-1980	Land pan evaporation measurements
	This Study	May 15, 2003 - present	Energy budget calculations from Diamond Lake weather Site
Surface Inflows	Lauer et al. (1979)	1972-1977	Collected by USGS
	Miscellaneous (See Eilers 2003b)	Scattered	
	This Study	June 2003-present	Continuous stage measurements from pressure transducer on Silent Creek; periodic measurements on Short Creek
Surface Outflow	USGS (#14312500)	1930-1953; 1971-1977; 1978-1984; 1999- present	Lake Creek; current gage is 600 ft downstream of Diamond Lake outlet
	This Study	June 2003-present	Continuous stage measurements from pressure transducer on Lake Creek at outlet
Groundwater Inflow	This Study (Breeden 2003)	August 2003-November 2003	16 observation wells installed in upper aquifer by USFS
Groundwater Outflow	This Study (Breeden 2003)	August 2003-November 2003	16 observation wells installed in upper aquifer by USFS

Annual precipitation at Crater Lake is greater than observed at Diamond Lake because of the higher elevation of the crater rim, but the data are still very useful for evaluating weather conditions at Diamond Lake in the 1990s. The Diamond Lake SNOTEL site is located near the sewage treatment lagoons to the northeast of the lake. There are two RAWS sites in the area; Cinnamon Butte and Tokatee. We have relied on data from the

Cinnamon Butte site for supplemental climate data for Diamond Lake because of its closer proximity to Diamond Lake and its similar elevation to the lake. The weather station at Diamond Lake was installed on May 15, 2003 on the northeast shore of the lake on the jetty extending into the lake at the north boat ramp. In addition to measuring rainfall (the precipitation bucket is not heated), the Diamond Lake weather station also measures air temperature, wind speed, wind direction, barometric pressure, and solar radiation (PAR), all at 15-minute intervals.

Historical land pan evaporation data are available for Odell Lake, located to the north of Diamond Lake at a similar elevation (4787 ft versus 5193 ft). However, we were able to supplement these observations with evaporation calculated using the data from the on-site weather station and relative humidity from the Cinnamon Butte RAWS site.

Surface discharge from Silent and Short Creeks was measured by USGS during the Lauer et al. (1979) study and only sporadic measurements of flow were made between 1977 and 2002. In June, 2003, we installed a recording stage recorder on Silent Creek to obtain continuous measurements of discharge. A staff gage was installed at Short Creek in 2002 to obtain periodic measurements of flow during the summers of 2002 and 2003. Discharge from the surface outlet of Diamond Lake has been conducted for several periods dating back to 1930. Most recently, the station has been operated continuously since 1999. We installed a back-up stage recorder at the outlet starting in June 2003.

Groundwater observations were collected from several wells by Lauer et al. (1979), but to our knowledge, these wells were not used in a flow-net analysis to compute groundwater flow paths or magnitude of flow. The only data for determining groundwater flow has been derived from the set of wells installed by the USFS in July/August 2003. Monthly groundwater level data are available for a limited portion of the annual hydrologic cycle. As of this date, no evidence of shallow groundwater discharge from Diamond Lake has been identified. However, there are plans to install additional observation wells on the north end of Diamond Lake in spring 2004 in a more concerted effort to better understand this component (R. Breeden, pers. comm. 2003).

3. Hydrodynamic Model:

The CE-QUAL-W2 model (Version 3.1, Cole and Wells [2001]) was selected to simulate daily fluctuations in flows and water temperature during the period from April 22 through September 22, 2001. The lake was partitioned into a series of 21 longitudinal segments 250 m in width and vertical segmentation of 1 m depths using the bathymetry developed by Eilers and Gubala (2003) as shown in Figure 2. The temperature data were calibrated to profiles collected on eight days in 2001 resulting in a model that simulated observed temperature values generally within 1°C (Appendix B). The model was run on a daily time-step. The key coefficients that were adjusted during the calibration process are presented in Table 2.

Table 2. Key coefficients that were adjusted during the hydrodynamic model calibration process.

Coefficient	Units	Value
Horizontal Eddy Viscosity	m^2/s	1.2
Horizontal Eddy Diffusivity	m^2/s	1.2
Sediment Heat Ex. Coefficient	$W/m^2/s$	2.00E-08
Sediment Temperature	C	5
Interfacial Friction	-	0.001
Fraction Solar at Surface To Water	0 - 1	1
Max. Vertical Eddy Viscosity	m^2/s	1.2
Wind Sheltering	0 - 1	0.15 - 0.2
Water Extinction Coefficient	0 - 1	0.27
Fraction of Solar Radiation Absorbed @ Water Surface	0 - 1	0.18 - 0.60

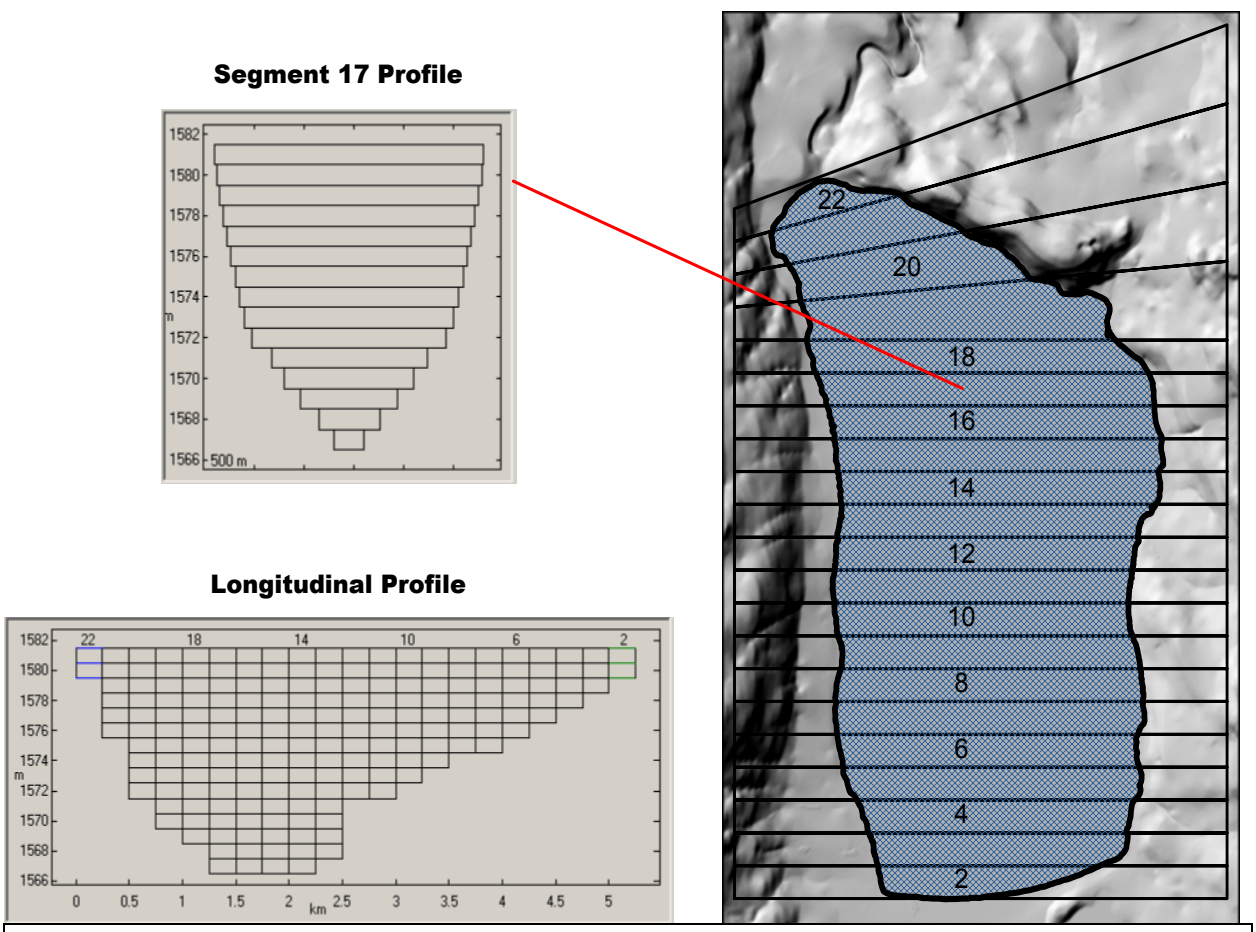


Figure 2. Segmentation of Diamond Lake bathymetry for running the hydrodynamic model, CE-QUAL-W2. The lake was partitioned into 21 longitudinal segments (250 m each) and up to15 vertical segments (1 m each), as shown for segment 17.

4. Assessment of Biological Communities:

The key biological communities selected for analysis, interpretation, and eventual representation in the biological model include fisheries, zooplankton, phytoplankton, zoobenthos, and macrophytes. The assessments of current status of these communities and evidence for how they may have changed were derived largely from data collected in several major studies and a number of miscellaneous activities as summarized in Table 3.

Table 3. Nature and sources of information for biological characterization in Diamond Lake.

Component	Source	Period of Data	Comments
Fisheries			
Creel Surveys	OSGC (1949-1961)	1946-1961 → up to 1975; 1993-2002	Trout caught
Netting	ODFW	1992-2003	CPUE
Hydroacoustics	Eilers and Gubala (2003); Eilers and Eilers (2003)	August 2002 and June 2003	Split-beam hydroacoustics; day and night surveys
Zooplankton			
Netting	A. Vogel (1995-2003)	1994-2003	Contained in Salinas reports
Hydroacoustics	Eilers and Gubala (2003); Eilers and Eilers (2003)	August 2002 and June 2003	Split-beam hydroacoustics; day and night surveys
Paleolimnology	Eilers et al. (2001b)	~1860 to 2001	Dated sediment core and reconstruction of cladocerans
Phytoplankton			
Lake Surveys	J. Sweet	1988; 1992-2003	Contained in Salinas reports; Community composition and chlorophyll
Paleolimnology	Eilers et al. (2001a,b)	~1860-2001	Dated sediment core; diatoms and cyanobacterial akinetes
Zoobenthos			
Peterson dredge	OSGC (1949-1961)	1946-1960	Community composition and density
PONAR	Lauer et al. (1979)	1972-1977	Community composition and density
PONAR	Eilers (2003a)	2002	Community composition and density
Paleolimnology	Eilers (2003a)	~1860-2003	Chironomid composition and density

Macrophytes			
Visual	OSGC (1949-1961); Locke (1947)	1946-1960	Anecdotal
Divers	Lauer et al. (1979)	1972-1977	Some community composition
Hydroacoustic	Eilers and Gubala (2003)	August 2002	Macrophyte extent, density and canopy height

The key questions used to focus the biological assessments were:

- What is the current status of the biological community?
- What was the status prior to the introduction of fish?
- What are the limiting factors influencing the biological communities?
- How are rates of nutrient cycling affected by specific groups of organisms?

5. Paleolimnology:

Long-term water quality monitoring programs on lakes in the United States are rare. Consequently, much of the information on past conditions in lakes must be derived from other methods. One technique is to model the changes in water quality, but to forecast future changes it is necessary to have historical information for calibration of the model. Fortunately, we have reconstructions of past conditions in Diamond Lake from several paleolimnological investigations (Meyerhoff 1977, Eilers et al. 2001a,b, Eilers 2003a). The nature of these reconstructions is summarized in Table 4.

Table 4. Nature and sources for paleolimnological data on Diamond Lake.

Component	Source	Period Represented	Comments
Diatoms			
	Meyerhoff (1977)	~1860 - 1973	Undated core
	Eilers et al. (2001a)	~1860 - 1996	Dated core
Sediment Chemistry			
	Meyerhoff (1977)	~1860 - 1973	Undated core
	Eilers et al. (2001a)	~1860 - 1996	Dated core
Cyanobacterial Akinetes			
	Eilers et al. (2001b)	~1860 - 2001	Dated core
Zooplankton			
Cladocerans	Eilers et al. (2001b)	~1860 - 2001	Dated core
Chironomids			
	Eilers (2003a)	~1860 - 2001	Dated core

6. Biological Model:

A number of investigators are now recognizing the need for incorporating biological processes in mathematical models for predicting water quality in lakes. Engineering approaches have a long history of incorporating biological processes in water quality modeling (cf. Phelps 1944), but in general only the net reactions (such as those involving BOD) were represented. It is now clear that accurate forecasting of conditions in many lakes requires knowledge of interactions at the species level and that water quality problems can be the product of species interactions rather than just the response to poor water quality. To this end, a number of biologically-based models are being developed (Schindler and Eby (1997), Portielje and Rijskijk (2003), Marion and Paillisson (2003), and Hakanson et al. (2003)). The Lake-Web model developed by Hakanson et al. (2003) is noteworthy because of its inclusion of nine functional groups of organisms, operating on a weekly time-step. However, the current Lake-Web model has only been tested for phosphorus and we believe that nitrogen interactions are also very important in Diamond Lake (cf. Eilers 2003b).

To meet the needs of assessing conditions in Diamond Lake, we developed a custom application using the STELLA® framework (Ver. 8.0; High Performance Systems [2003]). The model represents nine groups of organisms (trout, chub, rotifers, cladocerans + copepods, chironomids, amphipods, macrophytes, cyanobacteria, diatoms) and simulates both nitrogen and phosphorus transfers among the groups. The model in its present form is a series of linked differential equations to solve for the nutrient fluxes. The chub was represented by a population with a maximum life-span of seven years that was initialized by a "seed" population. The rate of chub increase was affected by the percentage of egg survival and food limitation factors. The food base was represented by phytoplankton (cyanobacteria and diatoms), zooplankton (rotifers and cladocerans/copepods), and benthos (amphipods and chironomids). The trout population was represented by stocked trout that also was connected to benthos and zooplankton for food sources. Trout abundance (not shown) was determined by stocking rates, fishing pressure, and out-migration. The nutrients (TN & TP) were connected to the fish and food cycles to compute inputs or losses from the water. A sediment compartment is not included in this version of the model. The complete model representation, linkages, and code are shown in Appendix C and are described below. The model was titled the **F**ish & **I**nternal **N**utrients –**S**TELLA (**FIN-S**).

a. Tui Chub

The exotic tui chub population is the focus of this model. The chub population was partitioned into bins representing the chub at each year of their life. The most recent population estimates by ODFW suggested ~23 million chub over age 2 in the lake (Jackson et al. 2003). By examining the population distribution, it was evident that the chub in age group 2 were greatly under-represented (Table 5).

Table 5. Age class distribution of tui chub used on the biological model for current conditions.

Chub Age Class	ODFW Pop Est	% 2+ Pop	Modified Pop Est	% 2+ Mod
0	210,000,000	887.03	117,789,000	497.00
1	53,000,000	223.87	29,790,900	125.70
2	2,000,000	8.45	13,563,652	57.23
3	17,500,000	73.92	9,813,696	41.41
4	4,000,000	16.90	224,676	0.95
5	150,000	0.63	84,182	0.36
6	21,000	0.09	11,778	0.05
7	3,500	0.01	1,967	0.01
Total 2+	23,674,500		23,699,951	

A new population distribution was generated with more chub in age class 2 than in 3, which is how a normal population distribution would be expected to appear. Mortality rates of the chub were developed from Bird (1975) and modified to reflect a population, which is currently limited by food. The chub death rate for chub age class 0 was set to zero, and all mortality from egg until the fry have reached 1 year old has been incorporated into the Chub Egg Survival variable. The fecundity of the chub was based on egg counts performed by ODFW during 2003, and eggs per female numbers were classified by age, starting at chub 3 years old (Table 6).

Table 6. Eggs per female tui chub by egg class used in the biological model.

Chub Age Class	3	4	5	6	7
Eggs per Female	911	14,207	26,294	37,172	44,424

The ratio of males to females was 57 to 43 (Jackson et al. 2003). The eggs per female for the age 3 class is half of what was measured by ODFW, this is because only half of the age 3 chub were considered to be of reproductive age. The mass of the chub population was computed from averaged data collected by ODFW (Table 7).

Table 7. Weight of tui chub per age class used in the biological model.

Chub Age Class	2	3	4	5	6	7
Mass (g)	7.9	16.4	29.6	45	52	88

The mass of all chub less than 2 years old was not included in this total due to limited mass estimates for the young fish. Preliminary estimates suggested that the chub younger than 2 comprised 5% to15% of the total population biomass.

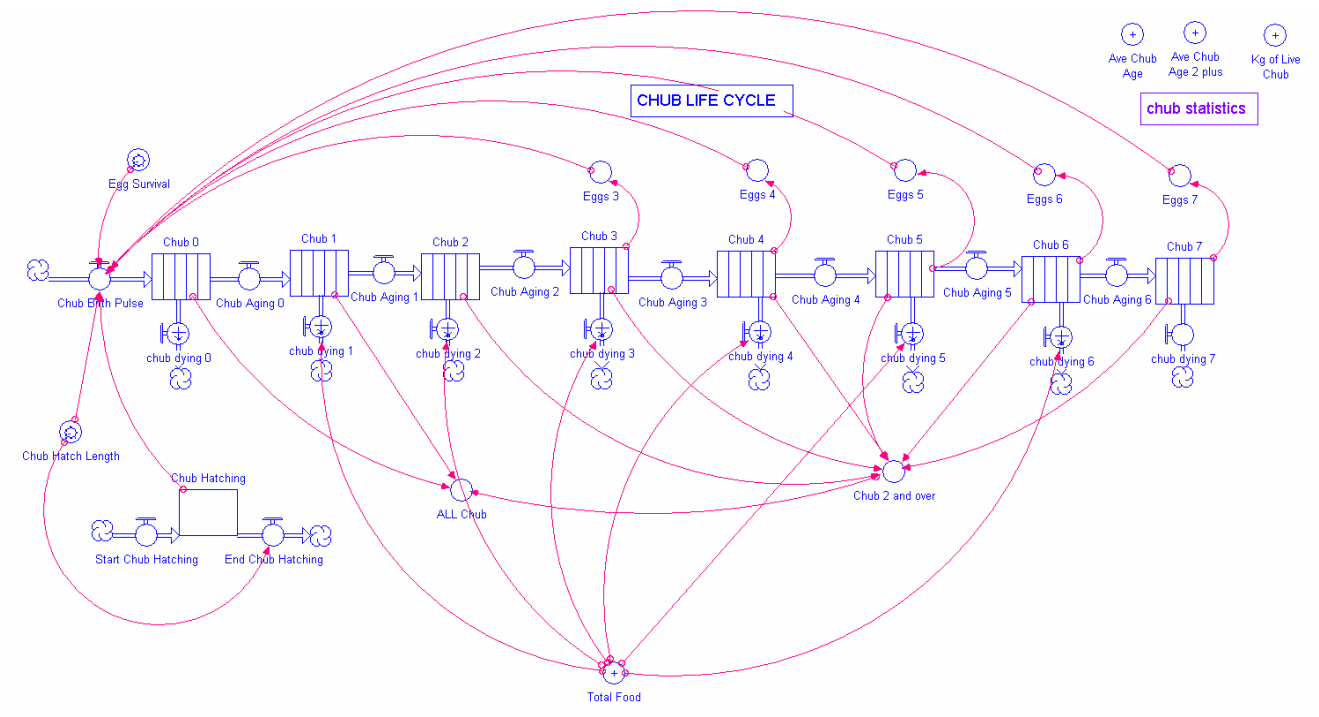


Figure 3. Representation of the tui chub population in FIN-S where valves represent flows, rectangles represent reservoirs (population size), and lines represent connections controlled by equations detailed in the code.

b. Trout

The trout population cycle is relatively simple when compared with the tui chub cycle. The cycle consists of one input (stocking) and three outputs (harvest, death, and migration). The trout fishery in Diamond Lake was modeled as a put and take fisheries where almost all of the stocked fished were caught and kept by anglers. There is a hold-over population equal to approximately 10% of the stocked population, or 5,000 hold-over fish with the modeled 50,000 stocked fish. The effect of varied trout stockings is examined in the sensitivity section.

c. Benthos and Phytoplankton

Each population reservoir in this section has one input (growth), and two outputs (death and consumption). The populations are represented in terms of biomass in the lake, with initial concentrations calculated from averaged measurements throughout 2003 and corrected for a mid-winter model start. Each population has different governing equations based on its location in the food chain. Examination of the dependency connections provide an idea of the general influences on the flows and populations. Further detail is available through the model or the equations in Appendix C.

All populations except for diatoms are sensitive to the seasons and subject to die-off in winter. The DT counter, summer indicator, and winter indicator specify the driving seasonal changes. The winter indicator influences the natural mortality rate of the various biological communities, whereas the summer indicator influences most growth processes. The benthos is represented as two groups, amphipods and chironomids. Among the benthos, amphipods are considered the preferred food source of both the chub and the trout, but once amphipod populations have been sufficiently depleted the chironomids act as a significant source of food.

The next group represented is the zooplankton, consisting of rotifers and non-rotifers. . The biomass of rotifers was based on the biomass of non-rotifers, so as the non-rotifer population decreases, the biomass of rotifers increases. Non-rotifers are a significant part of both the chub and trout food intake and are highly influenced by predation.

Phytoplankton was divided into two groups, cyanobacteria and diatoms. All other phytoplankton distinctions were disregarded in this model. The cyanobacteria were modeled as phosphorus-dependant. The grazing pressure on cyanobacteria was assumed to be insignificant until the non-rotifer population exceeded 1500 kg. To limit false spikes of the cyanobacteria population, a bloom time variable was defined whereby the cyanobacteria was only allowed to grow in the later part of the summer. This is the typical condition in Diamond Lake, although there have been exceptions to this pattern (Sanville and Powers 1973). The diatoms were assumed to be N-limited in Diamond Lake and the primary loss term for diatoms was grazing from non-rotifer zooplankton. These linkages are summarized in Figure 4.

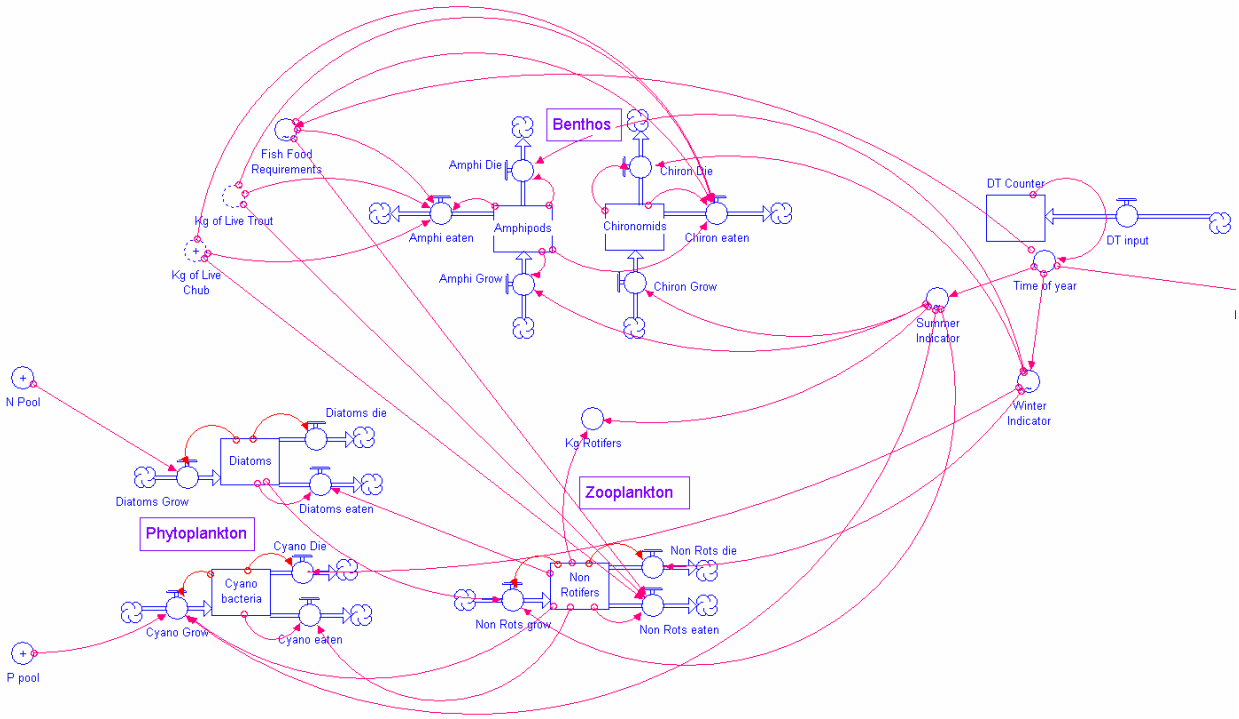


Figure 4. Representation of the "food" components of the FIN-S model.

d. Nutrients

Nitrogen and phosphorus were modeled as distinct reservoirs in the lake water; no sediment pool was depicted in this version of the model. Silica was assumed to be in sufficient supply to not limit the diatoms. The operation of the N and P cycles differ in that there is a net input of phosphorus into the lake through the tributaries, whereas there is a net outflow of nitrogen through Lake Creek. Initial concentrations and fluxes were calculated from *in situ* measurements in 2003 (Table 8).

Table 8. Concentrations and fluxes of TP and TN for Diamond Lake used to guide the nutrient fluxes in the biological model.

	Total Phosphorus			Total Nitrogen		
Conc.	mean	min	max	mean	min	max
mg/l	0.030	0.021	0.040	0.69	0.40	0.90
kg/lk	2,478	1,764	3,360	58,254	33,600	75,600
Flux	net flux in from tributaries			net flux out through Lake Creek		
kg/lk*yr	1922			23645		

The input and output fluxes for both nutrients were based on population fluctuations and wet weight concentrations of the nutrients in all of the other biological populations in the lake. Turnover rates were calculated using the same nutrient densities along with average life spans of each of the other populations (Figure 5).

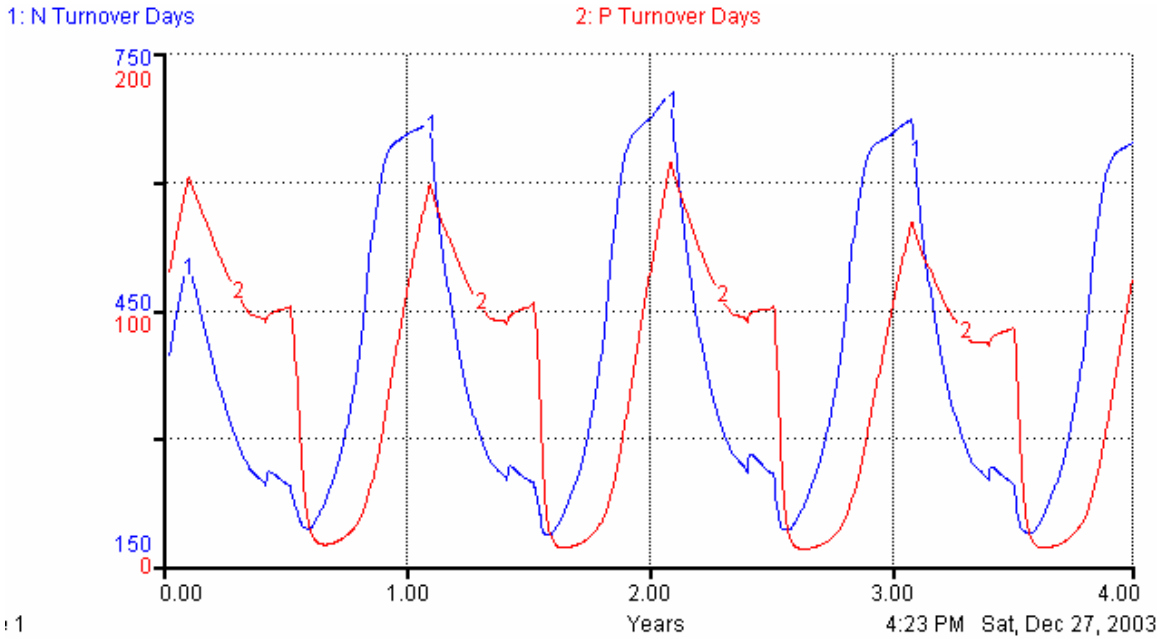


Figure 5. Cycling rates of nitrogen (blue) and phosphorus (red) for Diamond Lake base on modeled interactions and measured inputs.

The nutrients are being cycled through the system the quickest during the mid to late summer, which is what would be expected. The cycle rates should not be considered absolute values, only indicators of changes in the nutrient cycle time. Both the phosphorus and nitrogen inputs and outputs are connected to dials that can be accessed through the interface level of the FIN-S model.

RESULTS

1. Access Database:

The Access database was updated to include the data collected by DEQ in 2002 and 2003, water chemistry and plankton data collected in 2002 and 2003, benthic macroinvertebrate data from 2002, hydrology data from 2002 and 2003, and climate data from 2003 (Eilers 2003b). The database is provided as Appendix A as a CD attachment to this report.

2. Hydrologic Budget and Nutrient Fluxes:

a. Hydrology

The application of the most recent data to develop a water budget for Diamond Lake yields the following flows (Table 9). The inflows and outflows for Diamond Lake are shown as measured discharge rates and as a percentage of annual flows (Figures 6 and 7). On average, the majority of inflow is derived from Silent Creek and precipitation (on the lake surface), with substantially lower amounts from the remaining sources. Groundwater inputs were measured from August to November, 2003 at which point most of the wells indicated flow stagnation or slight flow reversals. It is believed that the groundwater flows will decline slightly through the winter, depending on the temperature regime during the winter. In the spring, we expect that groundwater discharge to the lake will increase substantially as the snowmelt is transformed into groundwater recharge. The only flow calculated as a residual (difference of measured flows) is the term for groundwater outflow. The hydrologic residence time for Diamond Lake based on these flows and the lake volume from the 2002 bathymetric study (Eilers and Gubala 2003) of 84 cubic hectometers is 1.64 years. This agrees closely with the estimates generated by Johnson et al. (1985) of 1.6 years and that of Lauer et al. (1979) of 1.5 years, even though the allocation of source flows varies among the investigations.

Table 9. Measured and estimated flows into and leaving Diamond Lake (X 10^6 m^3)

Component	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Total
Inflow													
Silent Creek	1.82	1.76	1.82	1.82	1.64	1.82	1.76	1.82	1.76	1.82	1.82	1.76	
Short Creek	0.53	0.51	0.53	0.53	0.48	0.53	0.51	0.53	0.51	0.53	0.53	0.51	
Misc. Tributaries	0	0	0.01	0.01	0.01	0.03	0.5	0.7	0.3	0.1	0	0	
Precipitation	1.02	1.64	2.16	1.17	1.78	2.03	3.52	0.43	0.72	0.12	0.29	1.37	
Groundwater	0.42	0.07	0	0	0	0	0.44	1.06	0.95	0.91	0.83	0.7	
	3.79	3.99	4.52	3.53	3.91	4.42	6.73	4.54	4.25	3.40	3.48	4.34	50.9
Outflow													
Lake Creek	3.43	6.04	7.29	5.99	4.42	4.04	2.64	2.00	2.02	1.62	0.57	1.20	
Evaporation	0.71	0.50	0.20	0	0	0	0.30	0.89	1.10	1.34	1.04	0.83	
Groundwater	0.10	0.20	0.33	0.66	.68	0.33	0.20	0.10	0.05	0.05	0.02	0.01	
													50.9

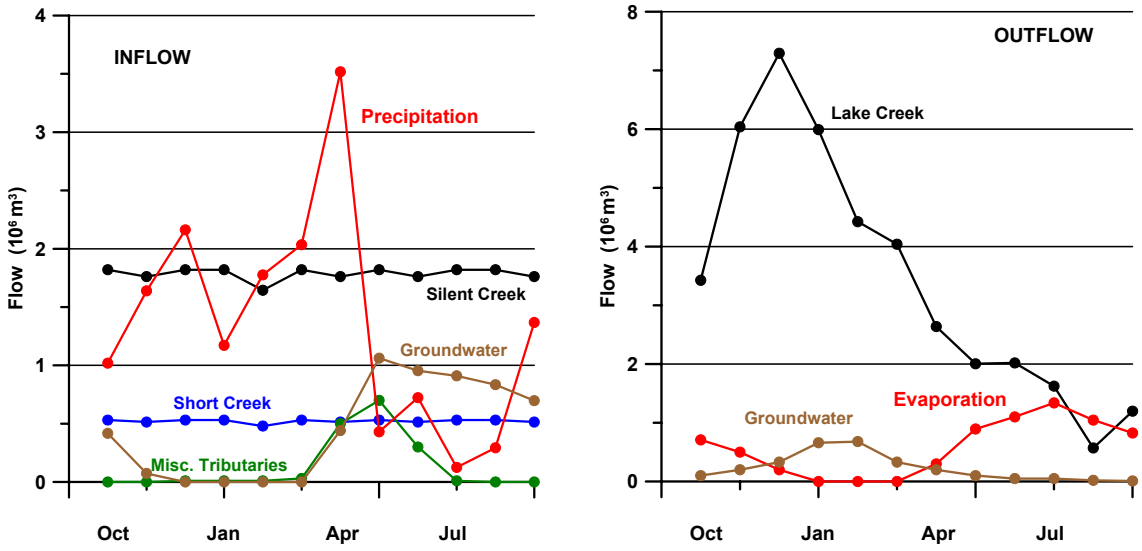


Figure 6. Flows (as $10^6 m^3$ per year) for inflow (left) and outflow (right) of Diamond Lake. Groundwater outflow is represented as a residual of measured components.

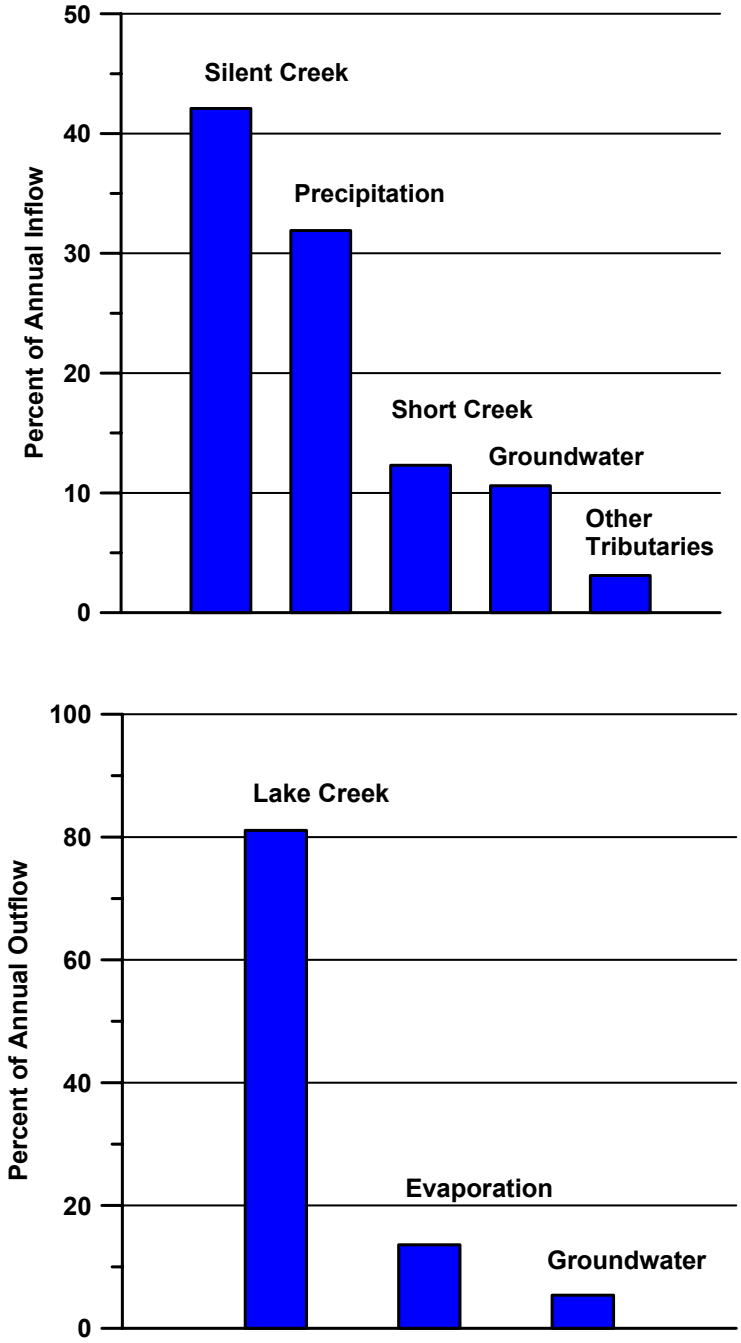


Figure 7. Annual hydrologic budget for Diamond Lake expressed as a percent of inputs (top) and outputs (top). Groundwater output is expressed as a residual of the other measured components.

b. Nutrient Fluxes

The flux of nutrients into and out of Diamond Lake is a function of the flows and the concentrations measured for each of the sources. The average nutrient concentrations measured for the Diamond Lake system are shown in Table 10 and the fluxes calculated from the flows (Table 9) and the concentrations (Table 10) are shown in Table 11. The concentrations for the major sources show that natural sources from stream inputs have very high concentrations of phosphorus and extremely low concentrations of nitrogen. Concentrations of silica inputs are high, except naturally for precipitation, which is largely devoid of silica in this region. The concentrations in the outputs are radically different than the inputs; the outlet has high concentrations of nitrogen and low concentrations of phosphorus. Silica concentrations in the outflows are substantially less than concentrations measured in the inflows. The estimates of nutrient flux derived from the flow and concentration data indicate that Diamond Lake retains about 50 percent of the total phosphorus inputs and about 60 percent of the silica inputs, but exports over six times more nitrogen than it receives from the watershed and precipitation. The lake retains about 99 percent of the nitrate input and presumably assimilates this into the macrophytes and phytoplankton. However, the retention of the nitrate is insufficient to account for the generation of 26 metric tons of nitrogen; the difference is presumably attributed to nitrogen fixation by cyanobacteria.

Table 10. Concentrations of nitrogen and phosphorus inflows and outflows for the Diamond Lake system. Concentration are expressed as micrograms per liter, except for silicon which is expressed in milligrams per liter.

