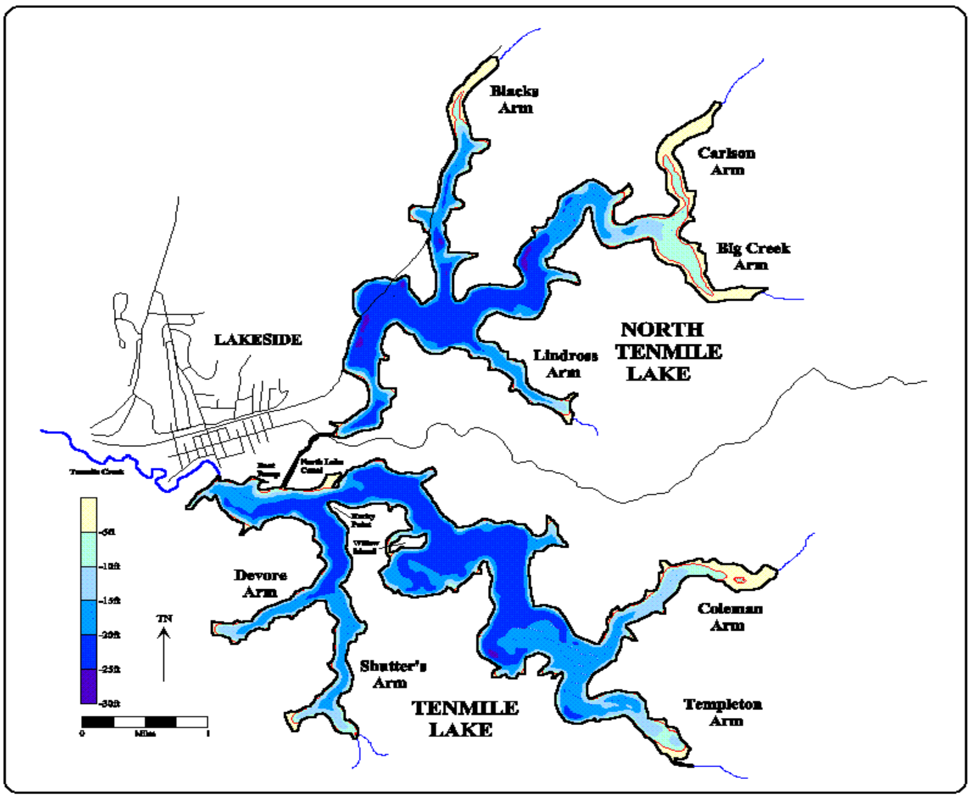
TENMILE LAKES NUTRIENT STUDY

Phase II Report



November, 2002

TENMILE LAKES NUTRIENT STUDY

Phase II Report

Submitted by

Joseph Eilers[a]
Kellie Vaché[b]
and
Jacob Kann[c]

E&S Environmental Chemistry, Inc.
Corvallis, OR

to the

Tenmile Lakes Basin Partnership
Lakeside, OR

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Current Affiliations

[a] JC Headwaters, Inc., 1912 NE 3rd Street, PMB 341, Bend, OR 97701
[b] Oregon State University, Dept. of BioResources Engineering, Corvallis, OR 97331
[c] Aquatic Ecosystem Sciences, LLC, 232 Nutley Street, Ashland, OR 97520

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Phase II of the Tenmile Lake Nutrient Study was funded under a grant from the Oregon Department of Environmental Quality (DEQ) to the Tenmile Lake Basin Partnership (TLBP). Matching support was provided by the City of Lakeside. The project was administered by Mr. Mike Mader, TLBP, through a contract with E&S Environmental Chemistry, Inc. The project contract representative for DEQ was Ms. Pam Blake, Coos Bay. The authors gratefully acknowledge the assistance in conducting the project from Mike Mader (TLBP) and Jim Brown (City of Lakeside). Local landowner, Ms. Sally Thomas, kindly granted permission to use her land to monitor Murphy Creek and assisted with site installation. The Coos County Road Department assisted with access to Benson Creek, and Merle and Sandy Cummings assisted with sampling logistics for Big Creek. Benn Eilers (E&S) assisted with field sampling and GIS analysis.

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EXECUTIVE SUMMARY

A study of Tenmile Lake and its watershed was initiated to better understand the role of the watershed and the lake in generating and processing sediment, phosphorus, and nitrogen. Field work was initiated in November 1998 and extended to August 1999 under Phase I of the study. Phase II of the work was funded to collect additional stream nutrient data, extend the sediment analyses, conduct additional phytoplankton sampling, and revise the SWAT watershed model using updated information on land use and stream chemistry. Three primary tributary sites were selected under Phase I (Big, Benson, and Murphy Creeks) for sampling during base flow and storm events. Johnson Creek was added as a primary site under Phase II. Additional sites were sampled throughout the watershed on a less frequent basis. Four sites were sampled in the lake (two each in North and South Tenmile Lakes) and at the outlet on Tenmile Creek. In addition to sampling for sediment and nutrients, the lake sites were also sampled for phytoplankton and profile conditions (dissolved oxygen, temperature, specific conductance). Sediment cores were collected from both lake basins and analyzed for rates of sediment accumulation, nitrogen, nitrogen isotope (^{15}N), cyanobacterial akinetes, and fossil diatoms. The stream discharge and chemistry data were used to calibrate a watershed model (SWAT) to estimate annual loads of sediment, nitrogen, and phosphorus to the lake.

The results of the stream monitoring showed that Big, Benson, and Johnson Creeks had discharge water with concentrations of sediment and nutrients about tenfold and threefold, respectively, greater than were measured at Murphy Creek. Murphy Creek is distinguished from the other two sites by the presence of a major restored wetland at the stream mouth extending up the channel for about 2.5 km. Maximum concentrations of total suspended sediments at Murphy Creek never exceeded 12 mg/L, whereas values of 580 mg/L and 423 mg/L were measured at Benson and Big Creeks, respectively. Concentrations of both total suspended solids and total phosphorus in the streams were strongly related to stream discharge, whereas stream nitrate concentrations were related to season. Nitrate concentrations were greatest in the fall and declined linearly through the winter and into spring.

Water quality in Tenmile Lake varied considerably in time and space. Water quality was generally the most favorable in winter, although the lake was visibly impacted by high inputs of suspended solids and nitrate. In spring, the lake experienced a major diatom bloom and produced chlorophyll *a* concentrations exceeding 60 ug/L. A second major bloom occurred in late summer; in this case the phytoplankton was dominated by cyanobacteria (blue-green algae).

Despite its relatively shallow depth, periods of quiescence were sufficient to allow significant oxygen depletion below depths of 4 m. In some cases, the bottom waters were anoxic. Secchi disk transparency varied from a high of 4.9 m in November to a low of 0.6 m following a storm. Total phosphorus averaged 25 ug/L in Tenmile Lake. Nearly all measures of water quality were indicative of eutrophic conditions. Water quality in the center of the lakes was generally better than that observed near the mouths of major tributaries. The "up-lake" sites were characterized by high concentrations of total phosphorus, higher chlorophyll, and lower transparency. This was particularly the case at site NTB located at the intersection of the Big Creek and Carlson Arms. The greatest increase in sediment accumulation rate (SAR) occurred in Coleman Arm, near Big Creek. The lowest rate of increase in SAR occurred in Lindross Arm, a site which lacks a major tributary.

Sampling the lake for *Microcystis aeruginosa* in 2000 showed that the populations were relatively low and never exceeded 13% pf the phytoplankton biomass. Despite these findings, concentrations of the toxin, microcystin, exceeded World Health Organization levels for drinking water. The nutrient concentrations in Tenmile Lake are sufficient to support robust populations of *Microcystis* and variations in climatic conditions will now determine its relative abundance at any given time. The single most significant factor (other than nutrient availability) affecting the *Microcystis* populations appears to be lake temperature. Because of the moderately high concentrations of microcystin measured in the lake, considerable caution should be exercised by lakeshore owners using the lake as a drinking water supply.

The analysis of the lake sediments showed that the sediment accumulation rate (SAR) has increased substantially over pre-development conditions. The greatest increase in SAR occurred in Coleman Arm, near Big Creek. The lowest rate of increase in SAR occurred in Lindross Arm, a site which lacks input from a major industry. The sediment shows both an increase in nitrogen and in the $^{15}N/^{14}N$ ratio, suggesting that there has been a qualitative change in the source of nitrogen to the lake. A small number of sediment intervals analyzed for akinetes (cell structures found on some cyanobacteria) showed a major increase in their deposition rate compared to pre-development rates. This is consistent with an increase in the biomass of cyanobacteria in the lake during the latter half of the 20th century. The diatom remains in the lake sediments show an increase in taxa such as *Asterionella formosa*, which are often associated with nutrient-rich waters. The relative abundance of benthic (bottom-dwelling) diatoms have greatly decreased in the lake, which is consistent with a reduction in lake transparency.

The watershed modeling using the SWAT model indicated loads of sediment and nutrients to the lake have increased throughout the watershed. Factors most strongly associated with increased loads include land use disturbances that are persistent and close to the lake or streams. In particular, livestock grazing near the base of the tributaries appears to provide the necessary elements for greatly increasing the sediment and nutrient loads to the lake. Despite the high concentrations of sediment and nutrients measured at the major tributaries, the model indicates that other tributaries may contribute more sediment and nutrients on a weighted-area basis.

The model estimates of nitrogen (N) and phosphorus (P) were compared with unit estimates of N and P associated with septic inputs from shoreline development. On an annual basis, the inputs of nutrients from the septic inputs are relatively modest, representing only about 20% of the watershed loadings. However, during the summer when tributary loads are small, the relative contribution of septic inputs increases to about 50%. Considering that the blooms of cyanobacteria occur in late summer and early fall, it is likely that septic inputs constitute an important component of the nutrient load.

The study consistently showed that sediment and nutrient loads to the lake are substantially elevated above pre-development conditions and that the water quality in the lake has declined accordingly. Measures of water quality in the lake point to eutrophic or near-eutrophic conditions, and the trends indicated in the sediments show that the level of deterioration is continuing.

A. INTRODUCTION

The Tenmile Lakes*, consisting of North Tenmile and Tenmile Lakes, are highly productive lakes located on the south-central Oregon Coast. The lakes have been the subject of fisheries investigations, related to a large degree to declining salmonid fisheries and possible interaction with introduced exotic species (Griffiths and Yeoman 1941, Schwartz 1977, Abrams et al. 1991, Dambacher et al. 1999). In addition to its importance as a noted recreational fisheries, Tenmile Lake serves as a drinking water supply for a number of lakeshore residents and has been investigated as a potential drinking water supply for the city of Coos Bay. However, the quality of the drinking water supply was jeopardized with high populations of the cyanobacteria, *Microcystis aeroginosa*, in 1997 when the lake was temporarily closed to use for potable water (Kann and Gilroy 1998). Lastly, Tenmile Lake has reported water quality problems that have caused it to be listed on the Oregon 303(d) list of impaired surface waters and were noted earlier in a statewide assessment of lakes (Johnson et al. 1985). In summary, a list of the recognized problems in Tenmile Lake includes:

(1) a major reduction of the historical anadromous fisheries (Abrams et al. 1991)
(2) presence of exotic fish species (Abrams et al. 1991)
(3) presence of exotic macrophytes (Systma 1995)
(4) toxic and nuisance algal blooms (Kann and Gilroy 1998)
(5) exceedences of water quality standards and guidelines (DEQ 305b Report).
(6) sediment accumulation at the mouths of streams (based on historical aerial photographs)

The water quality and fisheries problems in the lake have prompted management agencies and the City of Lakeside to take action to address the issues. The actions currently being taken include: (1) stream restoration activities by the Tenmile Lake Watershed Council in conjunction with the Oregon Department of Fish and Wildlife (ODFW) to improve habitat for spawning salmon and their progeny, (2) fisheries investigations by ODFW, and (3) surveys and corrective action of septic systems along the lakeshore by the Oregon Department of Environmental Quality (DEQ). In addition, there have been a variety of citizen efforts, including the Citizen

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* Tenmile Lake is actually two lakes, Tenmile and North Tenmile, which are connected by a channel. They are treated jointly as Tenmile Lake by the Tenmile Lakes Basin Partnership and we follow that convention in this report. Distinctions between the two systems are made here using “north” and “south”, whereas references to both systems are treated as “Tenmile Lake”.

Lakewatch Program, and volunteer efforts to assist in lake and watershed improvement. These actions illustrate the diverse nature of the problems associated with Tenmile Lake and the need to address the problems from multiple avenues. In recognition of the broad-based nature of the problems, the Tenmile Lake Watershed Council sought to develop a more comprehensive understanding of the lake and its watershed. One element of the attempt to better understand this resource involves this nutrient study.

By better understanding important watershed processes affecting the lake, it was hoped that management decisions regarding lake and watershed restoration could be guided by quantitative information. To this end, we developed a study plan designed to provide the following:

(1) measure inputs of nutrients and sediment from selected tributaries to the lake during baseflow and storm runoff,

(2) monitor nutrient concentrations and water quality in the lake,

(3) assess algal community composition in the lake,

(4) measure sediment accumulation in the lake, and

(5) model watershed nutrient and sediment inputs to the lake under current land use, pre-development conditions, and possible future scenarios.

The scope of the study was further influenced by a need to provide the Oregon Department of Environmental Quality (DEQ) information to support an analysis of Total Maximum Daily Loads (TMDLs) for Tenmile Lake. TMDLs are required under the Clean Water Act which stipulates that surface waters listed on a states' 303(d) list of impaired water bodies need to have an analysis completed and action taken leading to an improvement in water quality. Although this study does not constitute the formal TMDL analysis, it was intended to provide useful information in completing such an activity which is scheduled for the area in the near future.

The Phase II portion of this project sought to improve on all five of the primary technical goals listed above by:

(1) monitoring additional stream sites in WY* 2001;

(2) continued monitoring of existing lake and stream sites in WY 2000-2001;

(3) providing more intensive sampling of phytoplankton communities with special attention focused on *Microcystis*;

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* WY = Water Year, which is the period beginning on October 1 and ending on September 30.

(4) conducting additional sediment sampling and analysis of sediment accumulation rates; and

(5) revising the SWAT model output using updated land use information and the additional water quality data presented here.

B. STUDY AREA

Tenmile Lake watershed is located on the south-central Oregon Coast between Coos Bay and Reedsport (Figure 1). The municipality of Lakeside is located on the western shore of the lake at the outlet to Tenmile Creek. The study area is bounded by the Pacific Ocean 4 km to the west and the watershed divide occurs at an elevation of 550 m in the Coast Range. The lake was formed by dunal encroachment on Tenmile Creek during a period of glacial activity extending from about 18,000 BP to 6000 BP (cf. Cooper 1958). The resulting lake is highly dendritic and superficially resembles a reservoir (Figure 2). Average depth is 4.5 m in the north lake and 5.0 m in the south lake. Morphometric properties of the lake and watershed are presented in Table 1 and Figure 3.

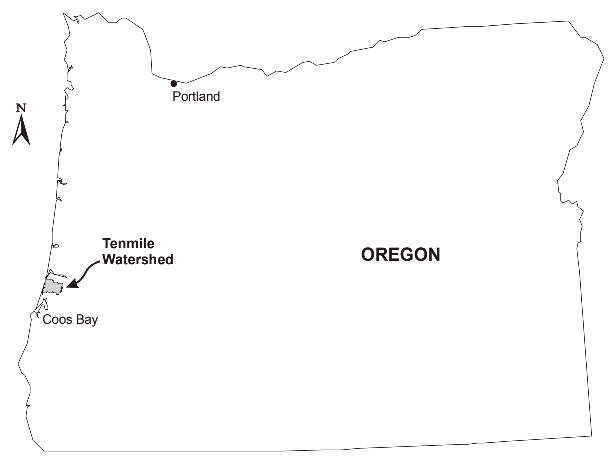


Figure 1. Location of Tenmile Lake.

The climate in the area is characterized by cool, wet winters and mild, dry summers. Annual average precipitation ranges from 200 cm in Lakeside to 245 cm in the eastern uplands of the Coastal Range. The higher elevations occasionally receive snow, although accumulations are brief. Nearly all precipitation in the watershed occurs as rain.

The watershed is characterized by flat, narrow valley floors that extend up to 10 km into the watershed leading to abrupt changes in slope to the surrounding hillsides. Soils in the uplands are derived from uplifted marine sediments and are highly erodible when exposed. The dominant pre-settlement vegetative types are Douglas fir in the uplands and wetland communities in the flat lowlands.

Table 1. Lake morphometry statistics for North Tenmile and Tenmile Lakes based on the May 1995 GPS/SONAR survey. (Source: Eilers et al. 1996a)

	North Tenmile	Tenmile	Combined
Area[a] (ha) (acres)	335.6 (829.3)	457.3 (1,129.9)	792.9 (1,958.2)
Perimeter[b] (km) (mi)	31.83 (19.78)	37.28 (23.17)	69.11 (42.95)
Depth,[c] maximum (m) (ft)	8.17 (26.8)	8.23 (27.0)	-
Depth,[c] mean (m) (ft)	4.50 (14.75)	4.98 (16.33)	-

[a] Excluding an area for Willow Island. Two previous digitized versions of the lake area are within 1% of these values (J. Kelsey, pers. comm.).
[b] Includes Willow Island and the channel connecting the two lakes. Digitized from 1:24,000-scale USGS topographic map. Even considering variation associated with scales used on the source maps, actual miles of shoreline are probably within 2% of these measurements.
[c] Normalized to a lake elevation of 9.0 ft MSL

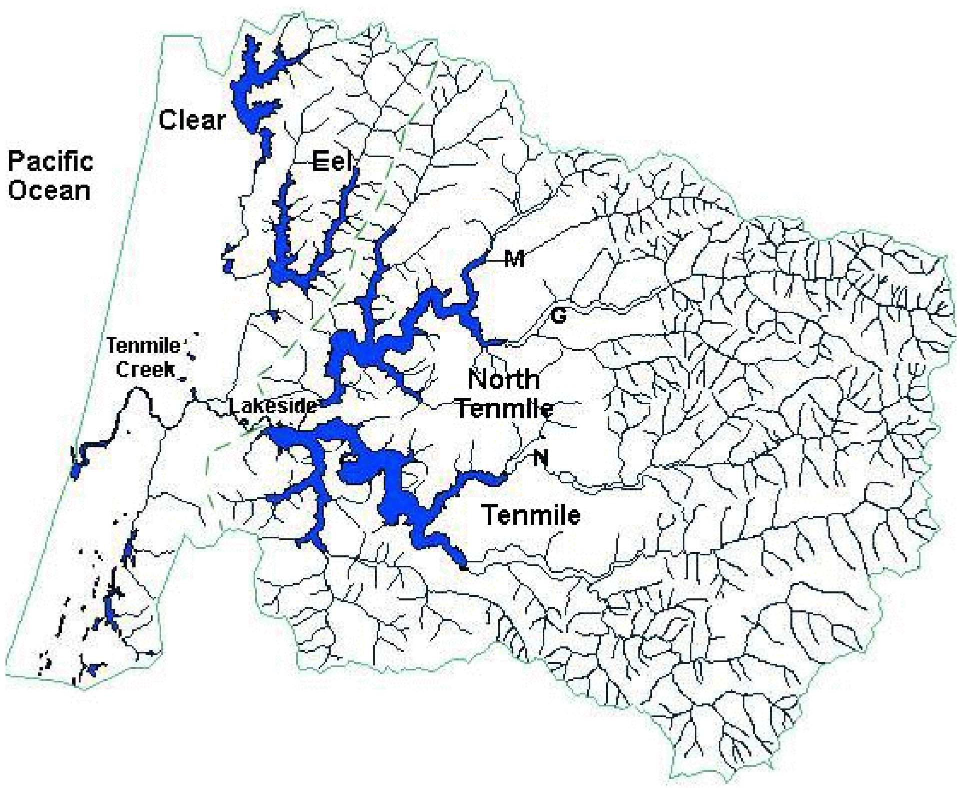


Figure 2. The Tenmile Creek watershed including the Tenmile Lake watershed shown as the dashed line. Clear Lake and Eel Lake discharge to Tenmile Creek west of Lakeside and downstream of Tenmile Lake. Three of the major stream sampling sites are identified as “M” (Murphy Creek), “G” (Big Creek), and “N” (Benson Creek).