Source	Flow ($10^6 m^3$)	TP	PO_4	TN	Org N	NO_3	NH_3	Si
Inflow								
Silent Creek	21.432	69	58	31.8	27	3.5	1.3	18.16
Short Creek	6.251	72	57	56.3	37	18.9	0.4	21.11
Groundwater	5.389	69	58	31.8	27	3.5	1.3	18.16
Precipitation	16.258	10	5	190	5	160	25	0
Other Tribs	1.57	69	58	31.8	27	3.5	1.3	18.16
Outflow								
Lake Creek	41.261	25	3	518.4	508	0.4	10	4.88
Evaporation	6.909	0	0	0	0	0	0	0
Groundwater	2.73	25	3	518.4	508	0.4	10	4.88

Table 11. Annual fluxes of nutrients entering and leaving Diamond Lake. Units are in kilograms per year, except where noted.

Source	TP	PO_4	TN	Org N	NO_3	NH_3	Si
Inflow							
Silent Creek	1478.8	1243.1	681.5	578.7	75.0	27.9	389205
Short Creek	450.1	356.3	351.9	231.3	118.1	2.5	131958
Groundwater	371.8	312.6	171.4	145.5	18.9	7.0	97864
Precipitation	162.6	81.3	3089.0	81.3	2601.3	406.5	0
Other Tribs	108.3	91.1	49.9	42.4	5.5	2.0	28511
	2571.6	**2084.3**	**4343.8**	**1079.1**	**2818.8**	**445.9**	**6475392**
Outflow							
Lake Creek	1281.5	153.8	26573.7	26040.6	20.5	512.6	250153
Evaporation	0.0	0.0	0.0	0.0	0.0	0.0	0
Groundwater	68.3	8.2	1415.2	1386.8	1.1	27.3	13322
	1349.8	**162.0**	**27988.9**	**27427.4**	**21.6**	**539.9**	**263476**
Net	**-1221.9**	**-1922.3**	**23645.1**	**26348.3**	**-2797.2**	**94.1**	**-384063**
Net %	**-47.5**	**-92.2**	**544.3**	**2441.6**	**-99.2**	**21.1**	**-59.3**

The loads to Diamond Lake presented in Table 11 do not reflect anthropogenic inputs from other sources. The anthropogenic sources of nutrients to Diamond Lake that are not reflected in the estimates shown above include septic inputs from the Summer Home Tract, artificial bait used for trout fishing, and fish stocking. Estimates from these sources are addressed below. Inputs from waterfowl were not estimated because of the absence of data on waterfowl populations.

The estimates of nutrient contributions from the Summer Home Tract were prepared for a range of inputs reflecting the uncertainty regarding what proportion of the human wastes entered the lake from the septic systems (Table 12). One of the factors associated with the uncertainty in delivery of the nutrients is the range in sophistication of the types of septic treatments present. Some of the systems are little more than temporary holding and overflow tanks, whereas others are professionally designed and installed septic systems with leach fields. Another component of the uncertainty is the direction of groundwater flow in the vicinity of the Summer Home Tract. Analysis of groundwater flow paths for the period August to November, 2003 indicated that a flow reversal (away from the lake) occurred by November (Breeden 2003). Consequently, not all of the water discharged from the septic systems will reach the lake. The area around Diamond Lake exhibits poor stream development, which reflects the rapid infiltration of precipitation.

Table 12. Estimates of nutrient loading to Diamond Lake from septic systems

									TN Load		TP Load	
Month	# Days	Percent	Density	TN Load	TP Load	Homes/lake	TN[a]	TP[a]	20%	80%	20%	80%
		Occupancy		g	g		g/cap/d	g/cap/d	kg	kg	kg	kg
Jan	31	5	2	7201.3	720.1	101	23	2.3	1.44	5.761	0.144	0.576
Feb	28	5	2	6504.4	650.4	101	23	2.3	1.301	5.204	0.13	0.52
Mar	31	5	2	7201.3	720.1	101	23	2.3	1.44	5.761	0.144	0.576
Apr	30	5	2	6969	696.9	101	23	2.3	1.394	5.575	0.139	0.558
May	31	20	2	28805	2881	101	23	2.3	5.761	23.04	0.576	2.304
Jun	30	30	3	62721	6272	101	23	2.3	12.54	50.18	1.254	5.018
Jul	31	40	4	115221	11522	101	23	2.3	23.04	92.18	2.304	9.218
Aug	31	40	4	115221	11522	101	23	2.3	23.04	92.18	2.304	9.218
Sep	30	30	3	62721	6272	101	23	2.3	12.54	50.18	1.254	5.018
Oct	31	15	2	21604	2160	101	23	2.3	4.321	17.28	0.432	1.728
Nov	30	10	2	13938	1394	101	23	2.3	2.788	11.15	0.279	1.115
Dec	31	5	2	7201.3	720.1	101	23	2.3	1.44	5.761	0.144	0.576
Total	365			455308	45531				**91**	**364**	**9**	**36**

[a] After Chapra (1997)

The next anthropogenic source of nutrients considered was the input of nutrients to the lake derived from the use of artificial baits. Lauer et al. (1979) considered this source and calculated an input of 70 kg of P to the lake in 1977 from the use of cheese. Although cheese is still used, the current favored artificial bait is PowerBait®. We have no specific information regarding the nutrient content of this material, but it may be similar to that of cheese. Our estimates for the input of nutrients from this source are even more modest (Table 13). We believe that even this estimate is high because much of the bait lost to the water will sink to the substrate and will be incompletely recycled into the water column.

Table 13. Estimate of phosphorus loading to Diamond Lake from artificial bait and the assumptions used to generate the estimate.

Fishing Days/yr	150
Fishing Pressure (people/day)	200
% Fishermen using Bait	50
g/day (4 oz jar)	113
% P	1
% lost to water	50
Kg P/yr	8.5

Lastly, we have the mass balance associated with the nutrients contained in the fish that are released into Diamond Lake. Lauer et al. (1979) estimated that the contribution of fish to Diamond Lake averaged about 15 kg P/yr compared to a removal of 550 kg P (as fish) during 1973. The estimates, generated by Lauer et al. (1979), were based on the stocking of about 400,000 trout fingerlings annually. The Diamond Lake trout stocking program has changed radically in recent years in response to proliferation of tui chub. Legal-size trout are now stocked because of the poor survivability of the trout fingerlings. An updated estimate for nutrient loading from fish stocking shows how the nutrient fluxes have been altered from this source (Table 14). Whereas the stocking program in the 1970s had a net export of nutrients associated with stocking (Input – Output; this does not reflect loading from internal cycling which is discussed later), the current stocking program has a net input of nutrients to the lake.

Table 14. Estimates of phosphorus and nitrogen loading to Diamond Lake based on data from ODFW for 2002 stocking. The estimates of out-migration are ours. Fingerlings are generally about 3 to 4 inches in length, ”legals” about 8 to 10 inches, and “trophys” about 14 to 16 inches.

Stocked	Number	Wet Wt (g)	Dry Weight(g)	%P	%N	Kg P	Kg N
Fingerlings	90000	18	3.6	1.7	10	6	32
Legals	49404	181	36	1.7	10	30	178
Trophys	15317	907	181	1.7	10	47	227
						83	487
Caught							
legals	11108	181	36	1.7	10	7	40
trophys	4991	907	181	1.7	10	15	90
Migrated							
legals	20000	181	36	1.7	10	12	72
trophys	10000	907	181	1.7	10	31	181
						65	383
				Net Loading		18	104

When we compare the anthropogenic sources of nitrogen and phosphorus with the natural loads, we find that anthropogenic sources represent about 16 percent of the total nitrogen load to the lake and about 1.9 percent of the total phosphorus load (Figure 8, Table 15). Based on these estimates of anthropogenic loading to Diamond Lake, it would appear that there is no basis for controlling nutrient loading. However, the deterioration of water quality in Diamond Lake has developed as a consequence of a change in internal loading of N and P.

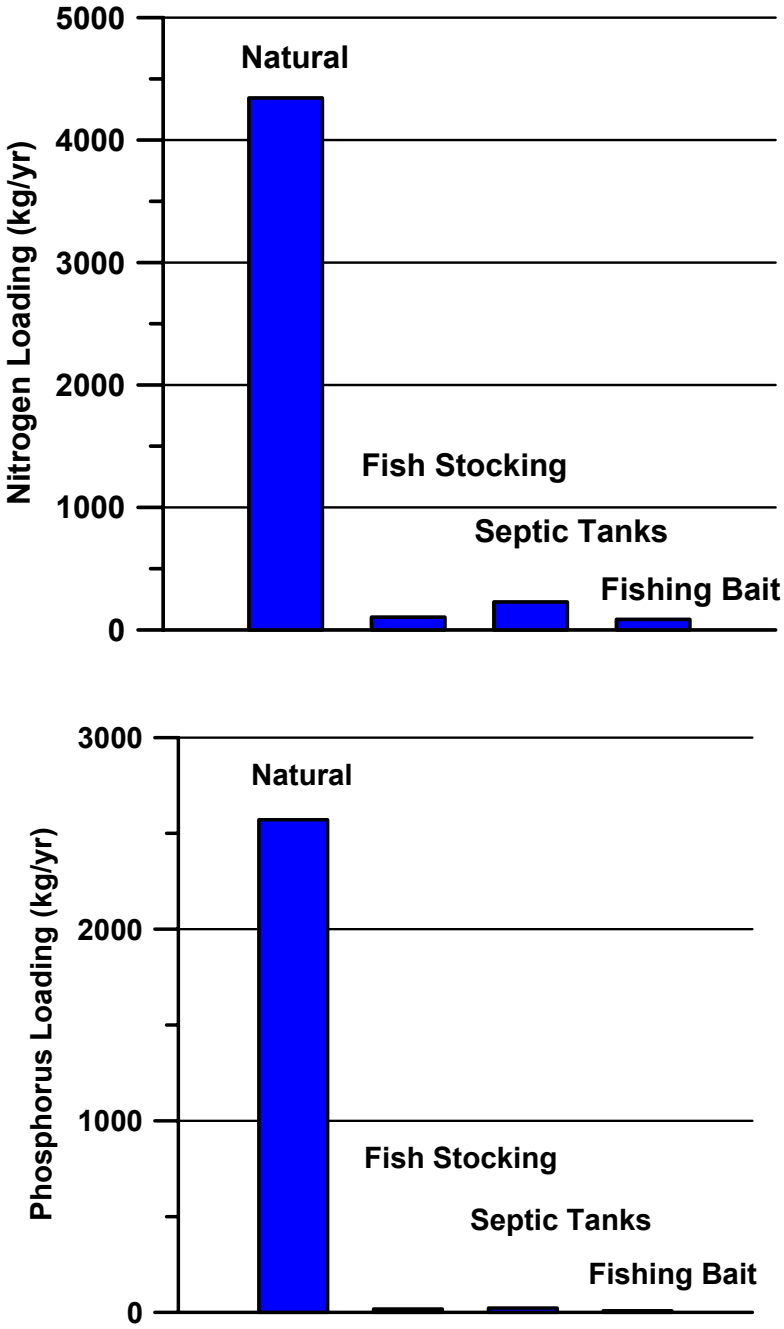


Figure 8. Annual loads of nitrogen and phosphorus to Diamond Lake for 2002-2003.

Table 15. Comparison of natural and anthropogenic sources of nitrogen and 0phosphorus annual loads.

Source	N (kg/Yr)	N (%)	P (kg/yr)	P (%)
Natural[a]	4344	91.4	2572	98.1
Anthropogenic				
Fish Stocking[b]	104	2.2	18	0.7
Septic Systems[c]	228	4.8	22.5	0.9
Fish Bait[d]	85	1.8	8.5	0.3
Subtotal	417	8.6	49	1.9
Total	4751		2621	

[a] From Table 7.
[b] From Table 10.
[c] From Table 8, using the median of the range of loading estimates
[d] From Table 9.

3. Hydrodynamic Model (CE-QUAL-W2);

The calibrated fit to the measured lake temperatures yielded an average difference of 0.05 °C and an average absolute difference of 0.57 °C (+/- 0.43; Appendix B). Static images of the model output for May through September illustrate the presence of the thermocline by mid-July, the increase in epilimnion temperature into August, and the destabilization of the thermocline in September (Figure 9). The continuous model simulation illustrates the rapid mixing of the cold inputs from Silent Creek in the surface waters of Diamond Lake (Appendix C). The model indicates that the thermocline structure in Diamond Lake is moderately stable and in most years, the thermocline should be functional for eight to 10 weeks. The current simulation is based on estimates of net groundwater flow that are less than the recent monitoring estimates. However, the thermal regime in the lake is resistant to minor modification in groundwater inputs even though these inputs are cold (~3-5 °C).

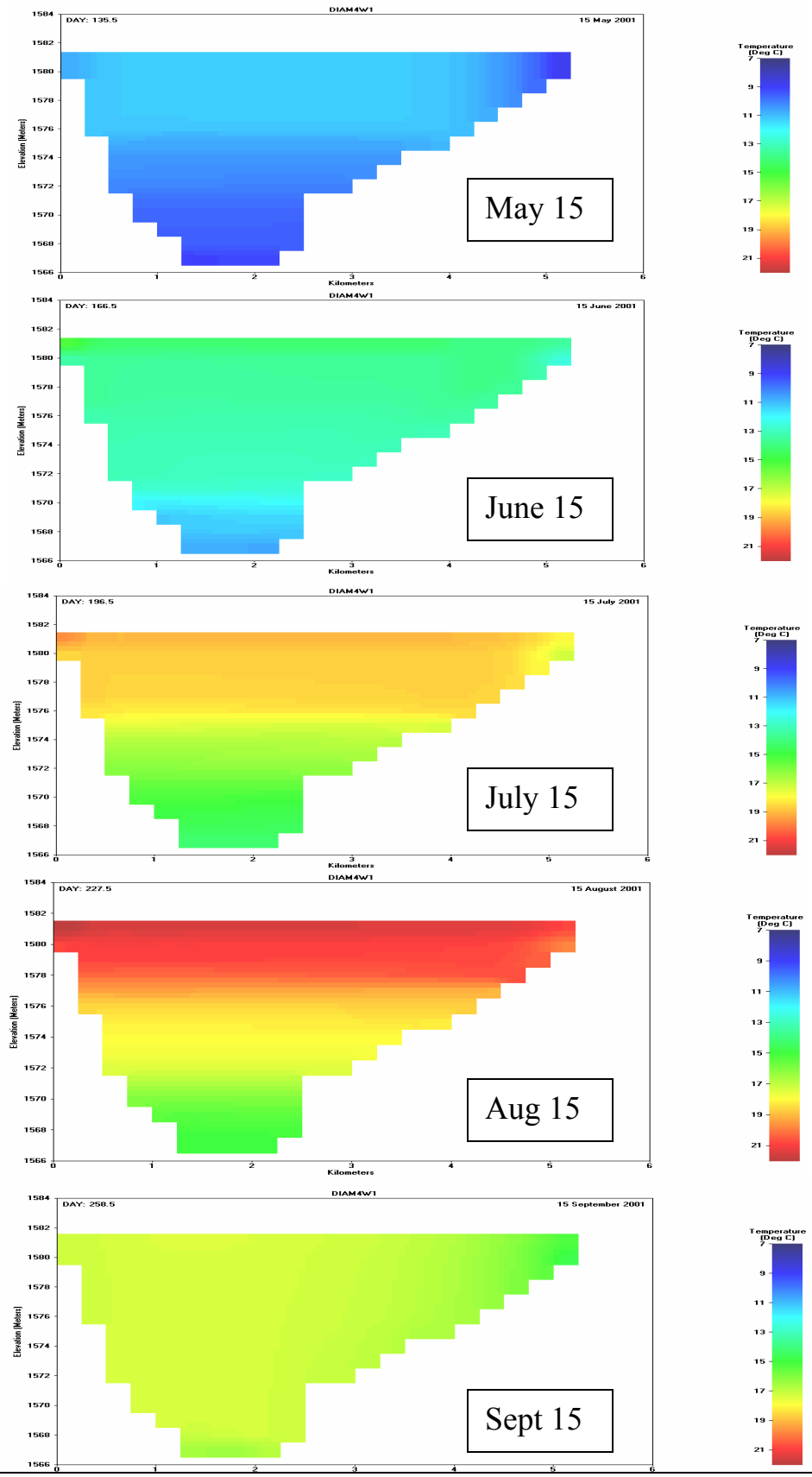


Figure 9. CE-QUAL-W2 model output for lake temperature (C) from May to September, 2001. The cross section of the lake is shown viewing the lake from the west looking east. The temperature range in the key is 7 °C (blue) to 22 °C(red).

4. Biological Assessment of Diamond Lake;

a. Dominant Communities and Community Interactions

Prior to the introduction of fish into Diamond Lake it had an abundant benthic population, which was likely dominated by amphipods, dipterans, non-dipteran insects (especially mayflies and caddisflies), and significant populations of leeches and molluscs (Eilers 2003a; Figure 10). The amphipods had no significant predators and enjoyed access too much of the lake. Macrophytes extended out to a depth of at least 8 m and thus over half of the lake was exceptionally good habitat for amphipods and non-dipteran insects. The benthic population was supported by a moderately abundant phytoplankton population of centric planktonic and epiphytic/epibenthic diatoms (Figures 11 and 12). The pre-fish zooplankton population was likely dominated by herbivorous copepods such as *Diaptomus kenai* and large cladocerans. The phytoplankton population had low densities of N-fixing cyanobacteria, most likely because of P-limitations associated with comparatively low rates of recycling of P within the lake and high rates of herbivory.

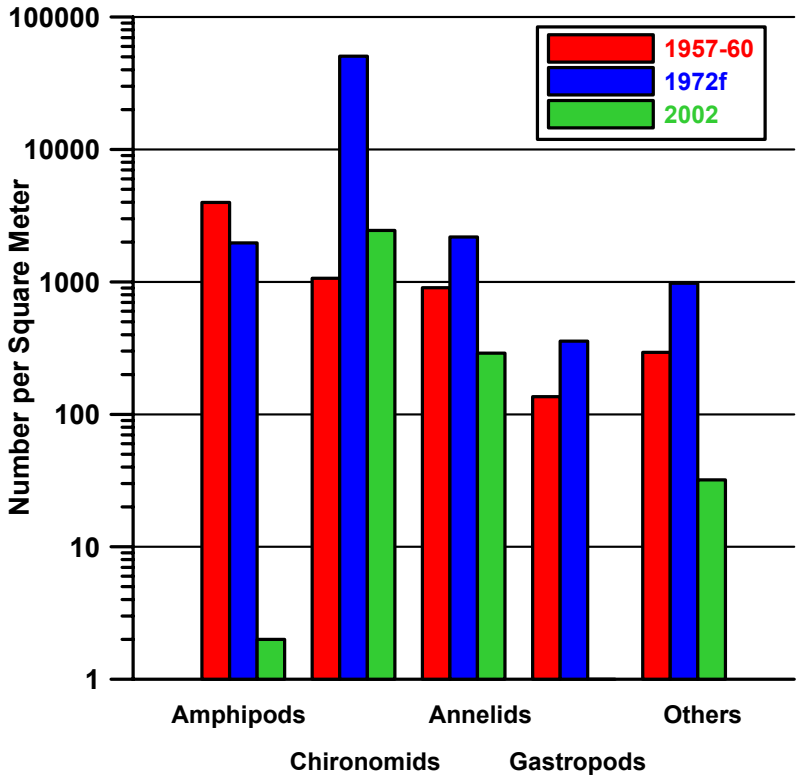


Figure 10. Densities of dominant zoobenthos in Diamond Lake from three different periods (from Eilers 2003a).

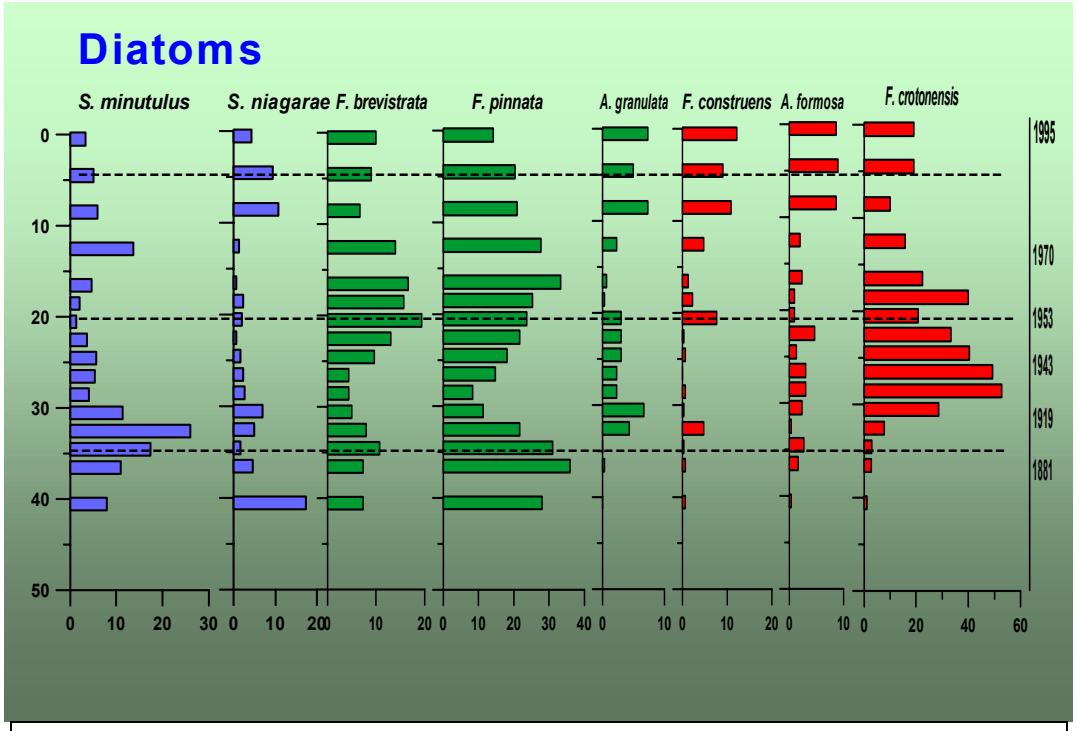


Figure 11. Stratigraphy of the dominant diatom taxa present in Diamond Lake from circa 1860 to 1995 (after Eilers et al. 2001a).

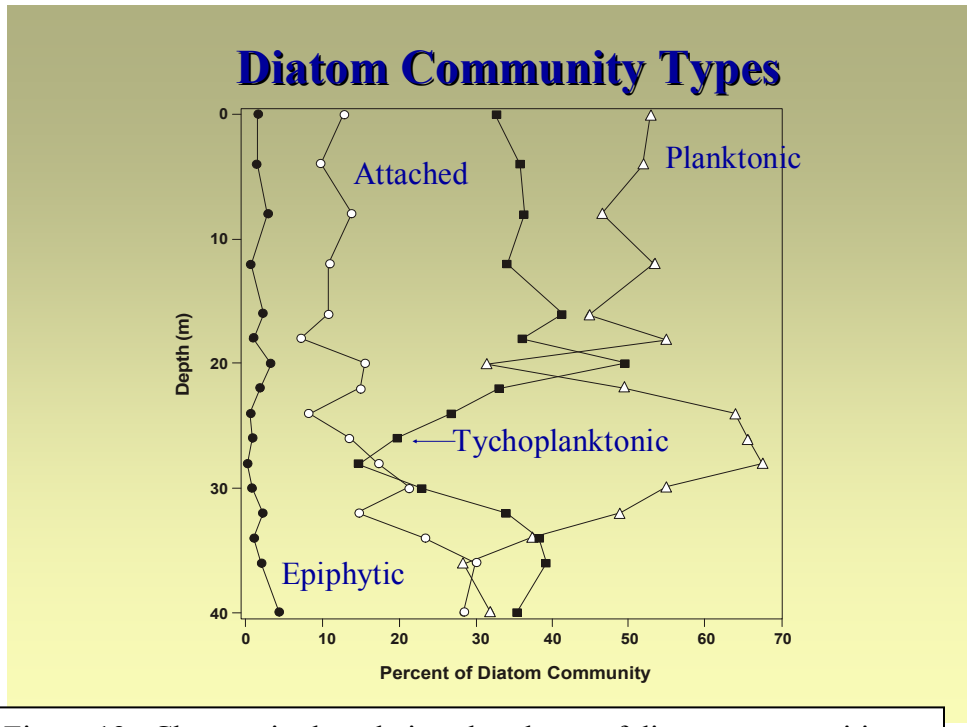


Figure 12. Changes in the relative abundance of diatom communities in Diamond Lake during the 20th century. Tycoplankton are those taxa that can operate as planktonic under conditions of high turbulence (after Eilers et al. 2001a).

With the introduction of trout circa 1910, aspects of the lake ecology changed rapidly. It is likely that the largest copepods and cladocerans experienced significant population reductions with the introduction of trout fry. Once beyond the fingerling stage, the trout probably converted to a benthic diet consisting of the larger benthos, especially amphipods, non-dipteran insects, and leeches (Eilers 2003a). The reduction in the large zooplankton herbivores likely decreased grazing rates on the phytoplankton (Sarnelle 1993; Nichols et al. 1996; Proulx et al. 1996; Pilliod et al. 2003). A decrease in average size of zooplankton taxa would increase the turnover rate of nutrients by reducing the average life-span of the zooplankton individuals (Lampert and Sommer 1997). As grazing rates on the phytoplankton decreased and nutrient availability increased, phytoplankton abundance increased and diatom taxa that were associated with macrophytes and substrate decreased (Figure 12). The increase in phytoplankton would be expected to decrease lake transparency and cause a reduction in the extent of macrophyte coverage in the lake.

However, the number of fry introduced into Diamond Lake generally did not exceed a million fish per year. With almost no natural reproduction, the trout population was limited to the numbers stocked. Growth rates of the surviving trout were extraordinary and rates of trout production were among the highest reported (Bauer 1976). The state responded by building an egg-collection facility at Diamond Lake and typically harvested about 18,000,000 eggs annually, while returning 1,000,000 fry to Diamond Lake. The trout harvested and type of fish stocking is summarized in Figure 13.

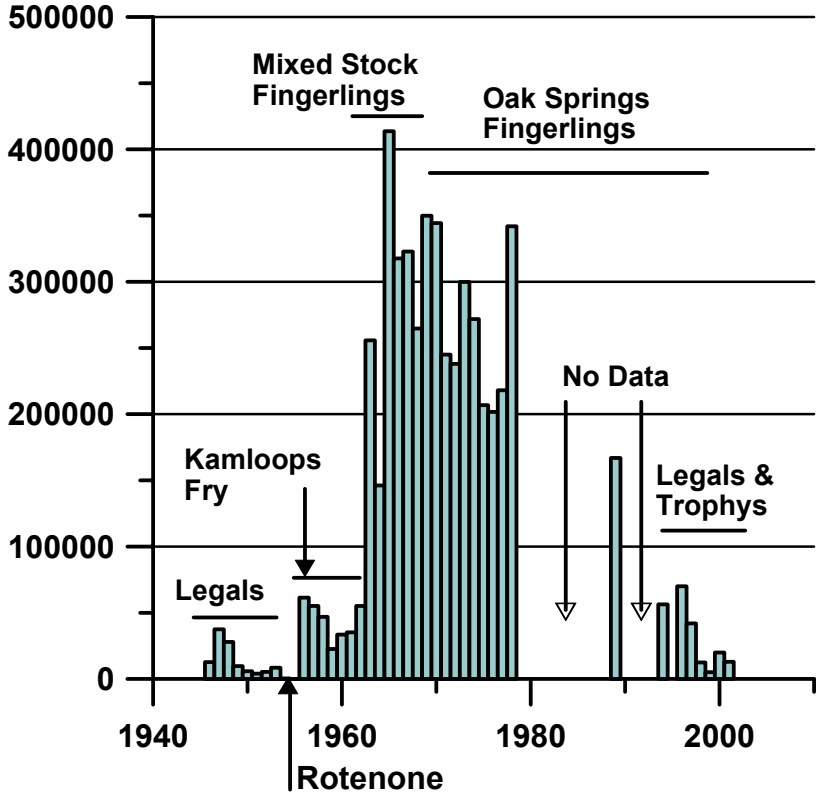


Figure 13. Number of rainbow trout caught in Diamond Lake from 1945 to 2002 and the dominant type of fish stocking employed during various periods (Source: OSGC 1945-1974; ODFW 1975-2002).

The impact to the phytoplankton was evident in a rapid increase in planktonic diatoms such as *Fragilaria crotonensis* and a decrease in attached taxa (Figure 11). Nitrogen-fixing cyanobacteria first became measurable in the lake, although the populations would not have been highly visible to the casual lake user. However, in the late 1930s or early 1940s tui chub were introduced into the lake (cf Locke 1947). The trout exhibited a precipitous drop in growth rates (OSGC 1949) and the OSGC had to abandon the stocking of fry and fingerlings and resort to stocking legal-size trout to sustain a sport fishery (Figure 13). Efforts to control the chub became more aggressive and included netting programs and partial treatment of the lake with rotenone. Trout production decreased and benthic production declined from 292 lbs/ac in 1946 to 2 lbs/acre in 1951 (Figure 14). The data in the late 1950s and during the 1970s indicate that amphipods were abundant and provided a substantial food source for the trout. By 2002, amphipods were rare and most of the benthic community was comprised of chironomids and Tubificidae (Figure 10). Again, this pattern of loss of the larger benthic individuals is consistent with effects of omnivorous fish observed elsewhere (Drenner et al. 1996)

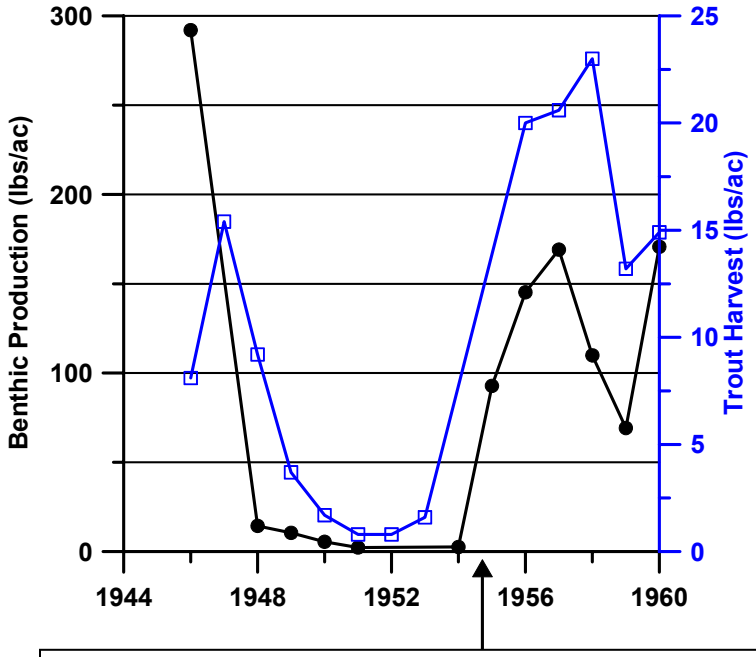


Figure 14. Benthic biomass and trout harvest for the period 1946 to 1960. Data from OSGC (1949-1961).

Although there were no measurements of algae or transparency recorded prior to 1971, we know from the paleolimnological results that during the 1940s, cyanobacteria populations increased greatly (Figure 15). OSGC staff reported that macrophyte populations decreased markedly (OSGC 1954), presumably because of light-limitation associated with the cyanobacteria blooms. Following treatment of Diamond Lake with rotenone in 1954, macrophytes began to return (OSGC 1958) and the benthic macroinvertebrate community began to recover in abundance and in desirable taxa (Figure 10). Again, there were no phytoplankton samples collected during this time, but

the paleolimnological data indicate that cyanobacteria populations greatly decreased following the rotenone treatment (Figure 15). A reconstruction of the fluctuations in macrophyte coverage for Diamond Lake based on the information reported by OSGC, the data reported by Lauer et al. (1979), and the hydroacoustic survey conducted in 2002 (Eilers and Gubala 2003) is depicted in Figure 16.

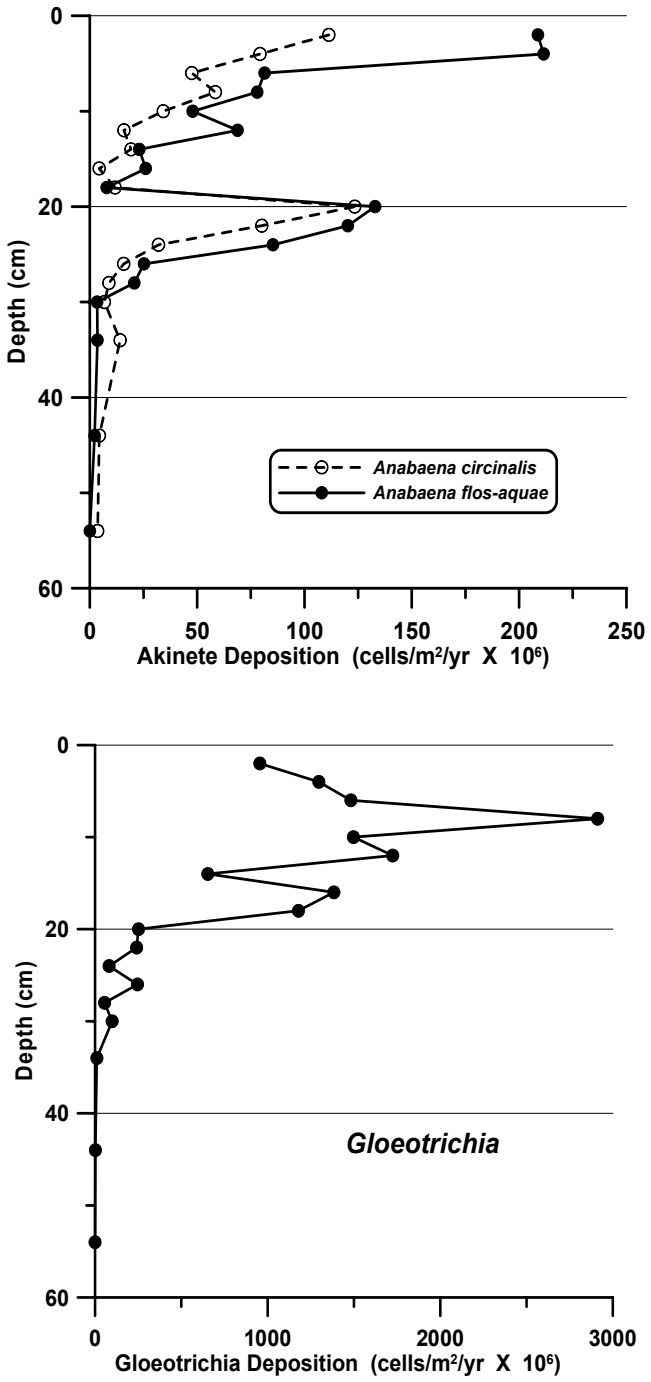


Figure 15. Deposition of cyanobacterial akinetes in Diamond Lake for *Anabaena flos-aquae* and A. *circinalis* (top) and *Gloeotrichia sp*. (bottom). From Eilers et al. 2001b.

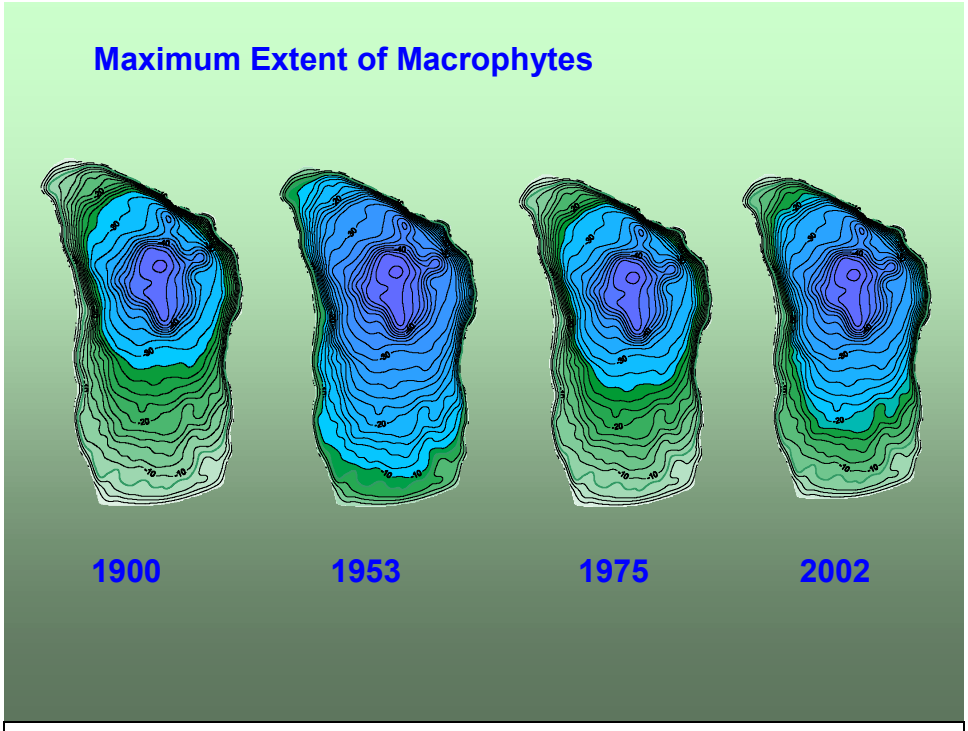


Figure 16. Estimated extent of macrophyte coverage (shown as green) in Diamond Lake for periods with reported data (1953, 1975, and 2002) and inferred distribution for 1900 prior to the introduction of fish.

Recreational development at Diamond Lake began in the 1920s. The Diamond Lake Lodge was constructed in 1923 and the Forest Service opened the Summer Home Tract consisting of 101 long-term leases for construction of private, seasonally-occupied dwellings. The Umpqua National Forest began building public campgrounds along the lakeshore and public use gradually increased, largely attracted by the bountiful trout fishery. Expansion of the public and private recreational opportunities at the lake increased most aggressively from the 1960s to the 1980s with expansion of the Diamond Lake Lodge, opening of additional public campgrounds, opening of a private trailer park, and completion of the construction in the Summer Home Tract. However, the lake was once again showing evidence of moderate phytoplankton growth and the Forest Service installed a sewage collection system on the east side of the lake to collect, transport, and treat the sewage. The only source not included in the collection system was the Summer Home Tract, which continued to use private septic disposal systems. In addition, Highway 138 was relocated away from the lake in 1978. Thus, it is expected that anthropogenic sources of nutrients to the lake increased up until the mid-1970s, but these loads decreased after this period (Eilers et al. 2001a).

Despite the reduction in watershed loads of nutrients to Diamond Lake, water quality through the 1970s exhibited summer pH values over 9.5 and the presence of substantial densities of cyanobacteria (Lauer at al. 1979). No response to the sewage diversion was observed by Lauer et al. (1979) and they concluded that most of the nutrient sources to

the lake were natural. Throughout this period, the trout fishery thrived, with an annual stocking of 300,000 to 500,000 Oak Springs rainbow trout fingerlings. Trout harvest averaged over 275,000 fish/yr compared to an average of less than 45,000 fish/yr from 1956 to 1962 (Figure 16). Despite the substantial net export of fish biomass during the successful trout fishery, water quality remained moderately poor. In 1992, ODFW staff recorded the first tui chub in their nets since the rotenone treatment of Diamond Lake in 1954. The presence of the adult chub in 1992 suggests that tui chub may have been introduced around 1988 or 1989. An alternative interpretation is that tui chub were present in the lake as a remnant population following the rotenone treatment in 1954 and only began increasing in measurable numbers in the 1990s. The increase in the tui chub population appeared to be rapid as documented by the ODFW (Figure 17). A water quality monitoring program was implemented by the Umpqua National Forest starting in 1992. The sampling generally consisted of sampling the lake and tributaries three times from June to September. Trends are difficult to establish in the data because of the intensity of sampling, the lack of annual synchronization in sampling, and some technical problems with the data collected. Nevertheless, the data exhibit some noteworthy changes in zooplankton and phytoplankton abundance, which are described below.

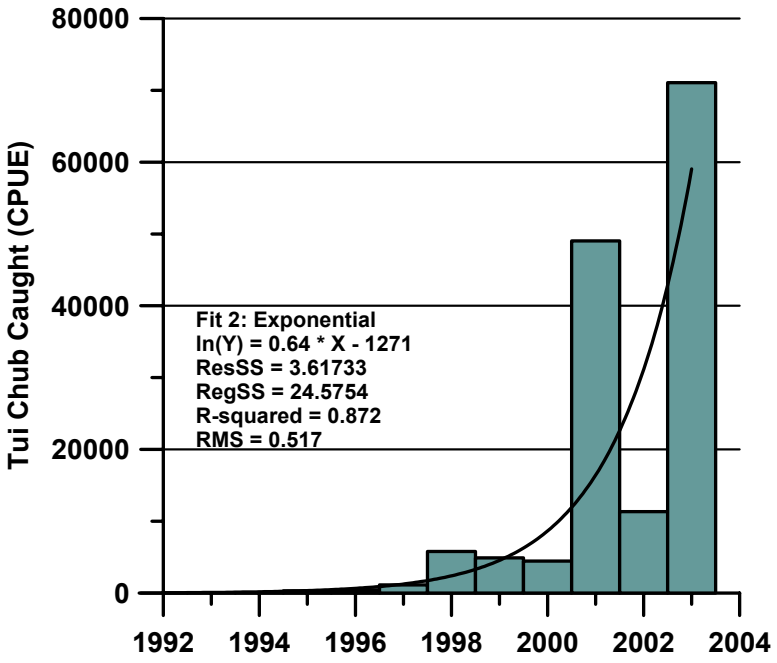


Figure 17. Tui chub caught per unit of effort in Diamond Lake during May from 1992 to 2003 (Data from Jackson et al. 2003). The exponential fit of the data is provided in the insert.

b. Zooplankton

The zooplankton data showed a conversion of a system with moderate-sized cladocerans in 1994 to smaller cladocerans in 1996 and a dominance of rotifers by 1999 (Salinas and Larson 1995, Salinas 1996-2002). This is exhibited in a major reduction in the percent of edible zooplankton (Figure 18) and an increase in the percent rotifers in the zooplankton community (Figure 19). Additional details of the changes in the zooplankton population since 1994 are described in Eilers (2003b). The potential influence of the zooplankton on chemical cycles in Diamond Lake is indicated by the apparent synchronization of the silicon concentrations in the lake with the percent rotifers in the zooplankton (Figure 20).

A low percentage of rotifers indicates that the zooplankton population is dominated by a relatively high percentage of larger herbivorous cladocerans and copepods. The grazers will consume large quantities of phytoplankton, especially diatoms. The diatoms will be excreted in particulate form (Lampert and Sommer 1997) and sink to the lake bottom, thus removing silica from the water column. As the number of planktonic herbivores declines (percent rotifers increase), the grazing on diatoms decreases allowing more silica to remain in the water. An alternate hypothesis is that as the cyanobacteria become dominant, the diatom population decreases and less silica is taken out of solution to support the formation of the frustules. Without additional data in the non-summer seasons for Diamond Lake it is difficult to determine the mechanism for the changes observed in lake silica concentrations. Although most of the available zooplankton data were collected from 1994 –2003, several samples were collected in 1971. Sanville and Powers (1973) reported that the majority of zooplankton individuals sampled were cladocerans.

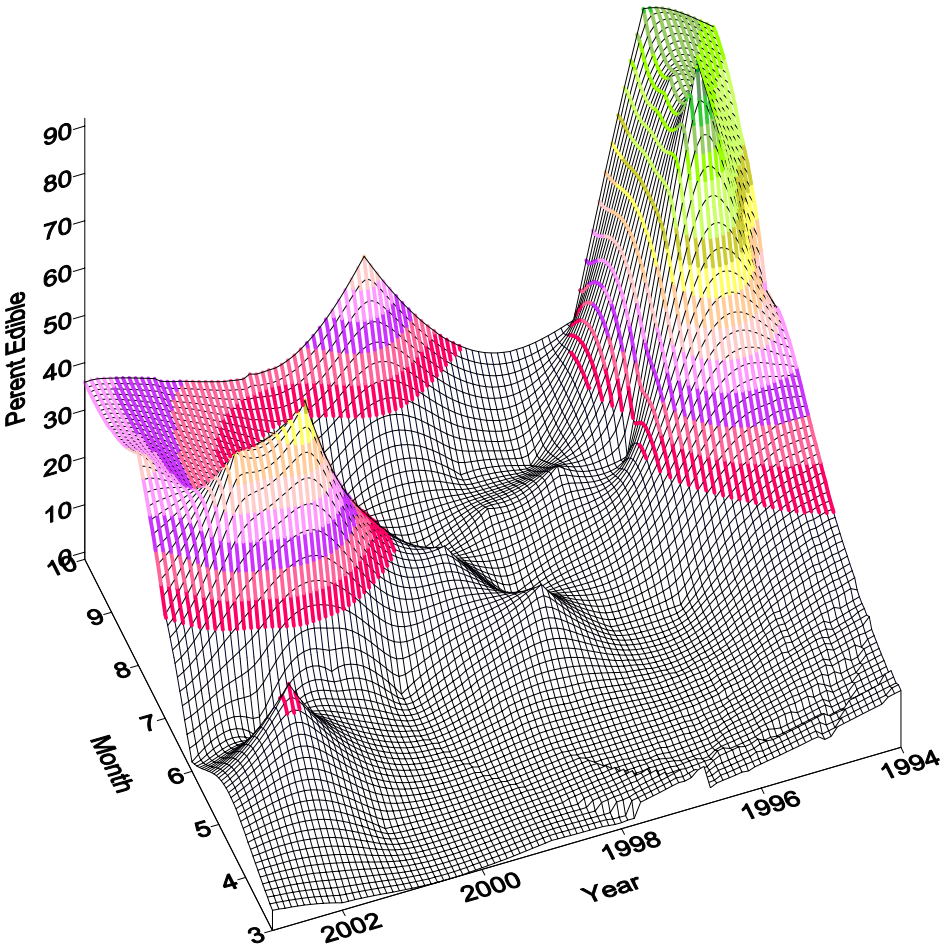


Figure 18. Changes in percent edible zooplankton in Diamond Lake from 1994 to 2003. The data and index of edible zooplankton is derived from A. Vogel (as presented in Salinas and Larson 1995; Salinas 1996-2002) and shown in Eilers (2003b).

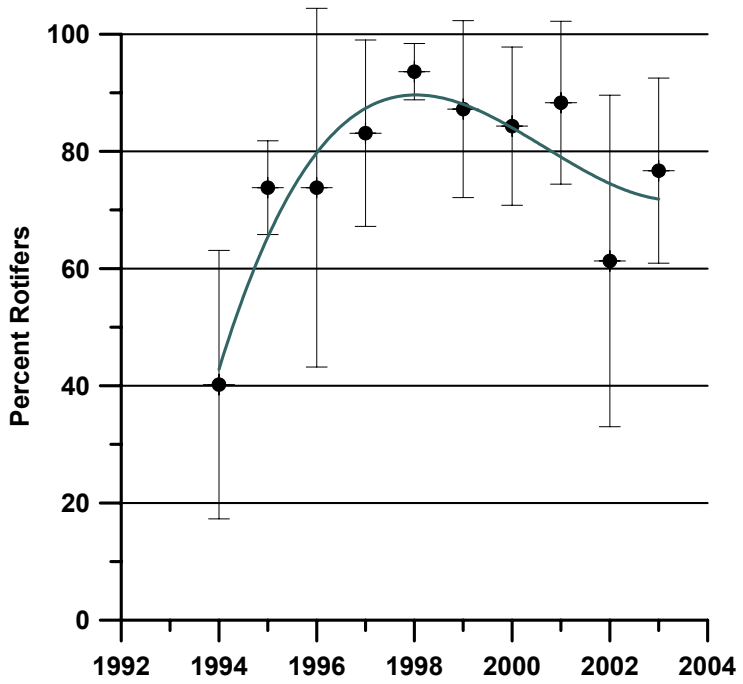


Figure 19. Percent rotifers in the zooplankton population in Diamond Lake from 1994 to 2003. The vertical bars represent the standard deviations for the samples within a year (Source: A. Vogel, In Salinas and Larson 1995; Salinas 1996-2003).

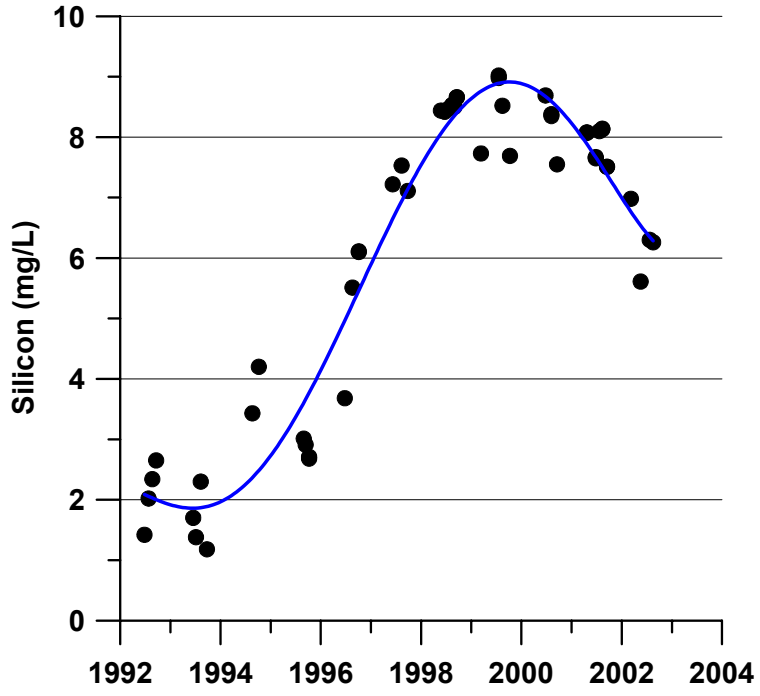


Figure 20. Dissolved silicon in the surface waters of Diamond Lake from 1992 to 2003 (from Eilers 2003b). The line is the best-fit polynomial for the observed data.

The conversion of the zooplankton population in Diamond Lake is significant in several aspects. First, the reduction in size of the zooplankton means that the zooplankton population is no longer available as a food source for trout. Although the trout historically only used the zooplankton population when the lake was stocked with fry in the period from 1910 to 1946 and again from 1955 to 1961 (ODFW, unpublished data), the zooplankton would have served as a supplemental source of food to replace the reduction in desirable food sources in the lake benthos. A second consequence of the loss of the larger zooplankton is that this included the large herbivorous zooplankton that grazed heavily on the phytoplankton. The loss of the grazing pressure released one of the controls on the phytoplankton (light, nutrients, temperature, grazing). Lastly, the reduction in size of zooplankton resulted in a population of rotifers that have a much shorter life span than the larger copepods and cladocerans. The life span of the larger herbivorous cladocerans and copepods can be several months (Williamson and Reid 2001; Dodson and Frey 2001) compared to an average life span of about 2.5 to 16 days for rotifers (Wallace and Snell 2001). The brief life span of the rotifers promotes a rapid cycling of nutrients as the biomass cycles much faster than when the system was dominated by larger copepods and cladocerans.

c. Phytoplankton

Diamond Lake water quality concerns that are currently tied to recurrent blooms of cyanobacteria (blue-green algae) and diatoms include hypolimnetic dissolved oxygen depletion, high pH, and the production of toxins by cyanobacteria. The later is of particular concern due to the potential public health threat to lake users. The two main species of toxin-producing cyanobacteria associated with the blooms in Diamond Lake are *Anabaena flos-aquae (*ABFA*)* and *Microcystis aeruginosa (*MSAE), with ABFA the more prominent toxin producer (Eilers and Kann 2002). ABFA is most frequently associated with the powerful neurotoxin anatoxin-a; however, it can also produce hepatotoxic microcystins. MSAE produces microcystins (Yoo et al. 1995). These species, like all cyanobacteria, also have compounds in their cell walls, which are probably responsible for adverse skin, eye, mucosal, and digestive reactions reported by people who have come in contact with them (Chorus 2001). Recent blooms of ABFA have caused the Umpqua National Forest to issue water quality alerts and to close the lake to recreational use in each if the past three years.

The purpose of this section is to analyze baseline phytoplankton bloom dynamics relative to inter-annual and seasonal trends, as well as potential driving variables. Phytoplankton and physical-chemical parameters were collected as part of annual sampling initiated chiefly by the USFS (Salinas 1992-2002; Eilers 2003; and DEQ 2002-2003). Such an analysis is intended to supplement TMDL empirical modeling efforts. However, due to inconsistency in sample collection until 2003, conclusions reached here are tentative and must be corroborated with more intensive data collection. For example, the lack of temporal resolution of phytoplankton and nutrient data, including important spring data, is necessary to understand spring diatom blooms and conditions leading up to summer

dominance by toxic cyanobacteria. In addition, important summer-growing season months were sampled erratically, such that two-month gaps often exist during this period. Moreover, data for total phosphorus, which is one of the most important driving nutrients for algal biomass, was not collected at all until 2001, and not consistently until 2002.

As noted above, numerous gaps and inconsistent sample timing exists for seasonal phytoplankton coverage in Diamond Lake (Figure 21). However, for those spring months where data do exist, (e.g., March 1999 and 2002; April 2001 and 2003; and May 1998, 2002 and 2003), the phytoplankton is dominated by diatoms in April and May; with occasional dominance by cryptophytes (March 1999) or chrysophytes (March 2002). During the summer growing season, dominance is by a mixture of cyanophyta and diatoms, with cyanophyta being more important in 1992, 1993, and 2001-2003 (see Appendix D for complete list of phytoplankton taxa). However, it is conceivable that significant cyanophyte peaks occurred in other years (e.g., July 1997), but were not detected due to inadequate temporal resolution. It is also apparent based on the more frequent sampling that occurred in 2001-2003 that monthly sampling may not be

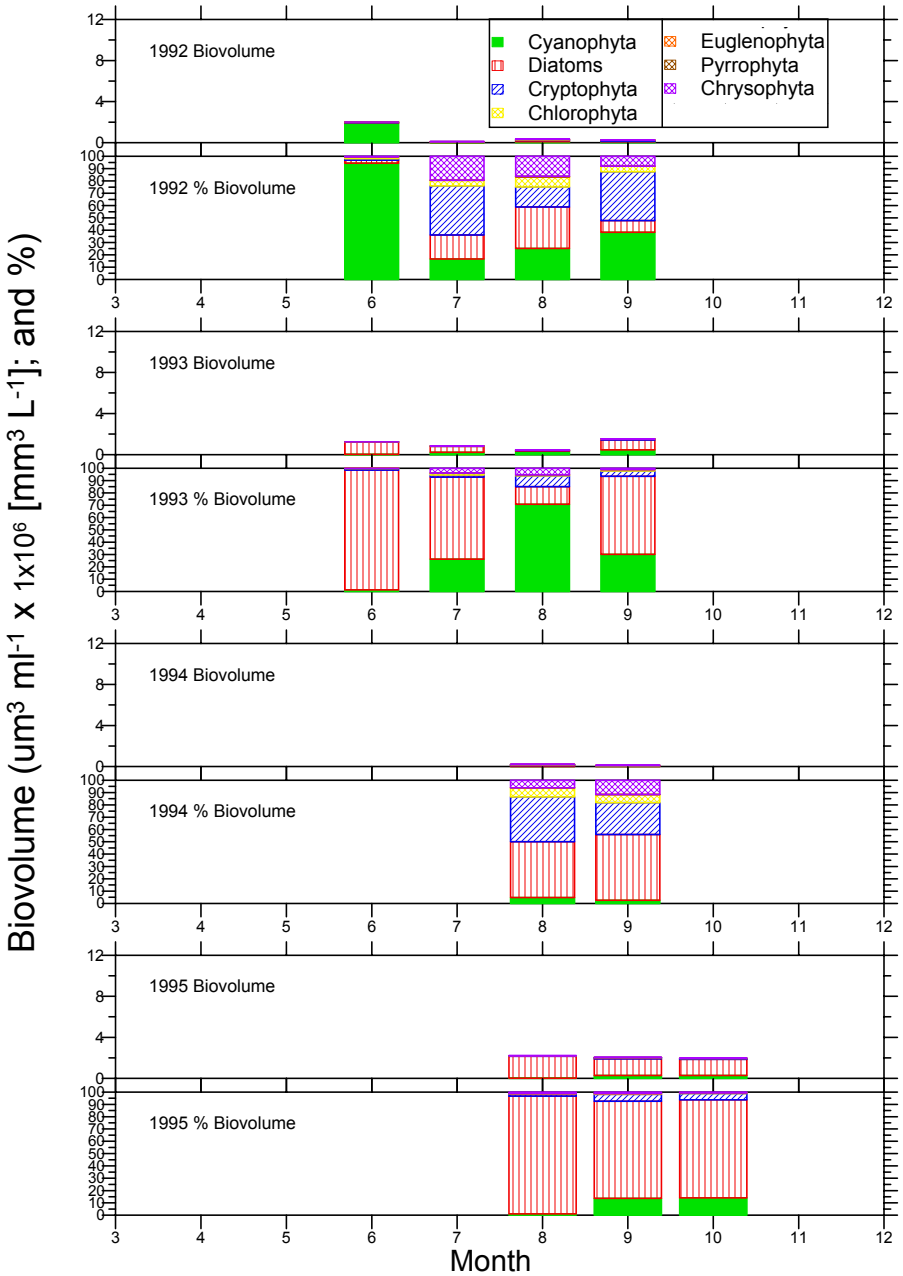


Figure 21a. Monthly depth-averaged biovolume and percent biovolume of major phytoplankton taxa for Diamond Lake station DLA, 1992-1995.

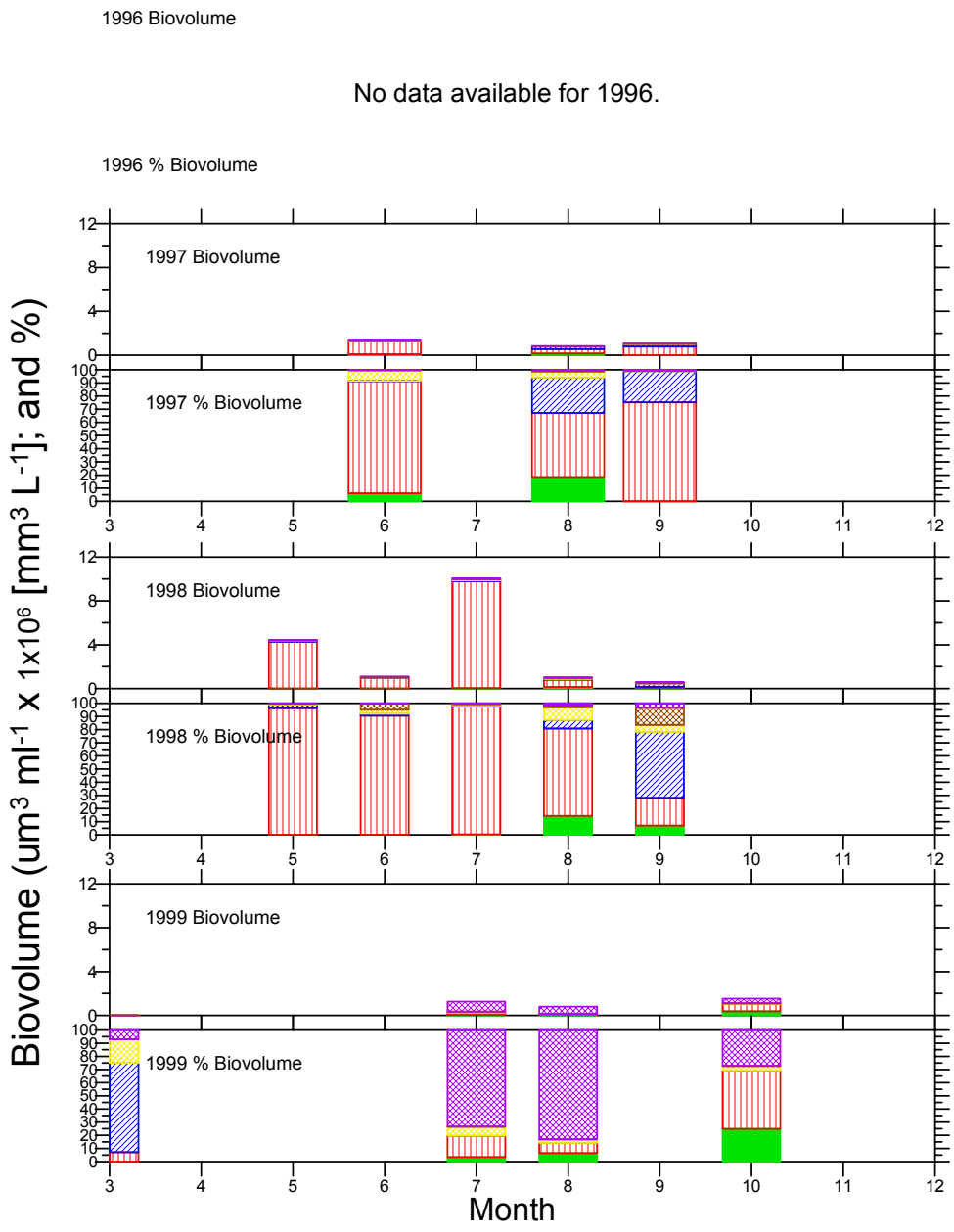


Figure 21b. continued (1996-1999).

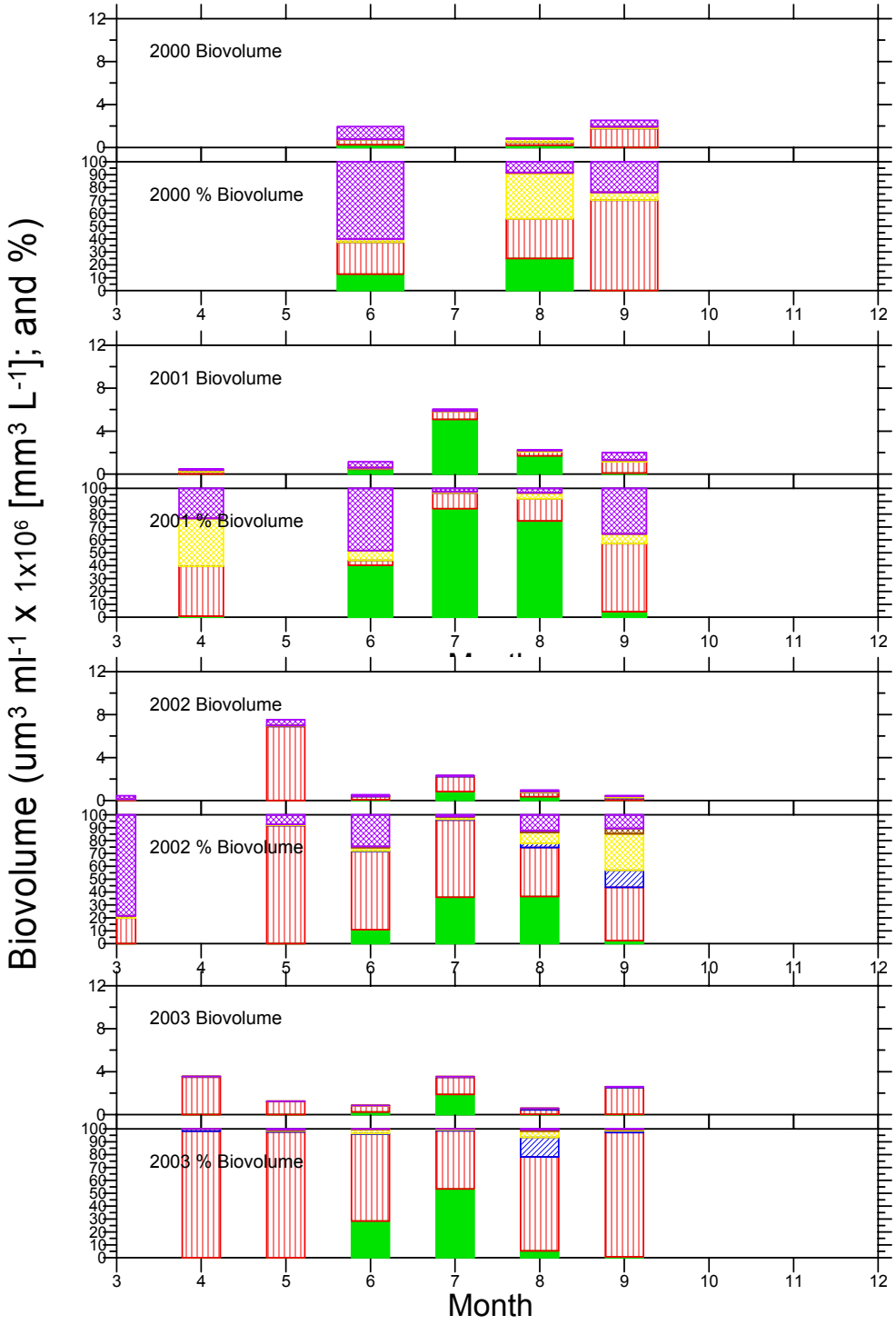


Figure 21c. continued (2000-2003).

frequent enough to detect rapidly changing growth trends in the phytoplankton, particularly bloom peaks for cyanophytes (Figure 22). Based on these figures it appears that at a minimum biweekly sampling is necessary to characterize phytoplankton trends in Diamond Lake. Occasional partial dominance by chrysophytes also occurred during the summer months (e.g., July-August 1999, June 2000, and June 2001). Chlorophytes and cryptophytes only occasionally represented a noticeable portion of the phytoplankton composition on a water-column average basis. Missing monthly samples precludes the ability to plot and compare average seasonal trends among years.

The depth distribution of major phytoplankton taxa indicates both a seasonal shift in the distribution of algae with depth, and that cryptophytes and chrysophytes can dominate at middle and deeper depths (Appendix E). For example, although highly variable, for years when cyanophytes were dominant, they tended to be mixed to a greater depth in the water column in June, but were more concentrated at the surface in July. This presumably indicates greater water column stability and concentration of buoyant cells at surface depths as the growing season proceeds. August was highly variable with respect to depth distribution of major taxa, and based on temperature and dissolved oxygen profiles, the water column often began seasonal mixing during this period.

Time series of dominant species (Appendix F) indicate that although biovolume of toxic ABFA was substantially higher during 2001-2003 (with 2001 the highest) than other years, ABFA was still prevalent, and represented >40% of the biovolume in 1992, 1993, and 1988. Other dominant cyanophyte species include *Anabaena circinalis* (ABCR), and *Anabaena planktonica* (ABPL) (Appendix F). Toxic MSAE represented a small portion of the total biomass in 2001-2003, nonetheless these levels were enough to exceed safe cell count levels in all three years. Although a clear trend of increasing ABFA and ABCR is difficult to definitively show due to inadequate temporal resolution, paleolimnological data does show an increasing trend of ABFA and ABCR akinetes during the past decade (Eilers et al. 2001).

Dominant diatom species include *Fragilaria crotonensis* (FRCR), *Synedra* sp. (SNPR, SNUL, and SNRD), Aulacoseira sp. (ALSP), and *Fragilaria construens* (FRCV). Blooms of *Synedra sp*., FRCR, and FRCV may be more prevalent during the later years, while *Stephanodiscus astrea* (STAS) and *Asterionella formosa* (ASFO) were more prevalent prior to 2000 (Appendix F).

Dominant chrysophytes include *Mallomonas* sp. (MMXX), *Chromulina* sp. (KMXX), and *Chrysochromulina* sp. (KKXX) (Appendix F). MMXX showed increased dominance from 1999 to 2002 (>60% of the total biovolume on several dates), while KMXX was more prevalent 1992-1993. A peak of KKXX occurred in 2002.

Dominant cryptophytes include *Cryptomonas erosa* (CXER) and *Rhodomonas minuta* (RDMN), with no apparent trend evident form the time series data (Appendix 3). The chlorophytes Cosmarium sp. (CSXX), and *Chlamydomonas* sp. (CHLK and CHXX) were occasionally dominant as well. While many of these chrysophytes, cryptophytes, and chlorophytes do not always represent a large portion of the biomass because cell size is

very small (e.g., KMXX and KKXX), they are nonetheless important numerically, and represent food sources for smaller zooplankton species.

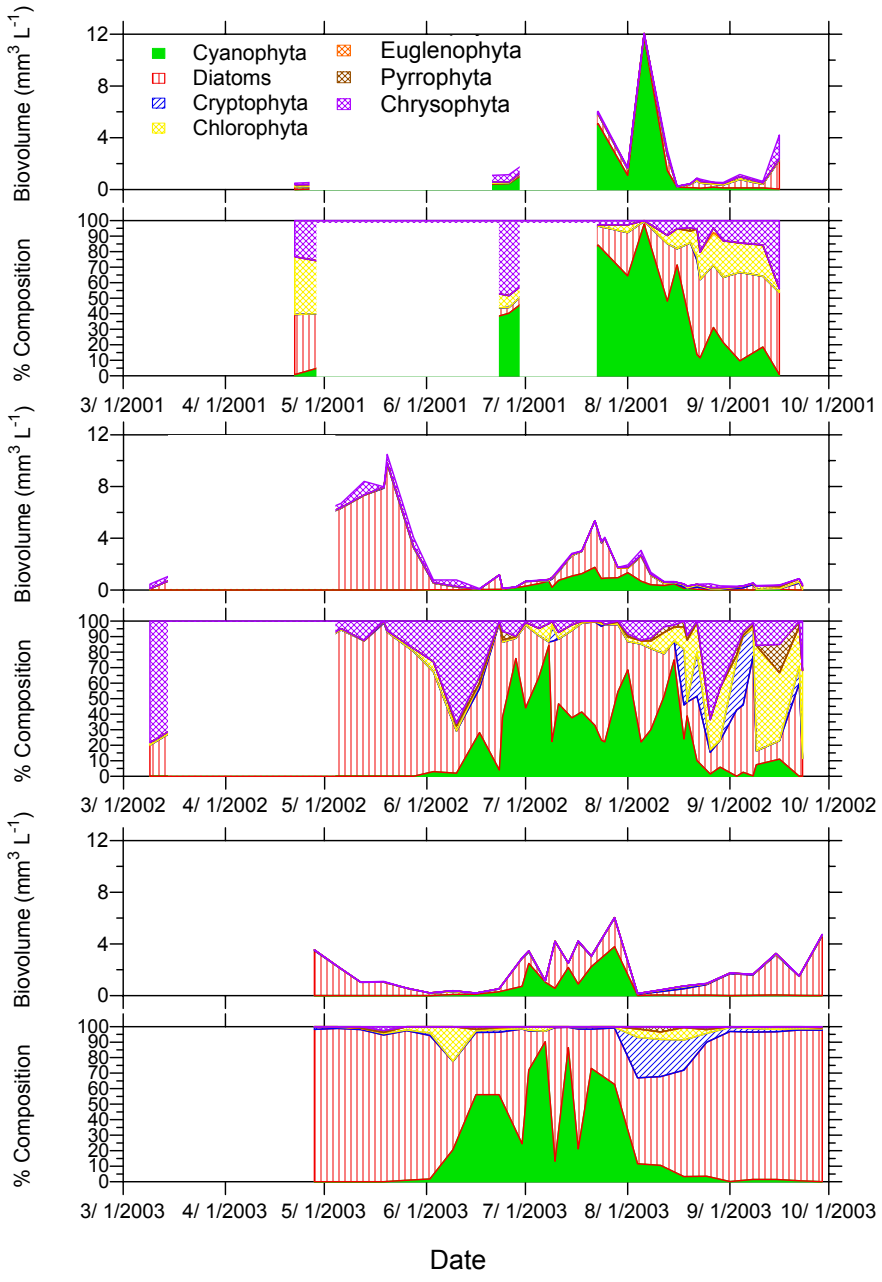


Figure 22. Time series of surface (0-1 m) biovolume and percent composition for major phytoplankton taxa at Diamond Lake station DLA for 2001 (a), 2002 (b), and 2003 (c).

In response to the large bloom of toxic ABFA in 2001, increased frequency of sampling for phytoplankton began in late July at several stations in Diamond Lake. Although spring coverage is poor, the years 2001-2003 provide the best coverage for observing growing-season dynamics. Because the most consistent coverage exists for surface depths at station DLA, further analysis will focus on this station and depth. Temporal trends indicate that the 2001 cyanophyte bloom was the largest of the three years (Figure 22), and that it consisted almost entirely of ABFA in all three years (Figure 23). The June-July algal blooms in 2002 and 2003 had, at times, almost equal dominance by diatoms, mainly *Fragilaria* sp. (FRCR and FRCV), whereas ABFA was more exclusively dominant in 2001 (Figure 23). A very large *Synedra* sp. (SNPR, SNRD) bloom occurred in May of 2002 and tended be the dominant spring diatom, becoming dominant again during the cyanophyte bloom decline in August (Figure 20b,c). *Mallomonas sp*. (MMXX) was also occasionally important in 2001 and 2002. The ABFA bloom consistently began in June in all three years and declined in mid August in all years (Figure 23).

Time series trends among years for major nutrient parameters are also difficult to evaluate due to the same inadequate temporal resolution for years prior to 2001 (Figure 23). Also, as noted above, total phosphorus was not measured in earnest until 2002. In addition, total nitrogen data was not collected until 2001. With this caveat aside, there appears to be a trend of increasing dissolved silica until 1999, and then decreasing through 2002 (Figure 23). The decreasing dissolved silica occurs as dominance by cyanophytes increases during 2001-2003; however, diatoms are also more prevalent during this period indicating that silica is likely not limiting, and that competition for light and other nutrients drives both overall biomass and composition. Reynolds (1994) also indicates that based on half saturation constants, the silica levels that occurred 1992-2003 are not expected to be limiting.

Another apparent trend (barring low sample frequency prior to 2001) may be increased levels of inorganic nitrogen (NO_3-N and NH_3-N) during years in which cyanophytes dominated (1992-1993, and 2001-2003; Figure 23). This may reflect conditions of increased water column stability that tend to favor buoyant cyanophytes such as ABFA, ABCR, and ABPL, but also foster hypolimnetic dissolved oxygen depletion and subsequent release of ammonia form decomposing algal cells and sediment. These cyanophytes are also nitrogen-fixers and tend to dominate when inorganic nitrogen is low (especially relative to phosphorus); however, the values in Figure 23 reflect the water column average, and as will be shown below, inorganic nitrogen is low near the lake surface where the cyanophytes are more prevalent.

A plot of monthly averages for these same nutrient variables for those years when the phytoplankton data were collected at an increased frequency (Figure 24) does little to elucidate trends, in part because, water chemistry sampling was not fully synchronized with collection of the phytoplankton samples, except for 2003. Dissolved silicon was greatest in 2001 (Figure 20) and may reflect lower dominance by diatoms (less uptake), however, as stated above, the silicon concentrations during 2001-2003 are not likely to be

limiting. One apparent trend is increasing ammonia-N in July and August in each of the years of cyanophyte

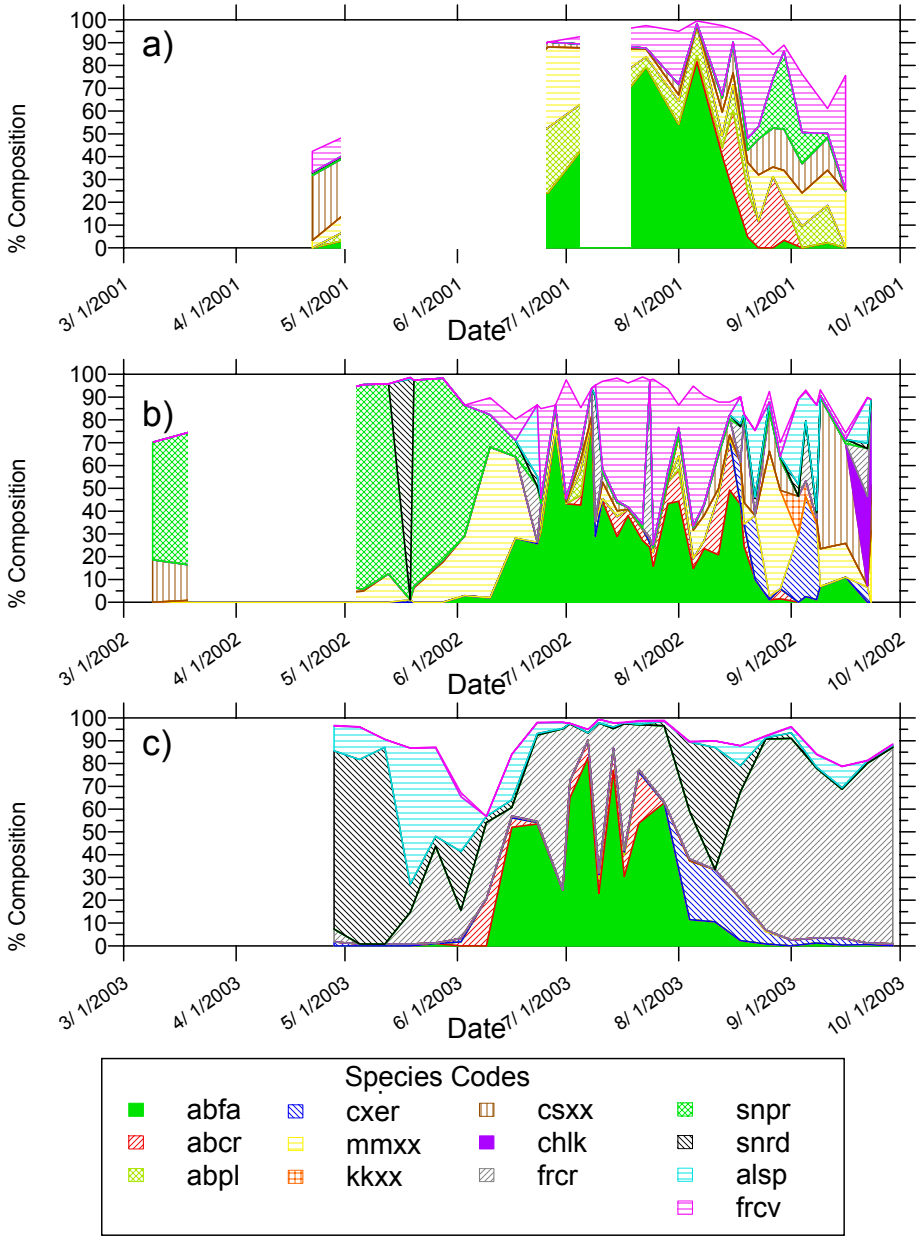


Figure 23. Time series of surface (0-1m) biovolume and percent composition for major phytoplankton species at Diamond Lake station DLA for 2001 (a), 2002 (b), and 2003 (c).