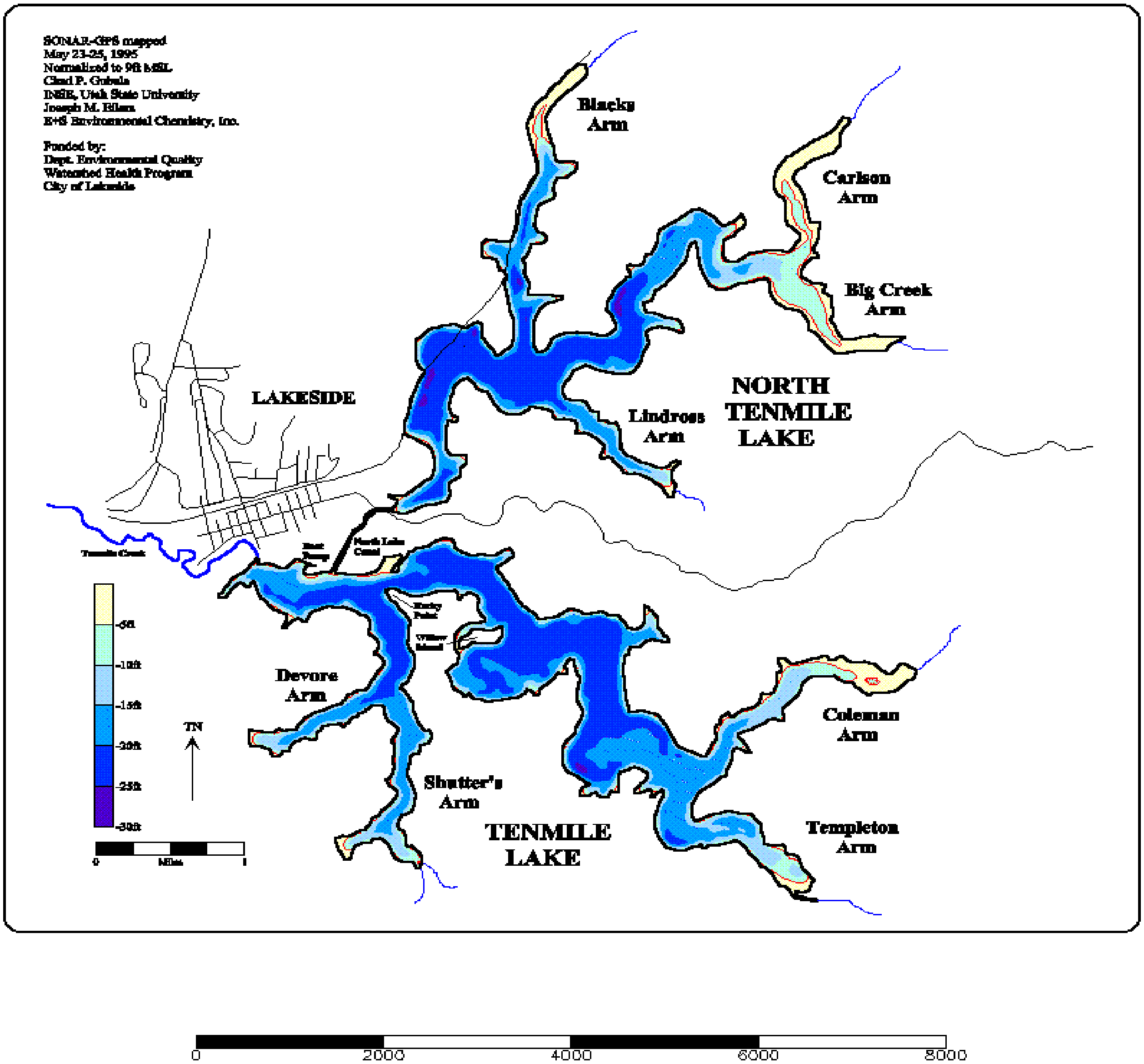


Figure 3. Bathymetric map of Tenmile Lake.

The present land use in the watershed is comprised of timber management in the uplands and agriculture in the valleys. The eastern portion of the watershed is within the Elliott State Forest and is actively managed for timber harvest. Private land holdings to the west of the state forest are also subject to timber harvest. Agricultural land is principally used for livestock grazing. The livestock used to consist primarily of dairy cattle. That has transitioned to beef cattle, although there are some sheep and horses present. High-density urban development is restricted largely to the City of Lakeside on the eastern shore. Lakeshore development is widespread on much of the accessible, upland portions of the shoreline. Approximately 500 dwellings are present on the lakes, divided nearly equally between the north and south basins.

The lake is used extensively for recreation, primarily fishing. The watershed historically was a major producer of coho salmon, with runs of over 75,000 fish (Figure 4). The lake also had substantial runs of steelhead and sea-run cutthroat. Following the introduction of largemouth bass, coho escapement into Tenmile Lake has remained below 10,000 adults and jacks (Abrams et al. 1991). Tenmile Lake is currently an important site for bass fishing tournaments in Oregon. The fisheries history is summarized in Table 2.

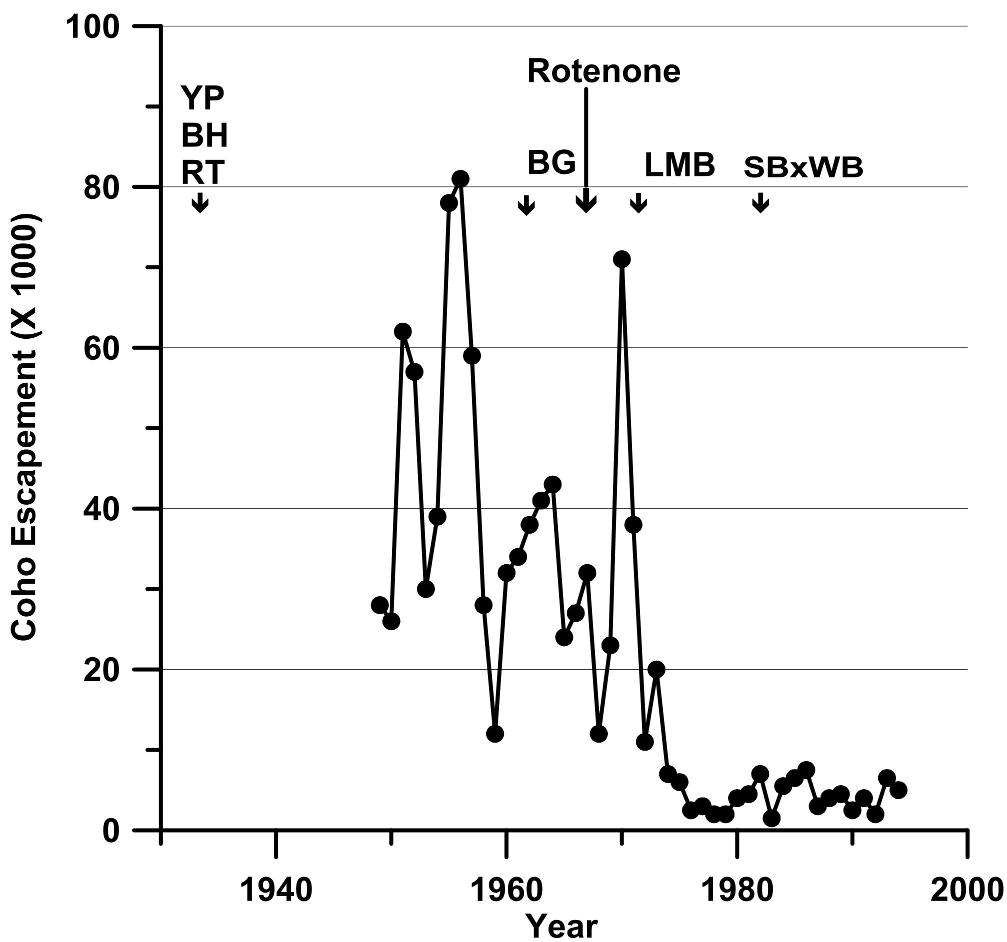


Figure 4. Coho escapement (return) to Tenmile Lake over the last 50 years (after Abrams et al. [1991]), where YP=yellow perch, BH=brown bullhead, RT=rainbow trout, BG=bluegill, LMB=largemouth bass, and SMxWB=smallmouth bass x white bass hybrid.

Table 2. Fisheries history of the Tenmile Lakes (derived from Griffiths and Yeoman [1941] and Abrams et al. [1991]).

Species	Native or Introduced	Year Introduced	Management
Coho Salmon	Native	—	Population declined from >75,000 adults/yr to about 4,000. Smolts and pre-smolts stocked.
Winter Steelhead	Native	—	Current population estimated at 20% of 19th century levels; currently stocked
Cutthroat Trout	Native	—	Historically abundant population; currently managed for wild stock
Rainbow Trout	Introduced [a]	1930's	Currently managed as a "put-and-take" fishery.
Brown Bullhead	Introduced[b]	1930's	High population was not impacted by a commercial fishery in 1952-53; not eradicated by rotenone; continued abundant population
Yellow Perch	Introduced[b]	1930's	Large populations developed
Bluegill	Introduced[b]	1960's	Currently most abundant fish in the lakes; attempt to eradicate the species in 1968 with rotenone was unsuccessful
—	—	1968	Rotenone treatment to eradicate introduced species (Montgomery 1969)
Largemouth Bass	Introduced[a]	1971	Highly successful fisheries
Hybrid Bass[c]	Introduced[a]	1982	Successful fisheries discontinued after 1988 because of concerns for hybrids straying into other river systems. No longer present.[d]
Miscellaneous Native Species eulachron (*Thaleichthys pacificus*) staghorn sculpin (*Eptocottus armatus*) threespine stickle (*Gusterosteus aculeatus*) green sturgeon (*Acipenser medirostris*) Pacific lamprey (*Lampetra tridentata*) western brook lamprey (*Lampetra richardsoni*) prickly sculpin (*Cottus asper*) shiner perch (*Cymatogaster aggregata*)			Populations unknown

[a] managed introduction
[b] illegal introduction
[c] Striped bass x white bass
[d] Michael Grey, ODFW, personal communication, Nov. 2001

C. METHODS

1. Sampling Design

The Tenmile Lake watershed is highly dendritic and the task of sampling all tributaries would be prohibitively expensive. We elected to use a model-based approach in which selected catchments would be monitored and the results from these systems would be used to calibrate a watershed model for computing annual loads of sediment and nutrients to the lake. In addition, we monitored the lake to assess the fate of the watershed inputs and used the information from the sediment history to infer changes in water quality in the 20th century. The sediment data are also used to constrain the model for historical inputs to the lake.

The sampling design focused on two classes of water quality constituents, sediments and nutrients. These are not necessarily mutually exclusive since the sediments transport particulate forms of nutrients, especially total phosphorus and organic nitrogen. The decision to focus the sampling on these sets of constituents was based on observations of erosion and mass failures in the watershed, extensive streambank erosion, siltation at the mouths of some of the tributaries, the presence of algal blooms which are often associated with excessive nutrient inputs, and previously collected data documenting water quality problems in the lake related to macrophytes, algal blooms, and high nutrient concentrations (Johnson et al. 1985; Systma 1995; Eilers et al. 1996a; DEQ, unpublished data).

2. Site Selection

The approach was to select stream monitoring sites that drained major portions of the watershed, were representative of the land cover types in the watershed, and were accessible. Two sites were initially selected that met these criteria, Benson Creek and Big Creek. The Benson Creek site is located at a bridge crossing on Benson Creek Road approximately 1.6 km from the south lake basin on Coleman Arm. The Big Creek site is located at the bridge crossing of North Lake Road about 1.8 km above the entrance to the north lake on Big Creek Arm. These two sites collectively represent 32 percent of the drainage to Tenmile Lake and appear to be representative of the combination of timber management found throughout the uplands and livestock grazing typical of the lowland valleys. A third site was selected that also included timber management, but lacked the influence of domestic livestock. This site was located on Murphy Creek about 200 m from the north lake on the Carlson Arm. There are no road crossings on Murphy Creek and access was by boat. This site is located on private land;

permission was obtained from the landowner, Ms Sally Thomas. Tenmile Creek, located at the outlet of Tenmile Lake also was sampled on a regular basis to evaluate changes occurring in the lake. Under Phase II, a site was established on Johnson Creek at a private bridge crossing about 15 km upstream of the mouth.

The above sites formed the core sites for the stream monitoring effort, but additional sites were sampled on one or more occasions to gather some basis for assessing the representativeness of these core sites and to attempt to characterize variability in water quality throughout the watershed. Water samples were collected from public roads at the sites shown in Figure 5. Unlike the core sites which were instrumented, no stream discharge or other real-time data were collected at the supplemental sites. The distribution of water quality samples collected during Phase I and Phase II of the study is shown in Table 3.

From the standpoint of morphometry and zonation, Tenmile Lake bears some resemblance to a reservoir. Because of this, we sought to measure not only differences between the two main basins, but also to evaluate changes that occur between the major arms and the mid-basin areas.

Table 3. Distribution of water quality samples collected during Phase I and Phase II of the Tenmile Lake study.

	Year				Total
	1998	1999	2000	2001	
Stream Sites					
Benson (BEN)	21	63	39	1	124
Big (BIG)	4	52	26	1	83
Murphy (MUR)	2	51	1	1	55
Johnson (JON)			21		21
Benson - trib (REX)		12	5	1	18
Tenmile Creek (TCO)	1	14			15
Others (N=13)	5	15	1		21
Lake Sites					
South Tenmile (A)	2	13	11	2	28
(B)	2	11	3		16
North Tenmile (A)	1	12	5	1	19
(B)	2	10	3	1	16
Others (2)	1	1	1		3

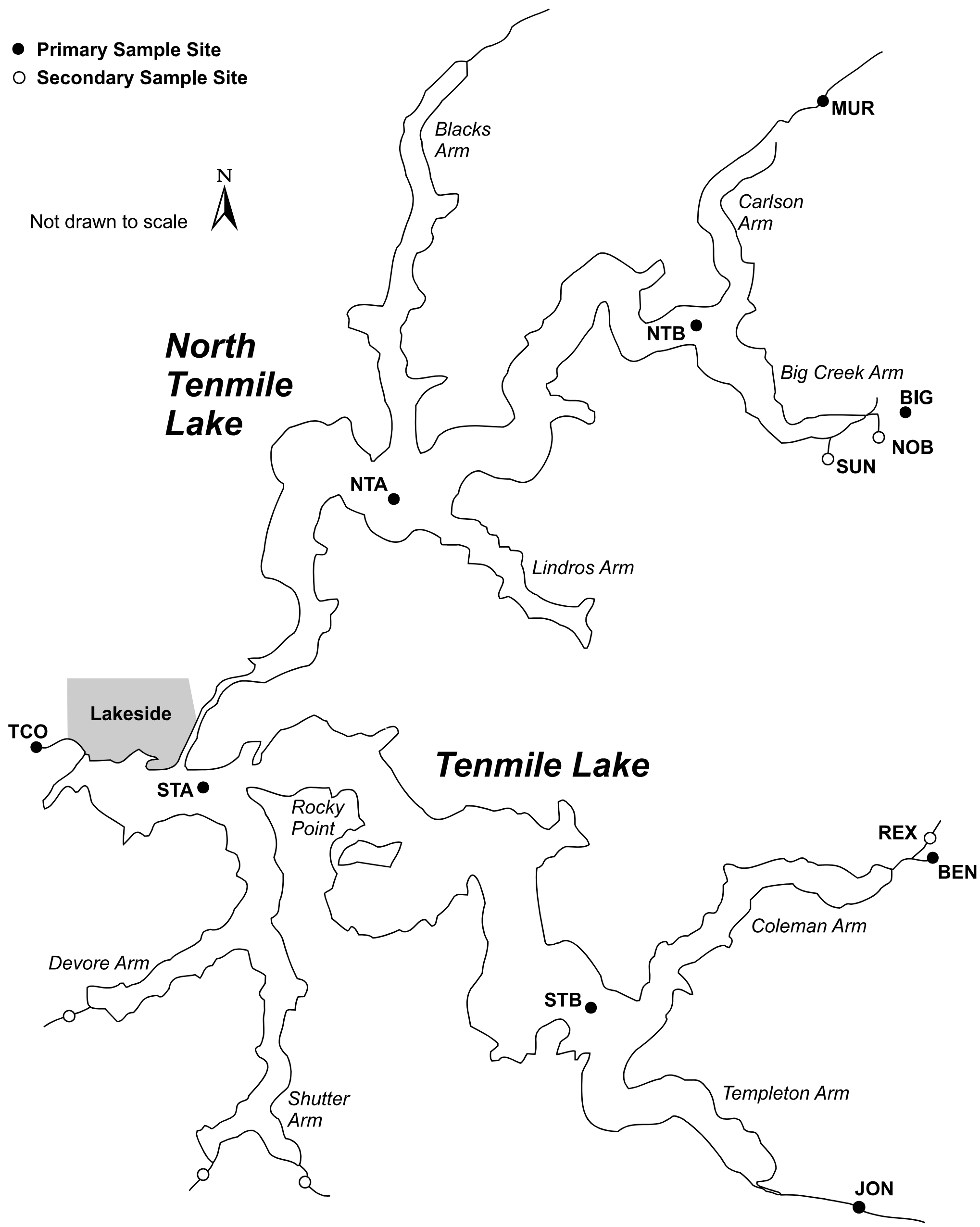


Figure 5. Phases I and II water quality monitoring sites, Tenmile Lake watershed.

Four lake sites were selected to represent conditions in both basins and to compare mid-lake and transitional zones (Figure 5).

3. Field Instrumentation

Most of the inputs of sediment and nutrients to lakes in western Oregon occur during storms. In some cases, 90 percent of the annual discharge of sediment from a watershed can occur within a several day period. The rapidity of response time in these watersheds required us to sample changing water quality conditions intervals of several hours during storm events lasting several days or longer. This frequency of sampling at multiple locations could only be accomplished with instruments designed to collect water samples at prescribed times and through use of continuously recording devices for temperature, precipitation, and streamflow.

Water sampling instruments, ISCO (Model 6700), were installed at Big, Benson, and Murphy Creeks and powered with deep cycle 12 volt batteries. The units were housed in plywood shelters located several feet above expected maximum stream elevation. Intake tubes extended into mid-channel and were secured using lead weights to insure that samples were always drawn from roughly the same positions in the channel and with respect to the stream bottom. After observing the general response times of the creeks, sampling intervals were programmed for every four hours. The units contained 24 1-L bottles which allowed us to sample continuously for up to four days prior to either processing the collected samples or retrieving the samples and resetting the ISCO for further operation.

Stream stage recorders were installed at Big, Benson, Murphy, and Tenmile Creeks. The Solinst pressure transducers were installed at Big and Benson Creeks. A Stevens Level Recorder was installed at Murphy Creek and a Global Water pressure transducer was installed at Tenmile Creek. Hobo Temps, temperature recording devices, were installed in Big and Benson Creeks. Precipitation gages were installed at Benson and Murphy Creeks (elevation ~ 6 m) and off Benson Ridge Road at an elevation of about 300 m.

4. Field Methods

ISCO samplers were prepared for use by thoroughly rinsing the sample bottles with stream water followed by either deionized or distilled water. Water samples from the ISCOs were transferred to 1 L Nalgene® bottles that were either new, or if used previously, had been acidwashed. Water samples were placed in coolers for transport back to Corvallis (or shipping to other laboratories), usually on the same day of collection.

Stream discharge was measured at the primary sites using a Marsh-McBirney (Model 2000; Flo-Mate) flowmeter with measurements of stream velocity at 0.6 of the depth at a given site at multiple sites across the channel. Discharge measurements within the cells were summed across the channel. Electronic data stored on the temperature, precipitation, and pressure transducer gages were periodically downloaded in the field to a laptop computer.

The lake was sampled at the four sites shown previously in Figure 5. Most lake samples were near-surface samples (0.5 m), except in cases of apparent stratification where samples 1 m above the bottom were also collected. Samples were typically collected using a trace-metal grade Plexiglas Van Dorn sampler and distributed to several aliquots. Samples collected for major ions, nutrients, and total suspended solids were placed in 1 L Nalgene® bottles. Samples of phytoplankton were placed in 250 mL bottles and preserved with Lugol's solution and samples of chlorophyll *a* were placed in 50 mL Nalgene® bottles and preserved with magnesium bicarbonate. Zooplankton samples were collected from each lake site in July and August 1999 using a plankton net (80 µ mesh size) equipped with a modified Wisconsin bucket and a 10 cm opening. Secchi disk measurements were made using a standard 20 cm disk. Vertical profiles were made of temperature, dissolved oxygen, and specific conductance using YSI field meters (Model 85) after calibrating according to manufacturers recommendations. All field activities were described in a written sampling plan that was provided to field personnel.

5. Analytical Methods

Water samples were analyzed according to the need to quantify key inputs to Tenmile Lake. High priority analytes were total suspended solids, total phosphorus, and nitrate-nitrogen. These analytes were generally run on most samples. Other analytes that were measured on a less-frequent basis include ammonia, total Kjeldahl nitrogen, major ions, pH, specific conductance, and silica (Table 4). Duplicate and blank samples were included among the routine samples as checks on the quality of the analytical results. Quality assurance protocols and analytical methods were detailed in a QA/QC plan submitted to the TLBP during the early phases of the project.

Table 4. Chemical methods and detection limits for analysis of samples.

Parameter[a]	Method	Detection Limit	Reporting Unit
pH, lab	Electrode	-	s.u.
Alkalinity, tot. as $CaCO_3$	Titration, double end-point	2	mg/L
Conductivity, lab	Platinum electrode	1.0	µS/cm
Calcium, as Ca^{2+}	AA flame	0.05	mg/L
Magnesium, as Mg^{2+}	AA flame	0.05	mg/L
Sodium, as Na^{+}	Flame emission	1.0	mg/L
Potassium, as K^{+}	Flame emission	0.5	mg/L
Sulfate, as SO_4^{2-}	Ion chromatography	1	mg/L
Chloride, as Cl^{-}	Ion chromatography	0.2	mg/L
Nitrogen, $NO_2 + NO_3$ as N	Ion chromatography	0.05	mg/L
Nitrogen, NH_4 as N	Perstorp (SM4500)	0.01	mg/L
Nitrogen, Kjeldahl as N	BD-40 auto. phenate	0.05	mg/L
Phosphorus, dis. react as P	Ascorbic acid	0.002	mg/L
Phosphorus, tot. as P	Digest./ascorbic acid	0.002	mg/L
Silica, as Si	AA flame	1	mg/L
Solids, tot. susp. (TSS)	Gravimetric 103C	2	mg/L

[a] Bolded parameters indicate core analytes; italicized parameters indicate supplement analytes.

Sediment samples were dated using ^{210} isotope. ^{210}Pb was analyzed using alpha spectroscopy (Eakins and Morrison 1978), which involves distillation of the sample, HNO_3 digestion, and plating onto silver prior to counting. The sediment ages and accumulation rates were calculated using the constant-rate-of-supply (CRS) model of Appleby and Oldfield with old age dates, using the method described by Binford (1990). Diatoms were analyzed according to protocols developed for the Paleoecological Investigation of Recent Lake Acidification (PIRLA) Program (Charles et al. 1990).

6. Plankton

Phytoplankton was collected as split samples with the lakewater samples. One 125 mL aliquot was preserved with Lugol's solution for analysis of phytoplankton community composition. Subsamples were withdrawn and counted on slides prepared as permanent mounts until 100 individuals were counted. Biovolume estimates were generated using published cell volumes.

Aliquots for chlorophyll *a* were collected in 60 mL amber Nalgene® bottles and preserved with magnesium carbonate. The samples were stored on ice and shipped via overnight courier. Both phytoplankton community composition and chlorophyll *a* measurements were conducted by Aquatic Analysts, Portland, OR.

Separate phytoplankton samples were collected for analysis of *Microcystis* and microcystin, the toxin produced by the cyanobacteria. Sampling was initiated during the summer of 2000 to coincide with other nutrient and phytoplankton sampling being conducted as part of the overall study (see description above). Four stations (2 in each lake) were sampled to cover a major arm and open-water location in each lake. The two open water stations, designated S8 and N16, coincide with stations STA and NTA of the overall study (Figure 5). Stations S3 and N11 were located near the terminus of Templeton Arm and Big Creek Arm, respectively. These stations were sampled four times beginning July 21st and ending September 4th, 2000. Because the goal of the *M. aeruginosa* sampling was to detect conditions that may pose human health hazards, samples were collected mid-day over the top 1 m of the water column. An integrated raw-water sample of the top one-meter of the water column was collected by compositing 6 hauls of a 4 cm diameter tube sampler. A subsample was drawn and shipped overnight air to the Wright State University laboratory of Dr. Wayne Carmichael, who performed the enzyme linked immunosorbent assay (ELISA) test for microcystin toxins. An additional sample was collected by combining 3 one-meter hauls of a Wisconsin style plankton net with 64 μm mesh. These samples were preserved in Lugol's Iodine and sent to BSA Inc. in Cleveland Ohio, where microscopic analysis was performed for *M. aeruginosa* colony abundance and average colony size (greatest axial linear dimension or GALD). An estimate of biomass was then computed as colony abundance multiplied by GALD.

Several zooplankton samples were collected to provide a characterization of the dominant taxa present. Samples were collected using a 10 cm conical net (Seattle Nets; 80 μ mesh) equipped with a modified Wisconsin bucket. Samples were preserved with ethanol and analyzed by ZPT Taxonomic Services, Salem, OR.

7. Watershed Modeling

A variety of models are available for estimating watershed loads of nonpoint source pollution ranging from simple unit-based loading coefficients to detailed simulation models that attempt to represent transport processes as realistically as possible. As the level of model sophistication increases, so do the requirements for increasingly detailed spatial and temporal data on the watershed. The object of using more complex models is to gain greater insight into the watershed processes and therefore more accurately assess the nature of the problems. However, if the level of data acquisition is not matched to the model, increasing the level of model sophistication can actually be counterproductive.

After evaluating a number of watershed models, we selected a USDA model called SWAT (Soil and Water Assessment Tool). SWAT is a model of intermediate complexity designed to be used in estimating sediment and nutrient loads in large watersheds. SWAT also has the capability to simulate pesticides, bacteria, and organic loads (not conducted on this project). SWAT is the product of model evolution associated with a long-term effort by the USDA to predict nonpoint sources of pollution. Some of the related models that preceded SWAT and whose code is incorporated to some degree in the current model include CREAMS (Chemicals, Runoff and Erosion from Agricultural Management Systems; Knisel 1980), GLEAMS (Groundwater Loading Effects on Agricultural Management Systems; Leonard et al.1987), EPIC (Erosion-Productivity Impact Calculator; Williams et al. 1985), AGNPS (Agricultural Nonpoint Source; Young et al. 1989), SWRRB (Simulator for Water Resources in Rural Basins; Williams et al. 1985) , and ROTO (Routing Outputs to Outlet; Arnold et al. 1995a).

SWAT has eight major components: hydrology, climate, sedimentation, soil temperature, crop growth, nutrients, pesticides, and agricultural management practices. Surface runoff is generated using a modification of the SCS curve number method (USDA Soil Conservation Service 1972), which incorporates non-linear watershed response to varying antecedent moisture conditions. Peak runoff is predicted based on a modified Rational Formula using Manning's Formula to predict time of concentration. The model accounts for routing of water through percolation into multiple soil layers and a shallow aquifer compartment. Processes reflected in the hydrology include lateral subsurface flow, groundwater flow, evapotranspiration, snowmelt, and temporary storage in ponds.

Sediment yield is estimated for each subbasin using the Modified Universal Soil Loss Equation (MUSLE; Williams 1975). Cropping factors and other factors affecting erosion follow the procedures described by Wischmeier and Smith (1978). Details of the SWAT model are described further in Arnold et al. (1995b).

The model was calibrated by sequentially and iteratively fitting observed versus simulated climate, hydrology, sediment transport, and nutrient transport. Data were withheld from the calibration to verify the robustness of the model output. Model inputs include data related to land use, topography, soils, and climate. Sources of these data sets are described below.

Soils: Natural Resources Conservation Service (NRCS) SSURGO data (1:24000) were used as the basic soils data. Data from Coos and Douglas counties were gathered and clipped to the Tenmile Lake basin. Currently, data from Coos County are still considered provisional by NRCS. A series of spot checks of the digital data against the hardcopy Soils Survey of the county was used as verification of the provisional data.

Land use: The composite land use data was developed from two separate data sources. Where available, detailed information from the Oregon Department of Forestry (ODF) was utilized. The dataset, called STANDS98, is available from the Oregon State Service Center for Geographic Information Systems (SSCGIS). In areas where this coverage was unavailable, land use information was photointerpreted from 1994 aerial photographs.

Elevation: A 30 meter digital elevation model (DEM) was developed as input for the SWAT model. The data originated at the SSCGIS as 18 United States Geologic Survey (USGS) quadrangle coverages. These datasets were integrated to generate a complete coverage for the watershed.

Climate: Model calibration was performed using daily rainfall and air temperature data collected near the Big and Benson Creek monitoring stations. Precipitation was collected using a tipping bucket attached to a digital data logger. Air temperature was collected at approximately hourly intervals using HoboTemp data recorders. Precipitation for Phase II was derived from the North Bend NOAA site.

D. RESULTS

1. Climate and Hydrology

Precipitation in the region is characterized by large frontal systems that often last for three to five days. Precipitation events that were particular noteworthy during the study period included storms of December 27-28, 1998 (70 mm); January 14-23, 1999 (175 mm); February 2-9, 1999 (123 mm); February 21-28 (101 mm); and March 24-30, 1999 (78 mm). These storms were often accompanied by high winds, which in some cases caused temporary damage to collection devices and/or limited the access to the sampling sites. We sampled three of the five major storms during the study period under Phase I (January and both February storms).

These storms generated high discharge in Big and Benson Creeks which would sometimes increase in flow by an order of magnitude within eight to twelve hours following initial precipitation (Figures 6 and 7). The rapid increase in stream discharge at these sites would be accompanied by large loads of woody debris which also posed some problems for the sampling equipment. Stream flows would increase to bank-full discharge with concomitant increases in stream velocity. The streams would become visibly muddy from both runoff entering the streams and bank erosion, which was evident from measured stream velocity following high flow events. Mass failures were evident in a number of areas around the watershed, particularly along existing roadways and former logging roads. Other ungaged tributaries in the watershed appeared to respond similar to that observed in Big and Benson Creeks.

The discharge response in Murphy Creek was in sharp contrast to that observed in Big and Benson Creeks. The stream channel for Murphy Creek adjacent to the sampling site was narrow (2 m wide), relatively deep (1 m), and highly vegetated. Its capacity to carry discharge was very limited and consequently stream discharge would often exceed the stream channel, resulting in sheet flow extending across the broad marsh. Because of the channel geometry, maximum stream velocity in Murphy Creek was about 0.6 m/s, compared to values up to 1.2 m/s in Big and Benson Creeks.

Lake stage in Tenmile Lake fluctuated considerably in response to seasonal precipitation and possibly from backwater effects associated with high winter tides. The large seasonal fluctuation in lake stage corresponds with considerable variation in the hydraulic residence time in the lake. High discharge from the tributaries in the winter reduce the hydraulic residence time in Tenmile Lake to approximately 15 days, compared to a residence time of 30 days in the spring and 300 days in the summer.[*] These fluctuations result in important seasonal variations in lake chemistry that are described later.

In Phase II, climatic conditions were radically different than observed in Phase I (Figure 8). Water Year 2001 was very dry from fall 2001 through January 2001, the last date of sampling. Precipitation for WY 99 was 69.4 inches (at North Bend) compared to only 55.9 inches in 2000

[*] $\tau_w = V/Q = 37.9 \times 10^6 m^3 / 2.44 \times 10^6 m^3/d$ at $28.3\ m^3/s$ (1000 cfs)

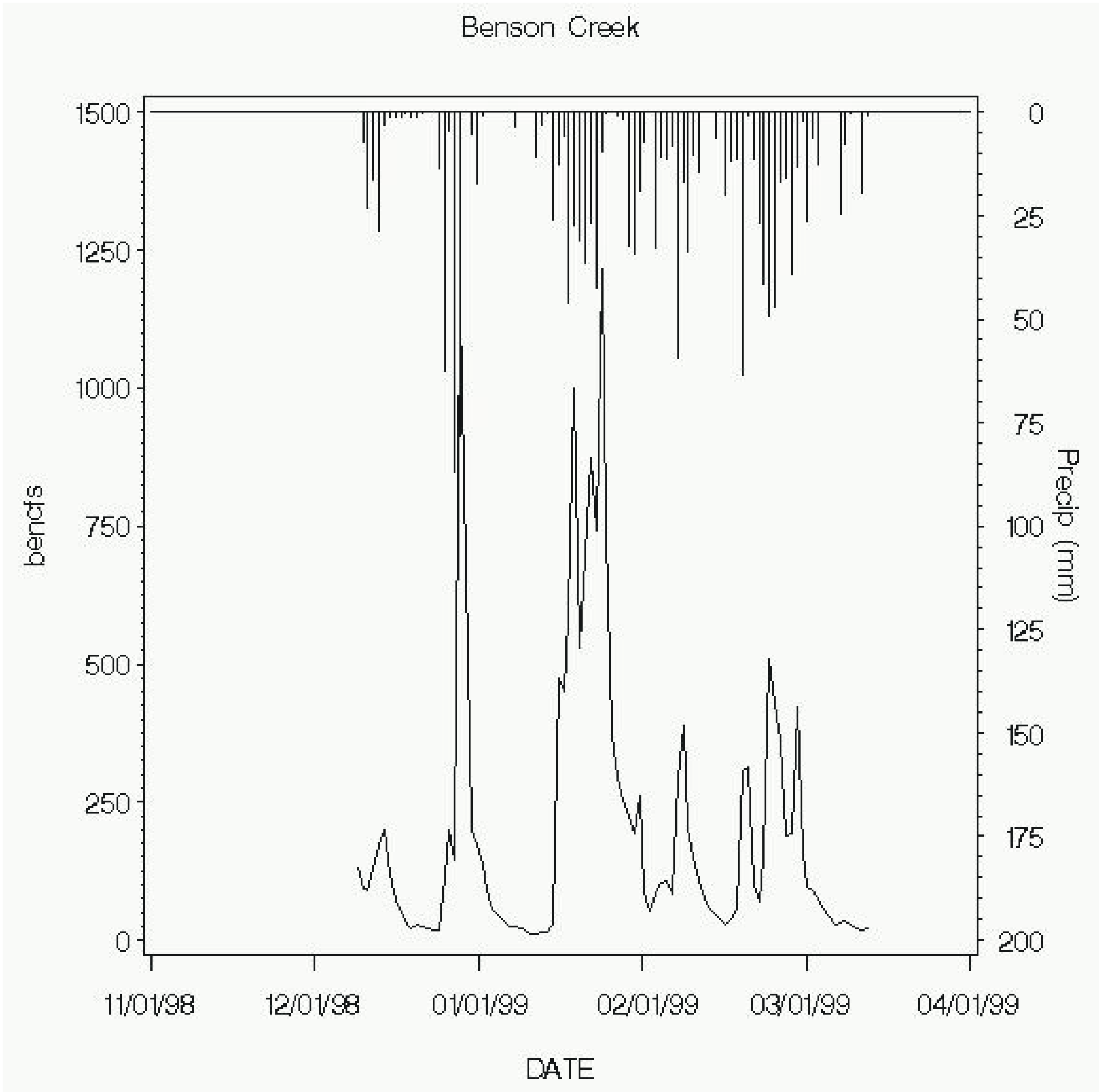


Figure 6. Stream discharge in Benson Creek (cfs) and precipitation collected on-site.

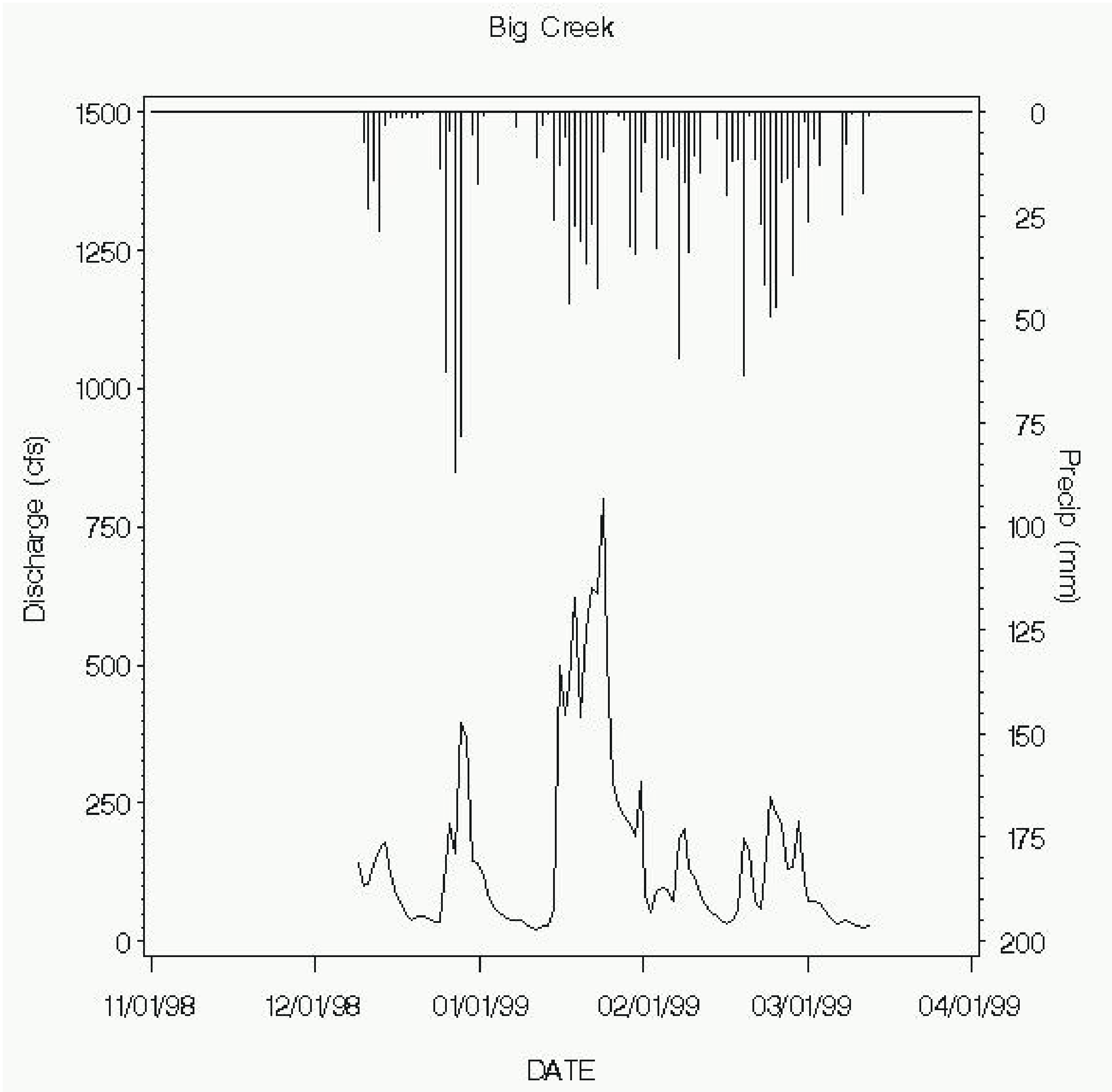


Figure 7. Stream discharge in Big Creek (cfs) shown with precipitation data measured at the Benson Creek site.