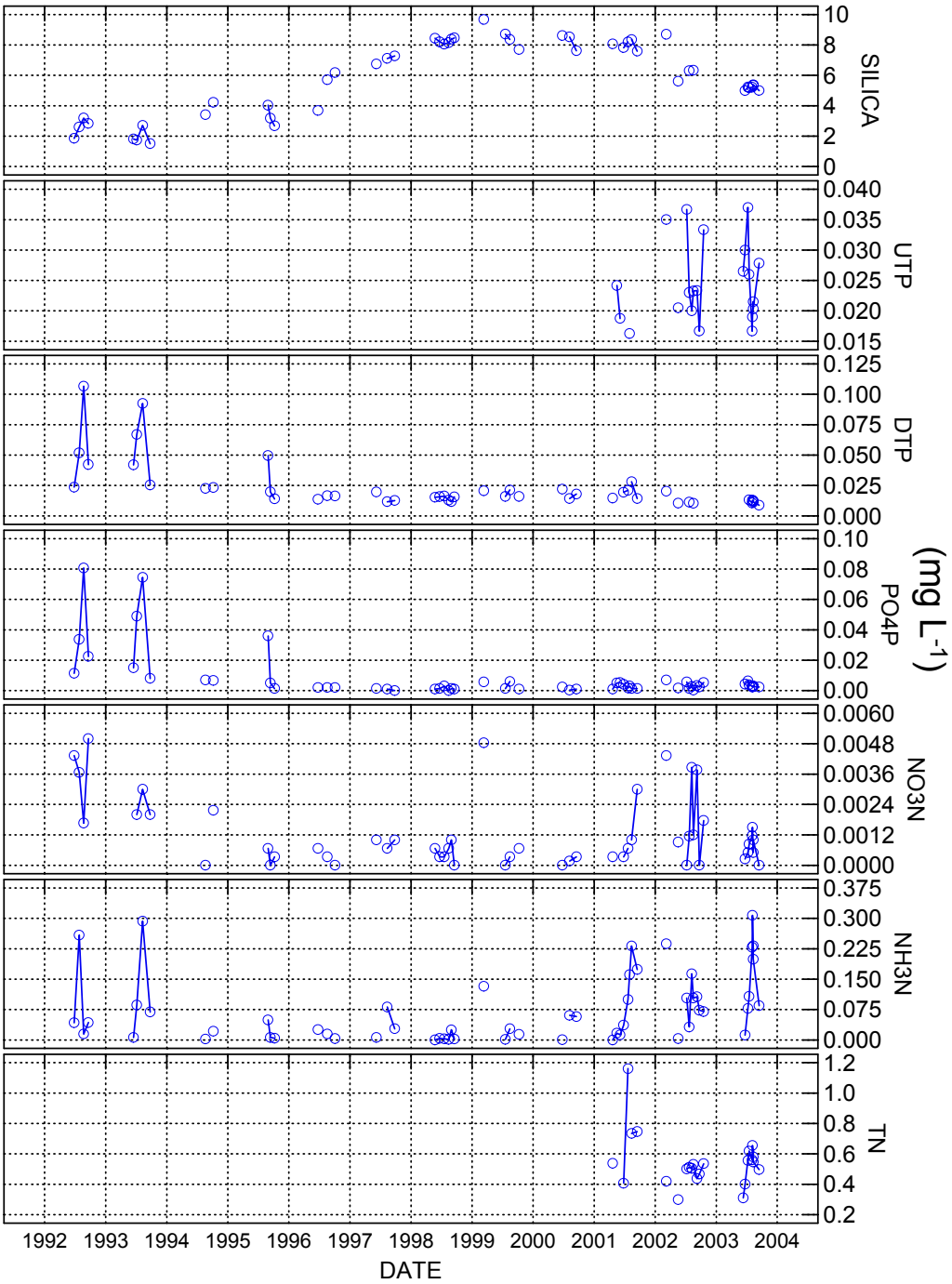


Figure 24. Time series of depth-averaged nutrient parameters at Diamond Lake Station DLA, 1992-2003. UTP = unfiltered total phosphorus (TP); DTP = dissolved (filtered) TP; PO4-P = phosphate-P; NO3-N = nitrate-nitrite nitrogen; NH3-N = ammonia nitrogen; TN = total nitrogen.

bloom development (Figure 24). This increase in ammonia-N translates to an increased ratio of inorganic nitrogen (TIN) to PO_4-P in July and August of all three years and September in 2001 (Figure 25). These ratios are high and generally not in the range (<5:1) that would favor nitrogen-fixing cyanobacteria. However, as mentioned above, cyanobacteria in July tend to be concentrated near the surface and the water-column averages in Figure 21 may not reflect actual availability of ammonia. A plot of ammonia at individual depths in 2001 (the year of greatest cyanobacteria biomass) does shows that surface levels of ammonia are low and that values increase with depth (Figure 25). Moreover, TIN:PO_4-P ratios at the surface for all three years are in the range expected to favor nitrogen-fixing cyanobacteria during June and July (Figure 26).

Given that both increased ammonia and dominance by cyanobacteria are both favored by stable water column conditions, an evaluation of stability was conducted by evaluating cumulative frequency plots of dissolved oxygen values taken from depth profiles collected between 1992 and 2003 (Figure 27). This plot shows that four of the years with high blue-green dominance, 1992 and 2001-2003 all had a greater proportion of profile dissolved oxygen concentrations at the lower end of the range. For example, for 2001, ~17% of the values were less than 4 mg L^{-1} dissolved oxygen; for 1992 and 2002 ~20% of the values were less than 4 mg L^{-1}; and for 2003 it was 25% (Figure 27). Because lower dissolved oxygen (DO) values occur both higher in the water column and increase in duration during periods of increased stratification, a greater frequency of low DO values indicates increased stability.

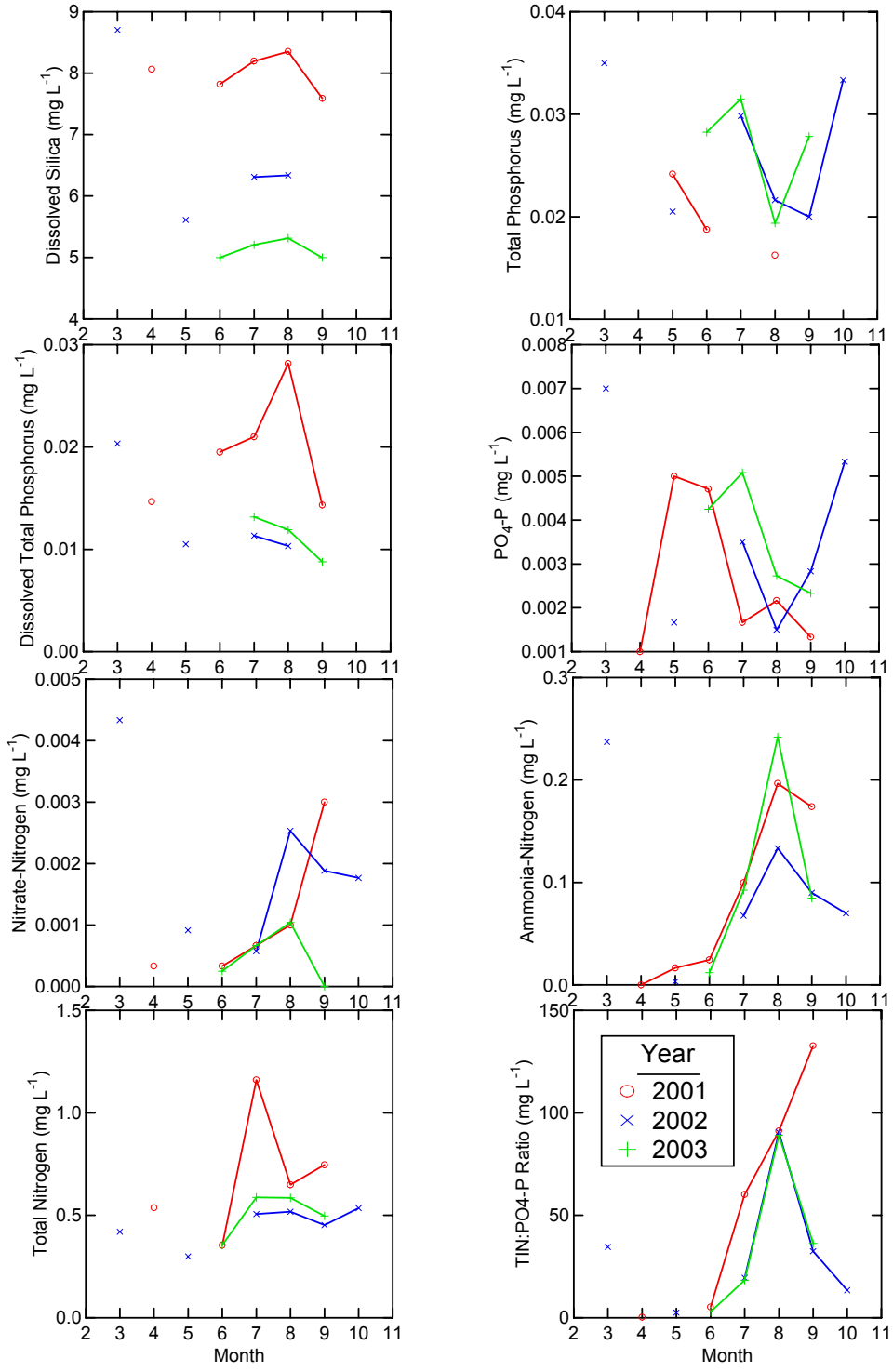


Figure 25. Monthly average for major nutrient parameters (depth-averaged) compared for years 2001-2003, Diamond Lake station DLA.

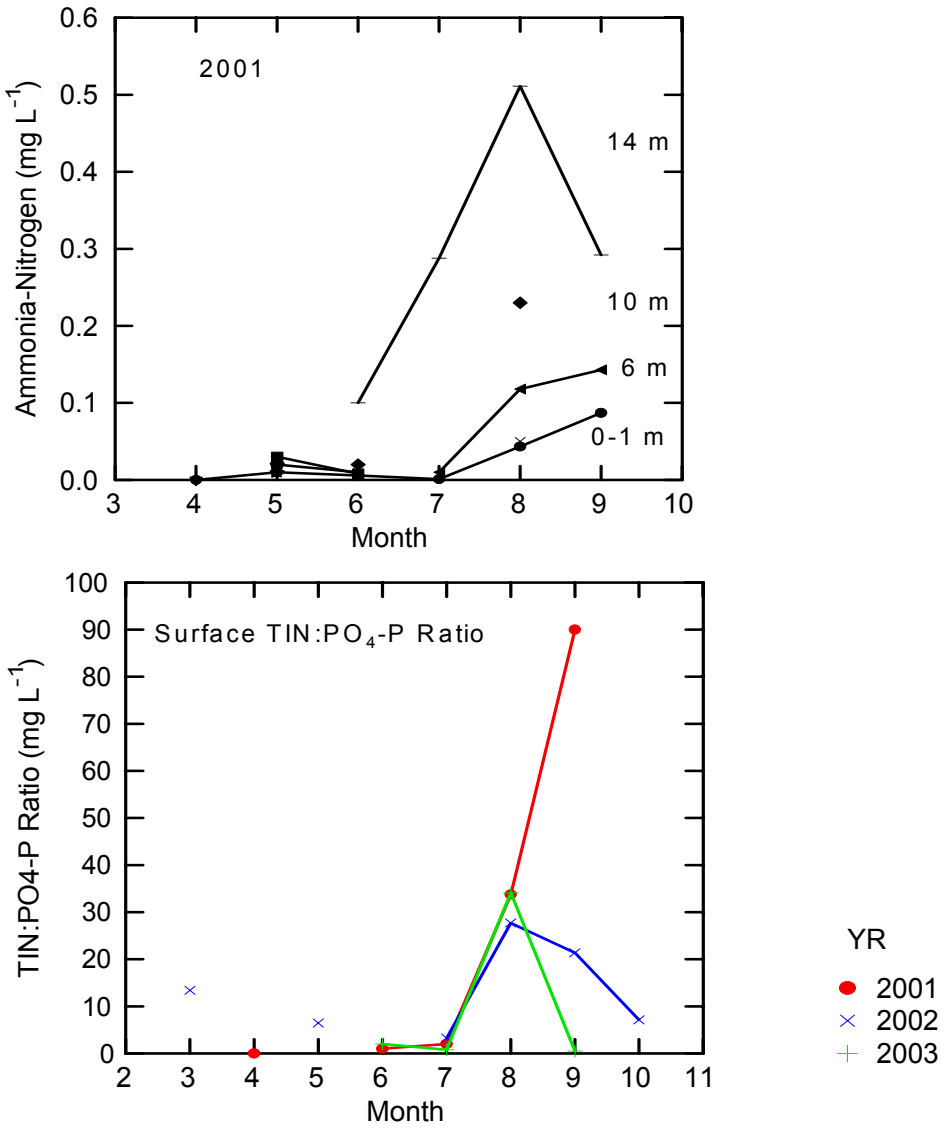


Figure 26. Monthly depth distribution of ammonia-nitrogen at Diamond Lake station DLA (the primary deep site) for 2001, and total inorganic nitrogen (TIN) to phosphate-P (PO_4-P) ratio at DLA surface (0-1 m) for 2001-2003.

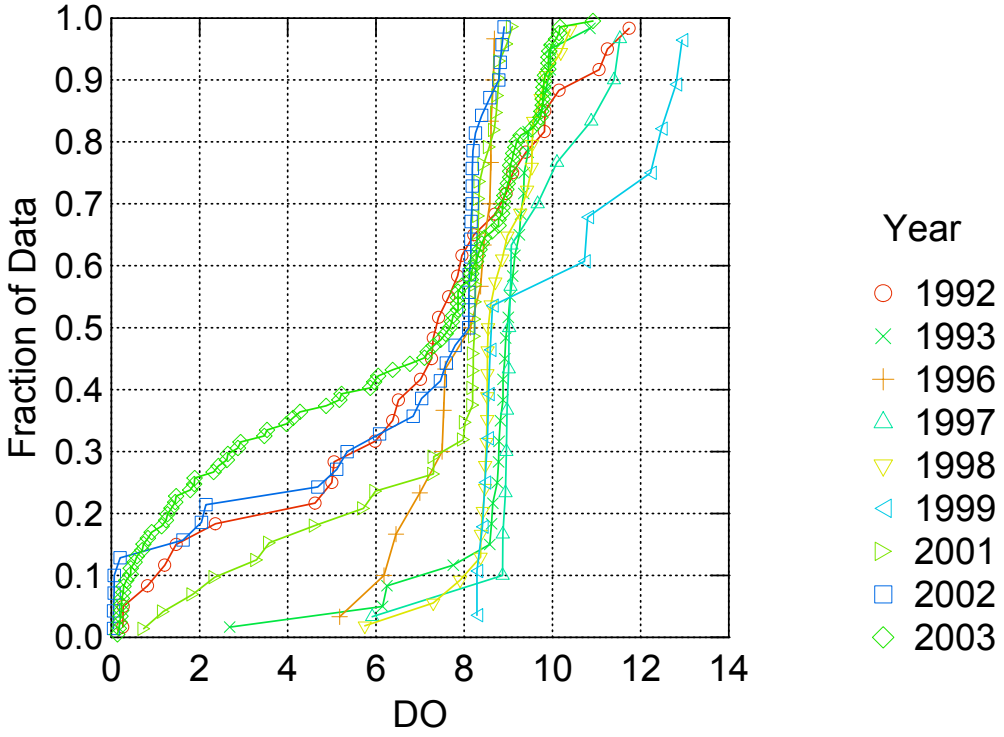


Figure 27. June-July quantile plot (cumulative frequency) of dissolved oxygen (DO; mg L^{-1}) values from depth profiles taken and Diamond Lake station DLA, 1992-2003 (note: samples not available for these months in all years).

In summary, the toxic cyanobacterial algal blooms that occurred in Diamond Lake during the summers of 2001-2003 are likely the result of optimal climatic conditions in conjunction with trophic changes stemming from large populations of non-native tui chub. Two possible pathways by which the currently large populations of tui chub can enhance the blooms are: (1) by reducing the number of larger sized zooplankton that can efficiently filter algal cells from the water column, and (2) by increasing the water column nutrient concentration through excretion of nitrogen and phosphorus in forms available for algal growth.

Predicting the effect of tui chub removal on the magnitude and annual trends of toxic algal blooms in Diamond Lake is difficult due to uncertainty in such factors as inter-annual climatic variability, restocking of the lake with rainbow trout, internal nutrient recycling from the sediments, and response of the zooplankton and benthic communities. Nonetheless, it is clear that based upon paleolimnological data that the lake began changing shortly after being stocked with trout, and that the greatest increases in *Anabaena* were associated with increased populations of tui chub (Figure 15). It is likely then that under current conditions of extremely high numbers of tui chub, that toxic *Anabaena* blooms (and subsequent lake closures) as well as high biomass of other taxa, will continue annually, with the severity determined largely by inter-annual variability in climate. In other words, under current high nutrient excretion rates by tui chub, the main determinant of inter-annual bloom variability will be the recurrence interval of calm, sunny, and warm conditions that tend to favor blooms of cyanobacteria. However, under a given set of climatic conditions the likelihood of large cyanobacterial blooms would be diminished (although periodic blooms can still be expected) as available nutrients decrease in the water column with chub removal.

INTERNAL LOADING AND THE NEED FOR A BIOLOGICAL MODEL

Internal loading represents nutrients that are recycled within the lake. This source of nutrients has been recognized for a considerable period in association with anoxic hypolimnion (Nurnberg 1984). The model of internal loading by Nurnberg was first developed to better explain discrepancies in mass-balance models (eg. Chapra and Reckhow 1979). It is now recognized that the cycling of nutrients in lakes can be strongly affected by the biota through a process described by some as trophic cascade (Carpenter et al. 1985), although earlier recognized by Shapiro et al.(1975) in biomanipulation experiments. In this approach, it was postulated that nutrient cycles in some lakes could be greatly altered by manipulating the composition of the top predators. The predators would alter the composition of the zooplankton and/or benthos, which in turn would affect the composition of the primary producers. Considerable research has been conducted in the last two decades on this topic and a better understanding is emerging regarding the role that fish composition exerts on water quality in lakes, especially those lakes that were formerly fishless (Schindler and Eby 1997; Schindler et al. 2001; Pilliod and Peterson 2001; Parker et al. 2001). Thus, models that rely on assumptions of steady-state mass-balance, such as the input/output analysis shown in Table 11 can grossly underestimate the actual loading to the system for some lakes. These types of lakes are generally those with long hydraulic residence times, productive, and deep enough to stratify.

There are two aspects of Table 11 that are striking. One is the similarity of the N and P loads to Diamond Lake. The mass ratio of N:P in the inputs is 1.7, which would promote conditions of strong N-limitation (cf Chapra 1997). The second aspect of Table 11 is that the anthropogenic load is small relative to the natural loading of nutrients. Both of these characteristics appear inconsistent with the current conditions in the lake which shows that, at least during certain periods, the lake is P-limited (Miller et al. 1974; other unpublished bioassays by Shiroyama in 1972 and 1973 reported N-limitation, as did an EPA study in 1975[US EPA 1978]). In addition, the lake is highly eutrophic and has become more eutrophic in the absence of substantial anthropogenic loads of nutrients (Eilers 2003b). Both of these apparent discrepancies can be explained by internal processes in Diamond Lake.

A first step to understanding the internal loads to Diamond Lake is to examine the depletion of oxygen in the hypolimnion. Detailed oxygen profiles on Diamond Lake were first collected by Sanville and Powers (1973) in 1971 and were continued through 1977 by Lauer et al. (1979). These types of measurements were resumed in 1992 (Salinas and Larson 1995). By examining the maximum height of anoxia in the water column (defined as DO < 1 mg/L), we were able to estimate the area of the lake substrate that was exposed to anoxic conditions and express that as an aerially-weighted basis by the duration of the anoxia (Figure 28). The results indicate that the degree of anoxia in Diamond Lake increased considerably starting in 2000.

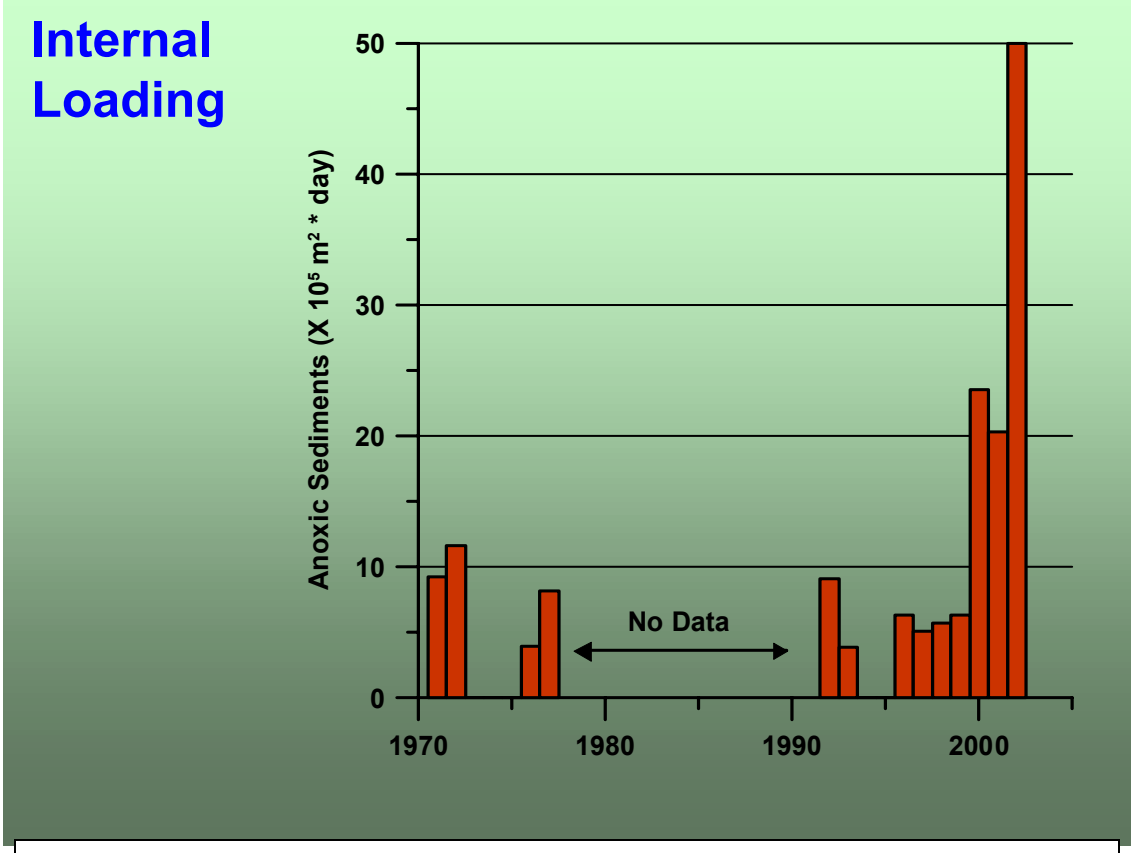


Figure 28. The duration and aerial extent of anoxia in Diamond Lake during the summer months based on the area of sediments/ per day exposed to anoxia from 1971-1977 and 1992-2002.

To assess the effects of anoxia and biological modifications on nutrient cycling in Diamond Lake, we reconstructed nutrient budgets for the lake for three periods: 1900 (prior to the introduction of fish), 1975, during the productive trout fishery, and 2000, during the dominance of the tui chub. By using the available water chemistry and paleolimnological data, it is possible to estimate the degree of internal loading of N and P in the lake (Table 16). The results indicate that internal loading of both N and P greatly exceed the external loads of nutrients. Thus, all of the external anthropogenic loads of nutrients do not approach the loads currently generated within the lake. Consequently, to meet water quality standards, it is necessary to address the internal loads of nutrients and the causes of the increases in internal loads.

Table 16. Measured and estimated annual inputs and outputs of TP and TN for Diamond Lake for circa 1900 (before fish), 1975, and 2000. The color indicate the degree of uncertainty associated with the estimates where green is very low, blue is low, yellow is moderate, gold is high , and red is very high.

INputs		Date	Flow[a]	TP conc[b]	TP Load[c]	TN conc[b]	TN Load[c]	N:P[c]
	Silent Creek		21.432	69	1479	31.8	682	0.5
	Short Creek		6.251	72	450	56.3	352	0.8
	Other Tribs		1.57	69	108	31.8	50	0.5
	Precipitation		16.258	10	163	190	3089	19.0
	Groundwater		5.389	69	372	31.8	171	0.5
	Fish Stocking	1900		0	0	0	0	
		1975		1.68%[d]	18	3%[d]	27	1.5
		2000		1.68%	43	3%	77	1.8
	Fish Bait	1900			0		0	
		1975		0.77%[d]	70	2.80%[d]	254	3.6
		2000		0.77%	10	2.80%	36	3.6
	Septic	1900			0		0	
		1975		50%[e]	23	50%[e]	228	9.9
		2000		50%	23	50%	228	9.9
	Internal	1900			0		0	
		1975			1556[f]		11342[f]	7.3
		2000			3202[f]		24588[f]	7.7
	TOTAL	1900			2572		4344	1.7
		1975			4239		15514	3.7
		2000			5867		28266	4.8
OUTputs								
	Lake Creek	1900	41.261	35[g]	1444	180[g]	7427	5.1
		1975	41.261	30[h]	1238	280[i]	11553	9.3
		2000	41.261	25[j]	1032	380[j]	15679	15.2
	Evaporation		6.909					
	Groundwater[k]	1900	2.73	35	96	180	491	5.1
		1975	2.73	30	82	280	764	9.3
		2000	2.73	25	68	380	1037	15.2
	Fishing	1900		0	0		0	
		1975			430		768	1.8
		2000			22		39	1.8
	Sediment	1900	0.015	0.04%[l]	490	0.5%[l]	6130	12.5
		1975	0.022	0.12%[l]	2158	1.0%[l]	17981	8.3
		2000	0.035	0.19%[l]	5435	1.5%[l]	42910	7.9
	TOTAL	1900			2030		14048	6.9
		1975			3907		31067	8.0
		2000			6557		59666	9.1

[a] Flows from Table 10
[b] Concentrations from Table 10 (ug/L)
[c] Loads in Kg/yr; N:P ratio based on mass
[d] From Lauer et al. (1979)
[e] Midpoint of ranges shown in Table 12
[f] Computed as the difference in the phosphorus accumulated in the sediments (Eilers et al. 2001a) less the anthropogenic inputs. Lauer et al. (1979) estimated an average of 2043 kg/yr and 1156 kg/yr for 1975.
[g] Estimated based on lower rates of sediment accumulation of P (Eilers et al. 2001a)
[h] From Lauer et al. (1979)
[I] Modified from Lauer et al. (1979); analytical measurement of organic nitrogen was problematic
[j] From Salinas (1992-2002) and Eilers (2003b)
[k] Assuming groundwater exists near the lake surface.
[l] After Eilers et al. (2001a)

By extracting selected information from Table 16, it becomes evident that internal loading of nitrogen has been increasing faster than internal loading of phosphorus, resulting in an absolute decrease in the export of P from Diamond Lake and a large increase in N export through Lake Creek (Figure 29). The consequence of these alterations in the flux of nutrients is that the ratio of N:P in the outflow has nearly tripled in the 20th century. Prior to the introduction of fish, it appears that the N:P ratio was still low enough for N to be limiting. Under the current conditions, the export of N has increased so dramatically that the water exported downstream is now P-limited.

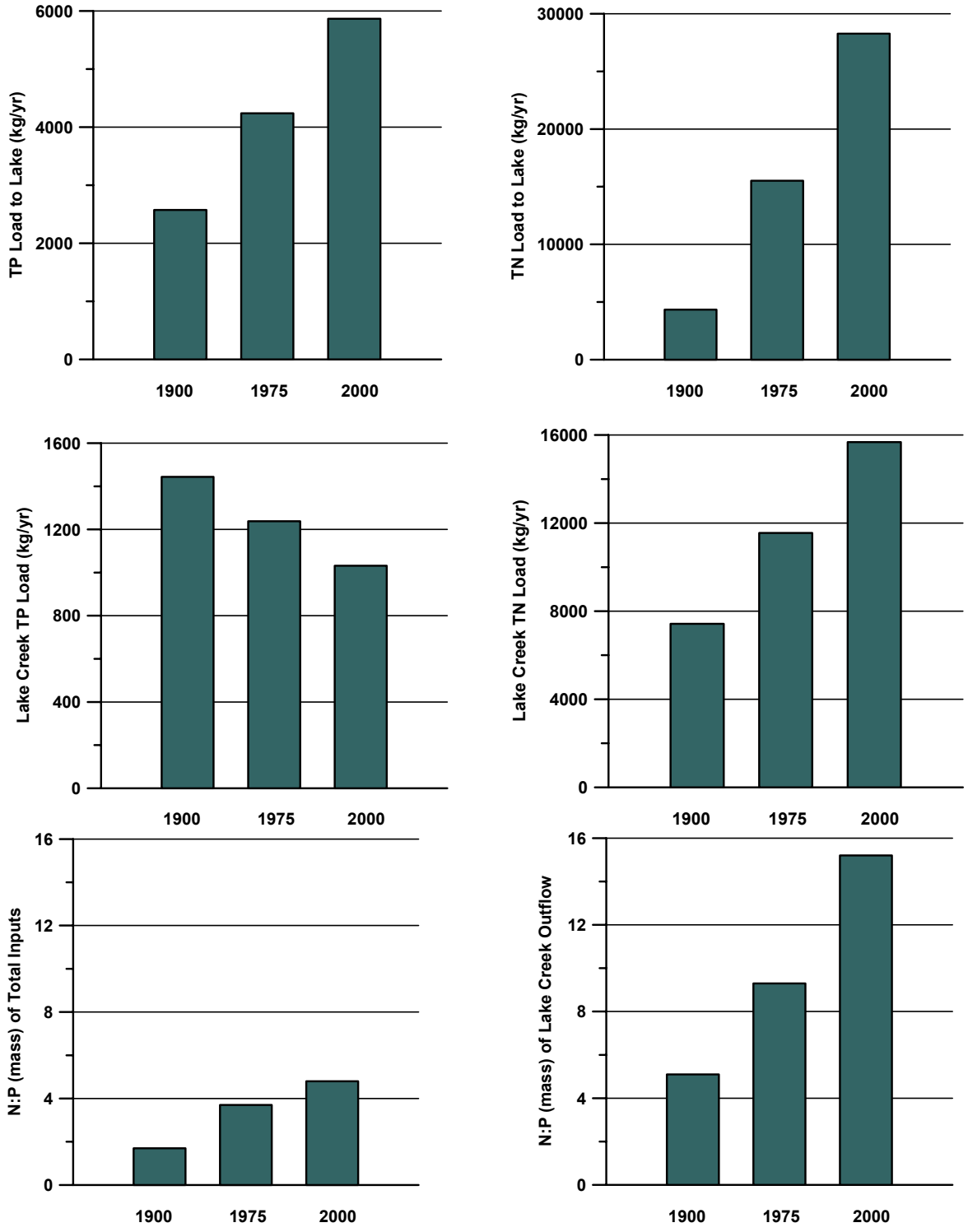


Figure 29. Changes in nutrient loads and N:P ratios for total inputs and outputs from Lake Creek. The top row shows the changes in the inputs of TP (left) and TN (right) for Diamond Lake from circa 1900, 1975, and 2000. The middle row shows the export of TP (left) and TN (right) from Lake Creek. The bottom row shows the changes in N:P ratios for total inputs (left) and Lake Creek outputs (right).

THE DEVELOPMENT AND APPLICATION OF THE FIN-S MODEL

The empirical biological model (FIN-S) was evaluated for sensitivity to a number of assumptions regarding the response of the chub population and the magnitude of the cyanobacterial response by varying the parameters through a broad range at five different levels. The first assumption that was test was the duration of a year required to complete the chub spawning and hatching. The period of time in which the chub are actively hatching was varied from 0.1 to 1 year in length (Figure 30). The smoothed orange line in the left figure represents juvenile chub being produced year round. While this is not a realistic situation (chub produce eggs only in the summer), it produces very similar results while maintaining a readable graph. For this reason all of the subsequent sensitivity analyses were performed for a chub population reproducing at a constant rate throughout the year.

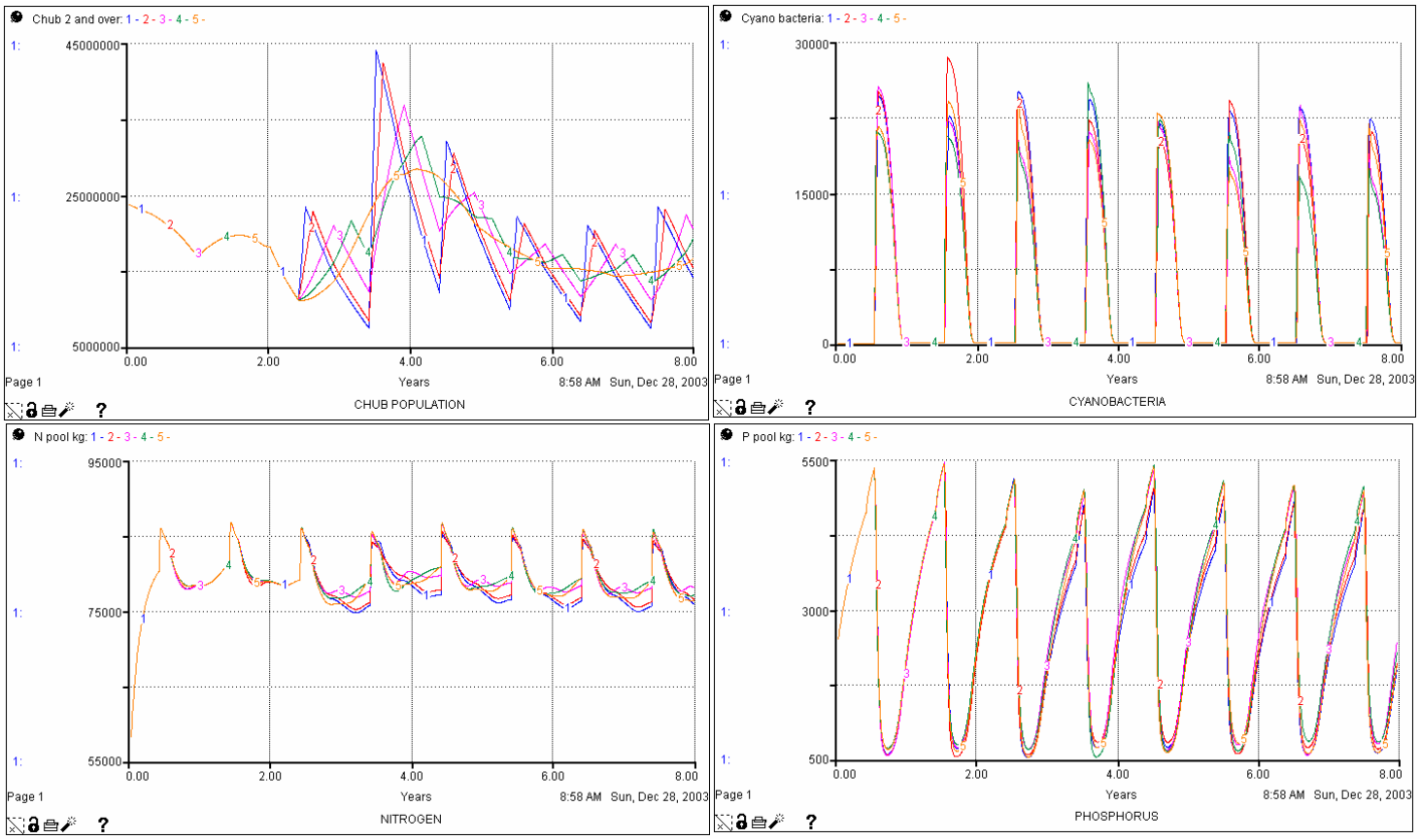


Figure 30. FIN-S model output for the response of the tui chub population (top left) and cyanobacterial response (top right) to assumptions in duration of chub egg hatching as a fraction of a year (1= 0.1, 2=0.2, 3= 0.5, 4= 0.75, 5 = 1). Total nitrogen (bottom left) and total phosphorus loads are shown in the lower left and lower right respectively.