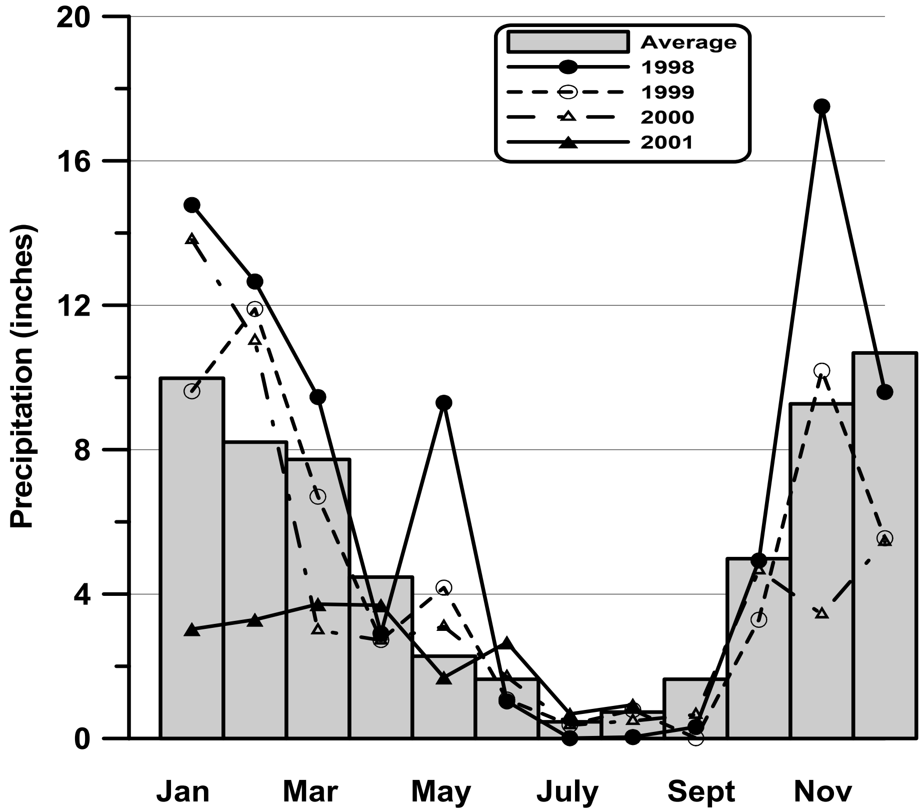


Figure 8. Monthly precipitation measured at North Bend for 1998 - 2001 compared with the long-term average (bars) for the site.

and 33.2 inches in 2001. The average precipitation for North Bend (1931-1993) is 62.1 inches. Note that the precipitation at North Bend probably underestimates precipitation for Tenmile Lake watershed of two factors: (1) greater precipitation occurs near the proximity of the Umpqua River as shown by a 25% greater precipitation at Reedsport and (2) the orographic effects of the Coastal Range intercepts more precipitation as the systems move inland. One of the objectives of Phase II was to obtain storm runoff data in the fall to observe and document the variations in precipitation in the watershed. However, vandalism and equipment failure prevented us from completing this task.

The long-term climate data for the Oregon Coastal zone illustrates that wide fluctuations in annual precipitation are typical for the region (Figure 9). However, it is important to recognize the climatic conditions under which the Phase I and Phase II data were collected. Precipitation in WY2001 remained well below normal and over-bank discharge at the monitoring sites was not observed. Nevertheless, the data collected during Phase II were useful in testing certain

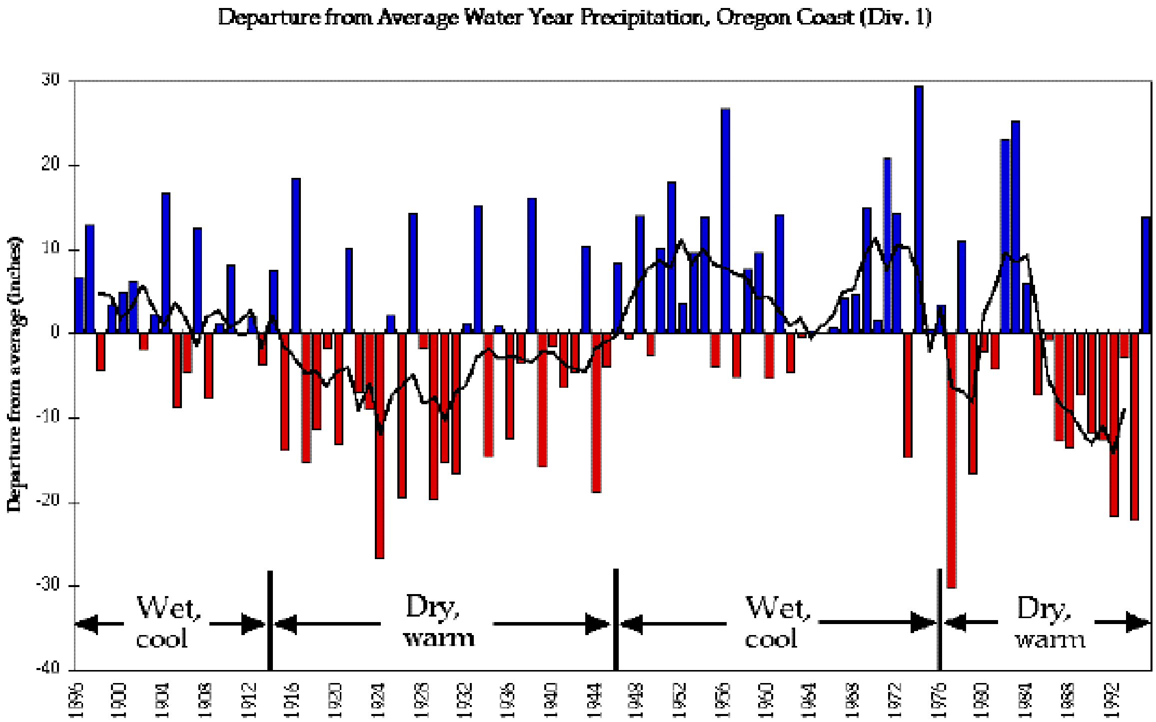


Figure 9. Long-term precipitation patterns for the Oregon Coast Division. (Source: Oregon Climate Service, Corvallis, OR)

assumptions about watershed processes that would not have been possible in a high-precipitation year. Phase I (WY1999) was much wetter and consequently most of the water quality data presented here are from Phase I.

2. Stream Water Quality

Stream water quality varied as a function of stream discharge in Big and Benson Creeks. Increased stream discharge resulted in increases in total suspended solids (TSS) and total phosphorus (TP; Figures 10 and 11). The largest increases in TSS and TP were associated with the greatest increases in flow. Pollutant concentrations varied not only as a function of stream discharge, but also varied in response to precipitation intensity, antecedent moisture conditions, position of the hydrograph (rising vs falling stage), and duration of the storm. Pollutant concentrations were generally greatest in high-intensity storms with rapidly rising hydrographs. As storms progressed, the pollutant concentrations generally declined, other factors being equal.

Whereas the concentration of pollutants varied by two orders of magnitude in Big and Benson Creeks during storm events, the water quality in Murphy Creek was comparatively

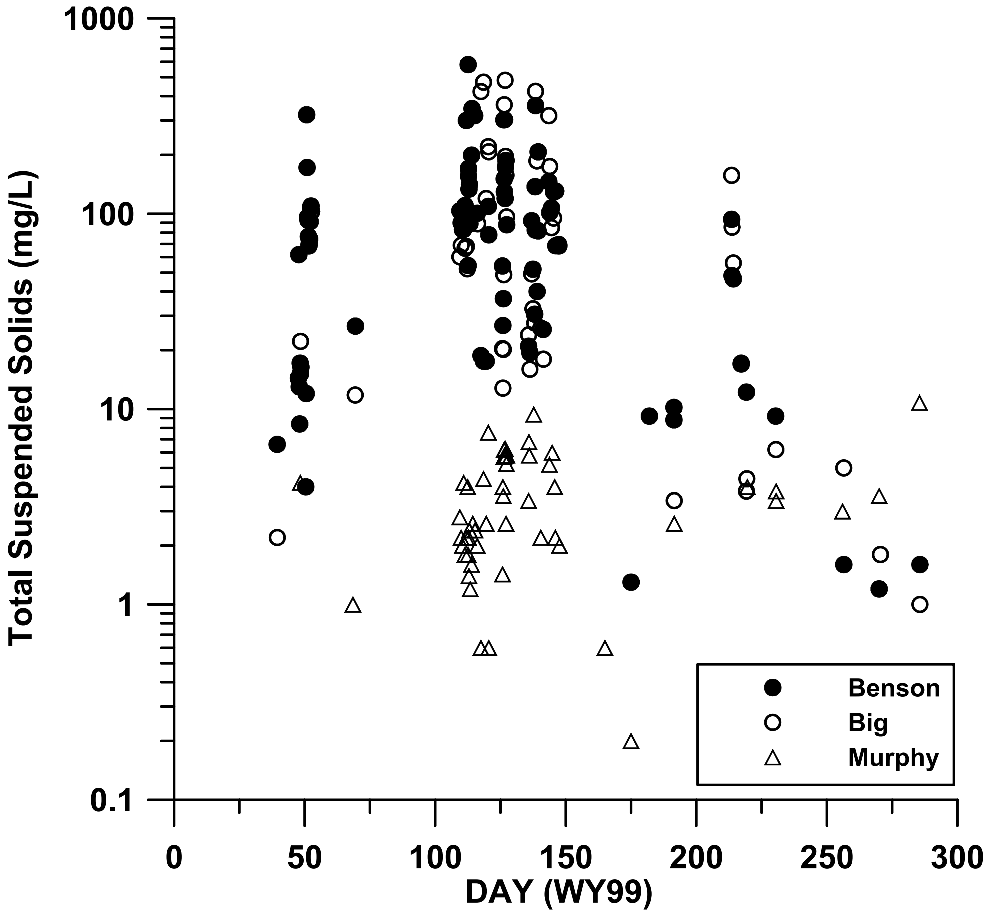


Figure 10. Total suspended solids (TSS, mg/L) for Big, Benson, and Murphy Creeks, WY1999. Note that the data are presented on a $\log_{10}$ scale.

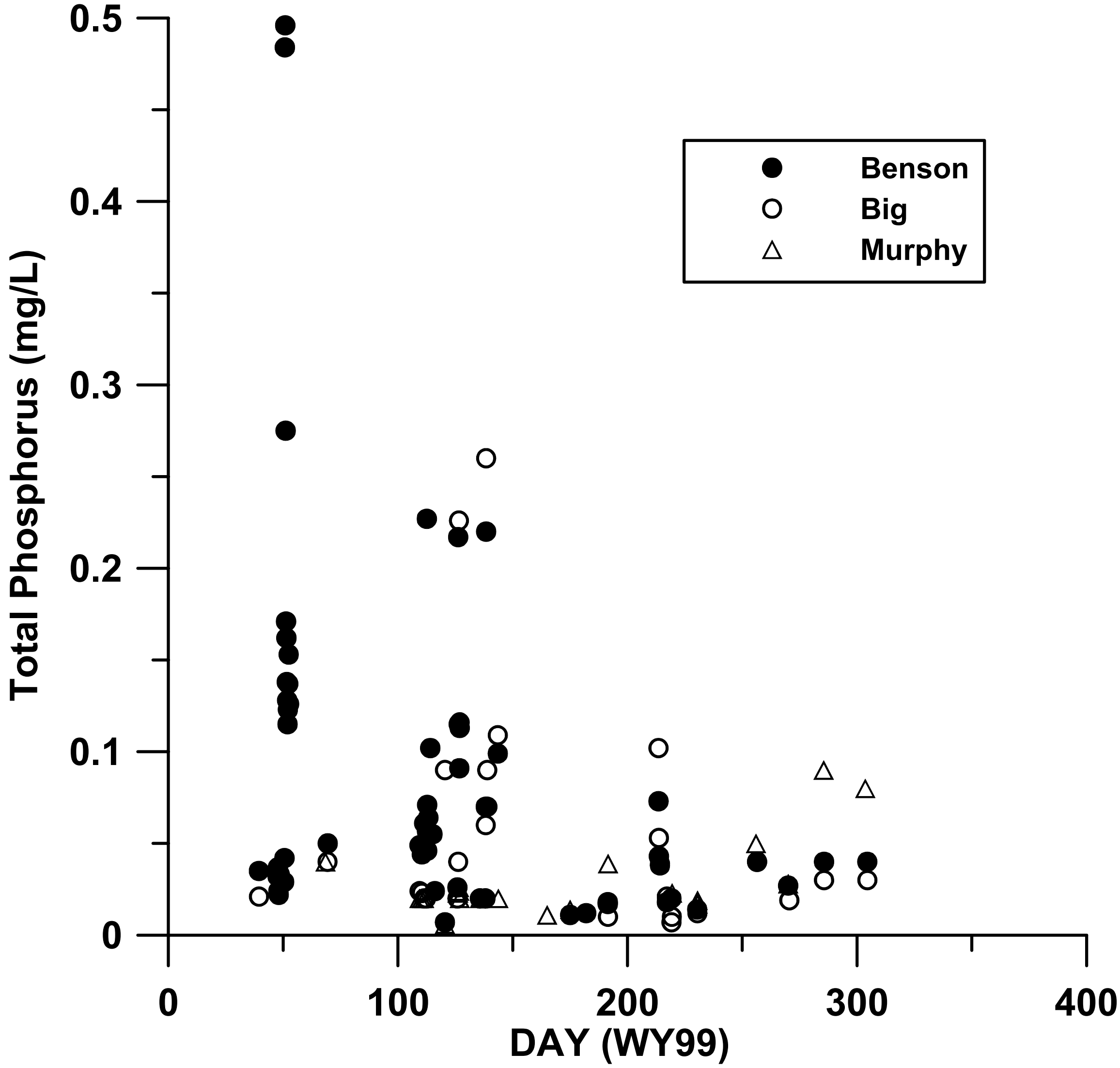


Figure 11. Total phosphorus (TP, mg/L) for Big, Benson, and Murphy Creeks.

constant. Concentrations of TSS never exceeded 12 mg/L in Murphy Creek compared to a maximum of 580 mg/L in Benson Creek and 423 mg/L in Big Creek. Concentrations of TP and nitrate (NO_3) were also much lower for most of the time in Murphy Creek.

Most of the phosphorus was attached to soil particles and was represented by TP. Ortho phosphorus (OP) represented a smaller proportion of the phosphorus delivered to the lake as indicated in the ratio of OP/TP (Figure 12). Concentrations of ortho phosphorus in Murphy Creek were greater than measured at the other stream sites, presumably because of release of P from wetland soils.

The dominant form of nitrogen in the streams was nitrate. Nitrate concentrations started out high at all sites in the fall and decreased to near zero in the summer (Figure 13). Ammonia was generally below detection limit (0.02 mg/L) in most samples. Total Kjeldahl nitrogen (TKN, reduced nitrogen) was generally well above detection limit (0.2 mg/L) in Big and Benson Creeks, but was often below detection limit in Murphy Creek. Many of the streams measured in the synoptic sampling also had measurable TKN.

Specific conductance values for the three tributaries show a decline through winter and gradual increases through spring and summer (Figure 14). These effects show the dilution of groundwater inputs caused by high surface runoff in winter and the return to a higher percentage of groundwater-dominated flow pattern in late spring. pH values in the streams were generally circumneutral and increased by about 0.4 pH units from fall to spring (Figure 15).

Concentrations of TSS, TP, and NO_3 elsewhere in the watershed were evaluated by collecting water samples from another twelve sites during the course of the study. These sites were generally sampled during storm events, although there was no effort to synchronize the sampling or sample at maximum discharge. Therefore, the results offer only a qualitative assessment of water quality at other sites in the watershed. The results suggest that the water quality observed at the other sites is most similar to that observed in Big and Benson Creeks, based on measures of central tendency (Table 5). The ranges of values measured at the synoptic sites do not approach the values measured at Big and Benson Creeks, which is to be expected given the transient nature of the nonpoint source loads and the need to sample intensively during storm events. The one synoptic site with a modest number of samples (an unnamed tributary to Benson Creek) exhibited high TSS and NO_3 concentrations. Again, this is consistent with a greater number of samples at this site (thus increasing the probability of measuring higher concentrations), but it also may reflect the recent clearcut in the catchment. In summary, the

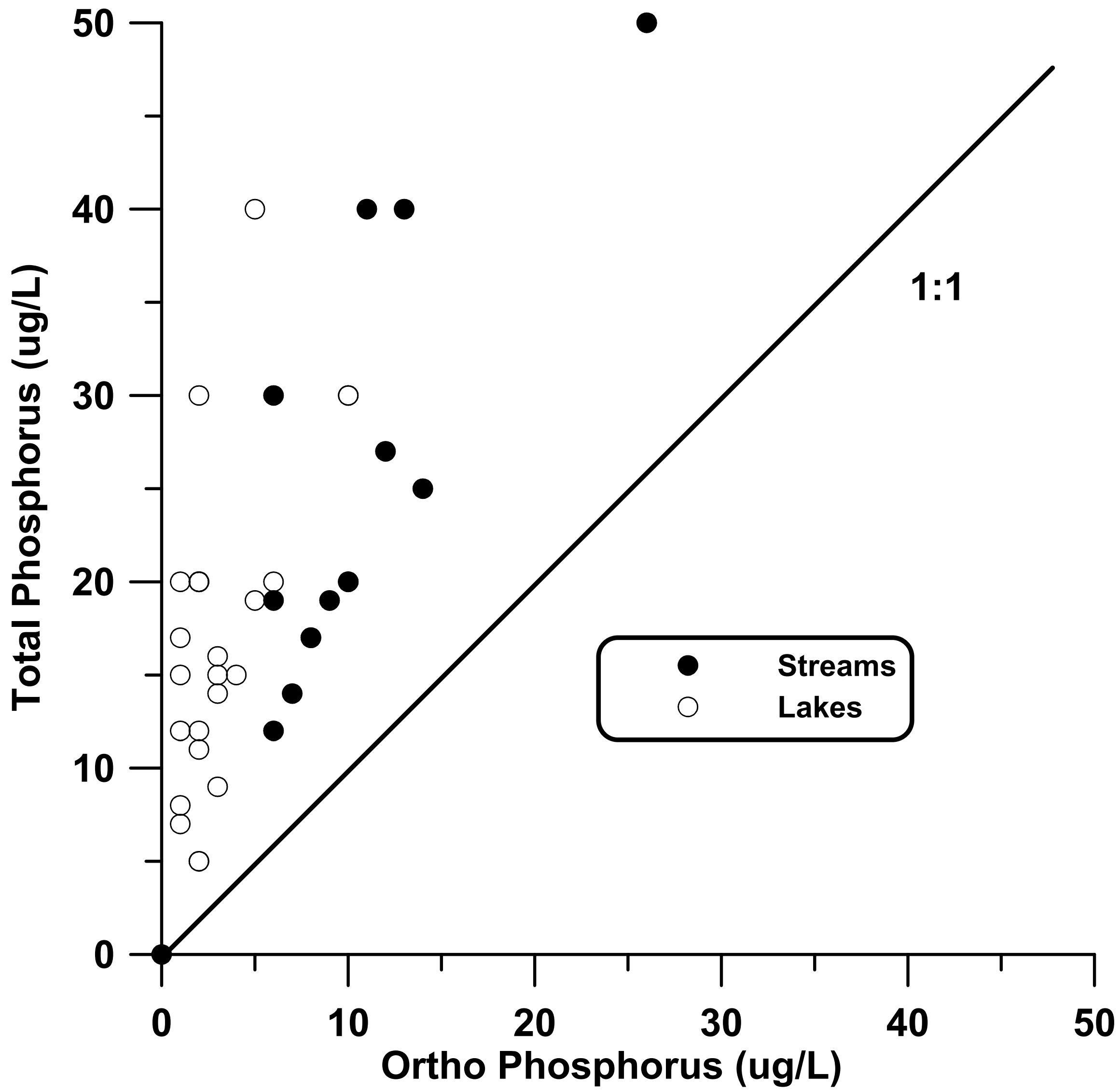


Figure 12. Total phosphorus and ortho phosphorus (µg/L) for lake and stream samples in the Tenmile Lake watershed.

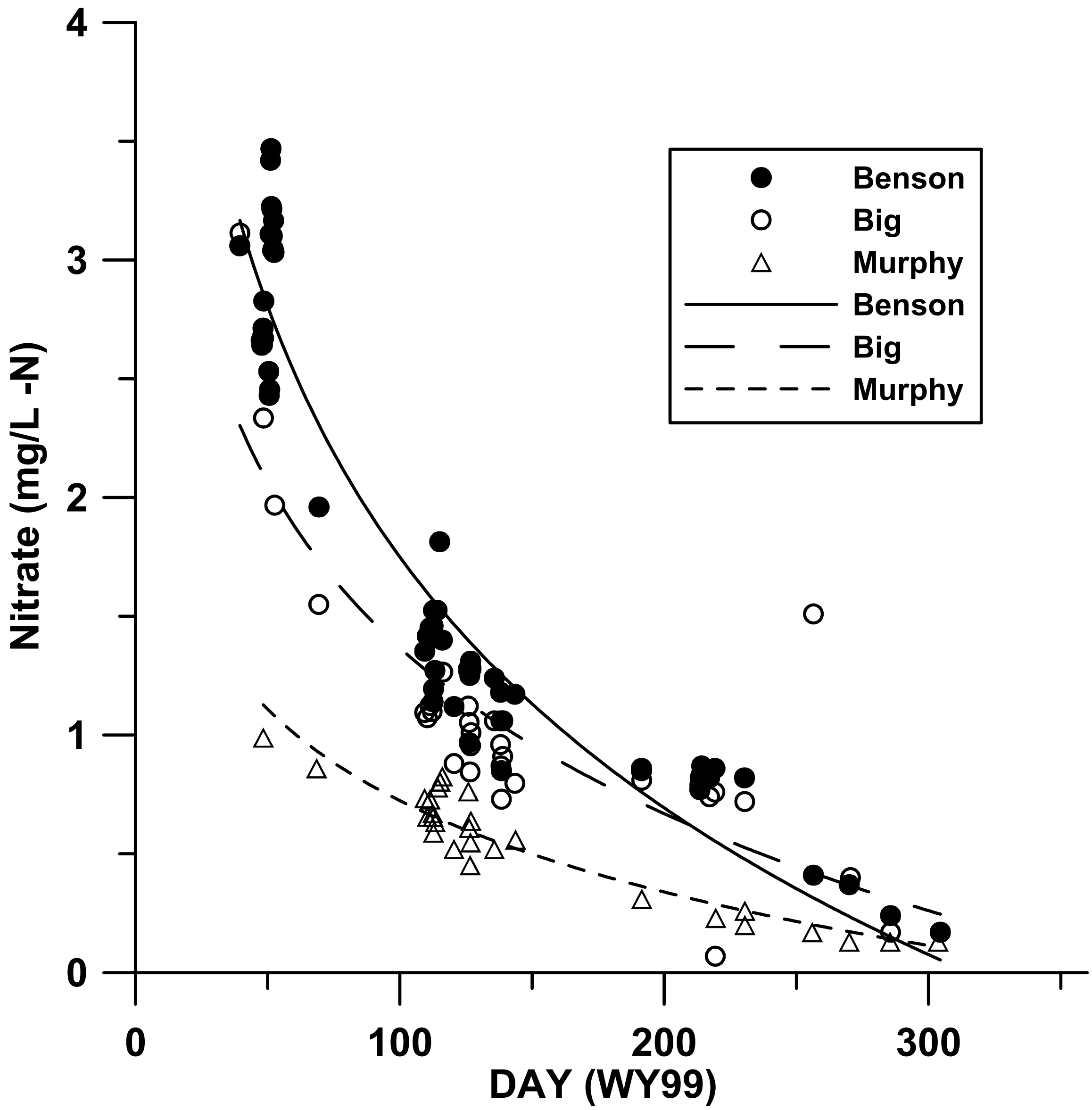


Figure 13. Nitrate (NO_3-N; mg/L) for Big, Benson, and Murphy Creeks. The curves represent log-fits to the observed data. The time period is represented by the water year which begins on October 1 (Day 0). For reference, Day 100 corresponds to January 8, 1999, Day 200 corresponds to April 18, 1999, and Day 300 corresponds to July 27, 1999.

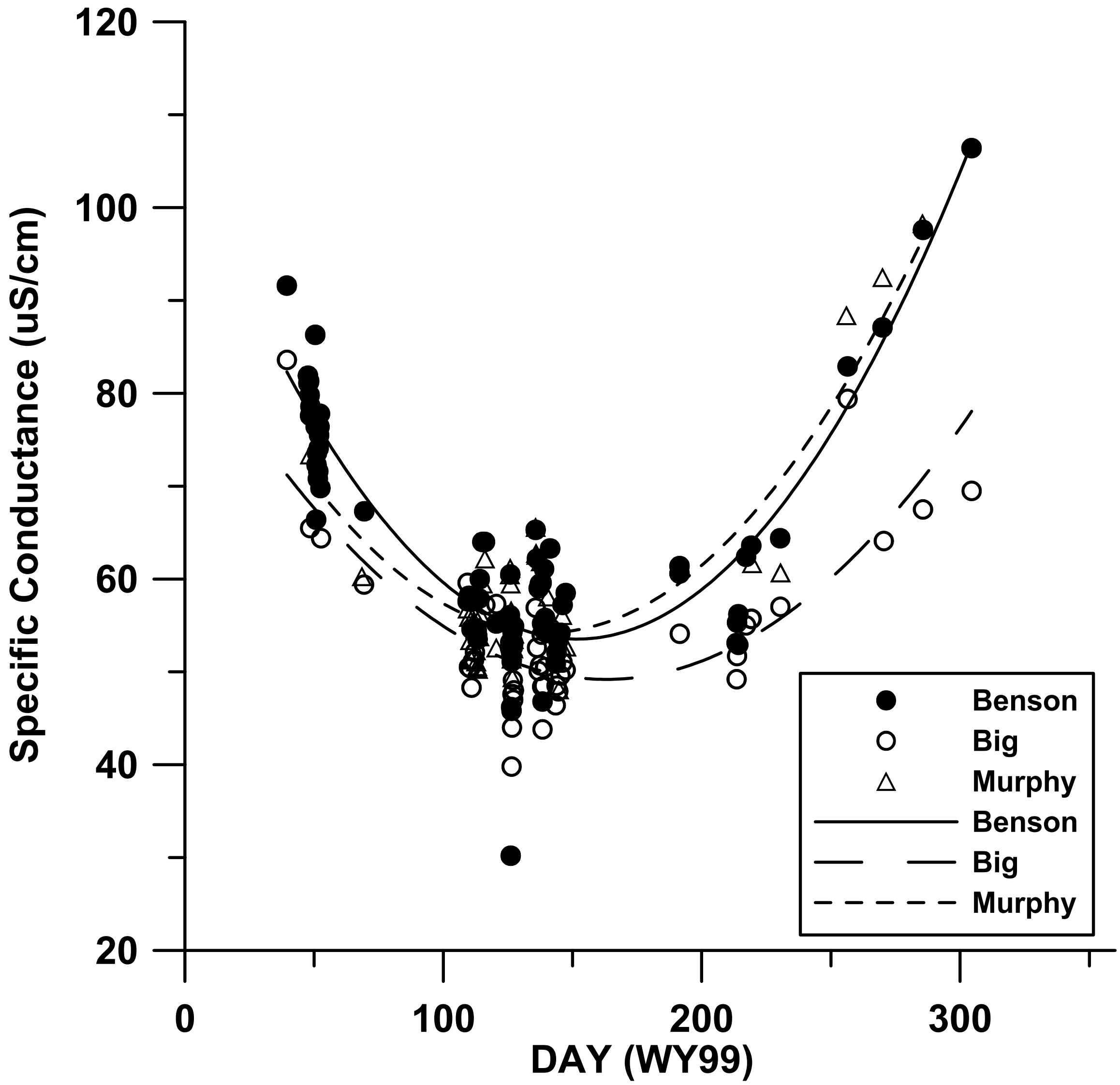


Figure 14. Specific conductance (μS/cm) for Big, Benson, and Murphy Creeks. The curves represent log-fits to the observed data. The time period is represented by the water year which begins on October 1 (Day 0). For reference, Day 100 corresponds to January 8, 1999, Day 200 corresponds to April 18, 1999, and Day 300 corresponds to July 27, 1999.

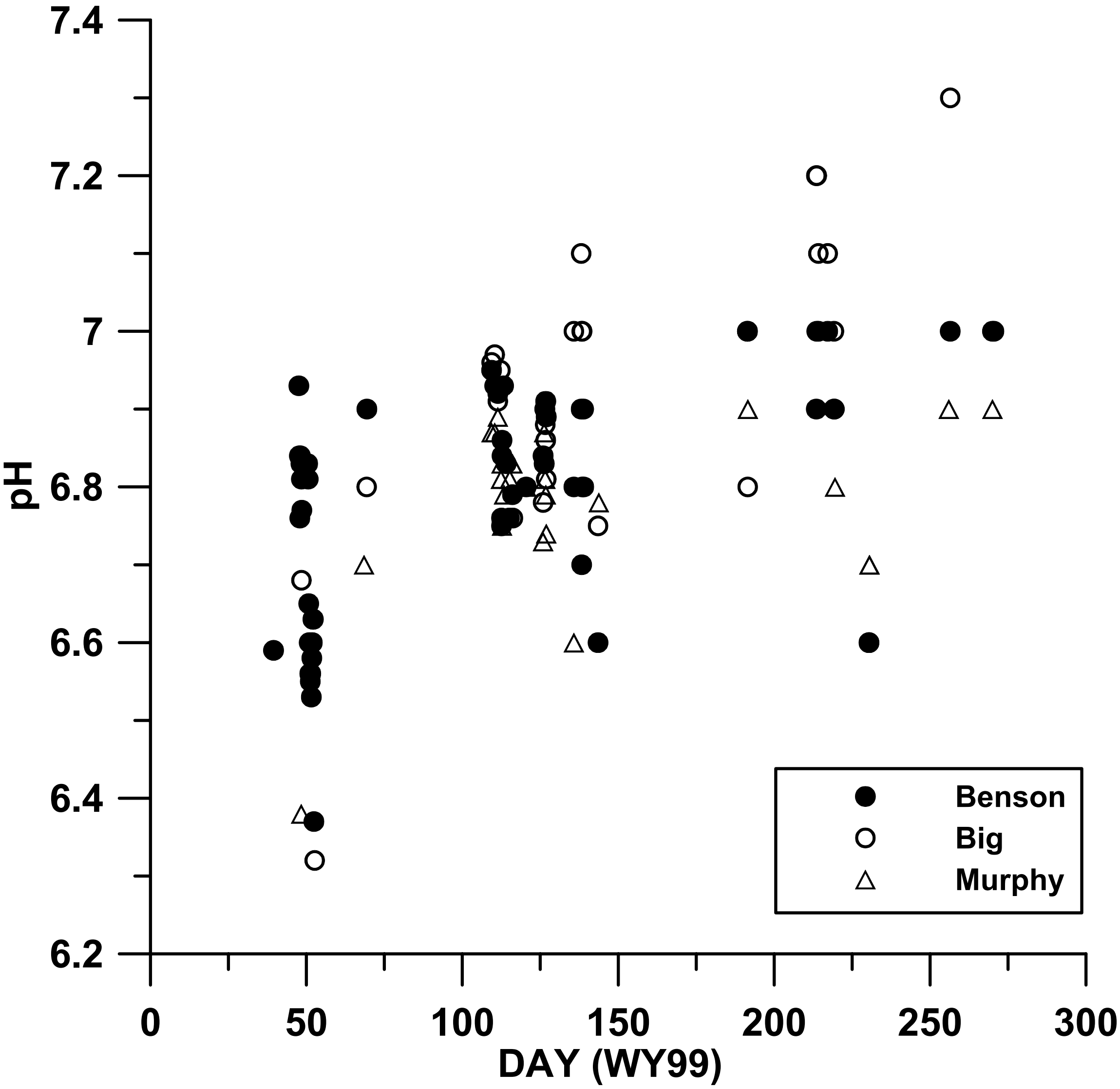


Figure 15. pH (standard units) for Big, Benson, and Murphy Creeks.

Table 5. Concentrations of TSS, TP, NO_3 (all in mg/L) for synoptic sites.					
Site ID	Site Name	Samples	TSS	TP	NO_3-N
ADA	Adams Creek (A)	2	62	0.102	1.21
ADB	Adams Creek (B)	1	1	0.034	2.03
BGD	Big Creek (Ditch)	2	177	0.167	0.87
BND	Benson Creek (Ditch)	1	21	0.085	0.10
JNA	Johnson Creek (A)	1	142	0.155	0.59
MIL	Johnson Creek (B)	1	100	0.124	0.47
MMR	Murphy Tributary	1	29	0.034	0.73
MUM	Murphy Marsh	2	4	0.030	0.76
NOB	Noble Creek	5	5	0.040	3.79
RAN	Rain	5	1	0.020	0.15
REX	Benson Tributary	12	252	0.066	2.93
SHB	Shutters Creek (B)	1	52	0.092	0.96
SHU	Shutters Creek	1	103	0.158	1.36
SUN	Ditch near Sunriver Marina	1	26	0.055	1.36

synoptic stream sampling suggested that the water quality in Big and Benson Creeks was typical of much of the Tenmile Lake watershed.

Major Ions

A small number of samples were analyzed for major ions to gain insight into the relative importance of some processes affecting Tenmile Lake (Figure 16). The major ion analyses for the late winter period reveals an apparent enrichment of Cl^-, NO_3^-, and K^+ for Big and Benson Creeks and the two lake sites. The Na:Cl ratio is lower for Big and Benson Creeks compared to Murphy Creek, which is explained by the higher Na^+ concentration at Murphy Creek. Perhaps of greater interest is the NO_3^-:Cl ratio which illustrates an altered ratio at Big and Benson Creeks and both lake sites. Not only do these four sites exhibit elevated NO_3^- concentrations, but they also display K^+ concentrations from 36% to 110% above that measured in Murphy Creek. Potassium (K^+) is a ration often associated with livestock, agriculture, and septic inputs. Although some inputs of K^+ from weathering are typical, the higher than expected concentrations of K^+ are suggestive of livestock and septic inputs.

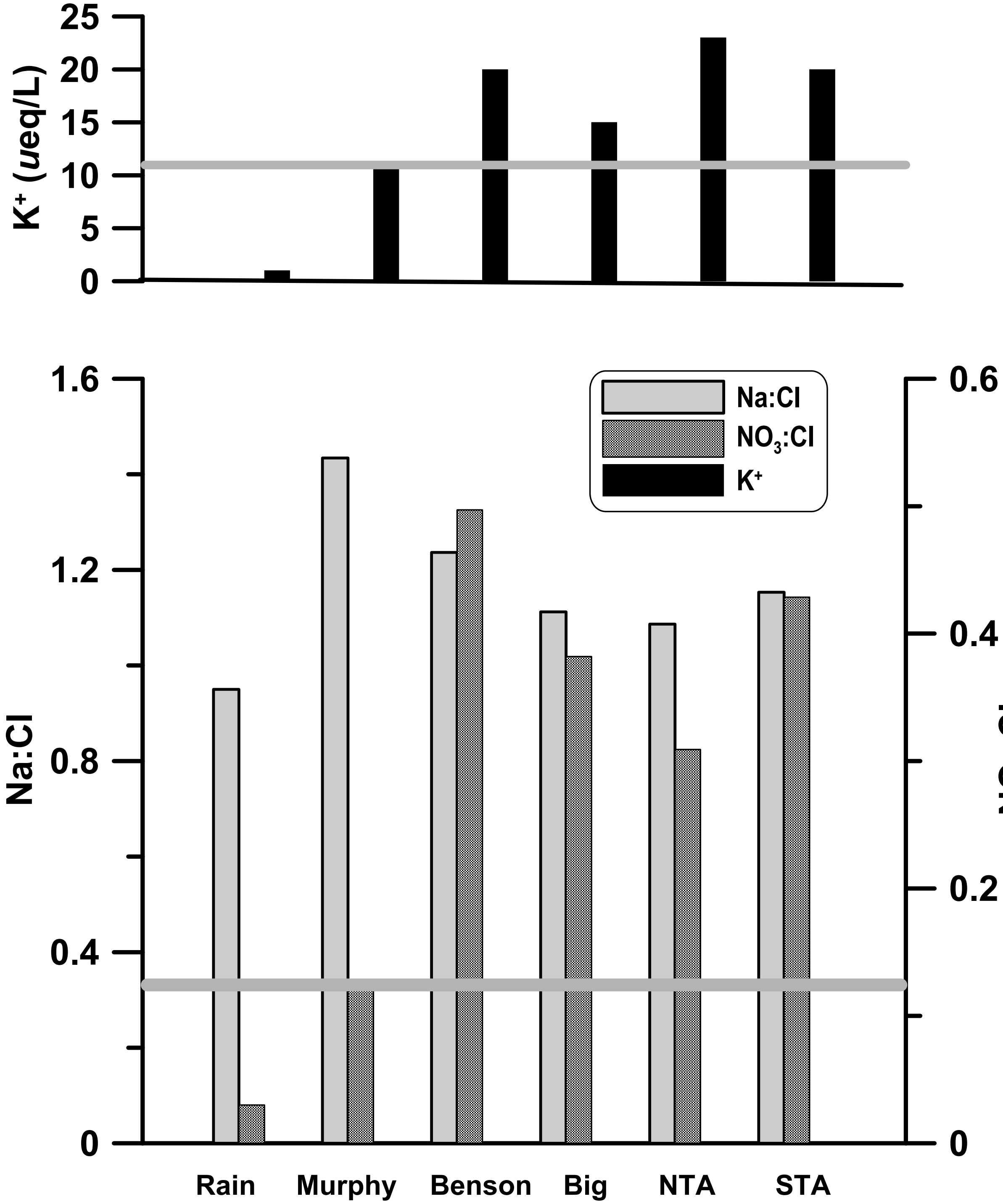


Figure 16. Concentrations of potassium (K^+, top) for representation sites and ratios of Na:Cl and NO_3:Cl (on an equivalent basis). The horizontal line in the ratio plot is the NO_3:Cl ratio for Murphy Creek.