There is very limited literature on the reproduction rates of tui chub, and the only available number for survivorship from egg to one year old was suggested to be 0.1% (0.001) (Bird, 1975). Based on this survivability and mortality rates (Bird, 1975), it was not possible to sustain or even achieve the population estimate of 23 million chub age 2 and older. The egg survivorship was tested for sensitivity (Figure 31). Scenario 1 also exhibits a sharp decline in cyanobacteria blooms. A relatively stable chub population was desired for sensitivity modeling and an egg survivorship value of 0.004 was used for all other model runs. To evaluate the chub population sensitivity to the amount of food available, the total food variable was varied as a percentage of the normal food total (Figure 32). As the percentage of food available to the chub decreases, the mortality rates of the chub rise, causing a sharp decrease in the population. This in turn leads to less nutrients being cycled causing a decrease in the amount of cyanobacteria present during the summer.

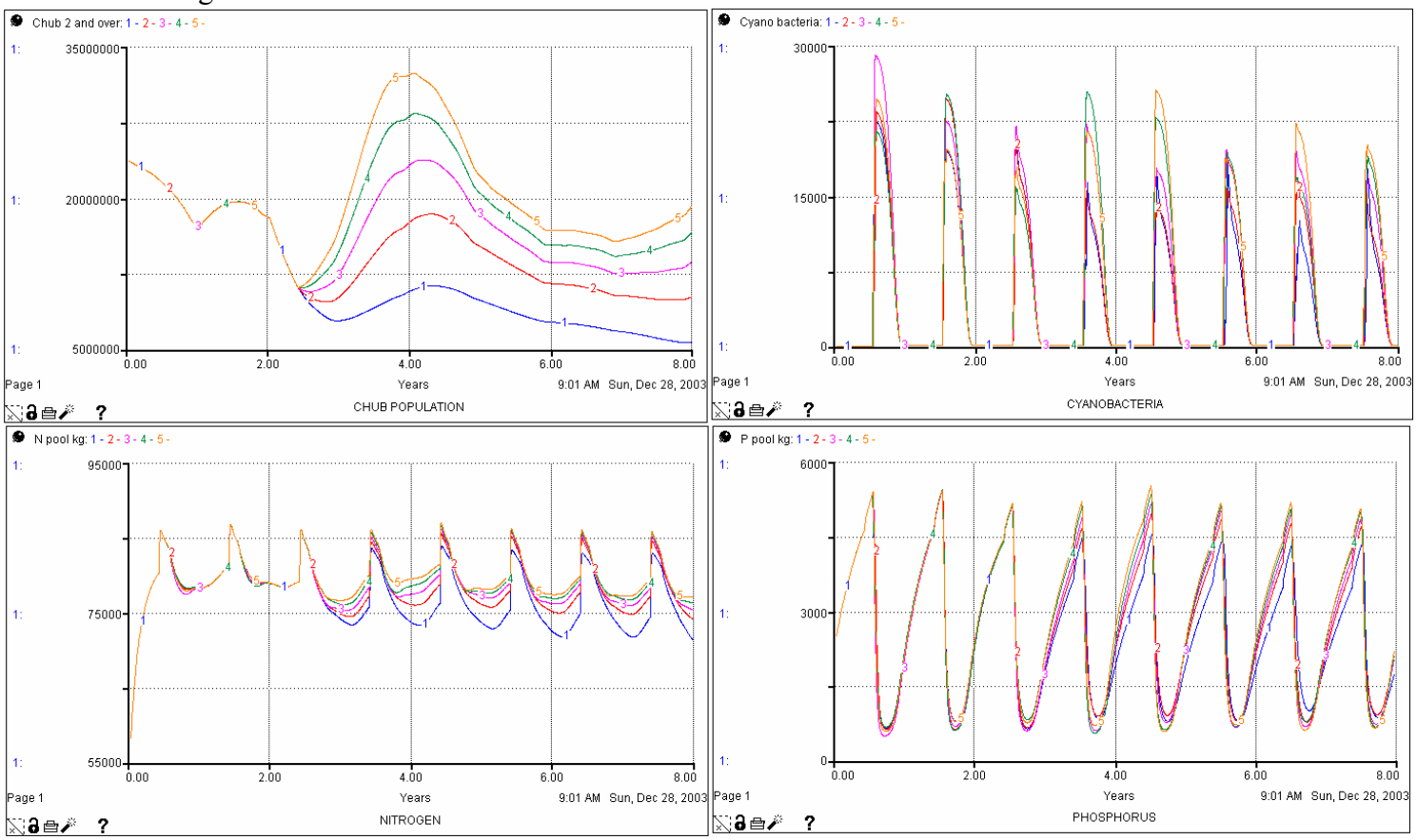


Figure 31. FIN-S model output for the response of the tui chub population (upper left) and cyanobacterial response (upper right) to assumptions in chub egg survival as a fraction of eggs laid (1= 0.001, 2=0.002, 3= 0.003, 4= 0.004, 5 = 0.005). Total nitrogen and total phosphorus loads are shown in the lower left and lower right respectively.

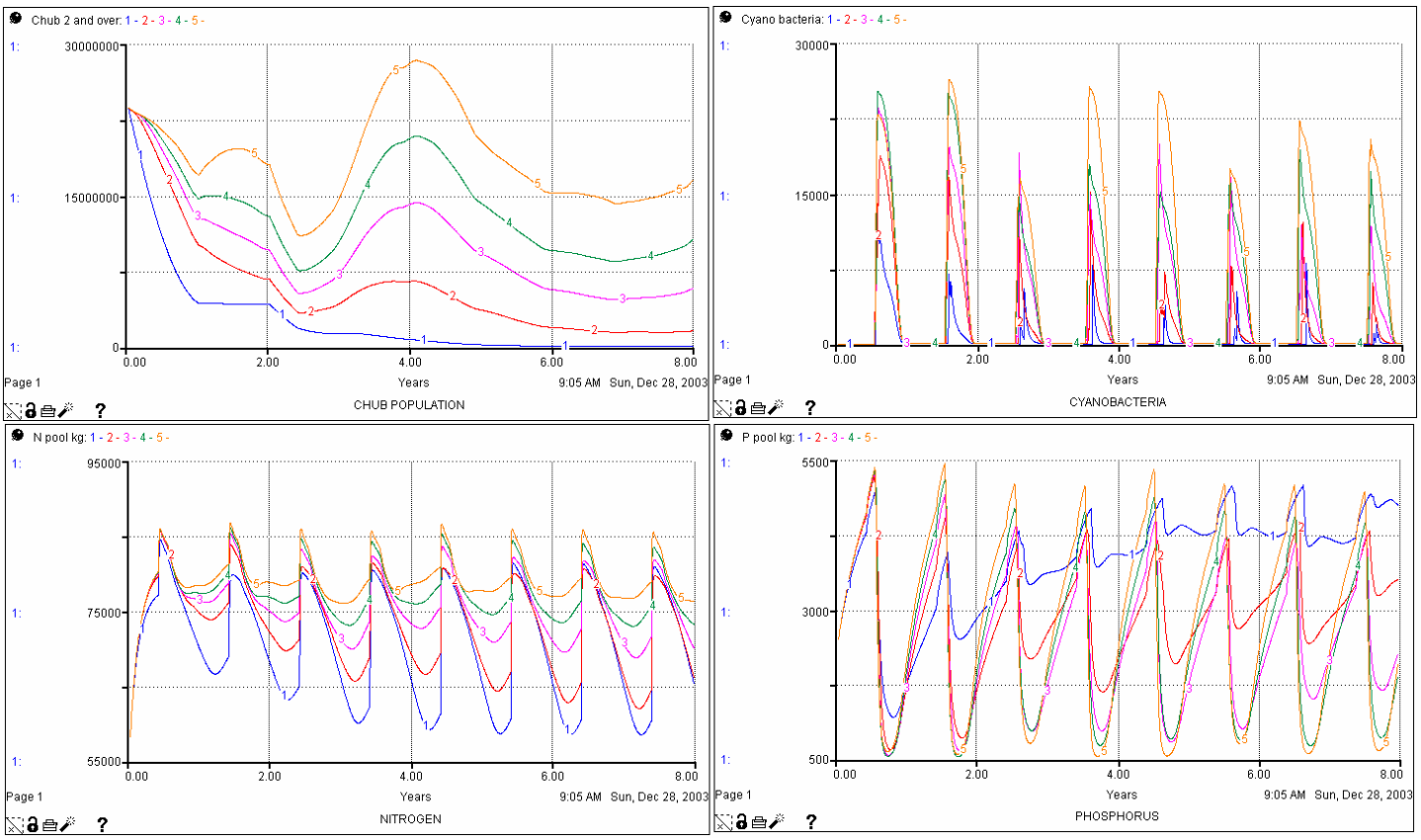


Figure 32. FIN-S model output for the response of the tui chub population (upper left) and cyanobacterial response (upper right) to assumptions in available zooplankton and benthic food sources for the chub, represented as a percentage of total food (1= 10, 2= 30, 3= 50, 4= 70, 5 = 100). Total nitrogen and total phosphorus loads are shown in the lower left and lower right respectively.

As a test of dependency on different food sources in the lake, the production rates of non-rotifers, amphipods, and chironomids was varied (Figures 33, 34, 35). The three previous food production figures were all run on the same specifications for the different runs; 1) 10%, 2) 50%, 3) 100%, 4) 200%, and 5) 500% production rate of normal. In all three figures the population decreases with a decrease in food production, with the most sensitive group being the amphipods (Figure 34). Chironomids are normally not the preferred food of chub or trout (when amphipods are present), but from the start, the system is operating at the limits of capacity and the chub population is experiencing food limitations resulting in consumption and limitation by the chironomid population.

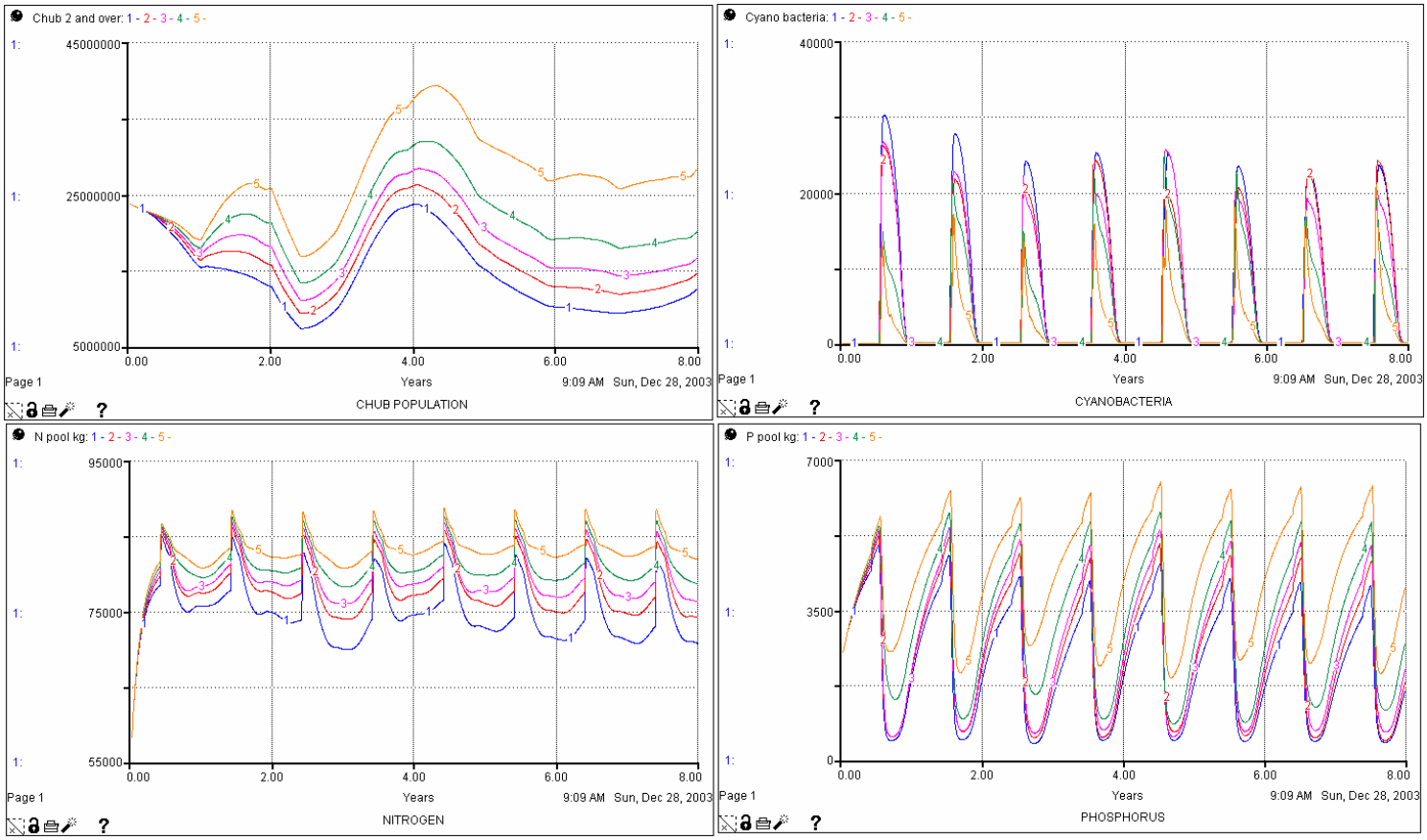


Figure 33. FIN-S model output for the response of the tui chub population (upper left) and cyanobacterial response (upper right) to assumptions in the percentage of non-rotifers (larger cladocerans and copepods) present in the zooplankton population as a percentage of current conditions (1= 10, 2= 50, 3= 100, 4= 200, 5 = 500). Total nitrogen and total phosphorus loads are shown in the lower left and lower right, respectively.

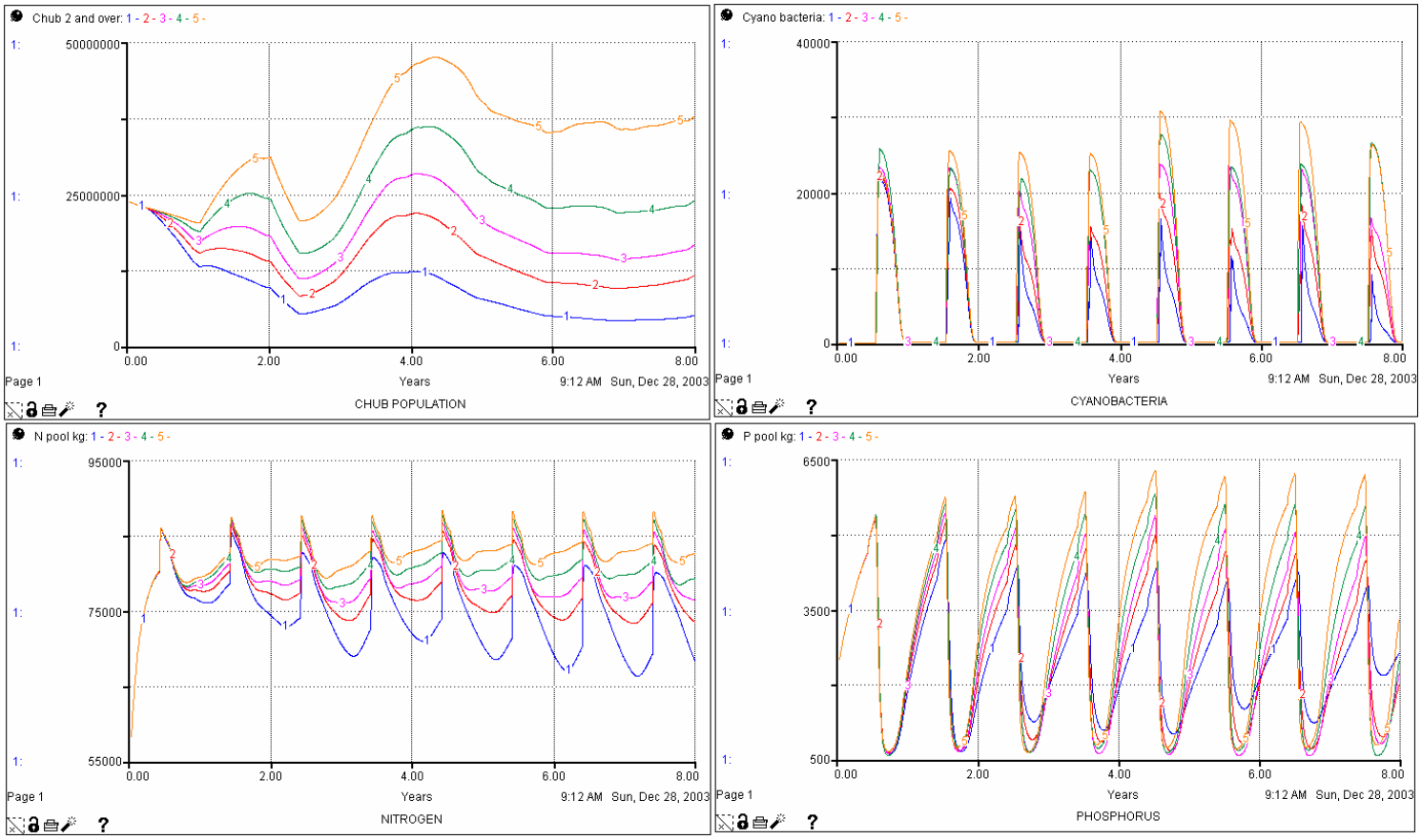


Figure 34. FIN-S model output for the response of the tui chub population (upper left) and cyanobacterial response (upper right) to assumptions in the percentage of amphipods present in the benthic population as a percentage of current conditions (1= 10, 2= 50, 3= 100, 4= 200, 5 = 500). Total nitrogen and total phosphorus loads are shown in the lower left and lower right, respectively.

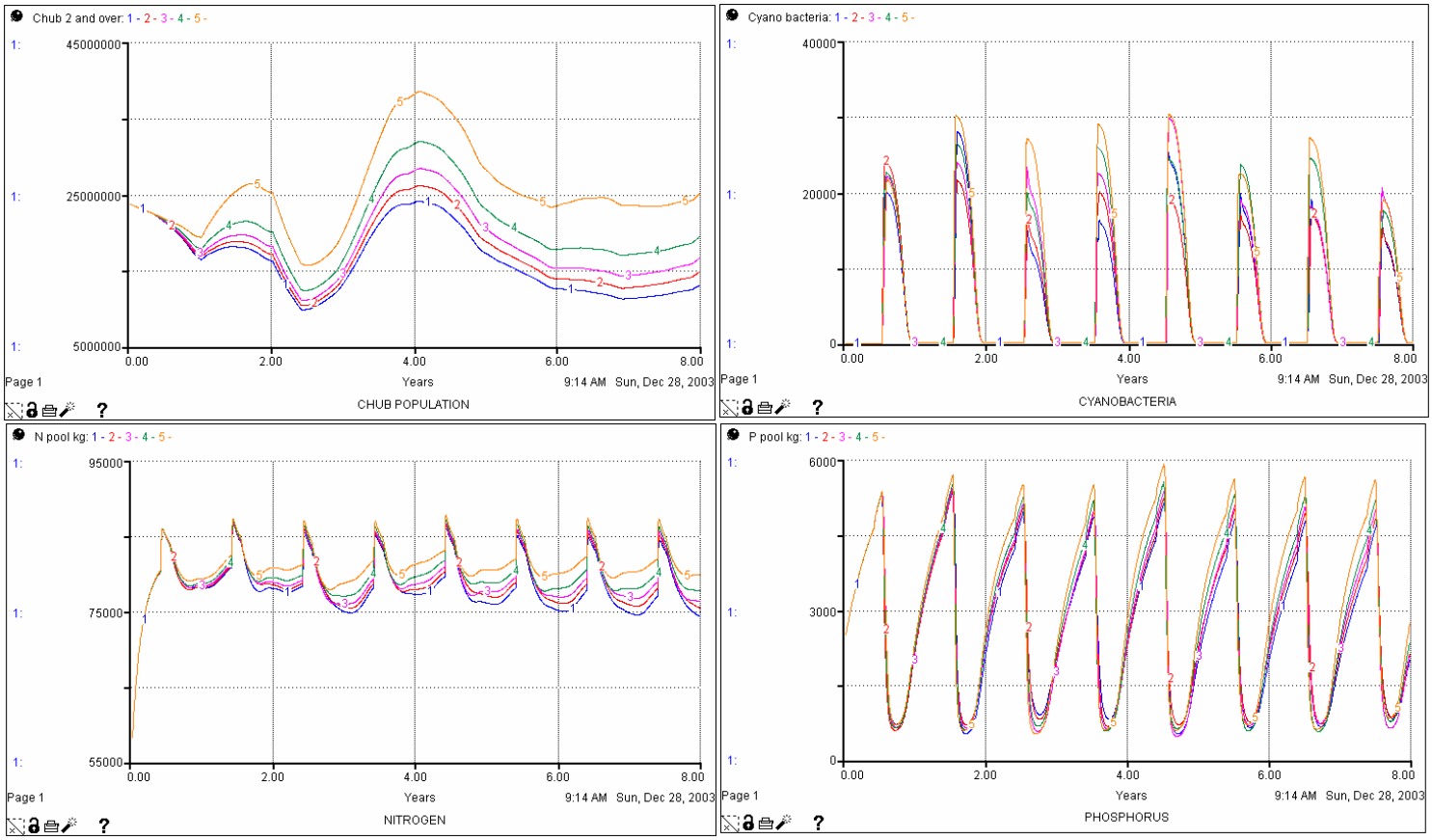


Figure 35. FIN-S model output for the response of the tui chub population (upper left) and cyanobacterial response (upper right) to assumptions in the percentage of chironomids present in the benthic population as a percentage of current conditions (1= 10, 2= 50, 3= 100, 4= 200, 5 = 500). Total nitrogen and total phosphorus loads are shown in the lower left and lower right, respectively.

Size of stocked trout and number of trout stocked was also varied to determine the extent of influence on the chub population (Figures 36 and 37). Neither parameter had much influence on the chub population or the cyanobacteria. Because there is believed to be very little predation of trout on the chub, this was not incorporated into the model. For this reason the only effect of increasing the overall trout biomass in the system was to add increased food pressure on the chub population.

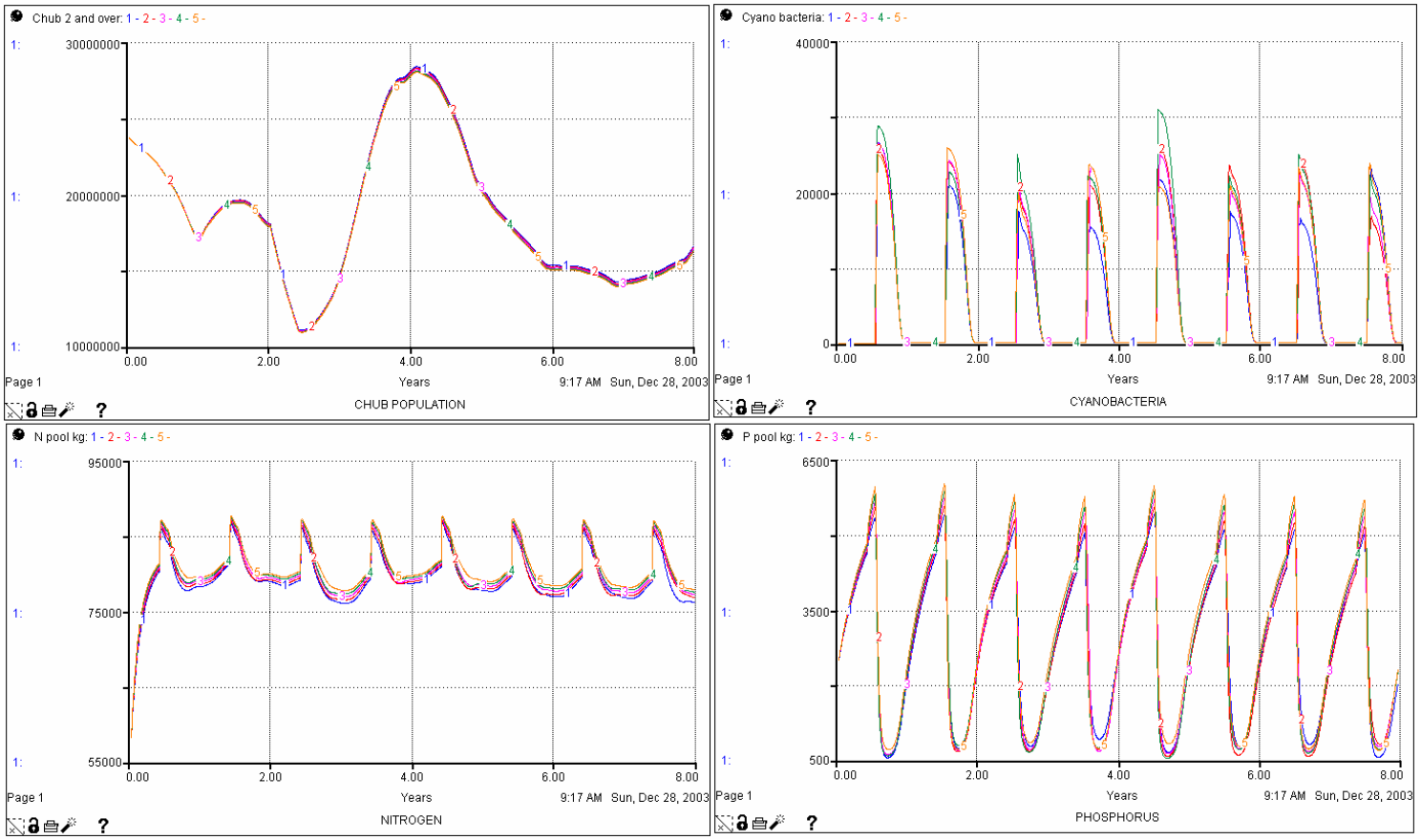


Figure 36. FIN-S model output showing the response of the tui chub population (upper left) and cyanobacterial response (upper right) to assumptions in the size of the stocked trout, expressed in kg/fish (1= 0.8, 2= 1.3, 3= 1.8, 4= 2.3, 5 = 2.8). Total nitrogen and total phosphorus loads are shown in the lower left and lower right, respectively.

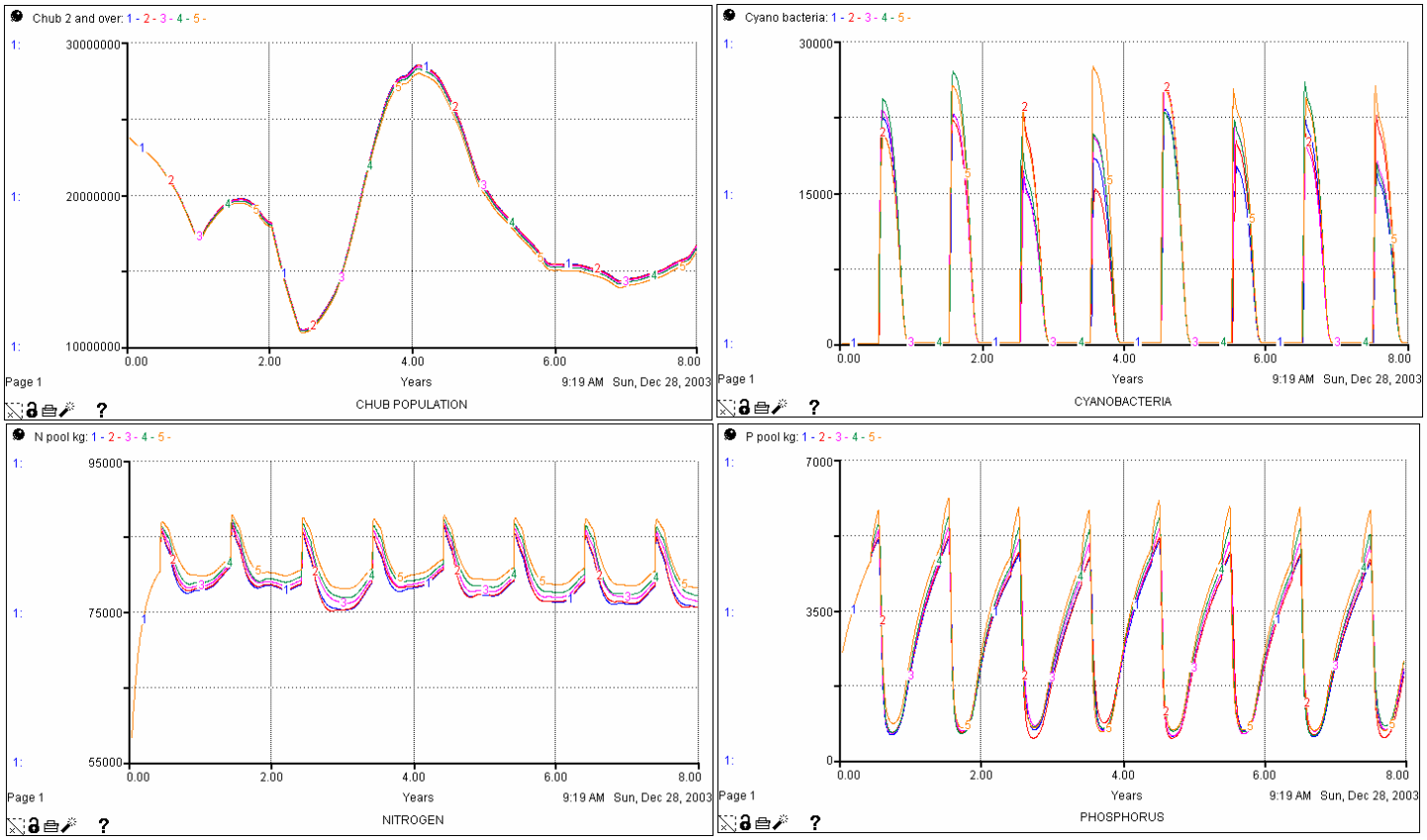


Figure 37. The response of the tui chub population (upper left) and cyanobacterial response (upper right) to assumptions in the number of the stocked trout, using a constant size trout of 0.8 kg/fish (1= 0, 2= 10,000, 3= 50,000, 4= 100,000, 5 = 200,000). Total nitrogen and total phosphorus loads are shown in the lower left and lower right, respectively.

The population dynamics of the chub were further tested by varying the initial chub population (Figure 38). A starting chub population higher than equilibrium is quickly killed off, while a low initial value takes many years to build to a sizeable population. In an effort to further explore the rate at which the chub population could build, a 60-year simulation was run with low initial chub populations (Figure 39).

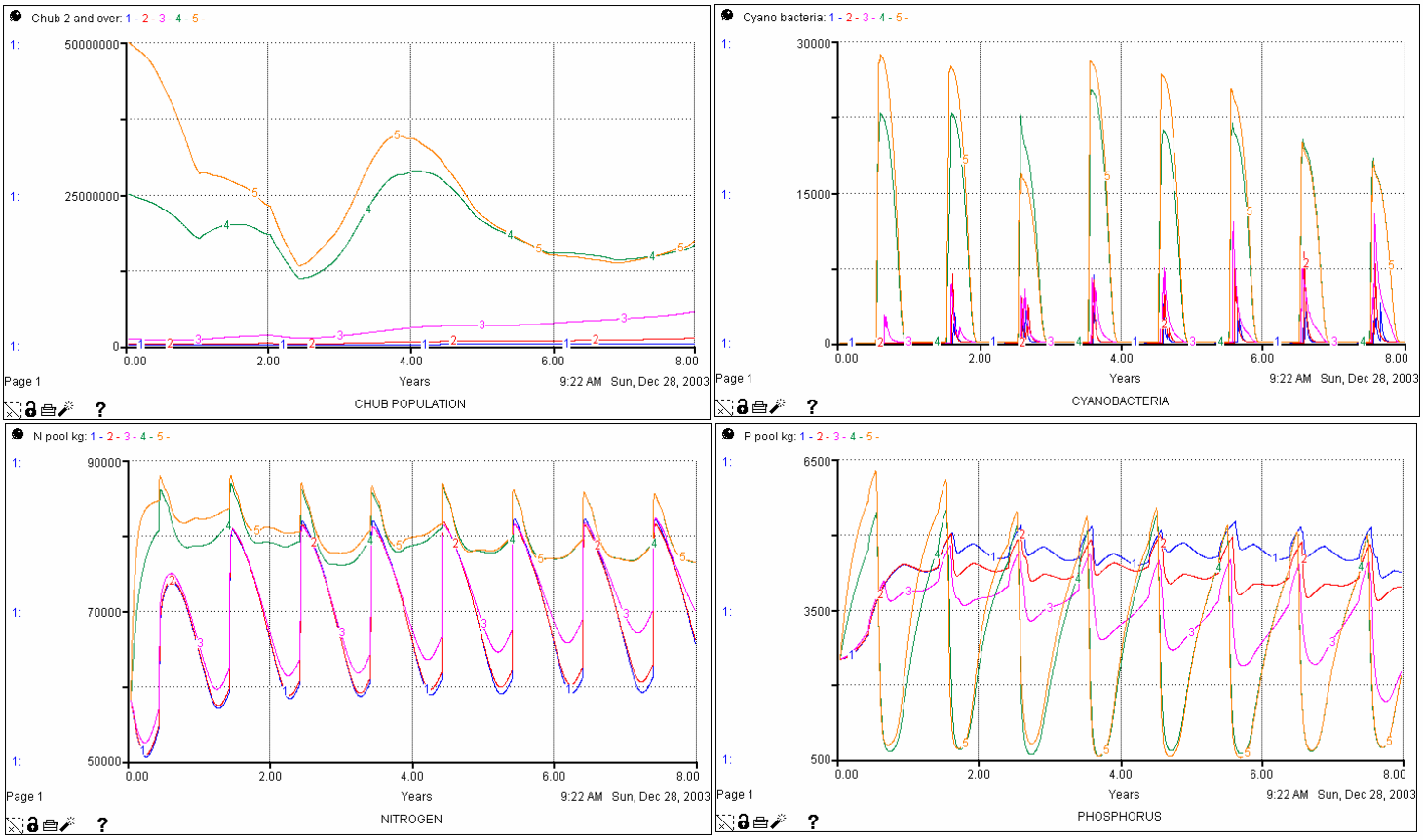


Figure 38. The response of the tui chub population (upper left) and cyanobacterial response (upper right) to assumptions in the number of tui chub (2+ years old) currently present in the lake (1= 50,000, 2= 200,000, 3= 1,000,000, 4= 25,000,000, 5 = 50,000,000). Total nitrogen and total phosphorus loads are shown in the lower left and lower right, respectively.

For very low initial chub populations, it requires over 45 years to attain a population size of five percent of the equilibrium population size simulated for the tui chub (~ 18,000,000). The principle governing equation for growing a non-food limited chub population is the rate of chub mortality at each life stage. The only information available on mortality rates for tui chub was for an established population (Bird 1975). Although it could be suggested that the rotenone treatment in 1954 was not completely successful, it is likely that the rapidly expanding chub population experienced much lower mortality rates than an established population due to an abundance of food and minimal predation. Regardless of the initial seed population, the cyanobacteria production always follows the sharp rise in the chub population.

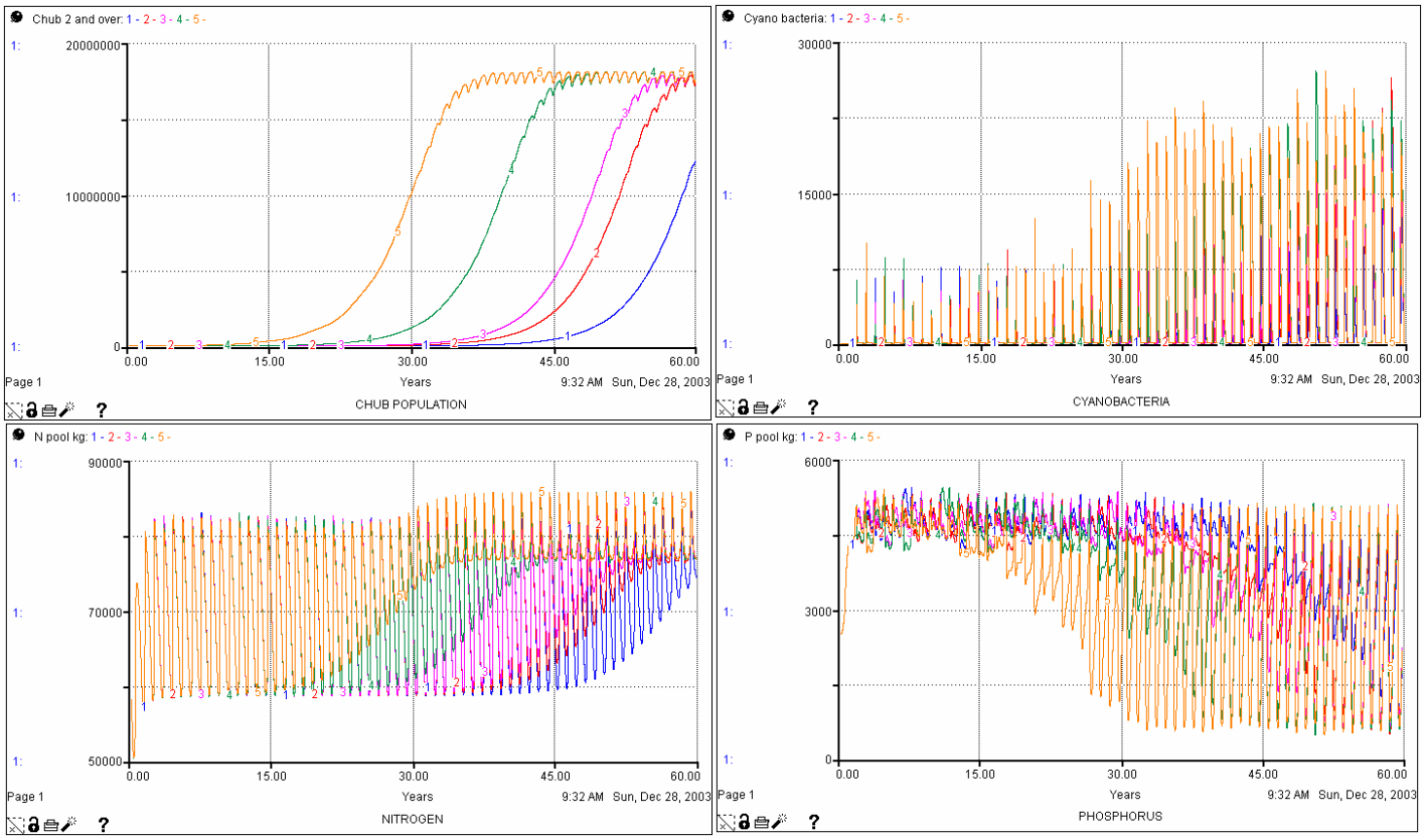


Figure 39. The response of the tui chub population (upper left) and cyanobacterial response (upper right) to assumptions in the number of tui chub (2+ years old) present initially (1= 10, 2= 50, 3= 100, 4= 1000, 5 = 10,000). Total nitrogen and total phosphorus loads are shown in the lower left and lower right, respectively.