3. Lake Water Quality

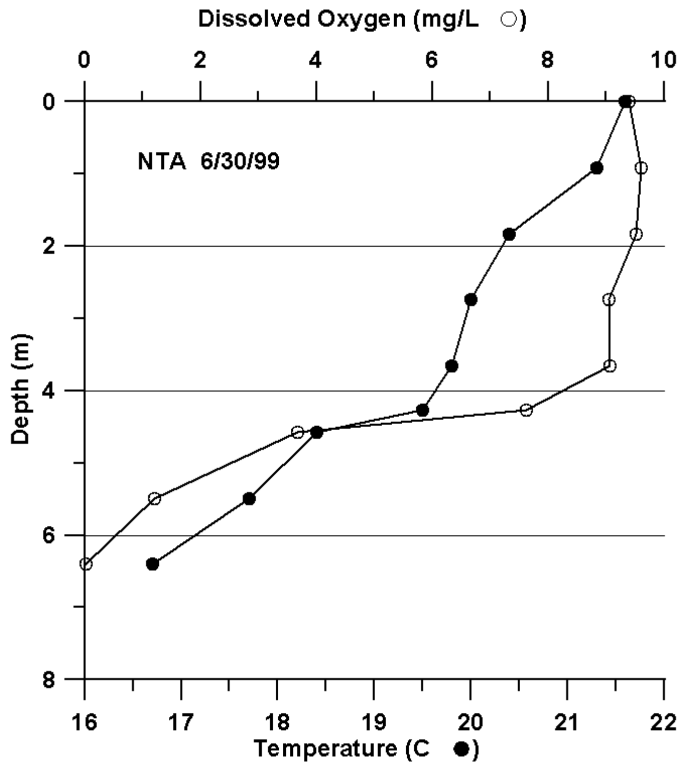
a. Field Results

Field measurements at the lake sites included temperature, specific conductance, dissolved oxygen, pH, and Secchi disk transparency. In most cases, the lake waters were thermally uniform resulting in relatively little vertical variation in dissolved oxygen and specific conductance. However, during summer some temperature stratification would occur when reduced wind conditions permitted establishment of a temperature differential of several degrees. The greatest temperature difference between the top and bottom waters occurred on June 30, 1999 at site NTA with a temperature difference of 4.9° C (21.6 to 16.7° C). Oxygen depletion in the bottom waters was rapid with DO approaching 0 mg/L at the bottom (Figure 17a). A similar depletion of DO was observed at the same site on August 26, 1999 even though the temperature difference between the top and bottom of the lake was only 3° C (Figure 17b). Oxygen depletion was equally intense at the shallower NTB site located at the confluence of Big Creek Arm and Carlson Arm during this August sample (Figure 18a,b). Oxygen depletion rates at the south lake sites (STA and STB) were generally less than those observed in the north lake sites, although there was still evidence of low DO (Figures 19-20).

Secchi disk transparency varied considerably between the deep and shallow sites (Figure 21). The sites closest to inflowing tributaries (NTB and STB) experienced lower transparency caused by suspended solids from watershed runoff during the winter and by intense algal blooms in the summer. This was particularly evident at site NTB which was proximate to the Big Creek inlet. Turbidity plumes from Big Creek were visible for over 1 km downstream of Big Creek during major runoff events. The transparency in the Big Creek Arm was 0.8 m on February 20, 1999 compared to 1.3 m in the Carlson Arm which drains Murphy and Wilkins Creeks. This difference of nearly 40 percent was typical of the transparency difference that existed in the winter between the two sites. In the summer, reduced transparency was associated with algal blooms. For example, on August 26, 1999 transparency ranged from 2.4 m at site NTA to 1.2 m at site NTB and down to 0.7 m at Sun Lake Marina located about 100 m from the mouth of Big Creek. Algal biovolume at site NTA was 4.9×10^5 μM^3/mL compared to 3.3×10^6 μM^3/mL at site NTB.

A similar event occurred in September, 2000 when Secchi disk transparency ranged from a low of 0.4 in the upper part of Big Creek Arm (NTC) to 1.3 m at NTB and 3.3 m at NTA. The maximum transparency at 4.4 m was recorded at South Tenmile Lake (STA). The chlorophyll *a*

a)



b)

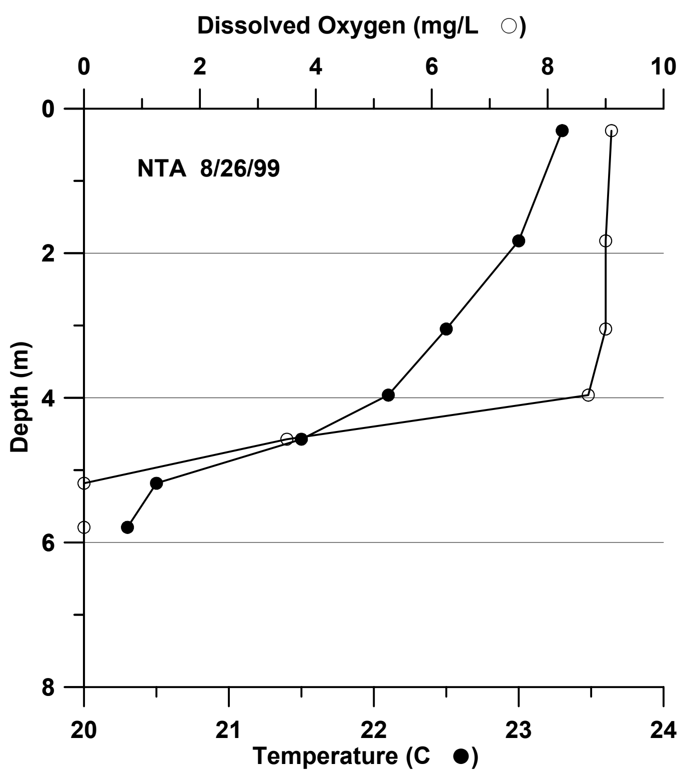
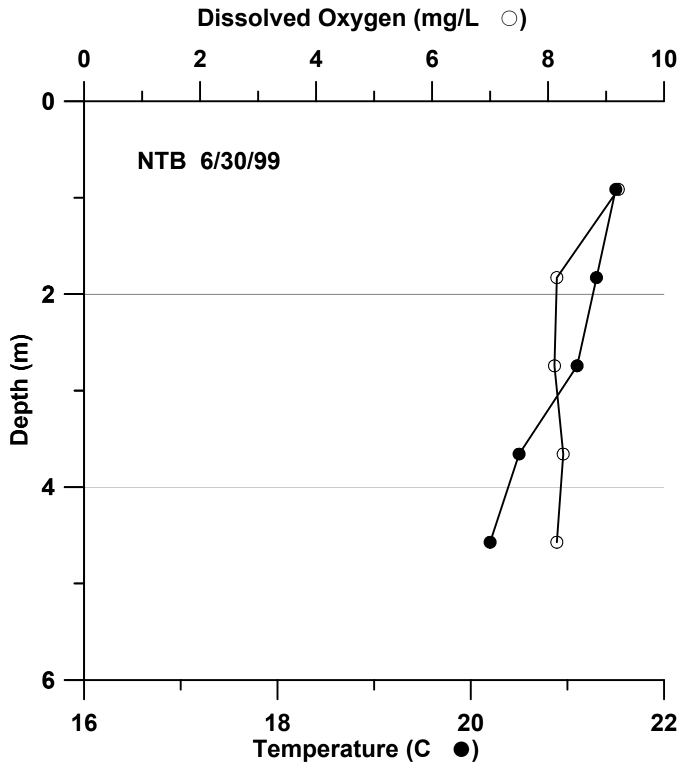


Figure 17. Temperature (ºC) and dissolved oxygen (mg/L) at NTA (North Tenmile-Site A) versus depth (m) on a) June 30, 1999 and b) August 26, 1999.

a)



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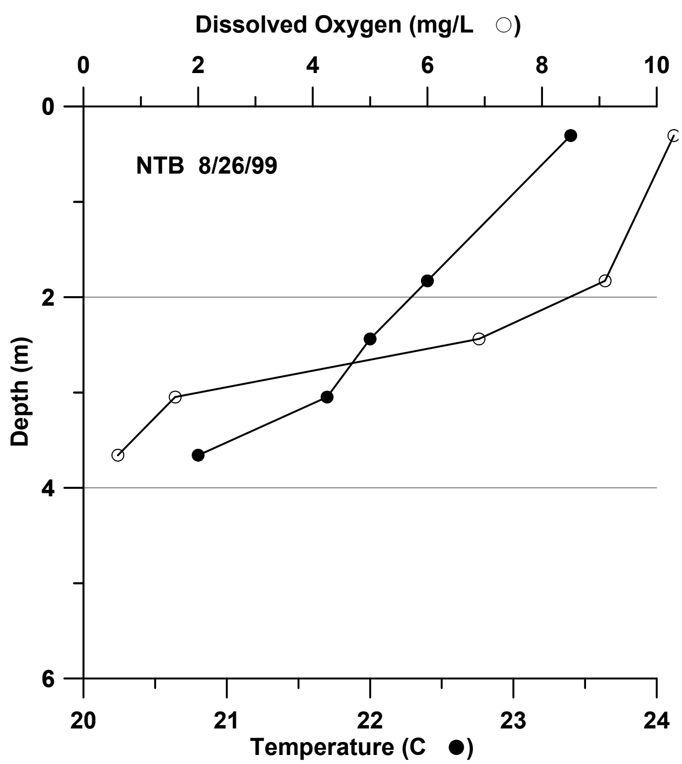
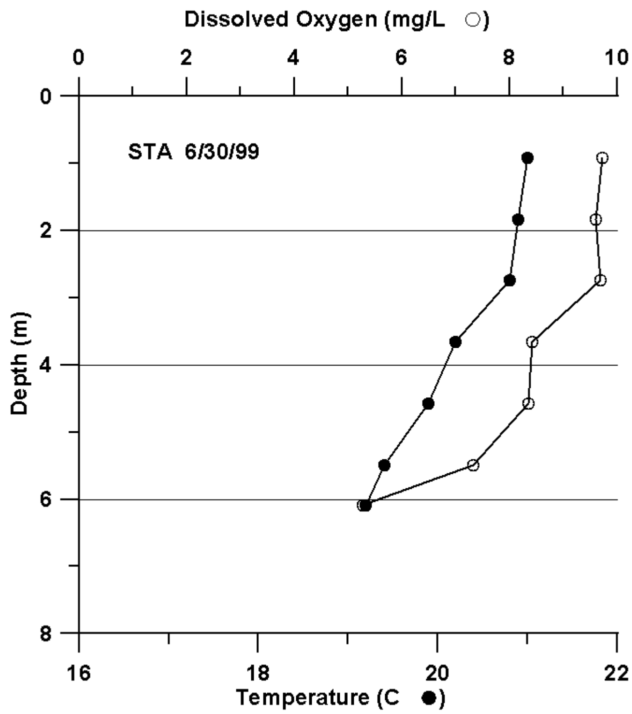


Figure 18. Temperature (ºC) and dissolved oxygen (mg/L) at NTB (North Tenmile-Site B) versus depth (m) on a) June 30, 1999 and b) August 26, 1999.

a)



b)

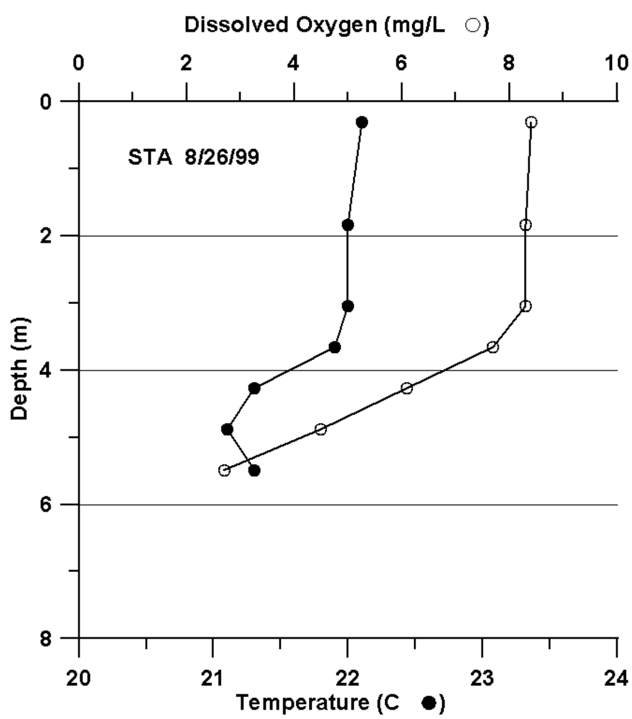
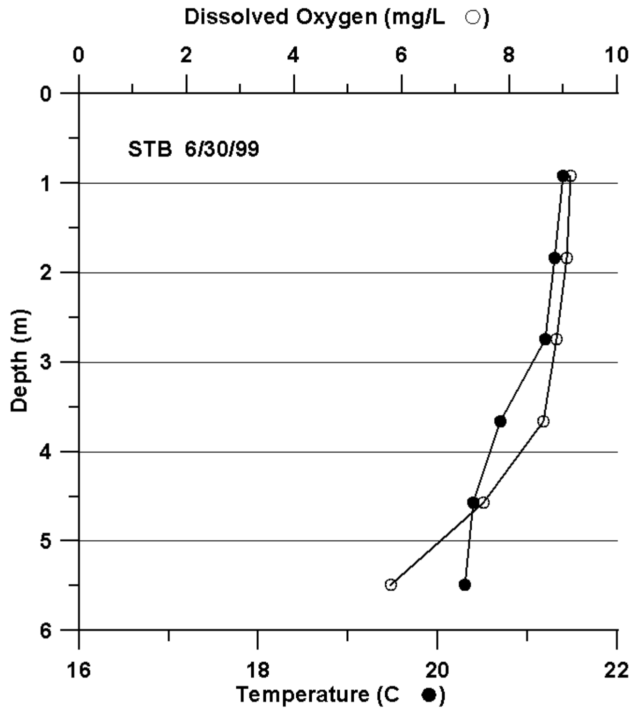


Figure 19. Temperature (°C) and dissolved oxygen (mg/L) at STA versus depth (m) on a) June 30, 1999 and b) August 26, 1999.

a)



b)

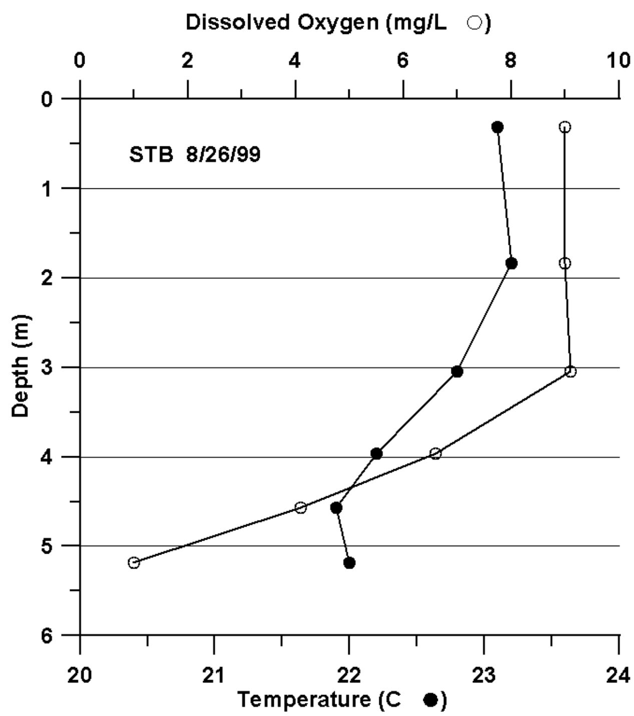


Figure 20. Temperature (°C) and dissolved oxygen (mg/L) at STB versus depth (m) on a) June 30, 1999 and b) August 26, 1999.

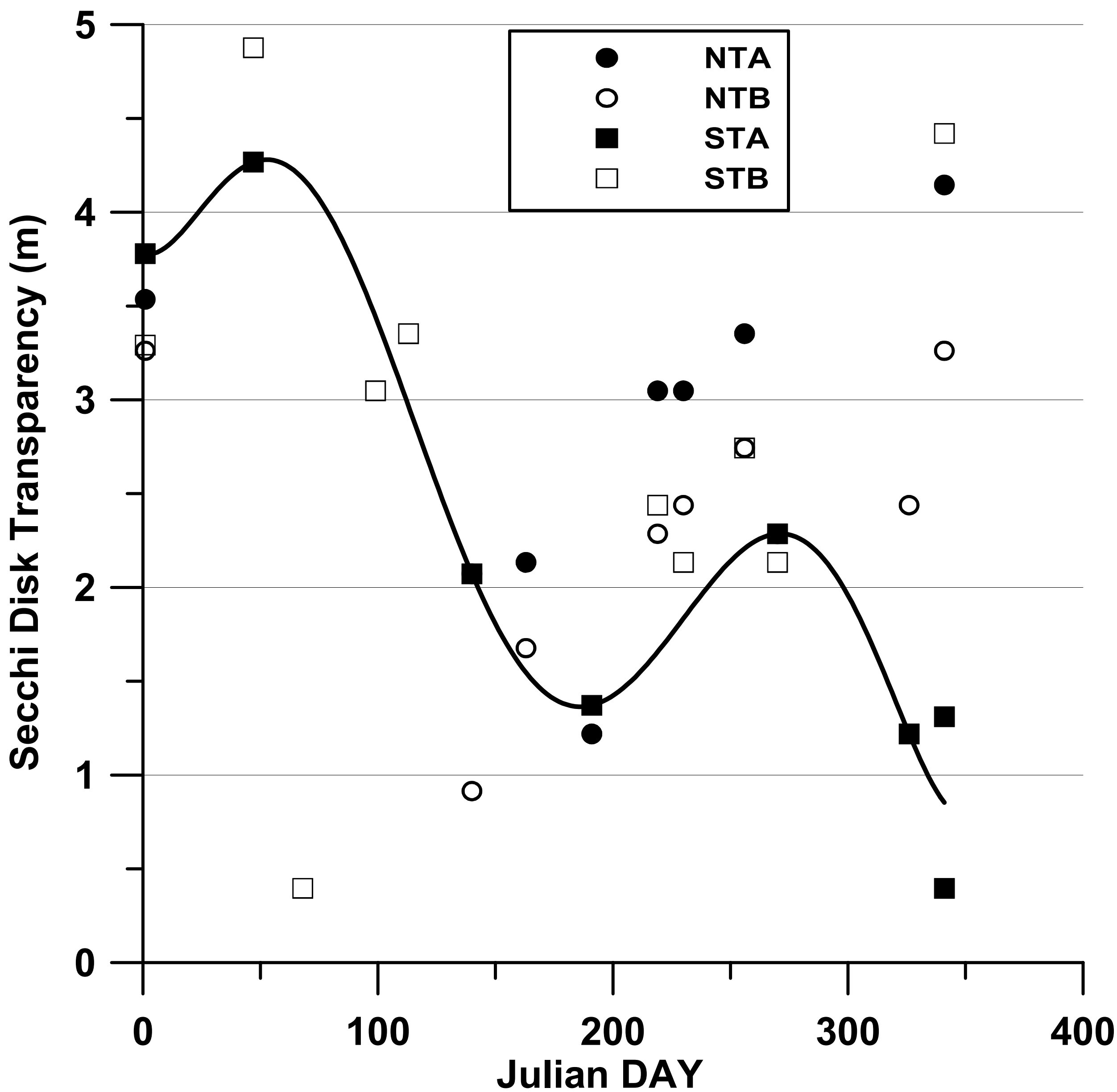


Figure 21. Secchi disk transparency (m) for the four sites on Tenmile Lake. The general pattern in temporal variations in the transparency is highlighted by the observation for site STB which have been linked together.

concentrations at these same sites ranged from 152 µg/L at NTC (presented in Section 4a) to 4.2 µg/L at NTA. Chlorophyll *a* concentrations generally above 10 µg/L are often associated with eutrophic conditions and values above 25 µg/L are typically found in hypereutrophic lakes.

b. Analytical Chemistry

Concentrations of TSS in the lake were generally less than 5 mg/L (Figure 22). The highest TSS value of 28 mg/L was measured at NTB on November 18, 1998 following a storm event. On any given date, TSS values were usually greater at the distal sites (NTB and STB) compared to the open water sites (NTA and STA). A comparison of TSS at the Benson Creek site compared to the outlet of Tenmile Lake illustrates the extent to which the lake serves as a settling basin for erosional inputs from the watershed (Figure 23).

Total phosphorus concentrations in Tenmile Lake generally ranged from 0.010 mg/L to 0.04 mg/L (Figure 24). The extreme value of 0.14 mg/L again was observed at site NTB on November 18, 1998. Concentrations of TP were highest during the winter when sediment loads would remain suspended and again in late summer when high TP was associated with planktonic algae. When we compare the inflowing waters from Benson Creek with the outlet from Tenmile Lake, Tenmile Creek, it is apparent that the lake also serves as a sink for phosphorus (Figure 25). However, concentrations of TP in the lake are comparable to those in the stream during the summer. Concentrations of ortho phosphorus were generally only 20 to 30 percent of the TP values for a given sample.

Inorganic nitrogen was present in the lake almost exclusively as nitrate; ammonia was seldom detected. Nitrate concentrations in the lake exhibit a striking seasonal effect with low concentrations in the fall followed by a sharp rise with input from winter runoff and a predictable decline in the summer (Figure 26). Concentrations of nitrate in the lake were below detection limit in summer. The pattern of nitrate decline in the lake matches the changes in tributary concentrations during the winter. However, during the spring, nitrate losses in the lake occur at a rate greater than the declines from the tributary inputs (Figure 27). The pattern in lake nitrate concentrations represents high watershed export of nitrate in the fall and simultaneous decreased rates of N-uptake in the lake. Watershed export of N continues to decline in the water year, yet demand for N in the lake increases in spring and summer.

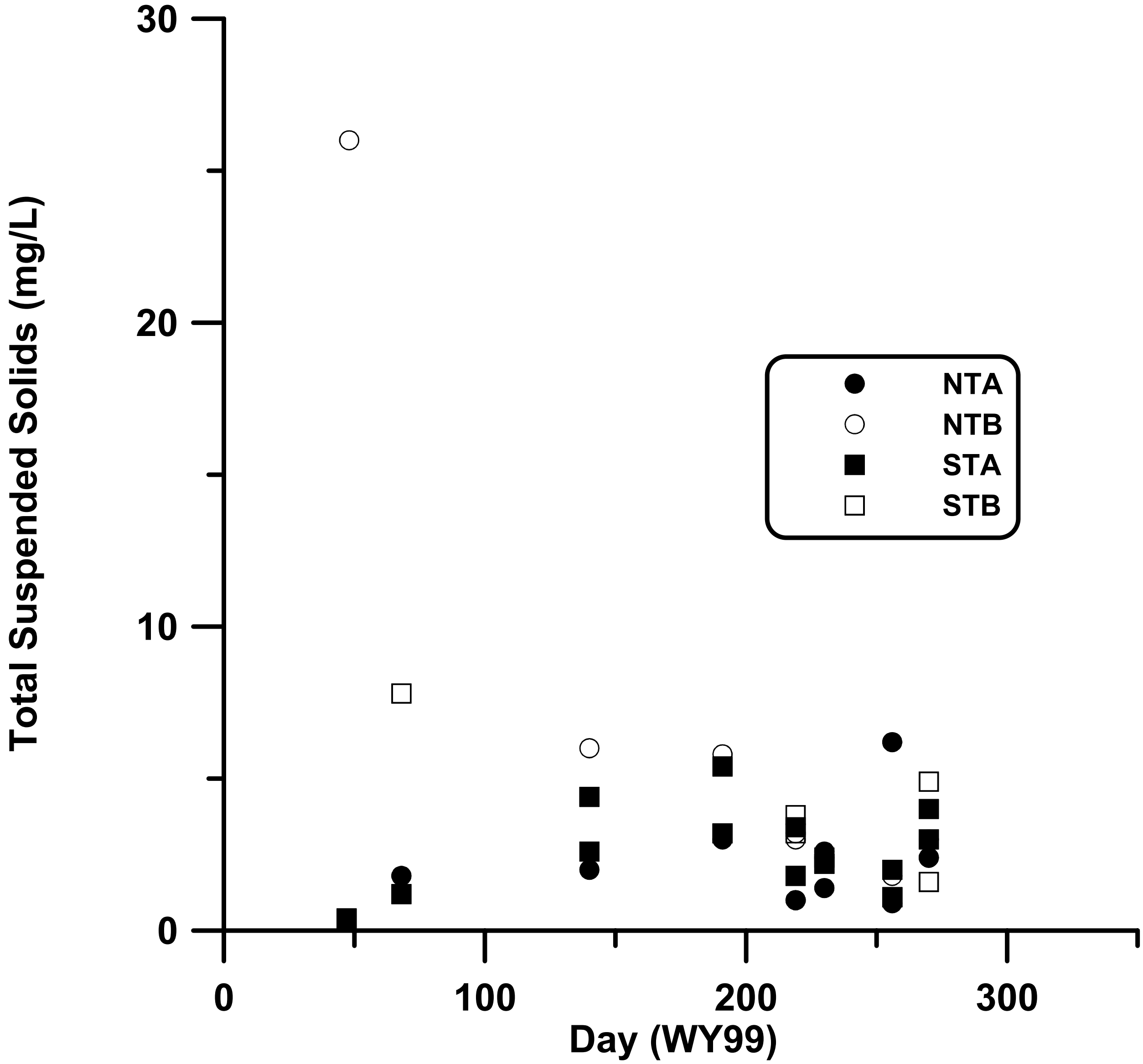


Figure 22. Total suspended solids (mg/L) for the four sites on Tenmile Lake.

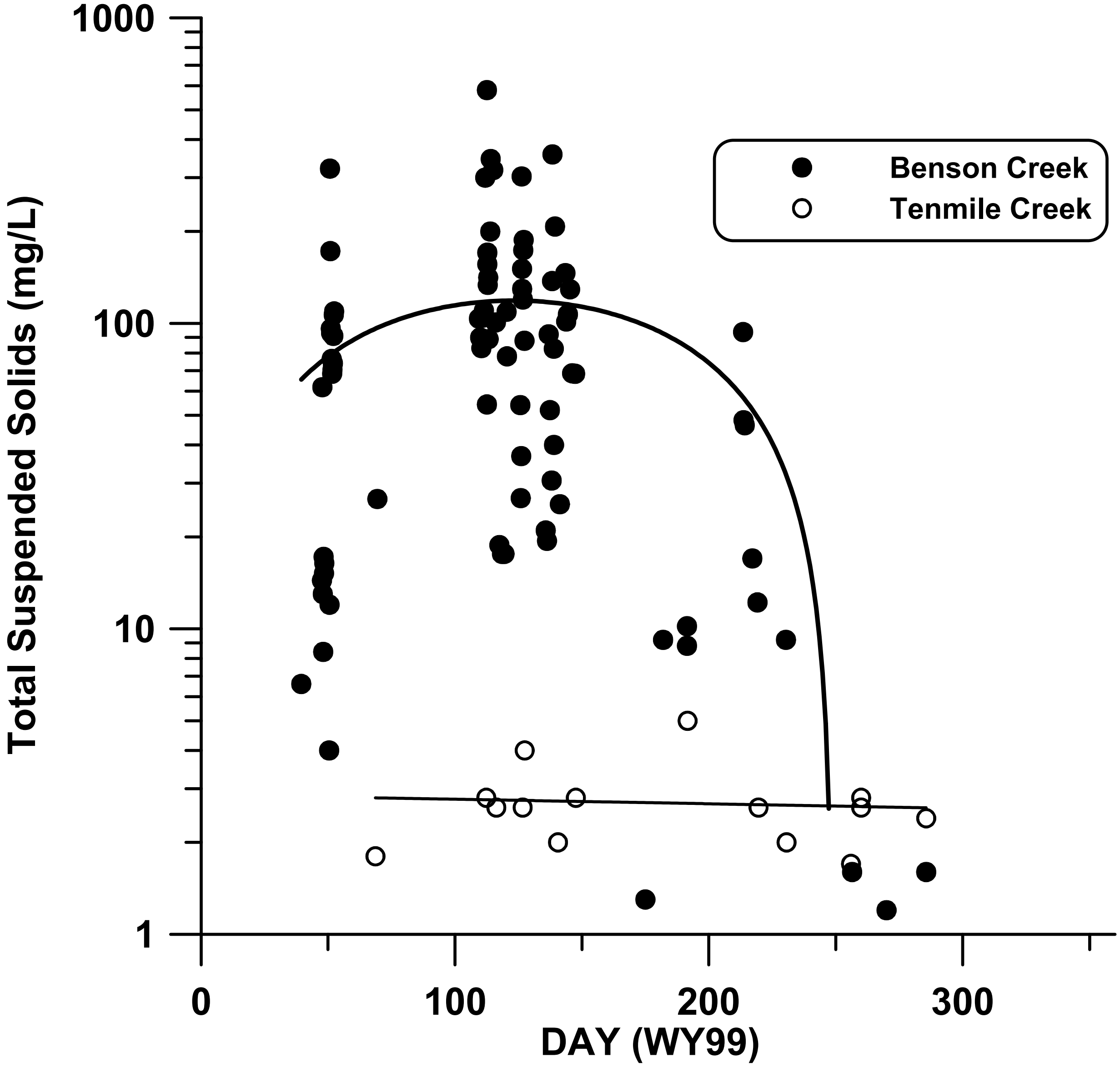


Figure 23. Total suspended solids (TSS, mg/L) versus Julian day for Benson Creek (●) and the outlet of Tenmile Creek (○). The lines represent the best fits for Benson Creek (power function) and Tenmile Creek (linear function).

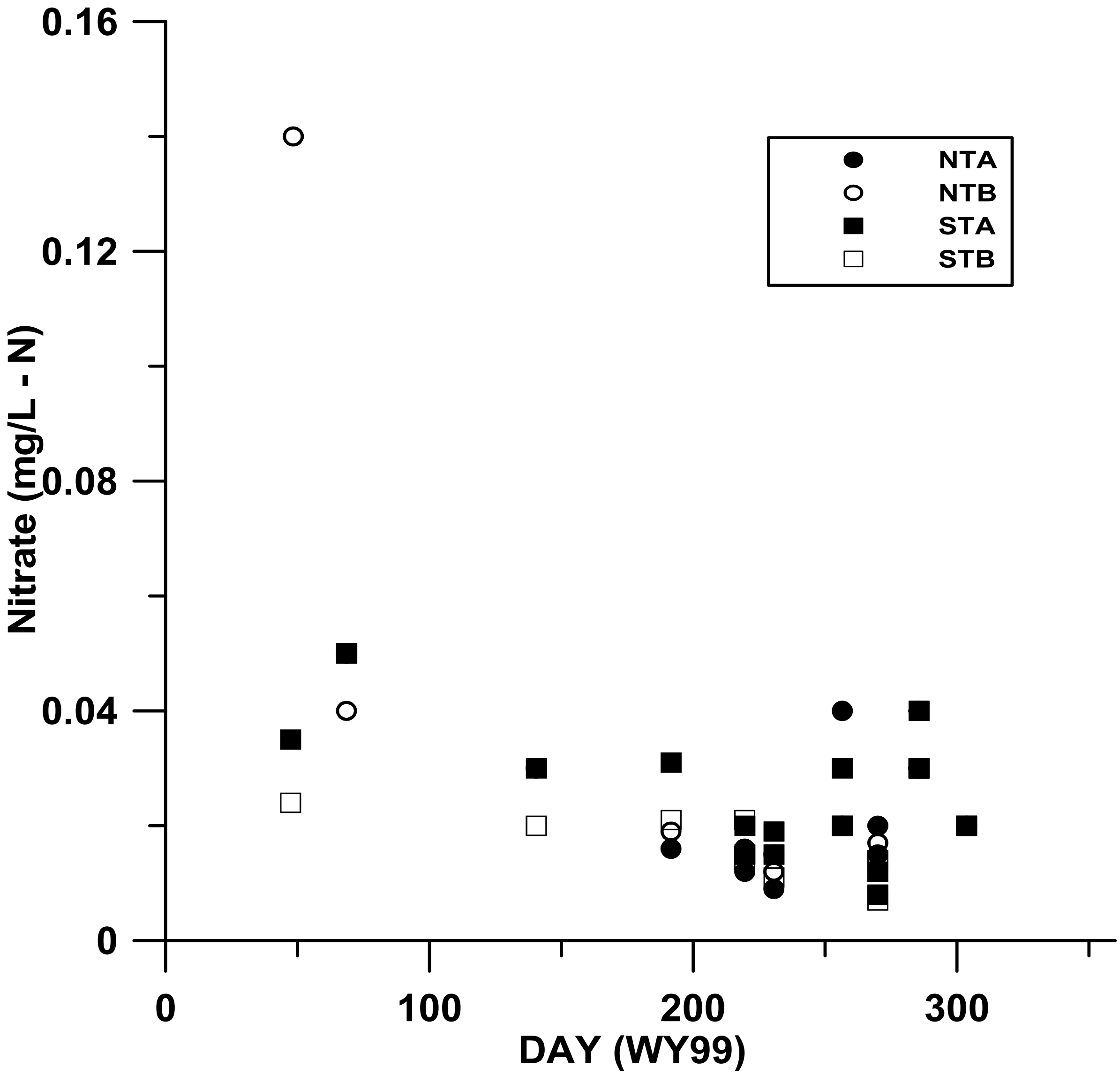


Figure 24. Total phosphorus (mg/L) at the four sites on Tenmile Lake.

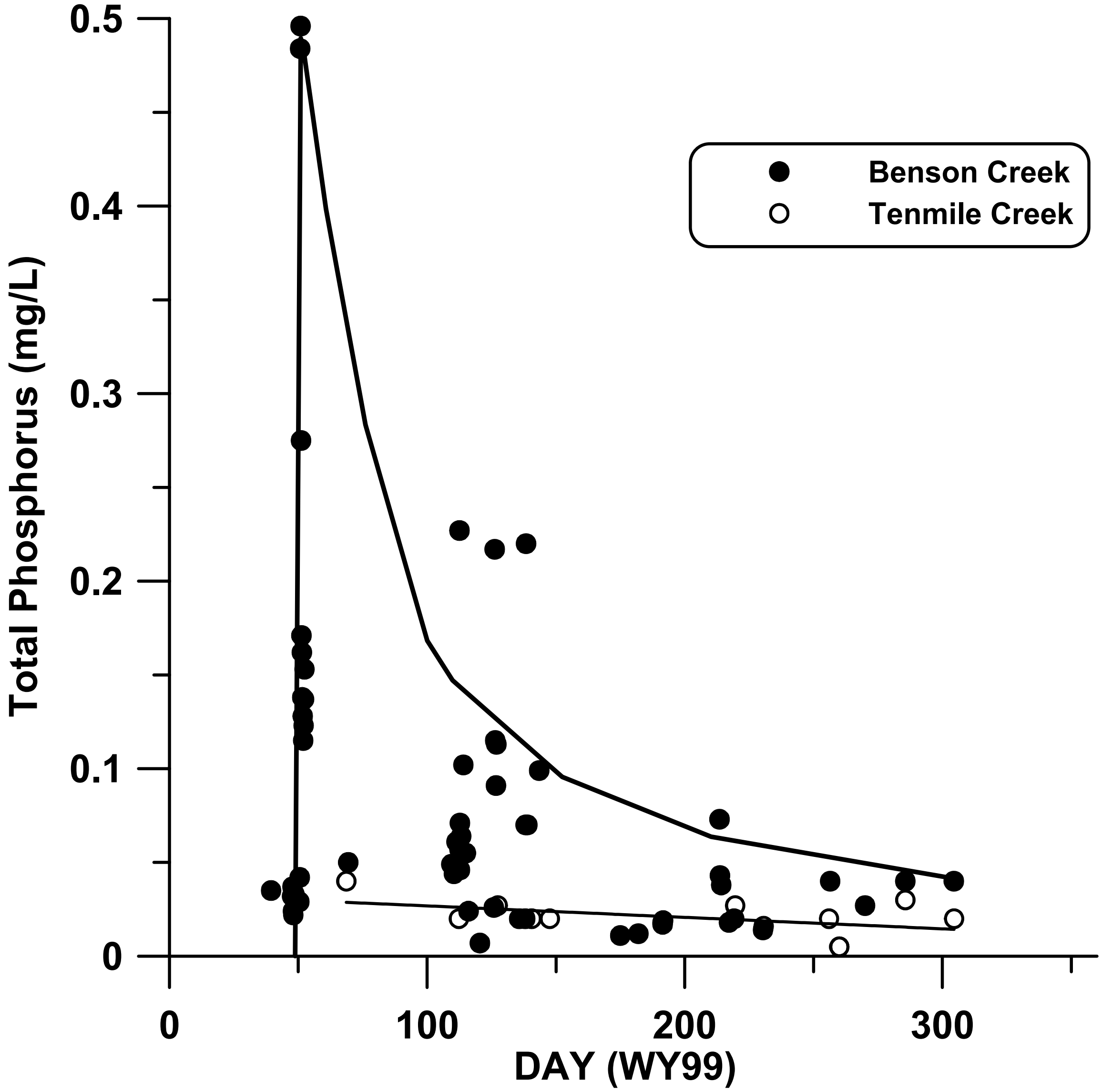


Figure 25. Concentrations of total phosphorus (TP; mg/L) versus Julian Day for Benson Creek (●) and Tenmile Creek (○).

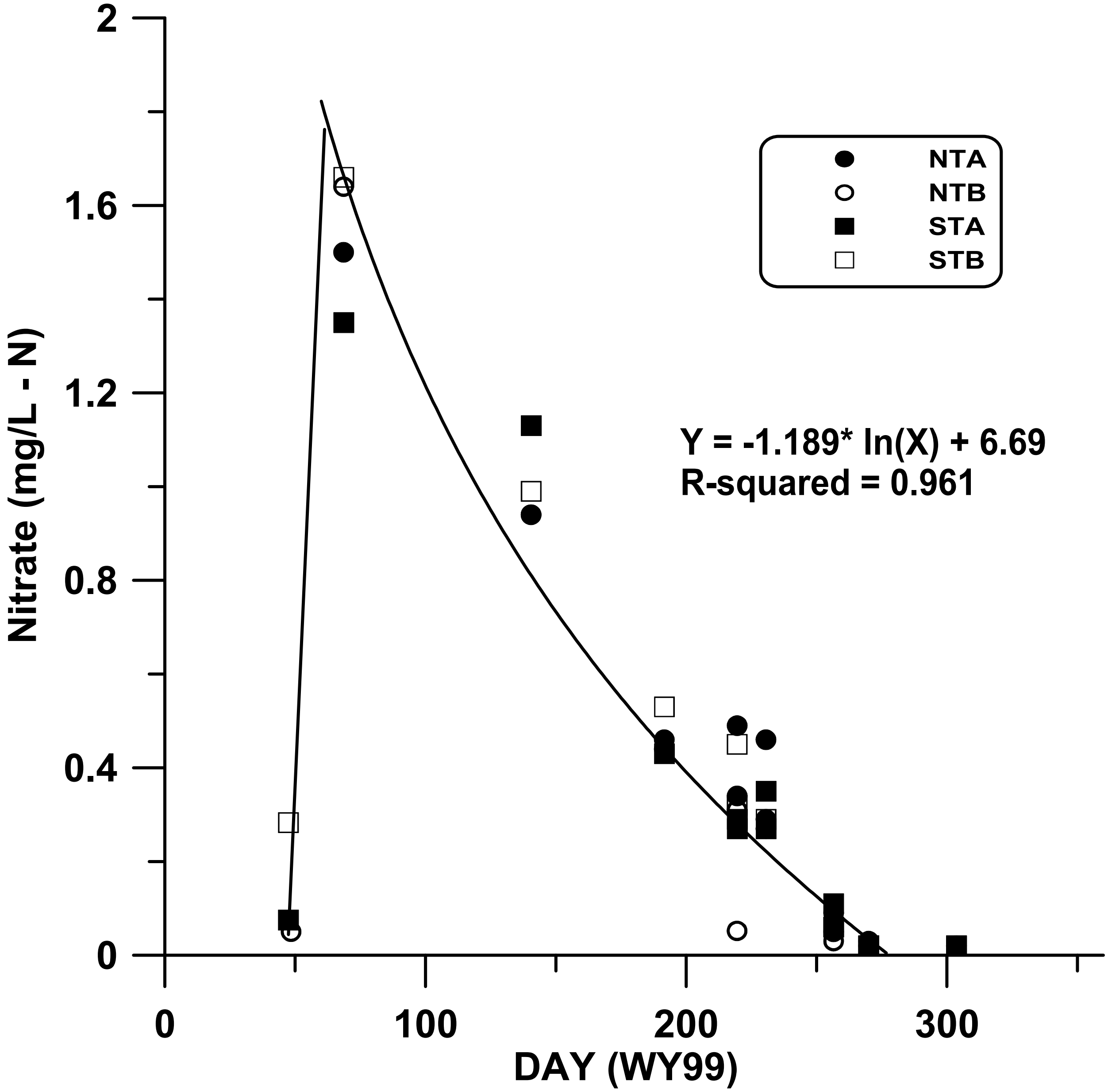


Figure 26. Concentration of nitrate (NO_3-N, mg/L) versus Julian day for the four lake sites.