TOTAL MAXIMUM DAILY LOAD

1. Fish-Based TMDL:

The Total Maximum Daily Load (TMDL) of phosphorus and nitrogen is designed to decrease the probability that water quality standards for pH and nuisance algae will not be exceeded for Diamond Lake. Because only 16 percent of the watershed load of phosphorus is anthropogenic and the internal load of TP is much greater than the external load, we have focused on the need to reduce the internal load. In our opinion, this can only be achieved in a realistic manner by removal of the tui chub and development of a modest trout-stocking program that is closely monitored for effects. The removal of the tui chub is expected to result in a reduction in the average peak biomass of cyanobacteria (average over 8 yrs) from about 20,000 kg to 4,000 kg. By reducing the internal load of phosphorus, we expect that this will further limit the growth of cyanobacteria, which in turn is expected to greatly reduce the internal load of nitrogen by reducing the amount of nitrogen fixed by the cyanobacteria. If the tui chub are completely removed, we anticipate that the frequency and intensity of cyanobacteria blooms will decline about 80 percent, although under favorable weather conditions some modest blooms will still occur (Figure 40).

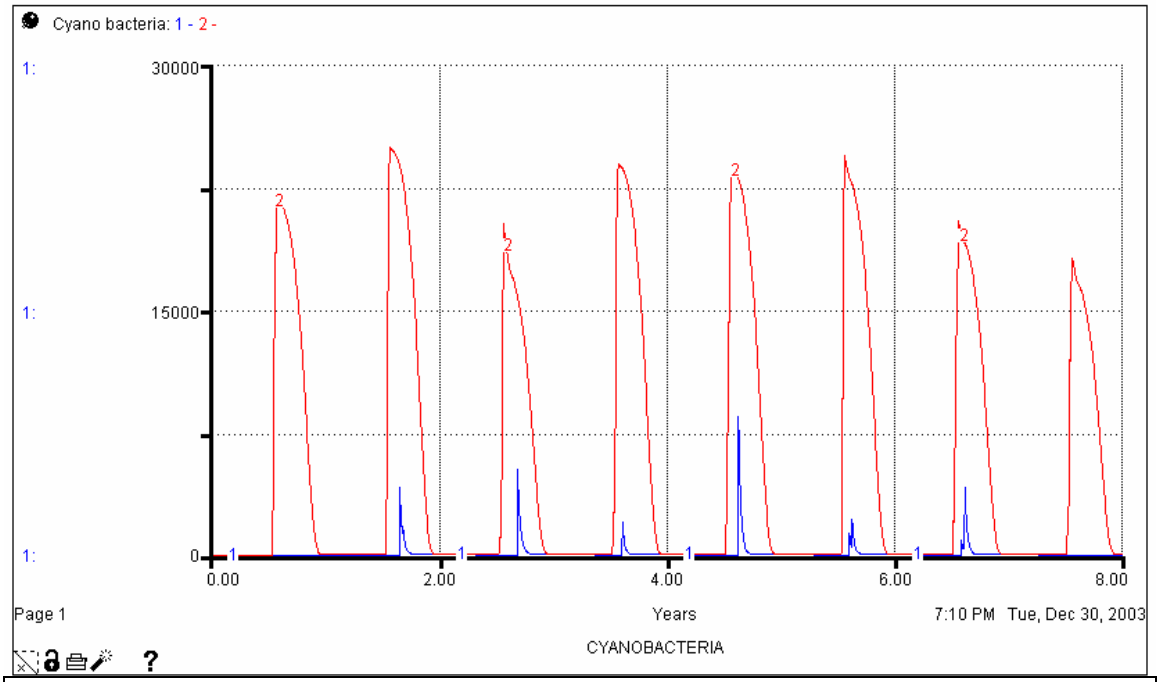


Figure 40. FIN-S model output displaying the biomass of cyanobacteria (kg) for the current population of tui chub (red) in Diamond Lake compared to a model simulation with no tui chub (blue).

Because of the uncertainty associated with the biological components in Diamond Lake, we recommend implementing a conservative stocking program (Table 17). The key to a successful effort will be to monitor key elements of the lake ecology to assess how the lake may be responding to the trout stocking. A monitoring strategy for evaluating the effects of a fish stocking program has been proposed by Eilers (2003c). The monitoring strategy includes components of lake chemistry, transparency, and phytoplankton, zooplankton, and zoobenthos.

Table 17. A possible trout stocking strategy for Diamond Lake that could be incrementally implemented.

Stocked Fingerlings	Number of Fish	Wet Wt (kg)	Wet Wt (g/fish)	Dry Wt (g/fish)	TP (%)	TN (%)	TP Load (kg)	TN Load (kg)
Stage 1	27,500	250	18	3.6	1.7	10	2	10
Stage 2	55,000	500	18	3.6	1.7	10	3	20
Stage 3	110,000	1000	18	3.6	1.7	10	7	40
Stage 4	220,000	2000	18	3.6	1.7	10	13	79
Stage 5	440,000	4000	18	3.6	1.7	10	27	158

2. <u>Uncertainty Analysis</u>:

The concept of Margin of Safety (MOS) was developed by EPA to acknowledge and quantify the uncertainties in the analytical approach for assigning pollutant loads under the TMDL. The concepts of MOS and uncertainty analysis have been controversial (cf Walker 2001) because of the uncertainties in the data used to develop the TMDL, the uncertainties in the biological components of a lake system, and our ability in quantifying these process under current and future conditions. These problems are particularly acute in Diamond Lake because of the paucity of data in the non-summer period, the unquantified inputs from waterfowl, the unknown inputs associated with the depressions in the lake (Eilers 2003d), the unquantified processes associated with the macrophytes, and the uncertainties in the interactions among the other biological communities. We suspect that there are substantial differences between processing of nutrients in the open water of Diamond Lake compared to those in the shallow areas covered by macrophytes. There are virtually no water and plankton samples from the shallow areas of the lake available to describe this habitat. This zone represents about one-half of the lake area. The spatial aspects of the nutrient processing are expected to be dramatic between the deep and shallow zones because of differences in utilization of habitat types by the trout and the chub. At this point, we are unsure what the partitioning of chub is between the deep and the shallow water and what area provides the major food source for the adults.

Thus, a formal analysis of uncertainty cannot be performed on the Diamond Lake TMDL. The best approach for dealing with this level of uncertainty is to manage the system conservatively and closely monitor the lake during any changes in management.

CONCLUSIONS

Multiple mathematical models were used to assess current, past, and possible future ecological and water quality conditions in Diamond Lake. The principal findings from these inquiries are summarized below.

- The development of the hydrologic model is in general agreement with previous investigations. The calculated hydraulic residence time for the 2003 water year is 1.65 years. Groundwater is a relatively small component of the inflows and outflows. The single largest hydraulic input is from Silent Creek.
- Diamond Lake is a major sink for phosphorus (with 92 percent retention for PO_4) and silicon (59 percent retention), but is a major exporter of nitrogen (6.4 fold increase), most of which is in organic form. The lake exports a large amount of nitrogen despite retaining 99 percent of the nitrate inputs.
- The phosphorus inputs to the lake are largely from natural sources; anthropogenic sources from the watershed represent only about 16 percent of the total phosphorus load to the lake. Nitrogen inputs to Diamond Lake from the watershed are only slightly greater than the phosphorus inputs, resulting in a system that would appear to be N-limited for periods of the year.
- Watershed inputs of N and P are currently much less than the internal load of nutrients and acceleration of nutrient cycling associated with loss of larger herbivorous zooplankton, loss of larger zoobenthos, persistent anoxia in the hypolimnion, excretion from fish, and nitrogen fixation by cyanobacteria.
- Most of these changes in the ecology and nutrient cycling of Diamond Lake appear to be the direct result of the large biomass of tui chub.
- Sensitivity analyses showed that, even if the survival of tui chub eggs was four times greater than that reported for East Lake, it is possible that some tui chub survived the rotenone treatment in 1954 and only reached a population size in excess of a million fish in the 1990's. If the birth rates and mortality rates are more favorable than reported for East Lake, the tui chub population currently in the lake can be explained by re-introduction of the chub circa 1990.
- The models show that the current biomass of tui chub can largely explain the frequency of the cyanobacterial blooms, although the intensity of the blooms appears to be strongly influenced by weather conditions associated with extended periods of high temperature, low average wind speed, and abundant solar radiation.
- To meet water quality goals, it appears necessary to remove a high percentage (~90 to 100 percent) of the tui chub from Diamond Lake.

- The uncertainty analysis of the nutrient fluxes is considerable, largely because of the unknowns associated with the precise role of the biological communities in nutrient cycling.
- Even under a no-fish scenario, it is likely that water quality standards for pH will be exceeded periodically under favorable weather conditions, however the frequency of exceedence based on past climate history is expected to be less than under any scenario with fish present.
- An incremental TMDL is proposed here based on fish biomass in which the tui chub is eliminated and a conservative trout-stocking program is implemented.

REFERENCES

Arnold, J.G., J.R. Williams, R. Srinivasan, and K.W. King. 1995. SWAT: Soil Water Assessment Tool. Texas A&M University, Texas Agricultural Experiment Station, Blackland Research Center, Temple, TX.

Bauer, J.A. 1976. Diamond Lake range management. *Oregon Wildlife*. 31:3-5.

Bird, F.H. 1975. Biology of the blue and tui chubs in East and Paulina Lakes, Oregon. MS thesis. Oregon State University, Corvallis, OR. 165 pp.

Breeden, R. 2003. Ground water analysis. *In* Draft Environmental Impact Statement for the Diamond Lake Restoration Project. Umpqua National Forest, Roseburg, OR.

Carlisle, D.M. and C.P. Hawkins. 1998. Relationship between invertebrate assemblage structure, 2 trout species, and habitat structure in Utah mountain lakes. *J. North Amer. Benthological Soc*. 17:286-300.

Carpenter, S.R. J.F. Kitchell, and J.K. Hodgson. 1985. Cascading trophic interactions and lake productivity. *Bioscience.* 35:634-369.

Chapra, S. C. 1997. Surface Water-Quality Modeling. WCB McGraw-Hill, New York. 844 pp.

Chapra, S.C. and K.H. Reckhow. 1979. Expressing the phosphorus loading concept in probabilistic terms. *J. Fish. Res. Bd. Can.* 36:225-229.

Chorus, I. (ed). 2001. Cyanotoxins, Occurrence, Causes, Consequences. Springer-Verlag. New York. 357 pp.

Cole, T.M. and S.A. Wells. 2001. CE-QUAL-W2: A two-dimensional, laterally averaged, hydrodynamic and water quality model, Version 3.1. Instruction Report EL-00-1. U.S. Army Corps of Engineers.

Dodson, S.I. and D.G. Frey. 2001. Cladocera and Other Branchiopoda. Chap. 21 *In* J.H. Thorp and A.P. Covich (eds). Ecology and Classification of North American Freshwater Invertebrates. Academic Press. New York. Pp. 850-914.

Drenner, R.W., J.D. Smith, and S.T. Threlkeld. 1996. Lake trophic state and the limnological effects of omnivorous fish. *Hydrobiologia*. 319:213-223.

Edmondson, W.T. 1991. The Uses of Ecology: Lake Washington and Beyond. University of Washington Press, Seattle. 329 pp.

Eilers, B.J. 2003. Access database for Diamond Lake. Version 3.2. MaxDepth Aquatics, Inc., Bend , OR.

Eilers, J.M. 2003a. Benthic macroinvertebrates of Diamond Lake, Oregon. A report to the Umpqua National Forest, Roseburg, OR. MaxDepth Aquatics, Inc., Bend, OR. 26 pp. + appendix.

Eilers, J.M. 2003b. Water quality monitoring of Diamond Lake for 2003. A report to the Umpqua National Forest, Roseburg, OR. 70 pp.

Eilers, J.M. and B.J. Eilers. 2003. Hydroacoustic investigations of Diamond Lake in spring 2003. A report to the Oregon Dept. Fish & Wildlife, Roseburg, OR. 18 pp.

Eilers, J.M. and C. Gubala. 2003. Hydroacoustic analysis of Diamond Lake, Oregon. Report to the Oregon Wildlife Heritage Foundation and the Oregon Department of Fish and Wildlife. 21 pp.

Eilers, J.M. and J. Kann. 2002. Diamond Lake database 2001. A report to the Umpqua National Forest, Roseburg, OR. 45 pp.

Eilers, J.M., K. Vache, and J. Kann. 2002. Tenmile Lakes nutrient study, phase II. A report to the Tenmile Lakes Basin Partnership. 136 pp.

Eilers, J.M., C.P. Gubala, P.R. Sweets, and D. Hanson. 2001. Effects of fisheries management and lakeshore development on water quality in Diamond Lake, Oregon. *J. Lake & Reserv. Mgmt*. 17:29-47.

Eilers, J.M., J. Beaver, A. St. Amand, and J. Cornett. 2001. Historical changes of zooplankton and cyanobacteria in Diamond Lake, Oregon based on analysis of the sediment record. A report to the Oregon Department of Fish & Wildlife,

Elser, J.J., R.W. Sterner, A.E. Galford, T.H. Chrzanowski, D.L. Findlay, K.H. Mills, M.J. Patterson, M.P. Stainton, and D.W.Schindler. 2000. Pelagic C:N:P stoichiometry in a eutrophied lake: responses to a whole-lake food-web manipulation. *Ecosystems* 3:293-307.

Hakanson, L., A.P. Ostapenia, and V.V. Boulion. 2003. A mass-balance model for phosphorus in lakes accounting for biouptake and retention in biota. *Freshwater Biology*. 48:928-950.

High Performance Systems. 2003. *ithink*/STELLA 8. Technical Documentation. Lebanon, NH.

Horppilla, J., H. Peltonen, T. Malinen, E. Loukkanen, and T. Kairesal. 1998. Top-down or bottom-up effects by fish: issues of concern in biomanipulation of lakes. Restoration Ecology. 6:20-28.

Jackson, L.S., D.H. Jackson, and D.W. Loomis. 2003. Analysis of Diamond Lake tui chub data collected by the Oregon Department of Fish and Wildlife 1992-2003. ODFW, Roseburg, OR.

Johnson, D. M., R. R. Petersen, D. R. Lycan, J. W. Sweet, M. E. Neuhaus, and A. L Schaedel. 1985. Atlas of Oregon Lakes. Oregon State University Press. Corvallis, Oregon. 317 pp.

Kalff, J. 2002. Limnology: Inland Water Ecosystems. Prentice Hall. Upper Saddle River, New Jersey. 592 pp.

Lampert, W. and U. Sommer. 1997. Limnoecology: the Ecology of Lakes and Streams. Oxford University Press. New York. 382 pp.

Lauer, W.L., G.S. Schuytema, W.D. Sanville., F.S. Stay, and C.F. Powers. 1979. The effects of decreased nutrient loading on the limnology of Diamond Lake, Oregon. U.S. Environmental Protection Agency, Research and Development Report, EPA-600/8-79-017a. USEPA Corvallis Environmental Research laboratory, Corvallis, OR. 60 pp.

Locke, F.E. 1947. A preliminary report on the Diamond Lake study. Oregon State Game Commission. 26 pp.

Marion, L. and J.M. Paillisson. 2003. A mass balance assessment of the contribution of floating-leaved macrophytes in nutrient stocks in an eutrophic macrophyte-dominated lake. *Aquatic Botany.* 75:249-260.

Meyerhoff, R.D. 1977. Sediment characteristics and the trophic status of four Oregon lakes. MS Thesis. Oregon State University, Corvallis, OR. 74 pp.

Miller, W.T., T.E. Maloney, and J.C. Greene. 1974. Algal productivity in 49 lake waters as determined by algal assays. *Water Research.* 8:667-679.

Nichols,K.H., M.F.P. Michalski, and W. Gibson. 1996. An experimental demonstration of trophic interactions affecting water quality of Rice Lake, Ontario (Canada). *Hydrobiologia*. 319:73-85.

Nurnberg, G.K. 1984. The prediction of internal phosphorus load in lakes with anoxic hypolimnia. *Limnology & Oceanography*. 29:111-24.

Oregon State Game Commission. 1949. Annual Report for 1948, Fishery Division.
_____.1951. Annual Report for 1950, Fishery Division.
_____.1952. Annual Report for 1951, Fishery Division.
_____.1953. Annual Report for 1952, Fishery Division.
_____.1953. Diamond Lake rehabilitation. Internal memorandum , March 24, 1953. Fishery Division.
_____.1954. Annual Report for 1953, Fishery Division.
_____.1956. Annual Report for 1955, Fishery Division.
_____.1958. Annual Report for 1957, Fishery Division.
_____.1959. Annual Report for 1958, Fishery Division.
_____.1960. Annual Report for 1959, Fishery Division.
_____.1961. Annual Report for 1960, Fishery Division.

Parker, B.R., D.W. Schindler, D.B. Donald, and R.S. Anderson. 2001. The effects of stocking and removal of nonnative salmonids on the plankton of an alpine lake. *Ecosystems*. 4:334-345.

Phelps, E.B. 1944. Stream Sanitation. John Wiley & Sons, New York. 276 pp.

Pilliod, D.S. and C.R. Peterson. 2001. Effects of introduced trout on persistence in historically fishless watersheds. *Ecosystems* 4:322-333.

Portielje, R. and R.E. Rijsdijk. 2003. Stochastic modeling of nutrient loading and lake ecosystem response in relation to submerged macrophytes and benthivorous fish. *Freshwater Biology.* 48:741-755.

Proulx, M., F.R. Pick, A. Mazumder, P.B. Hamilton, and D.R.S. Lean. 1996. Effects of nutrients and planktivorous fish on the phytoplankton of shallow and deep aquatic systems. *Ecology*. 77:1556-1572.

Salinas, J.T. and D.W. Larson. 1995. Diamond Lake, Umpqua National Forest, Limnological and Bacteriological Investigations: 1992-1994. A report to the Umpqua National Forest. 109 pp. + appendices.

Salinas, J.T. 1996. Diamond Lake, the Umpqua National Forest, a limnological survey, 1995. A report to the Umpqua National Forest. Rogue Community College, Final Report, CAS-9601. 59 pp. + appendices.

Salinas, J.T. 1998. A Cascade lakes limnological survey in the Rogue River, Umpqua, and Winema National Forests 1996/1997. Rogue Community College. Final Report, CAS-9801. 211 pp. + appendices.

Salinas, J.T. 1999. A Cascade lakes limnological survey in the Umpqua and Winema National Forests 1998. Rogue Community College. Final Report, CAS-9901. 112 pp. + appendices.

Salinas, J.T. 2000. A Cascade lakes limnological survey in the Umpqua and Winema National Forests 1999. Rogue Community College. Final Report, CAS-0001. 182 pp. + appendices.

Salinas, J.T. 2001. A Cascade lakes limnological survey in the Umpqua and Winema National Forests 2000. Rogue Community College. Final Report, CAS-0101. 135 pp. + appendices.

Salinas, J.T. 2002. Diamond Lake limnological survey Umpqua National Forest 2001. Rogue Community College. Final Report, CAS-0206. 62 pp. + appendices.

Sanville, W.D. and C. Powers.1973. Diamond Lake Studies – 1971. Progress Report No. 1. Working Paper #8. U.S. Environmental Protection Agency. Corvallis, OR. 13 pp. + appendices.

Sarnelle, O. 1993. Herbivore effects on phytoplankton succession in a eutrophic lake. *Ecol. Monogr*. 63:129-149.

Schindler, D.E., K.A. Knapp, and P.R. Leavitt. 2001. Alteration of nutrient cycles and algal production resulting from fish introductions into mountain lakes. *Ecosystems*. 4:308-321.

Schindler, D.E. and L. Eby. 1997. Stoichiometry of fishes and their prey: implications for nutrient recycling. *Ecology*. 78:1816-1831.

Shapiro, J., V. Lamarra, and M. Lynch. 1975. Biomanipulation: an ecosystem approach to lake restoration. *In*: P.L. Brezonik and J.L Fox (eds.). Water Quality Management Through Biological Control. Symposium Proceedings, University of Florida, Department of Environmental Engineering, Gainesville, FL.

Sherrod, D.R. 1991. Geologic map of part of the Cascade Range between latitudes 43°-44°, central Oregon. Miscellaneous Investigations Series. Map I-1891. US Geological Survey.

Simon, K.S. and C.R. Townsend. 2003. Impacts of freshwater invaders at different levels of ecological organization, with emphasis on salmonids and ecosystem consequences. *Freshwater Biology*. 48:982-994.

U.S. Environmental Protection Agency. 1978. Report on Diamond Lake, Douglas County, Oregon, EPA Region X. Working Paper No. 828. 7 pp. + appendices.

Vogel, A. 2002. Zooplankton. *In* J. Salinas. Diamond Lake Limnological Survey. Report to the Umpqua National Forest. Final Report CAS-0206.

Walker, W. W. 2001. Quantifying variability and uncertainty in phosphorus TMDL's for lakes. Presentation at the North American Lake Management Society, Madison, WI. Nov. 8, 2001.

Wallace, R.L. and T.W. Snell. 2001. Phylum Rotifera. Chap. 8 *In* J.H. Thorp and A.P. Covich (eds). Ecology and Classification of North American Freshwater Invertebrates. Academic Press. New York. Pp. 191-254.

Williamson, C.E. and J.W. Reid. 2001. Copepoda. Chap. 22 *In* J.H. Thorp and A.P. Covich (eds). Ecology and Classification of North American Freshwater Invertebrates. Academic Press. New York. Pp. 915-954.

Yoo, S., W.W. Carmichael, R.C. Hoehn, and S.E. Hrudey. 1995. Cyanobacterial (Blue-green Algal) Toxins: A Resource Guide. American Water Works Association Research Foundation. Denver, Colorado.

ACKNOWLEDGEMENTS

This project was funded by contract #040-03 from the Oregon Department of Environmental Quality. We thank Dr. Allan Vogel for consultation regarding recent changes in zooplankton populations in Diamond Lake.

APPENDICIES

A. Access database for Diamond Lake, Ver. 3.2, 2003 (CD)
B. CE-QUAL-W2 model application for Diamond Lake (CD)
C. FIN-S model for Diamond Lake (code and CD)
D. List of phytoplankton taxa for Diamond Lake
E. Distribution of algae and cyanobacteria in Diamond Lake
F. Time-series plots of dominant algae in Diamond Lake

Appendix D. Phytoplankton Species in Diamond Lake, 1992-2003

Species	Taxonomic Group	Species Code
Anabaena circinalis	B	ABCR
Anabaena flos-aquae	B	ABFA
Anabaena planctonica	B	ABPL
Aphanizomenon flos-aquae	B	APFA
Microcystis aeruginosa	B	MSAE
Oscillatoria sp	B	OSXX
Achnanthes clevei	D	ACCV
Achnanthes exigua	D	ACEX
Achnanthes hauckiana	D	ACHK
Achnanthes lanceolata	D	ACLC
Achnanthes linearis	D	ACLN
Achnanthes minutissima	D	ACMN
Achnanthes peragalli	D	ACPR
Amphora ovalis	D	AFOV
Amphora perpusilla	D	AFPR
Asterionella formosa	D	ASFO
Caloneis ventricosa minuta	D	CAVM
Caloneis sp	D	CAXX
Cyclotella comta	D	CCCM
Cyclotella meneghiniana	D	CCMG
Cymbella affinis	D	CMAF
Cymbella cistula	D	CMCL
Cymbella microcephala	D	CMMC
Cymbella minuta	D	CMMN
Cymbella sinuata	D	CMSN
Cymbella tumida	D	CMTM
Cymbellonitzschia diluviana	D	CNDL
Cocconeis disculus	D	CODS
Cocconeis klamathensis	D	COKL
Cocconeis placentula	D	COPC
Cocconeis pediculus	D	COPD
Diploneis elliptica	D	DPEL
Diatoma vulgare	D	DTVL
Epithemia sorex	D	EPSX
Epithemia turgida	D	EPTR
Eunotia pectinalis	D	EUPC
Eunotia sp	D	EUXX
Fragilaria brevistriata	D	FRBR
Fragilaria capucina	D	FRCA
Fragilaria capucina mesolepta	D	FRCM
Fragilaria construens	D	FRCN
Fragilaria crotonensis	D	FRCR
Fragilaria construens venter	D	FRCV
Fragilaria leptostauron	D	FRLP

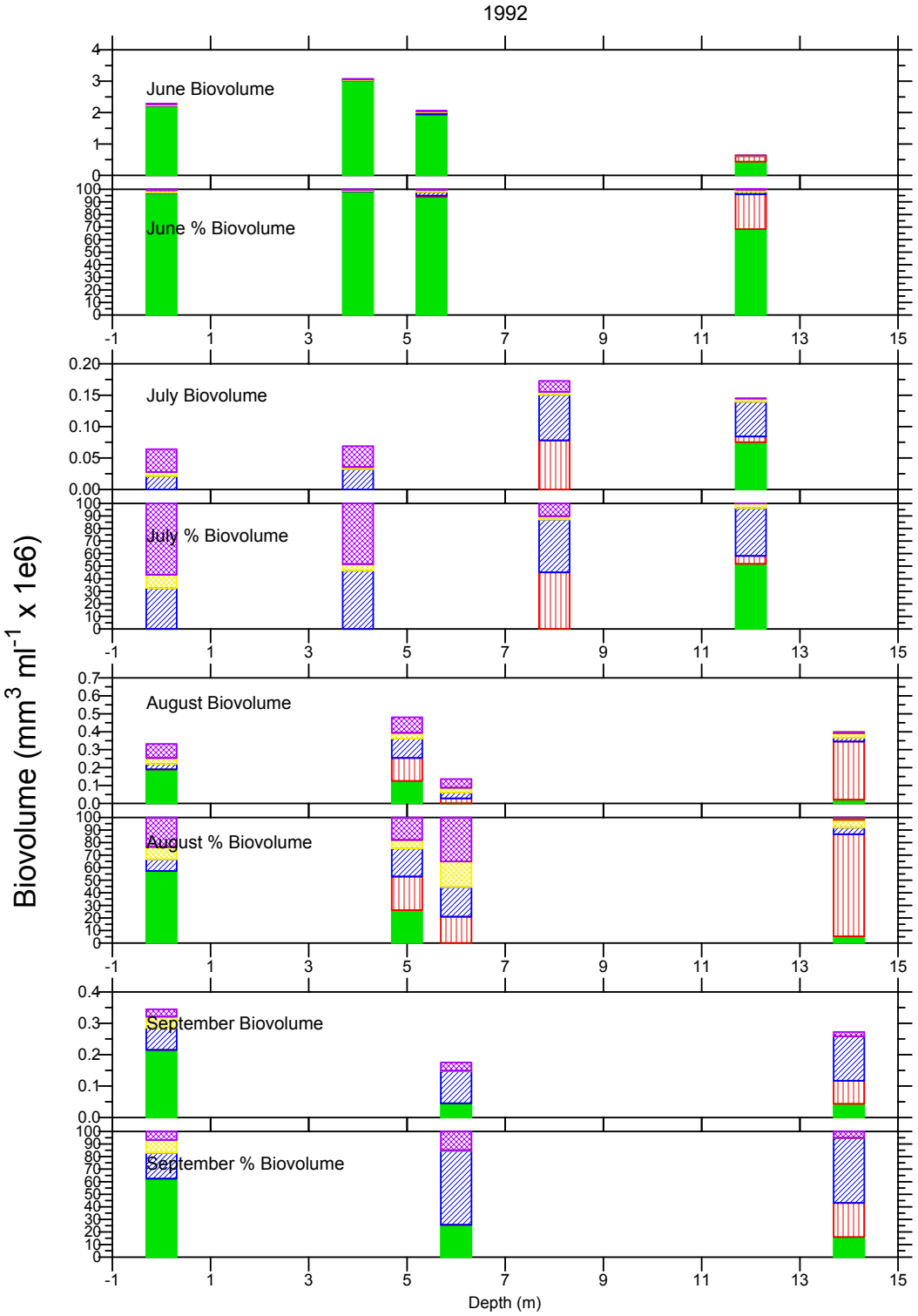
Taxonomic Groups	
B	Blue-green
D	Diatom
E	Euglenoids
F	Dinoflagelates
G	Green
K	Chrysophytes
Y	Cryptophytes

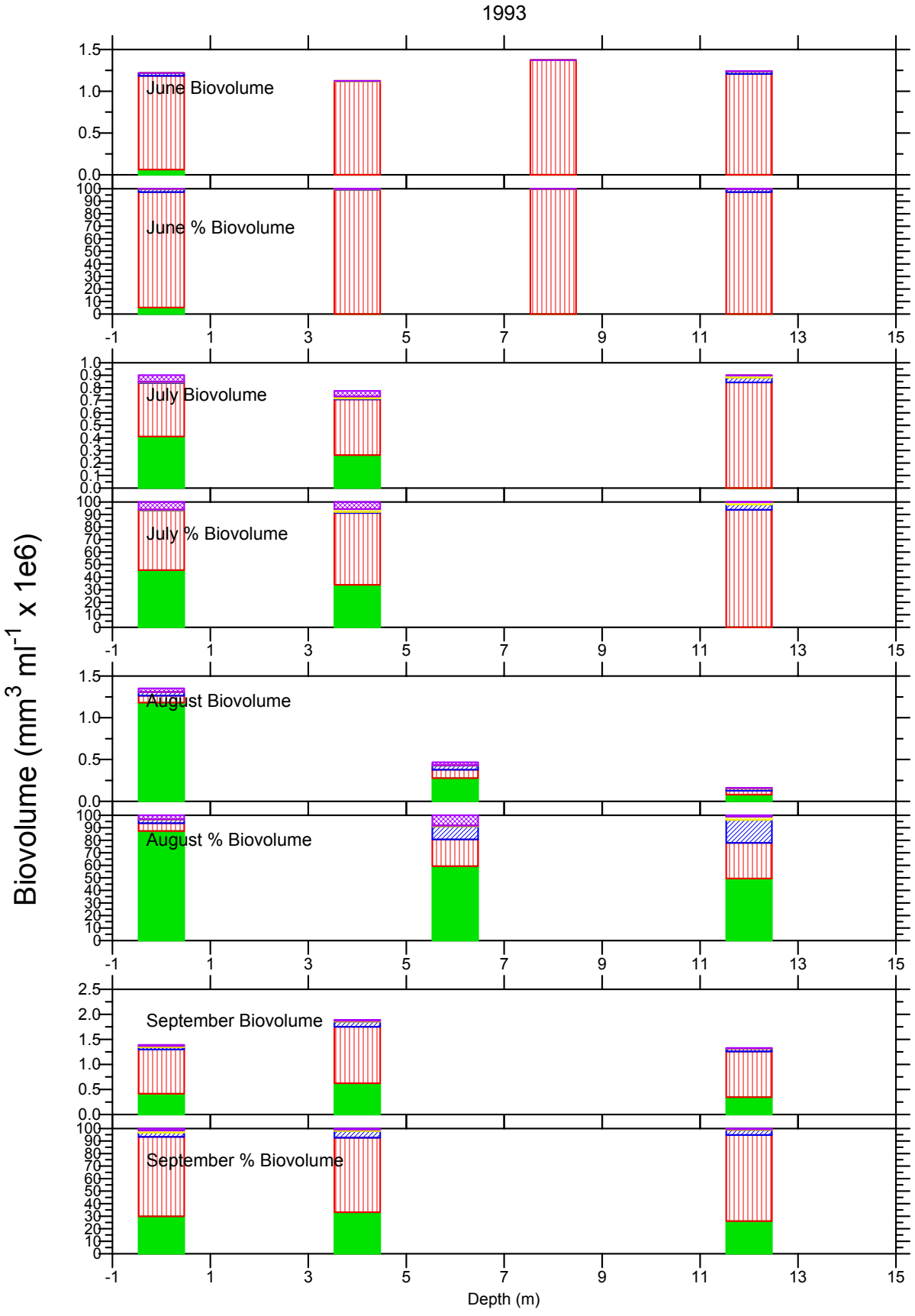
Species	Taxonomic Group	Species Code
Fragilaria pinnata	D	FRPN
Fragilaria vaucheriae	D	FRVA
Fragilaria sp	D	FRXX
Gomphonema acuminatum	D	GFAC
Gomphonema angustatum	D	GFAN
Gomphonema clevei	D	GFCL
Gomphonema gracile	D	GFGC
Gomphonema olivaceum	D	GFOM
Gomphonema subclavatum	D	GFSB
Gomphonema ventricosum	D	GFVT
Gomphonema sp	D	GFXX
Hannaea arcus	D	HNAR
Hemidinium sp	D	HZXX
Melosira ambigua	D	MLAM
Melosira granulata angustissima	D	MLGA
Melosira granulata	D	MLGR
Melosira italica	D	MLIT
Melosira varians	D	MLVR
Navicula anglica	D	NVAG
Navicula cryptocephala	D	NVCR
Navicula cascadensis	D	NVCS
Navicula cryptocephala veneta	D	NVCV
Navicula decussis	D	NVDC
Navicula graciloides	D	NVGC
Navicula minima	D	NVMN
Navicula pupula	D	NVPP
Navicula pseudoscutiformis	D	NVPS
Navicula radiosa	D	NVRD
Navicula rhynchocephala	D	NVRH
Navicula reinhartii	D	NVRN
Navicula scutiformis	D	NVSC
Navicula seminulum	D	NVSM
Navicula tripunctata	D	NVTP
Navicula sp	D	NVXX
Nitzschia acicularis	D	NZAC
Nitzschia amphibia	D	NZAM
Nitzschia communis	D	NZCM
Nitzschia capitellata	D	NZCP
Nitzschia dissipata	D	NZDS
Nitzschia frustulum	D	NZFR
Nitzschia fonticola	D	NZFT
Nitzschia holsatica	D	NZHL
Nitzschia innominata	D	NZIN
Nitzschia linearis	D	NZLN
Nitzschia microcephala	D	NZMC
Nitzschia paleacea	D	NZPC

Species	Taxonomic Group	Species Code
Nitzschia palea	D	NZPL
Nitzschia sigmoidea	D	NZSG
Nitzschia volcanica	D	NZVL
Nitzschia sp	D	NZXX
Pinnularia microstauron	D	PLMC
Pinnularia sp	D	PLXX
Rhoicosphenia curvata	D	RHCU
Rhopalodia gibba	D	RPGB
Rhizosolenia eriensis	D	RZER
Synedra cyclopum	D	SNCY
Synedra delicatissima	D	SNDL
Synedra mazamaensis	D	SNMZ
Synedra parasitica	D	SNPR
Synedra radians	D	SNRD
Synedra rumpens	D	SNRM
Synedra socia	D	SNSC
Synedra ulna	D	SNUL
Stauroneis sp	D	SRXX
Stephanodiscus astraea minutula	D	STAM
Stephanodiscus astraea	D	STAS
Stephanodiscus hantzschii	D	STHN
Stephanodiscus niagarae	D	STNG
Tabellaria fenestrata	D	TBFN
Euglena sp	E	EGXX
Trachelomonas charkowensis	E	TRAM
Trachelomonas cylindrica	E	TRCR
Trachelomonas hispida	E	TRHS
Trachelomonas scabra	E	TRSC
Trachelomonas volvocina	E	TRVL
Ceratium hirundinella	F	CJHR
Glenodinium sp	F	GDXX
Gymnodinium sp	F	GNXX
Peridinium cinctum	F	PRCN
Ankistrodesmus falcatus	G	AKFL
Chodatella wratislawiensis	G	CDWR
Chlamydomonas-like	G	CHLK
Chlamydomonas sp	G	CHXX
Chlorella sp	G	CLXX
Cosmarium sp	G	CSXX
Coelastrum microporum	G	CUMC
Dictyosphaerium ehrenbergianum	G	DCEH
Elakatothrix gelatinosa	G	ELGL
Gloeocystis ampla	G	GLAM
Gloeocystis sp	G	GLXX
Mougeotia sp	G	MGXX
Unident. desmid	G	MXDS

Species	Taxonomic Group	Species Code
Nephrocytium sp	G	NFXX
Oocystis lacustris	G	OCLA
Oocystis pusilla	G	OCPU
Oocystis parva	G	OCPV
Oocystis sp	G	OCXX
Pandorina morum	G	PDMR
Pediastrum boryanum	G	PSBR
Pediastrum duplex	G	PSDP
Pediastrum tetras	G	PSTT
Quadrigula closterioides	G	QDCL
Scenedesmus denticulatus	G	SCDT
Scenedesmus quadricauda	G	SCQD
Scenedesmus sp	G	SCXX
Sphaerocystis schroeteri	G	SFSR
Selenastrum minutum	G	SLMN
Staurastrum gracile	G	SMGC
Staurastrum sp	G	SMXX
Tetraedron minimum	G	TEMN
Tetraedron regulare	G	TERG
Ulothrix sp	G	ULXX
Unident green alga	G	MXGR
Volvox sp	G	VLXX
Chrysococcus rufescens	K	CYRF
Dinobryon bavaricum	K	DBBV
Dinobryon sertularia	K	DBST
Dinobryon sp	K	DBXX
Kephyrion littorale	K	KFLT
Chrysochromulina sp	K	KKXX
Chromulina sp	K	KMXX
Mallomonas sp	K	MMXX
Unident. chrysophyte	K	MXCY
Pseudopedinella sp	K	PKST
Tribonema affine	K	TNAF
Chroomonas sp	Y	CVXX
Cryptomonas erosa	Y	CXER
Cryptomonas ovata	Y	CXOV
Cryptomonas sp	Y	CXXX
Rhodomonas minuta	Y	RDMN
Rhodomonas sp	Y	RDXX

Appendix E. Distribution of algae by year, month, and depth (m).




1993
June Biovolume
June % Biovolume
July Biovolume
July % Biovolume
August Biovolume
August % Biovolume
September Biovolume
September % Biovolume
Biovolume (mm3 ml-1 x 1e6)
Depth (m)

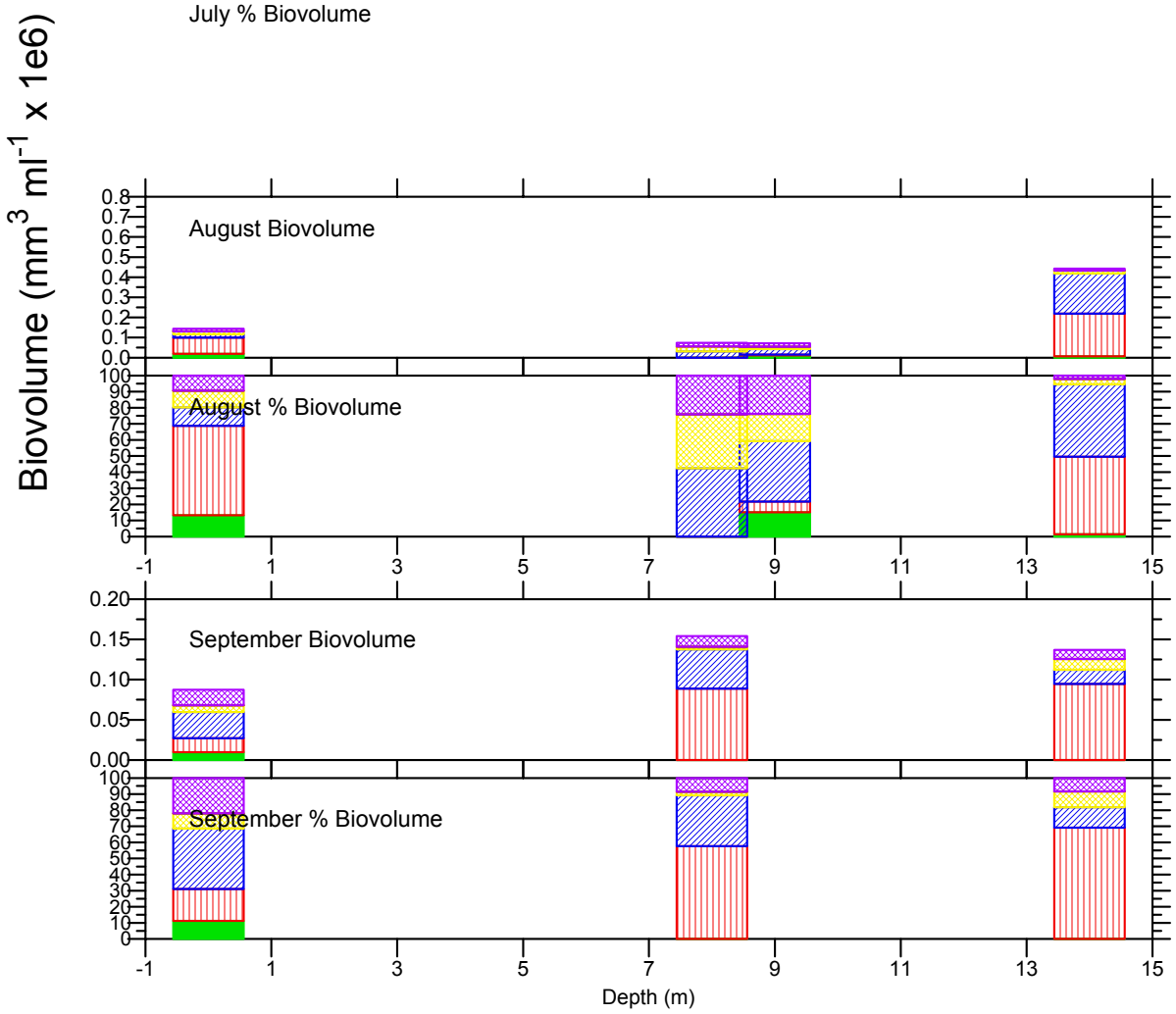
1994

June Biovolume

June % Biovolume

July Biovolume

July % Biovolume



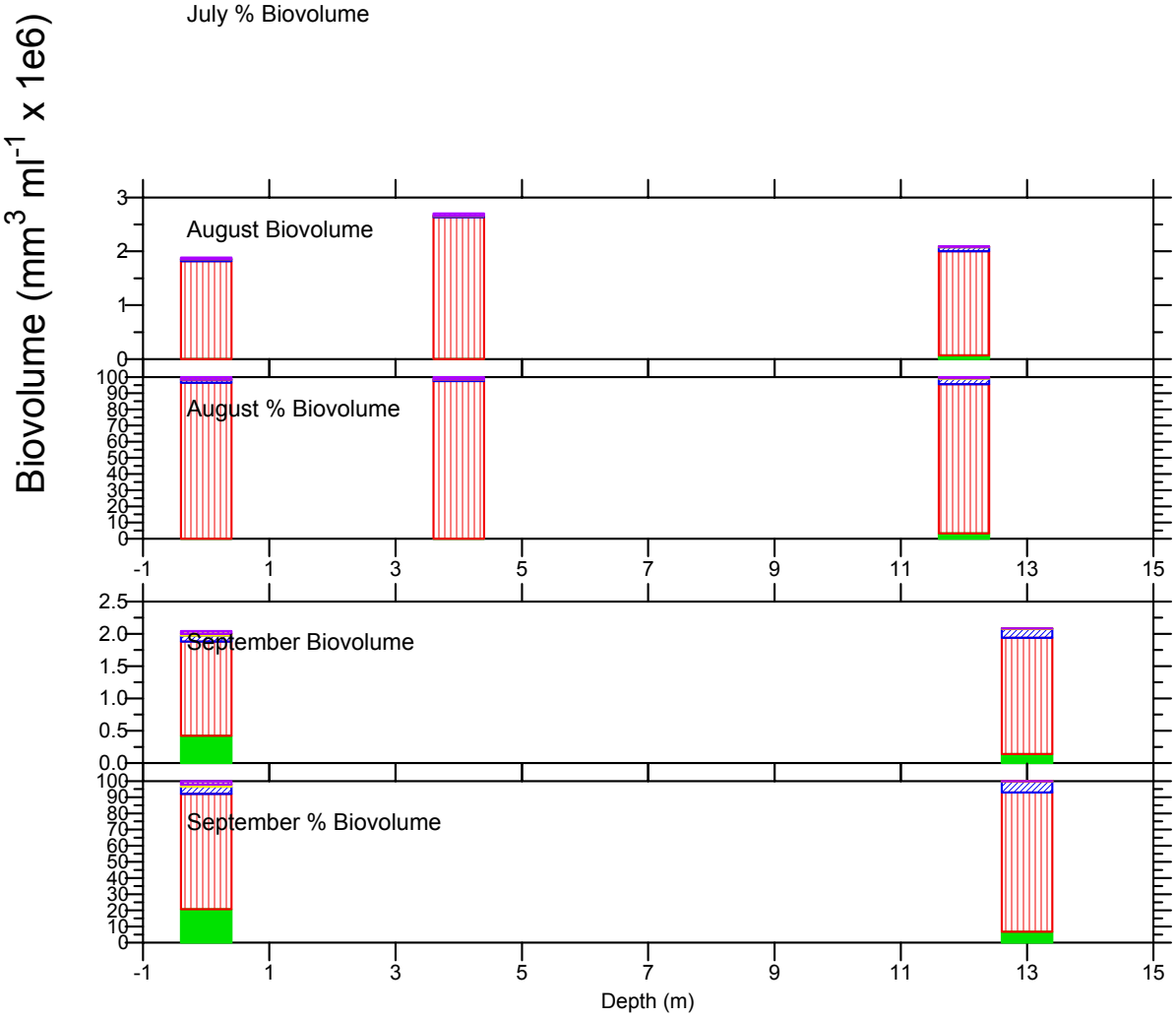
1995

June Biovolume

June % Biovolume

July Biovolume

July % Biovolume



1996

June Biovolume

June % Biovolume

July Biovolume

July % Biovolume

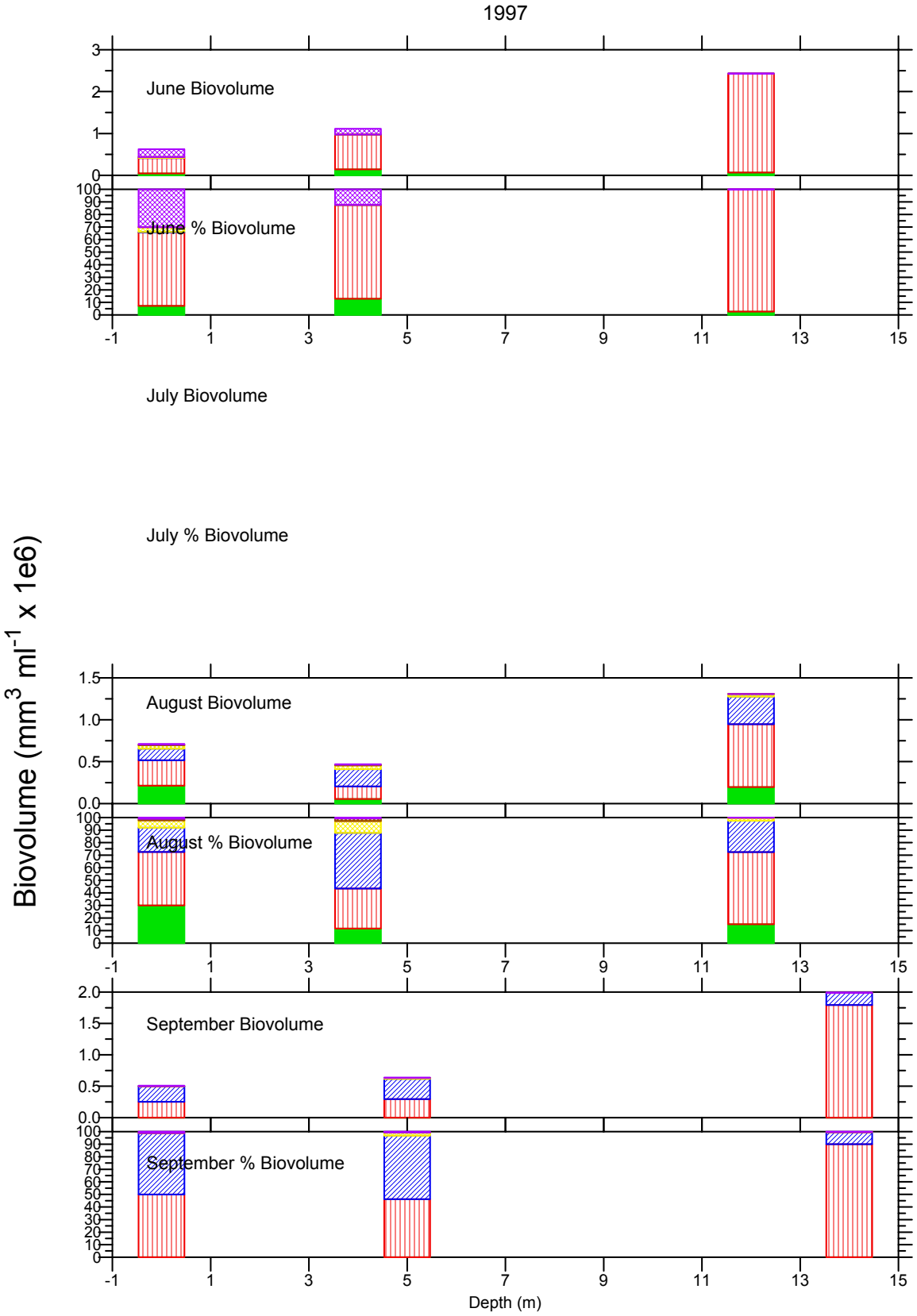
Biovolume (mm^3 ml^{-1} x 1e6)

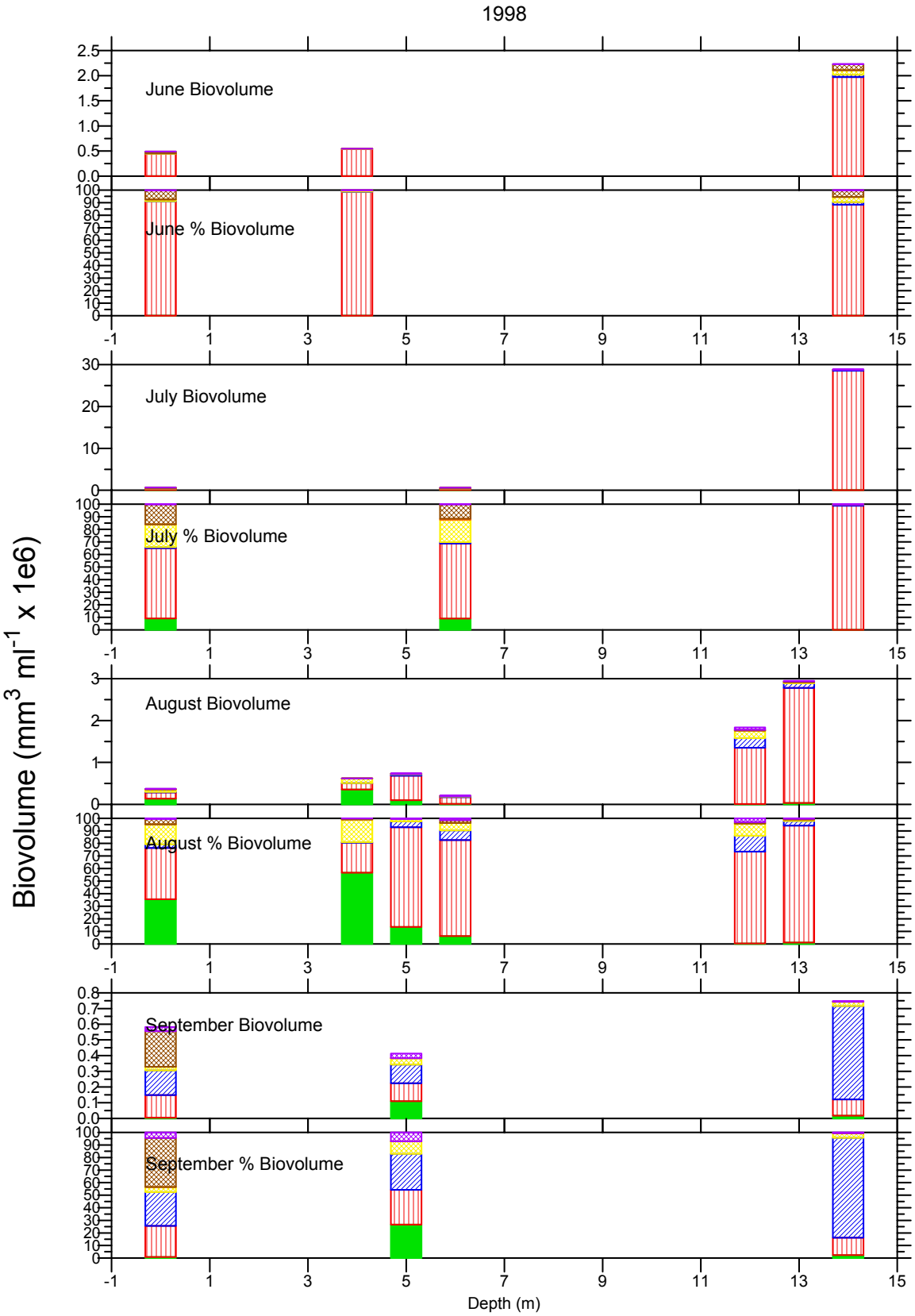
August Biovolume

August % Biovolume

September Biovolume

September % Biovolume

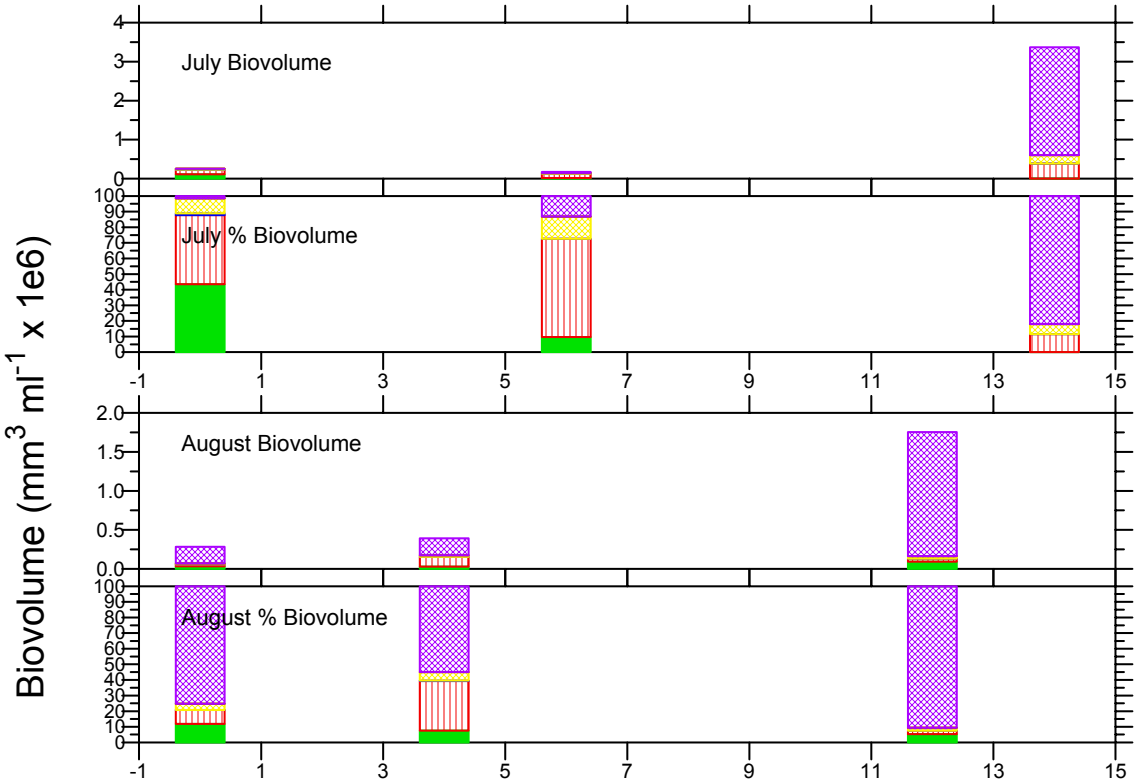

1997
June Biovolume
June % Biovolume
July Biovolume
July % Biovolume
August Biovolume
August % Biovolume
September Biovolume
September % Biovolume
Biovolume (mm3 ml-1 x 1e6)
Depth (m)


1998
June Biovolume
June % Biovolume
July Biovolume
July % Biovolume
August Biovolume
August % Biovolume
September Biovolume
September % Biovolume
Biovolume (mm3 ml-1 x 1e6)
Depth (m)

1999

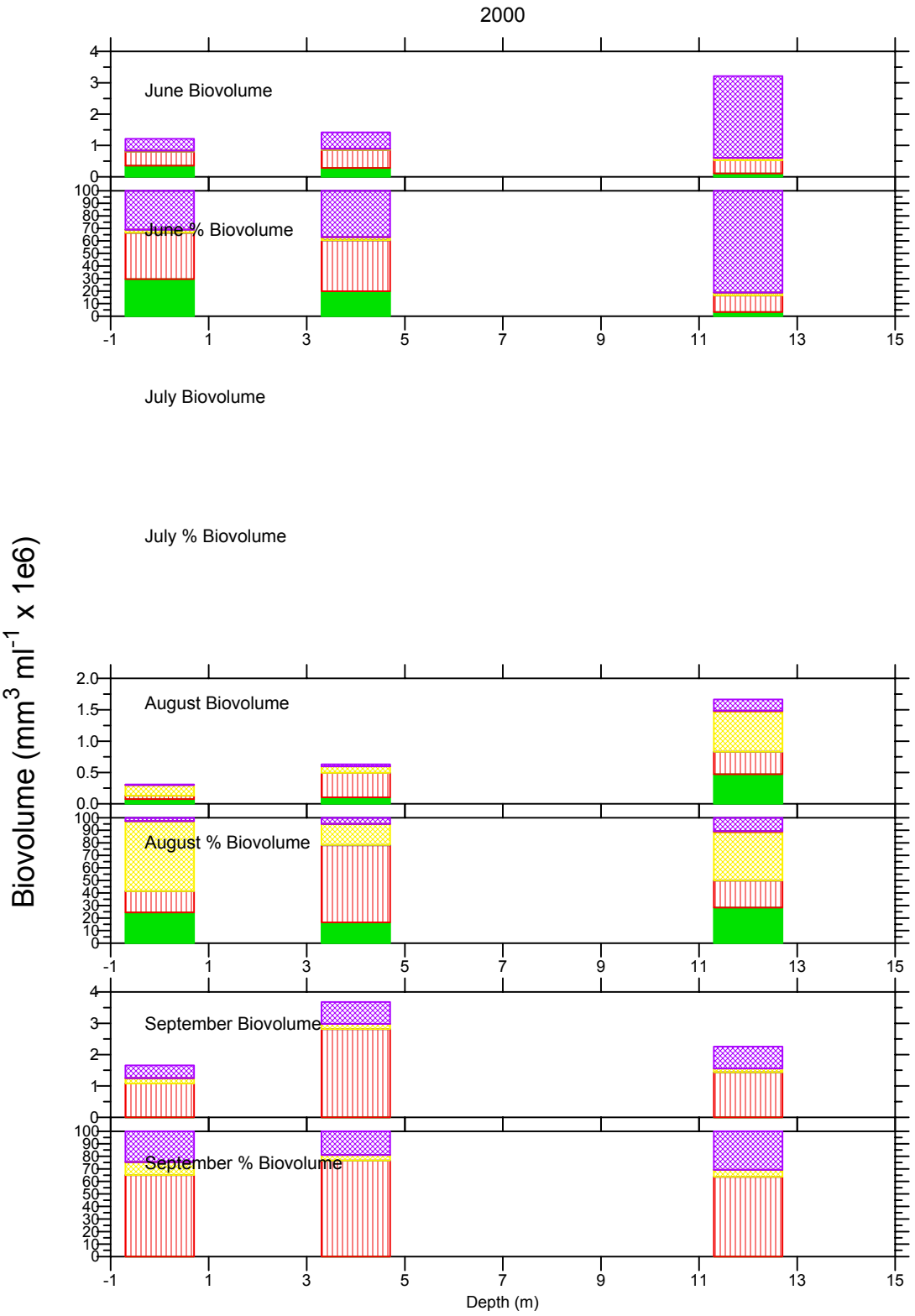
June Biovolume

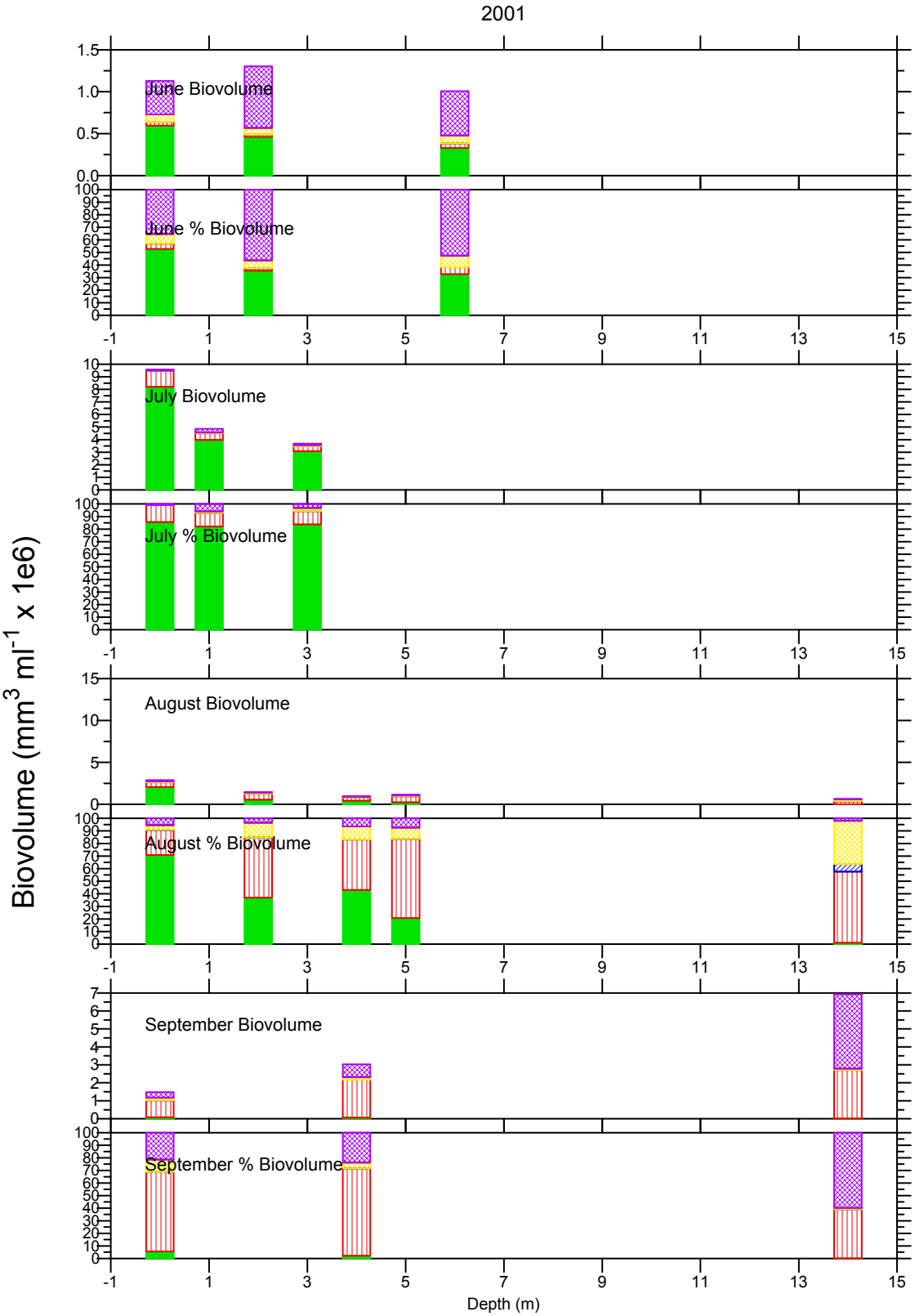
June % Biovolume

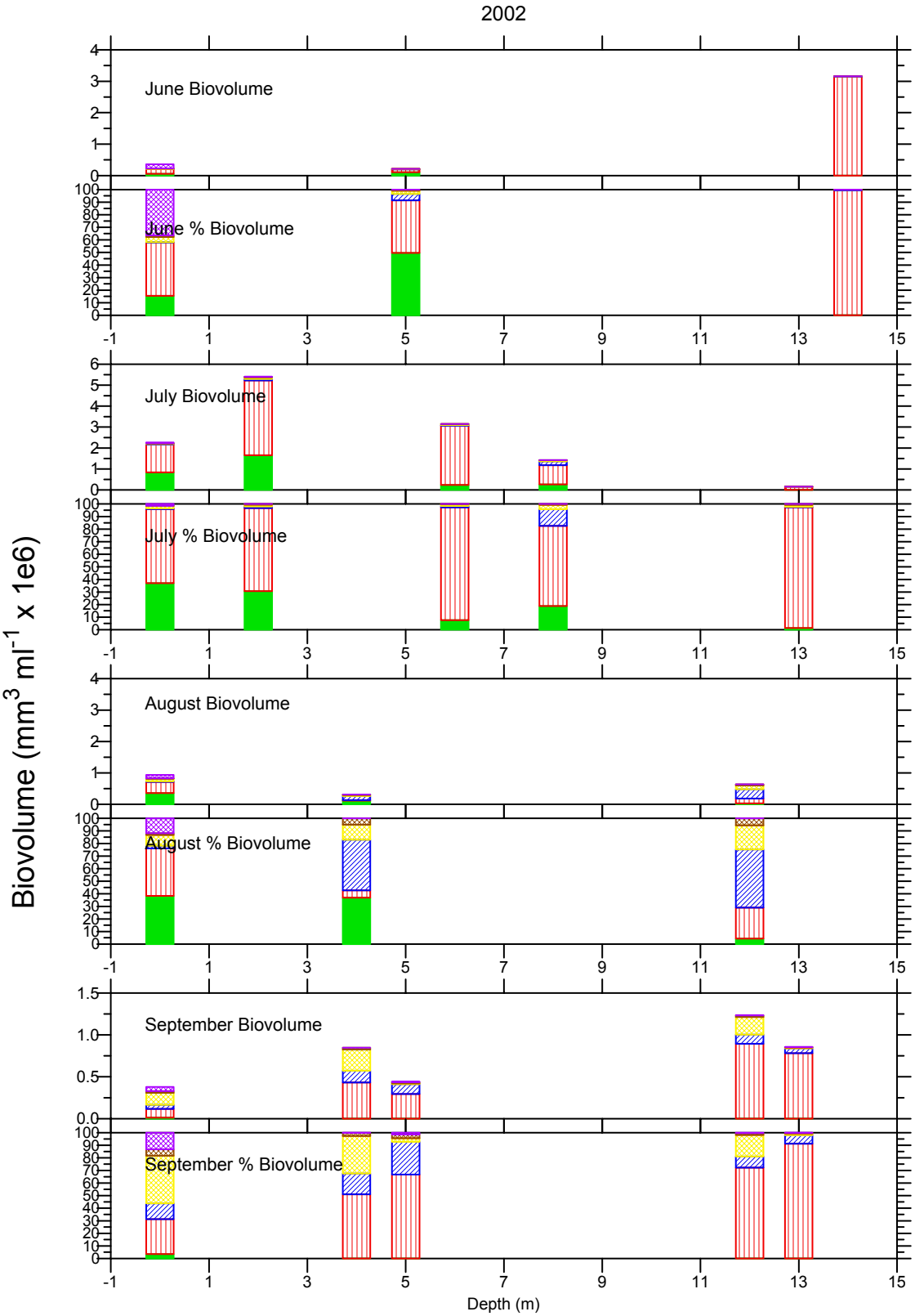


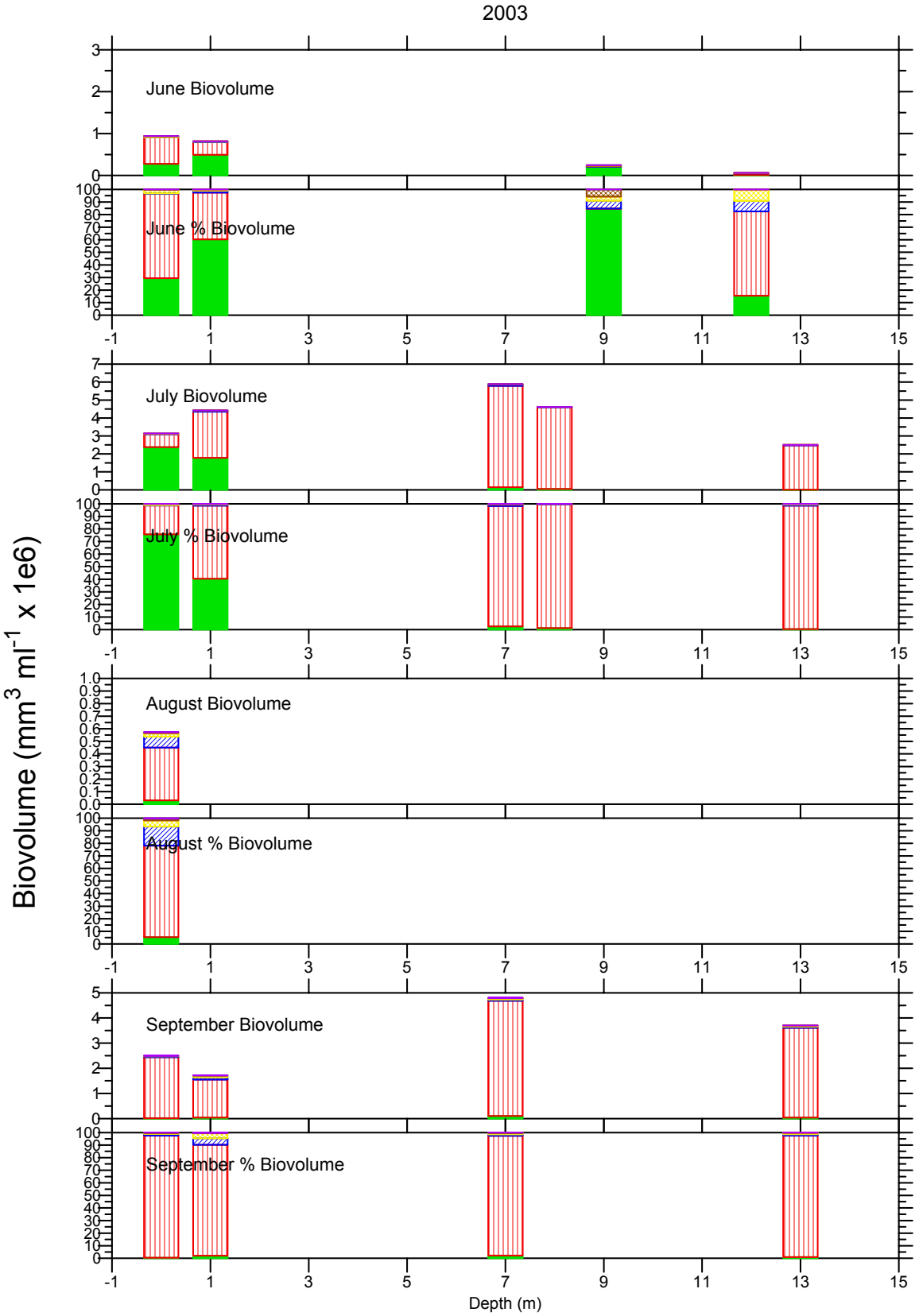
September Biovolume

September % Biovolume

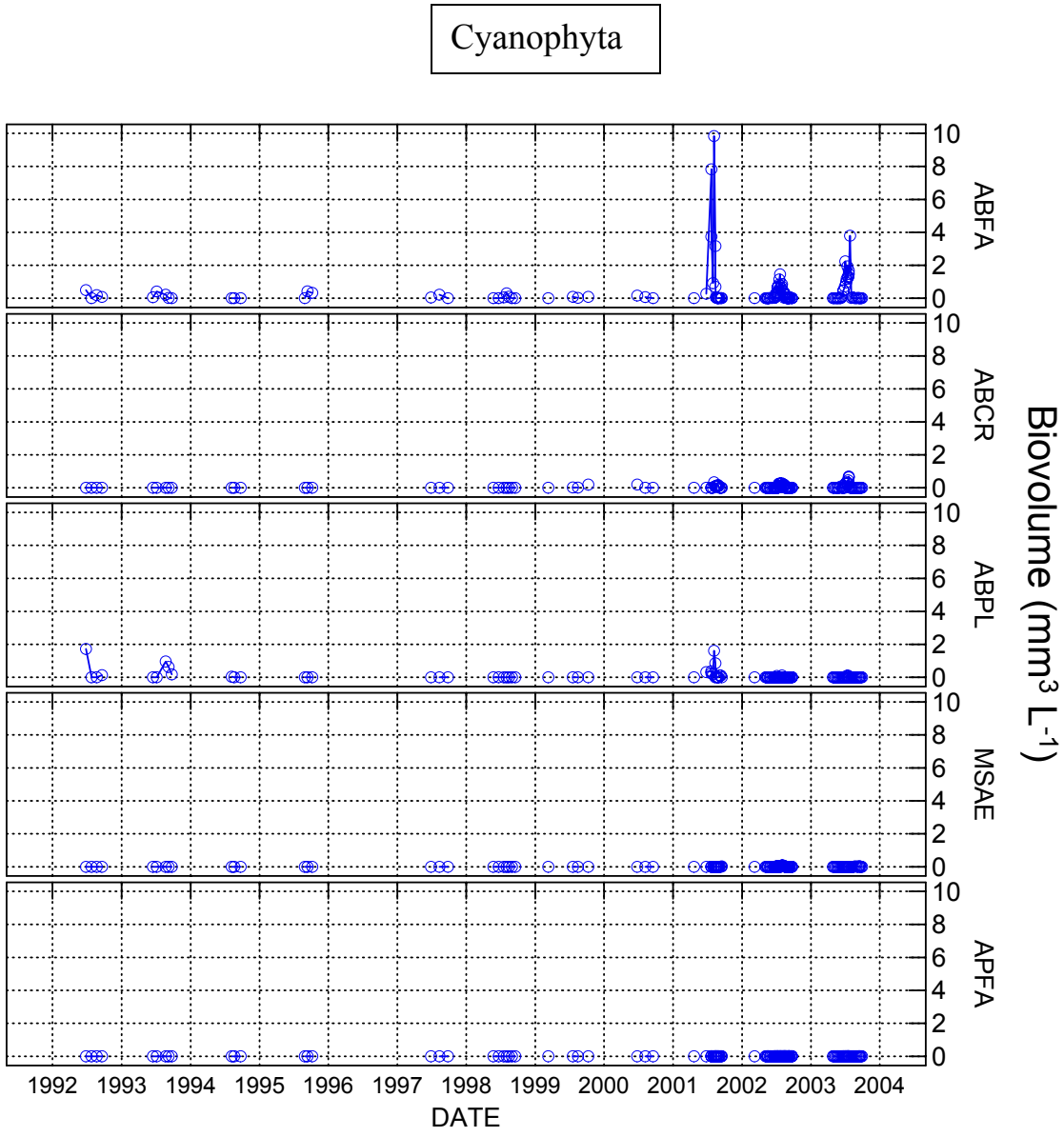

2000
June Biovolume
June % Biovolume
July Biovolume
July % Biovolume
August Biovolume
August % Biovolume
September Biovolume
September % Biovolume
Biovolume (mm3 ml-1 x 1e6)
Depth (m)


2001
June Biovolume
June % Biovolume
July Biovolume
July % Biovolume
August Biovolume
August % Biovolume
September Biovolume
September % Biovolume
Biovolume (mm3 ml-1 x 1e6)
Depth (m)

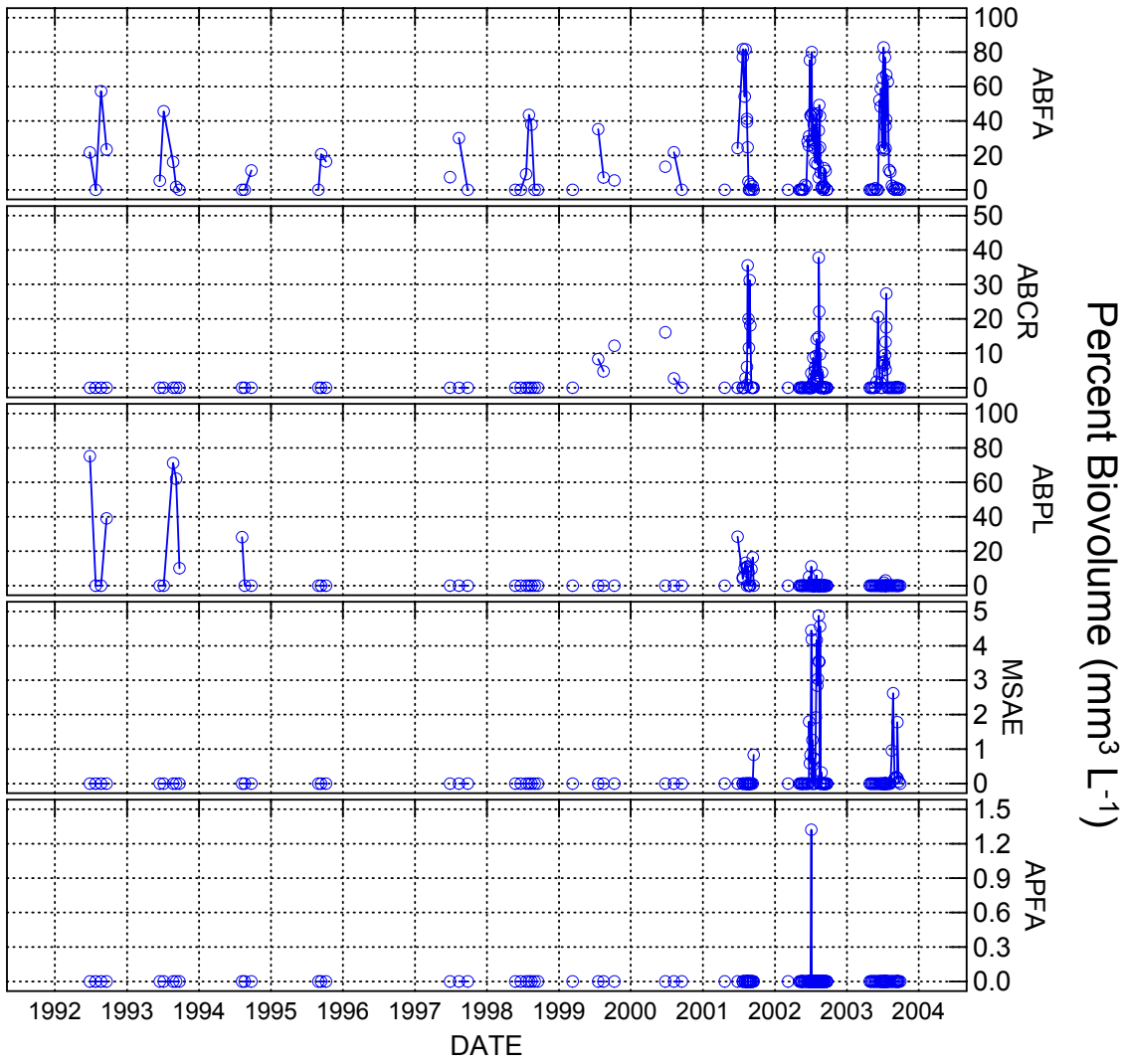

2002
June Biovolume
June % Biovolume
July Biovolume
July % Biovolume
August Biovolume
August % Biovolume
September Biovolume
September % Biovolume
Biovolume (mm3 ml-1 x 1e6)
Depth (m)


2003
June Biovolume
June % Biovolume
July Biovolume
July % Biovolume
August Biovolume
August % Biovolume
September Biovolume
September % Biovolume
Biovolume (mm3 ml-1 x 1e6)
Depth (m)

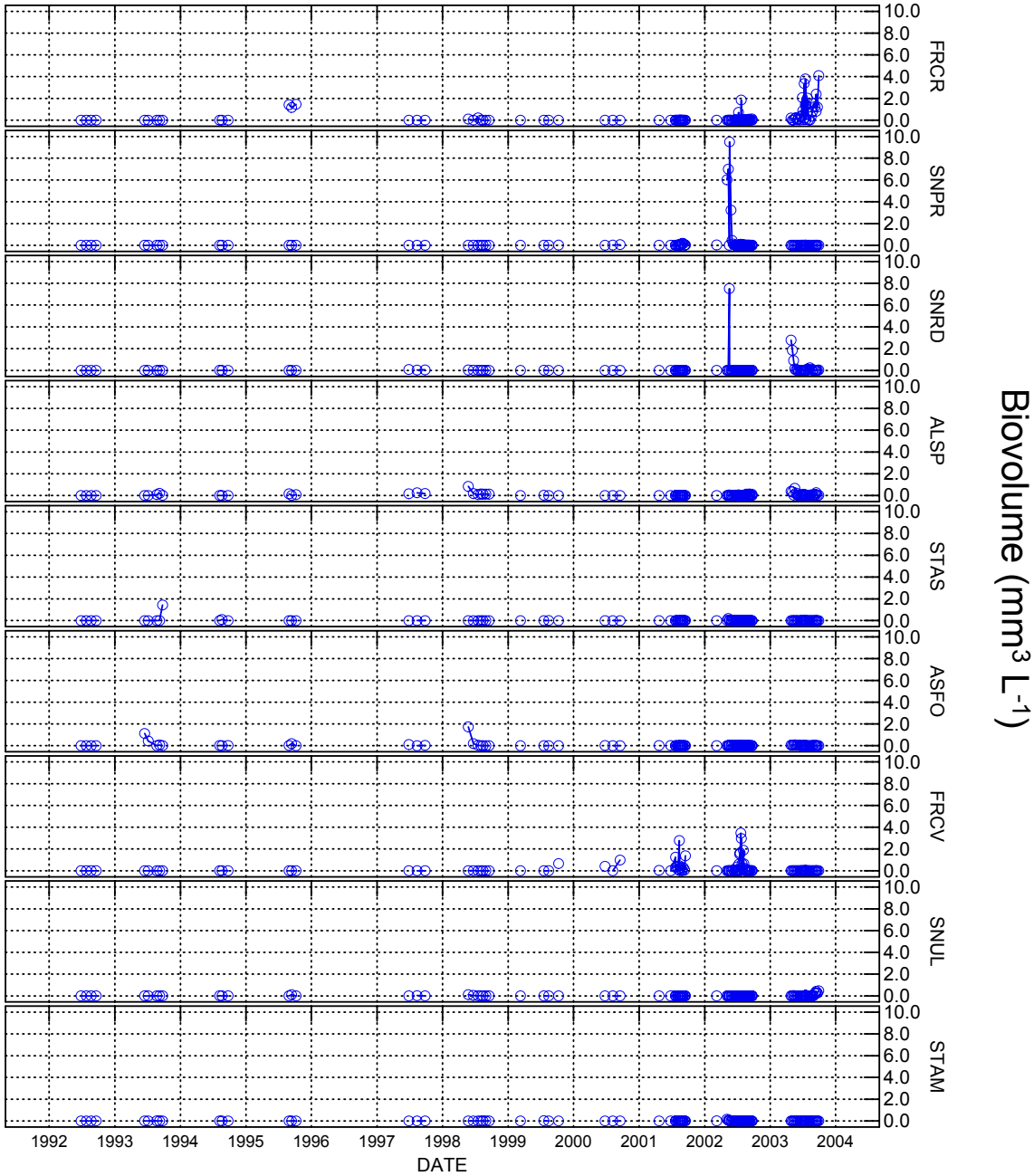
Appendix F. Biovolume and % biovolume of dominant phytoplankton species at Diamond Lake station DLA, 1992-2003.



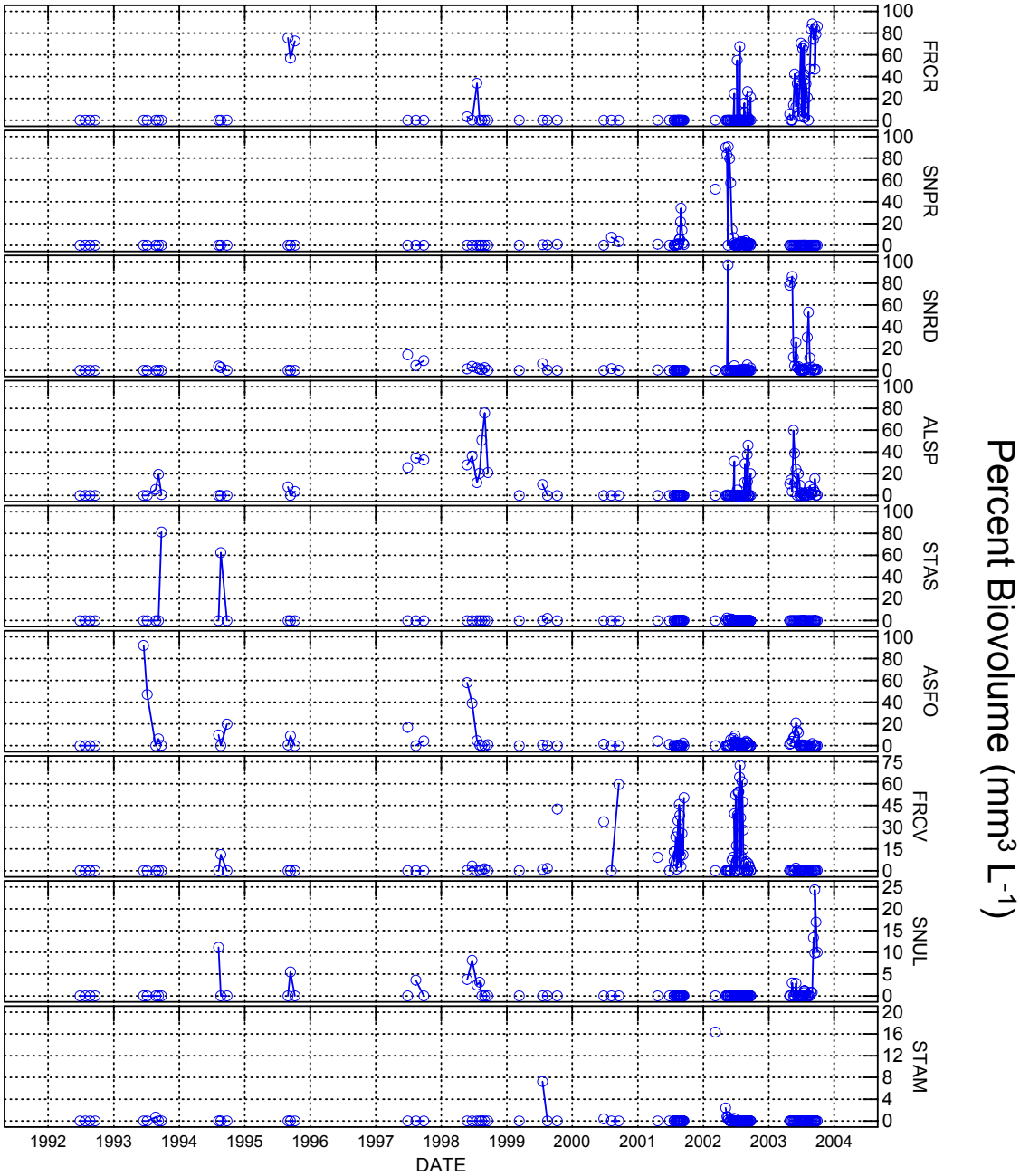
% Cyanophyta


ABFA
ABCR
ABPL
MSAE
APFA
Percent Biovolume (mm^3 L^{-1})
DATE
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004

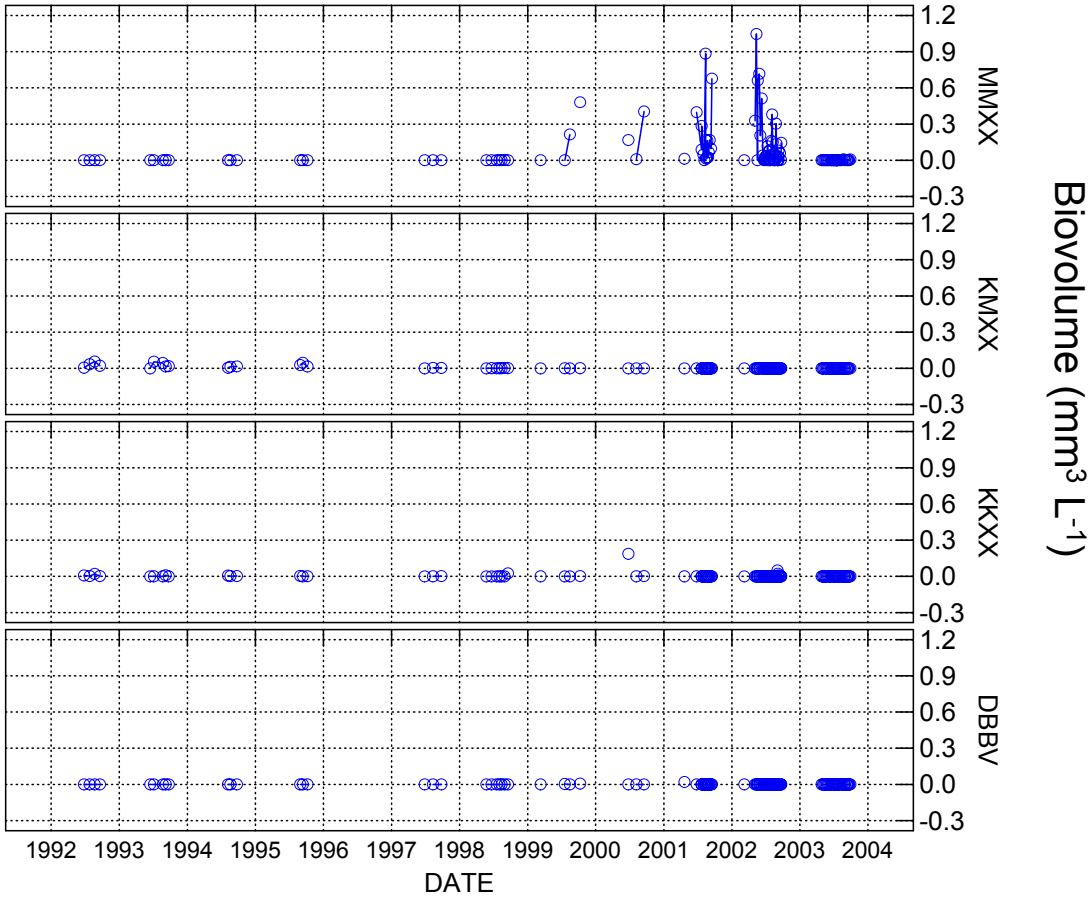
Diatoma


Biovolume (mm3 L-1)
FRCR
SNPR
SNRD
ALSP
STAS
ASFO
FRCV
SNUL
STAM
10.0
8.0
6.0
4.0
2.0
0.0
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
DATE

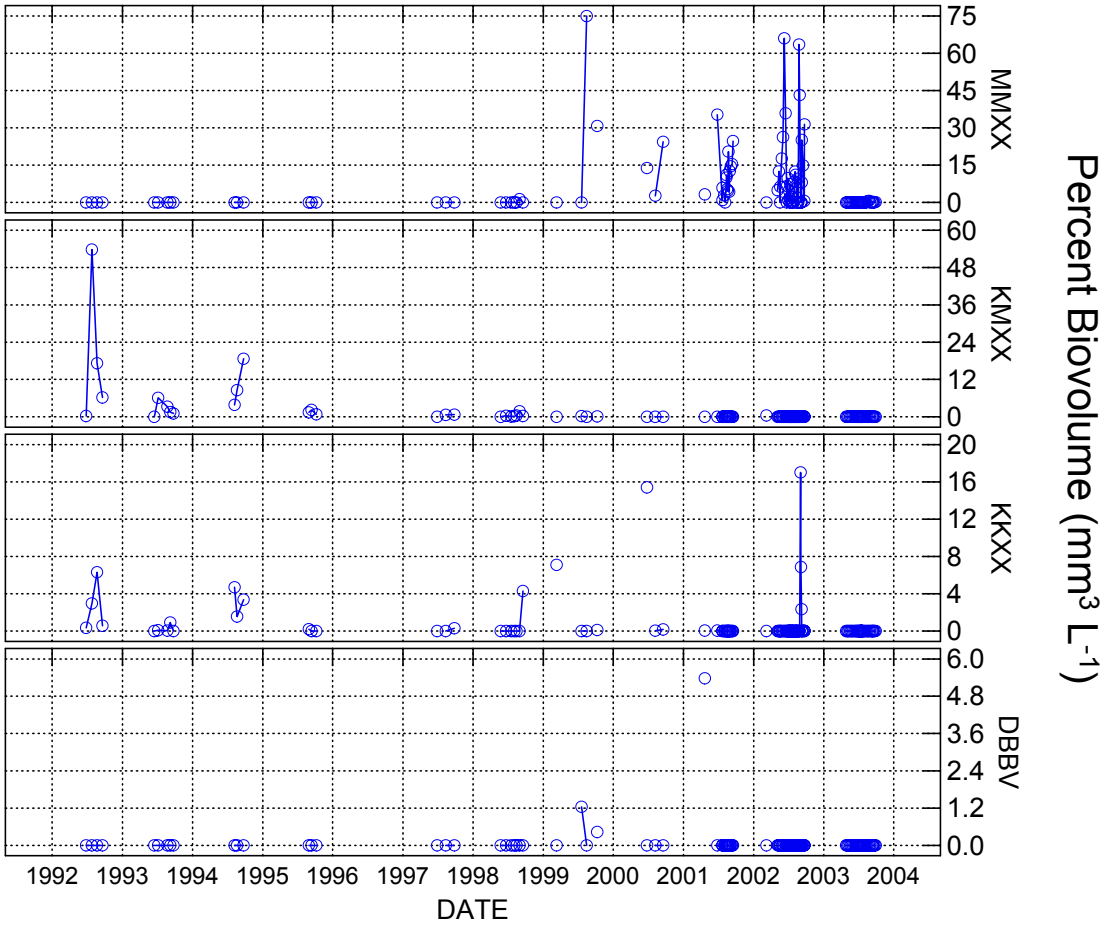
% Diatoma


Percent Biovolume (mm3 L-1)
FRCR
SNPR
SNRD
ALSP
STAS
ASFO
FRCV
SNUL
STAM
DATE
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004

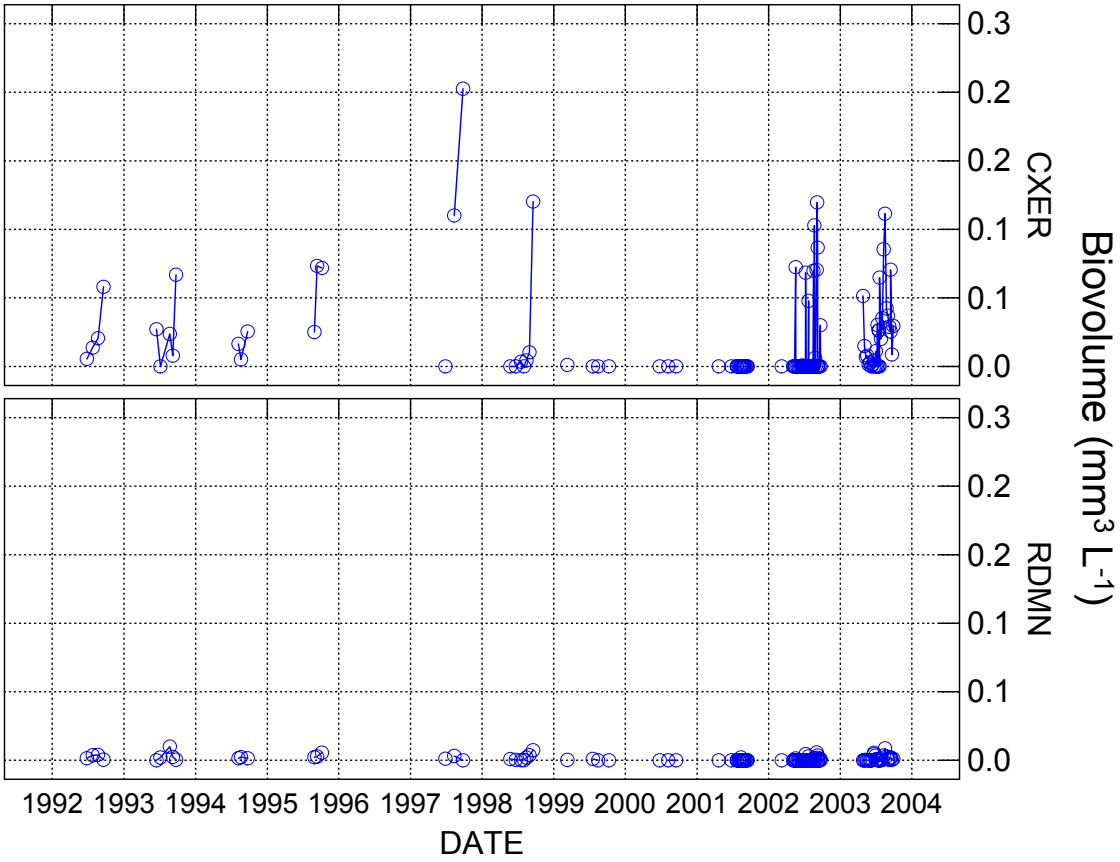
Chrysophyta



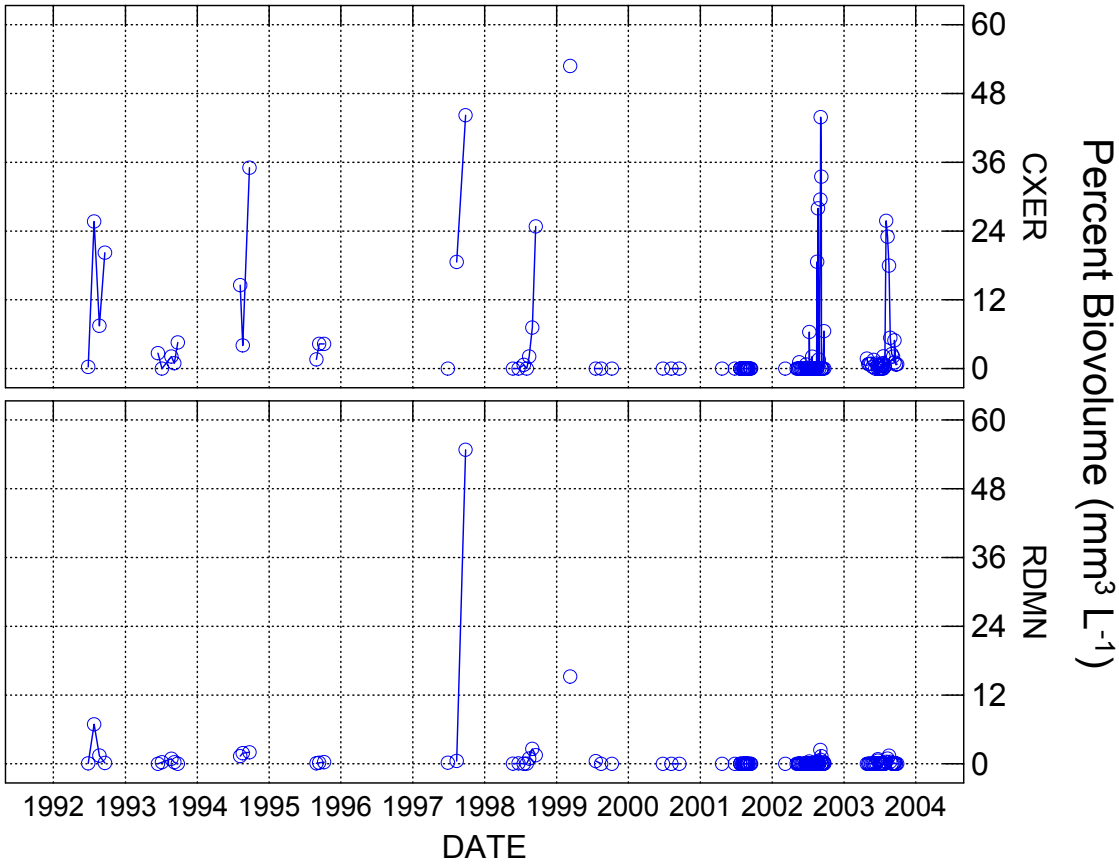
% Chrysophyta



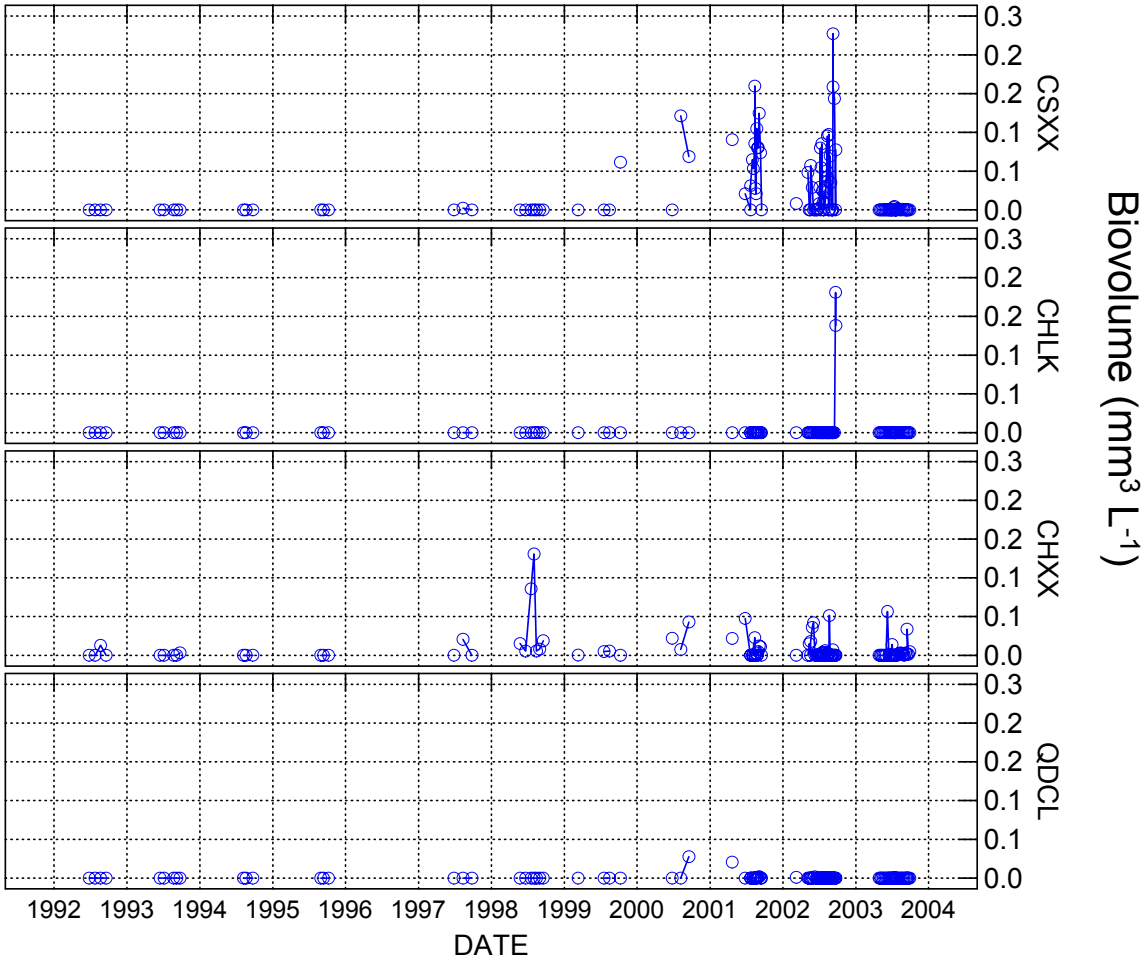
Cryptophyta



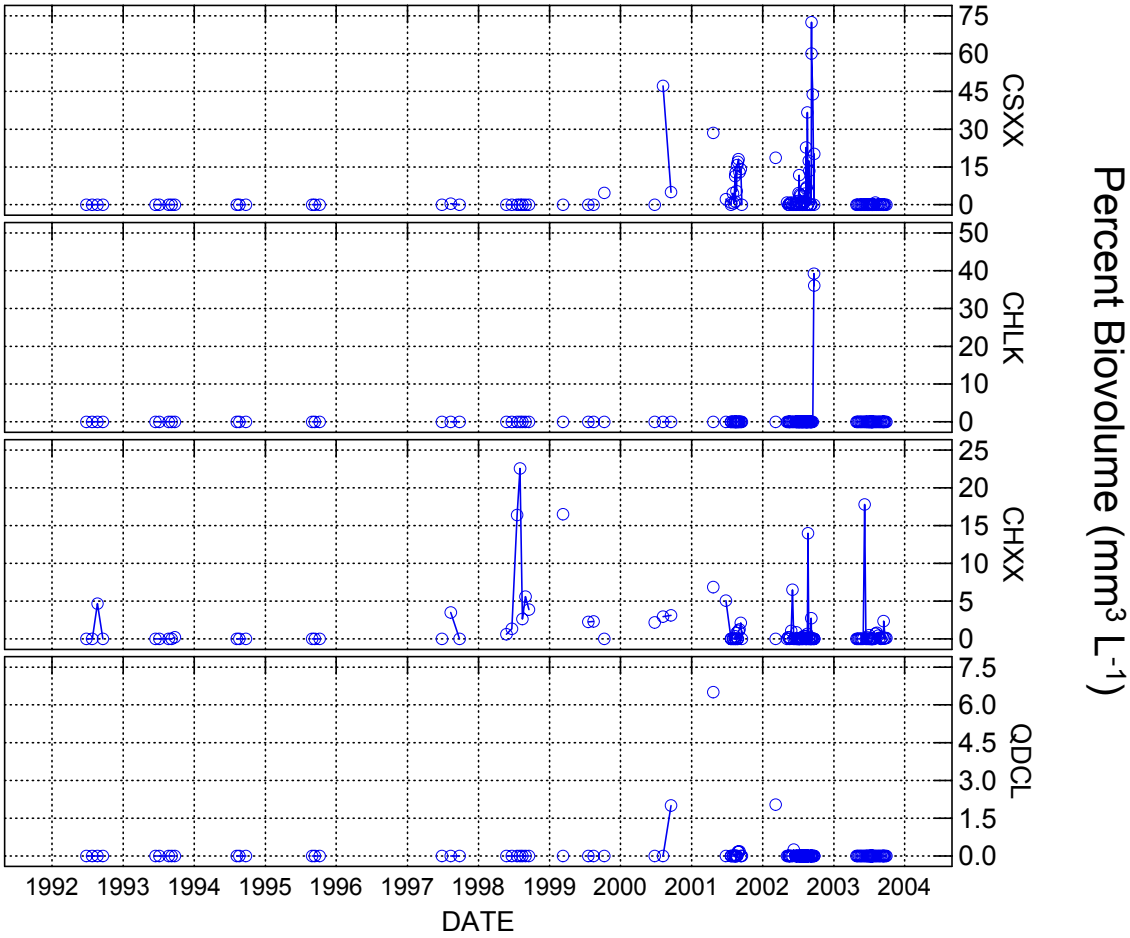
% Cryptophyta



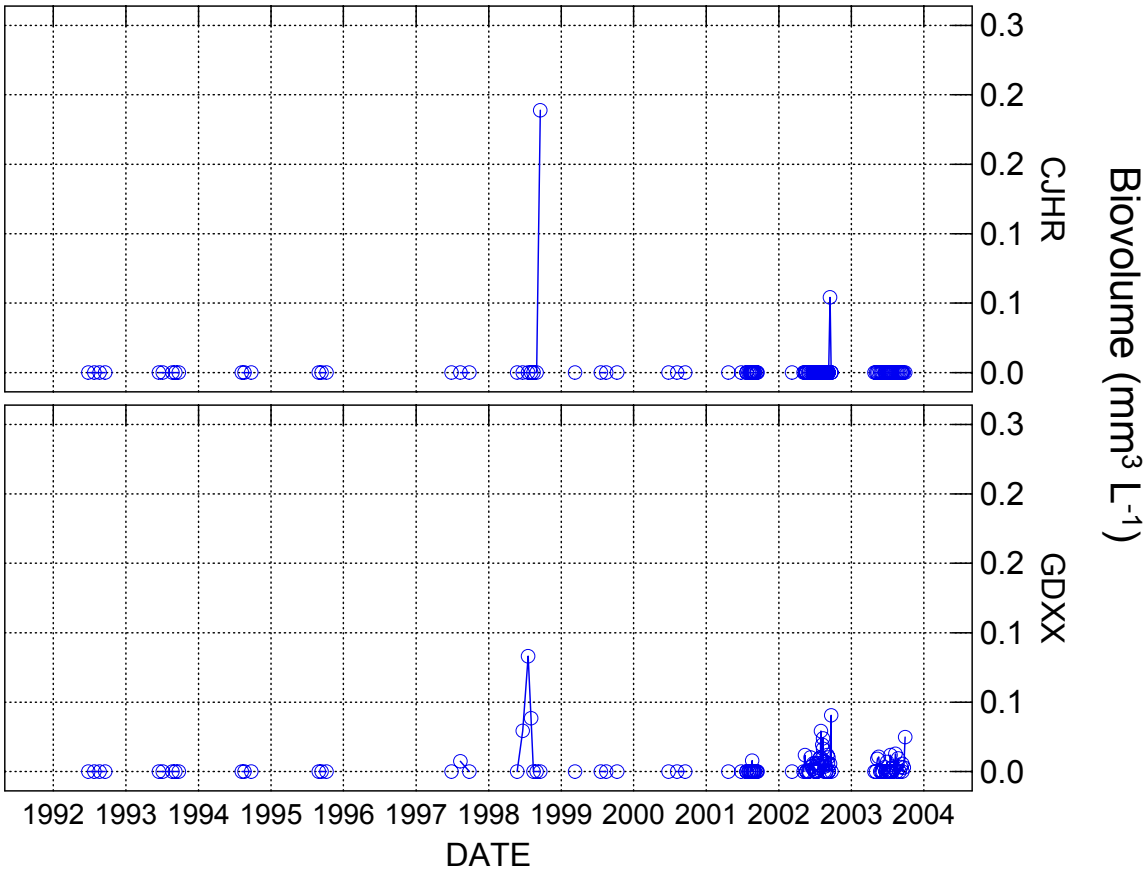
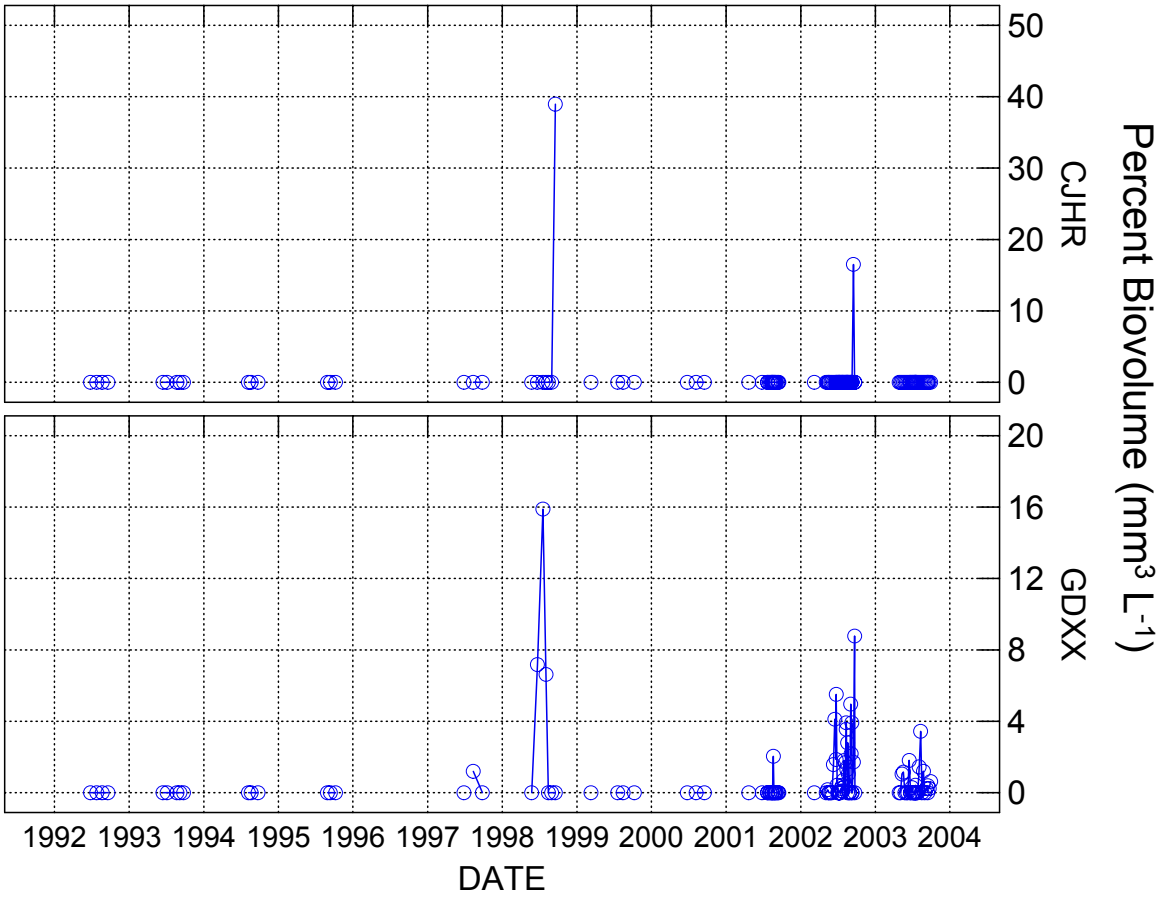
Chlorophyta



% Chlorophyta



Pyrrhophyta



% Pyrrhophyta