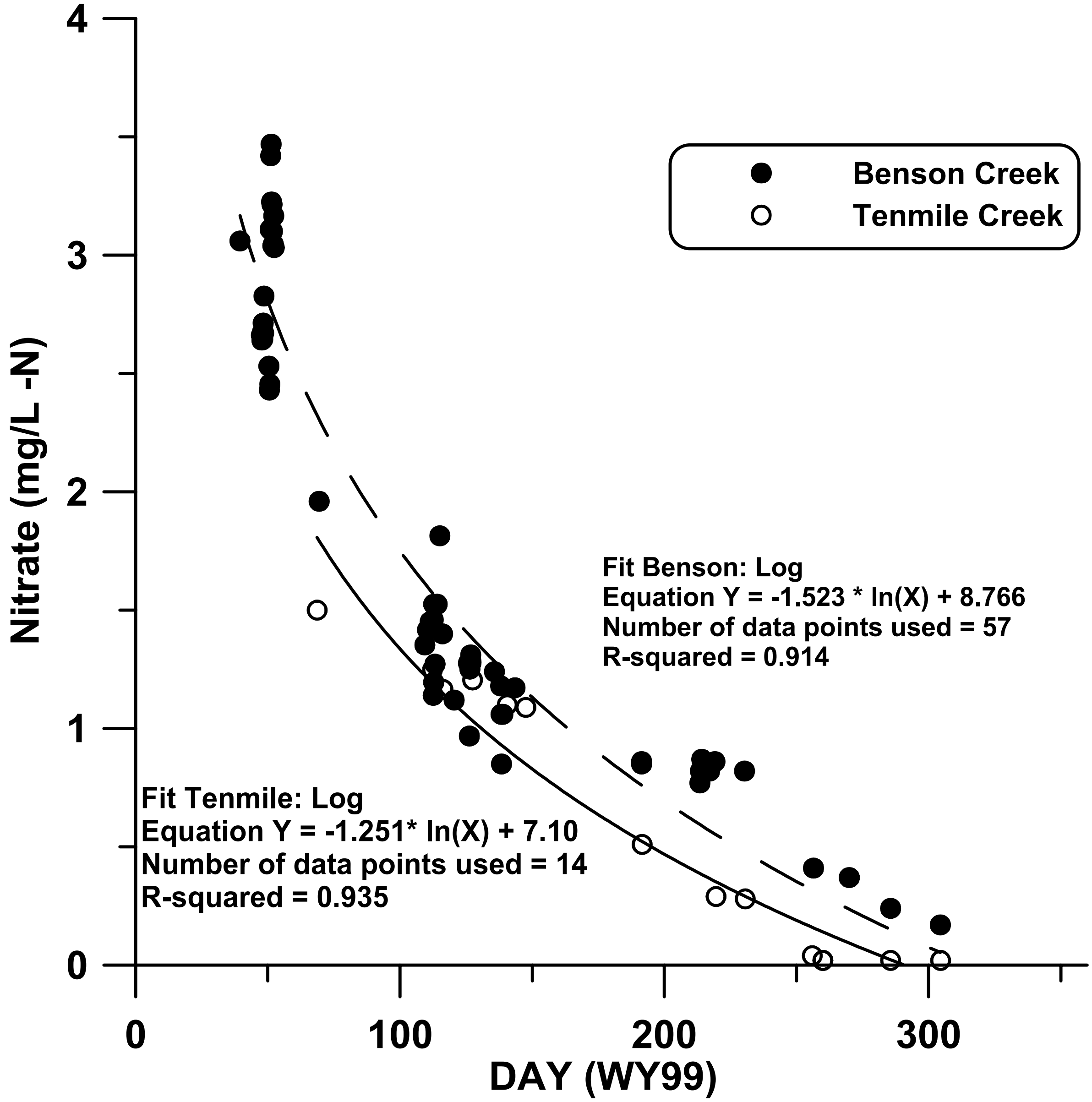


Figure 27. Nitrate (NO_3-N, mg/L) in Benson Creek (●) compared to the outlet of Tenmile Lake (○). The lines represent best fits of the observed data.

pH values (lab) in Tenmile Lake were typically from 6.8 to 7.6 (Figure 28). These values measured in the laboratory may have equilibrated with CO_2 in the atmosphere and may not reflect actual field conditions. However, the maximum field pH measured during Phase I did not exceed 8.0. As expected, there is a noticeable seasonality in the pH data where pH in the summer is higher than that measured in the winter. This reflects the increase in photosynthetic activity in the warmer months.

Specific conductance is proportionally related to the concentrations of major ions in the water. The values in Tenmile Lake closely reflect the pattern observed in the tributaries in which higher concentrations in the fall are diluted by winter runoff. As groundwater becomes a greater proportion of streamflow and evapotranspiration increases, specific conductance increases (Figure 29). The variation in specific conductance has no particular consequence with respect to water quality within the observed ranges, but does illustrate the importance of runoff versus groundwater on lake chemistry. The increase in the relative importance of groundwater in the spring and summer, however, does have significance with respect to transport of dissolved nutrients that enter the lake. Note that the specific conductance for site STB, located closest to the major tributaries, exhibits a greater dilution effect compared to the main lake site at STA.

4. Phytoplankton

a. Chlorophyll a and Phytoplankton

The aquatic plants in a lake reflect the trophic nature of the system and in turn affect the lake fisheries and water quality. Tenmile Lake has an abundant macrophyte population, which is evident in most areas of the lake less than a depth of 5 m. However, no analysis of the macrophyte population was included in the scope of this study[*]. The assessment of primary production for this report was based solely on phytoplankton composition and abundance.

Measures of phytoplankton abundance include chlorophyll, phytoplankton density, and phytoplankton biovolume. Chlorophyll *a* concentrations exceeded 20 µg/L on two occasions, in the spring and the late summer in 1999 (Figure 30). This bimodal distribution of chlorophyll is common in temperate lakes. Typically, the spring bloom is associated with diatoms, whereas the

[*] This does not reflect the potential importance of macrophytes to conditions in the lake, but rather reflects the focus of this study given the available funding. Additional assessment of the lake macrophytes and other important components of the lake are anticipated under future efforts.

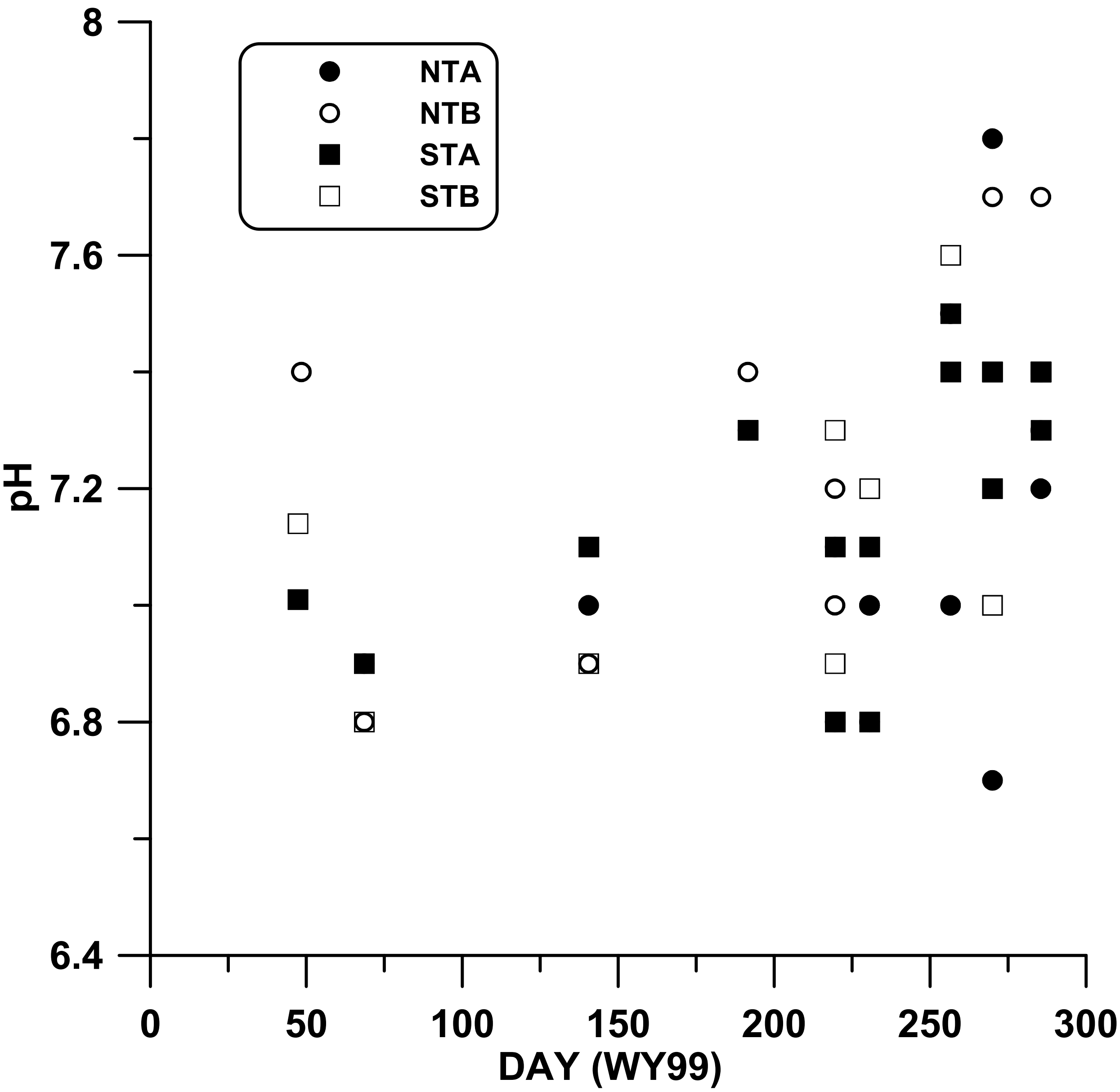


Figure 28. Laboratory pH values in Tenmile Lake. Multiple observations for a given site on the same day represent measurements taken on the surface and near the bottom. In all cases, top values were greater than bottom values.

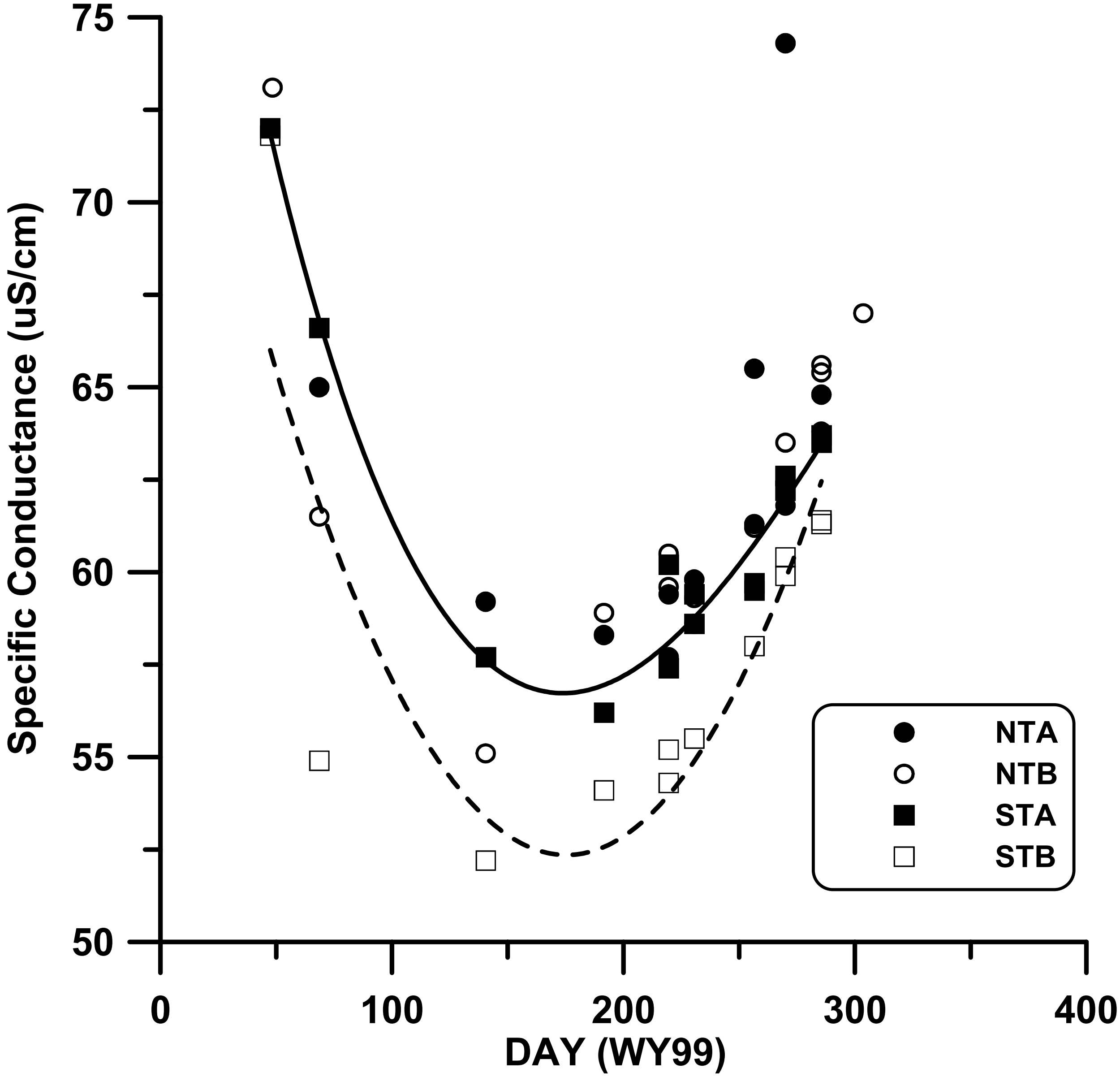


Figure 29. Specific conductance (µS/cm) for the four lake sites versus Julian day. The solid line represents the polynomial fit for site STA and the dashed line represents the fit for site STB.

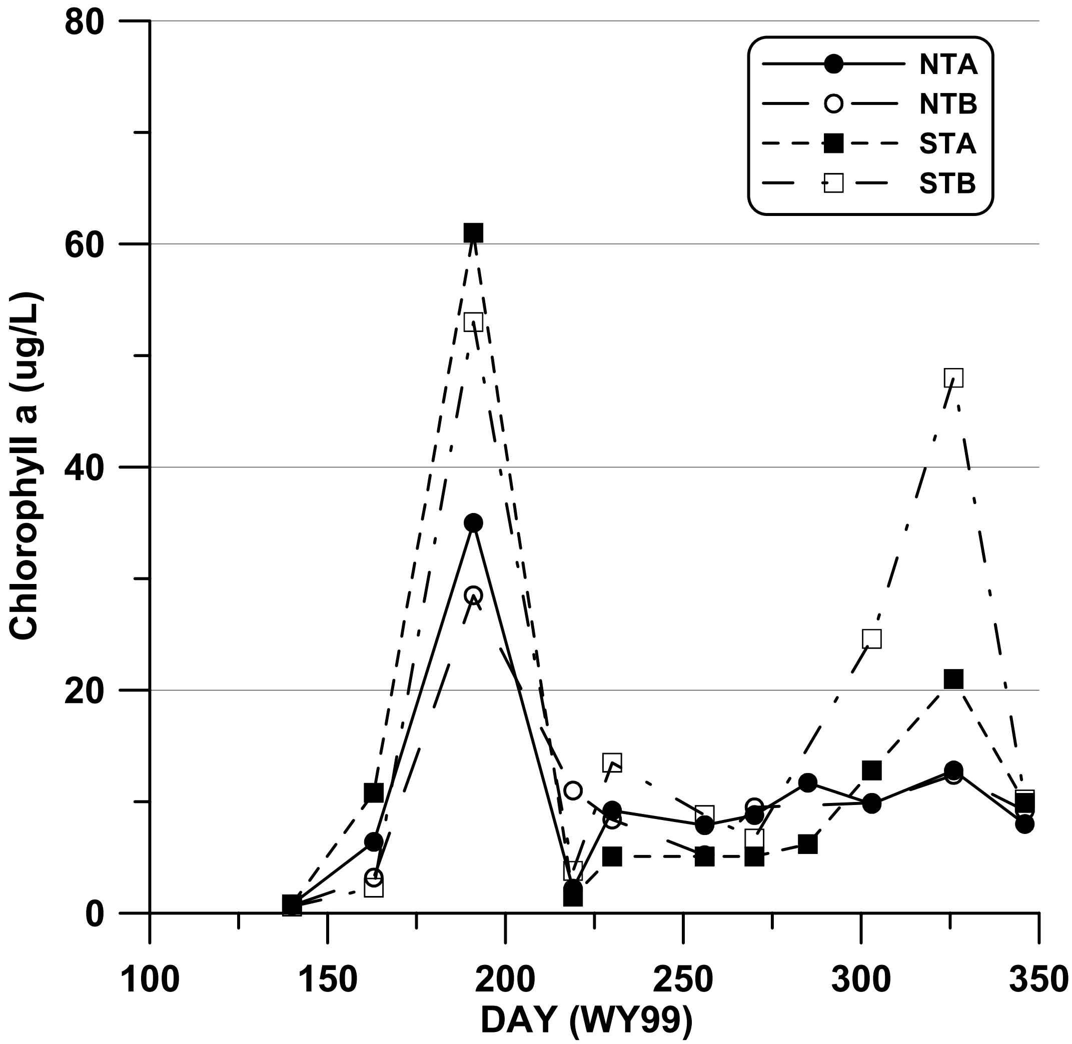


Figure 30. Chlorophyll *a* (µg/L) in Tenmile Lake. Extreme values measured at site NTC (152 µg/L), September 11, 2000 and sheltered locations such as the canal connecting the two lakes are not displayed. The peak value of 61 µg/L shown for STA in WY99 was repeated in WY00 for site STB.

summer bloom is caused by cyanobacteria (blue-green algae). In general, the south lake sites had higher concentrations of chlorophyll *a*. The trophic state index (TSI; Carlson 1977), computed based on the chlorophyll measurements, shows that the lake typically has a TSI value between 50 and 60 (Figure 31). These TSI values are generally indicative of eutrophic conditions.

Phytoplankton density, another metric of algal abundance, shows a similar pattern to that of chlorophyll with the exception that the density does not exhibit a late summer peak (Figure 32). This is consistent with a higher proportion of larger algal cells and colonies which cause a decrease in the cell density relative to the biovolume. This effect is consistent with a diatom (generally individual cells) bloom in the spring and a cyanobacteria (filamentous and clumped cells) bloom in the summer.

Phytoplankton biovolume appears to integrate the information provided in the chlorophyll and plankton density data. Biovolume is high in the spring at all four sites, corresponding to a lakewide diatom bloom (Figure 33). However, in the late summer it is primarily site NTB (Big Creek Arm) which exhibits a high biovolume. Again, this is consistent with a higher proportion of cyanobacteria at NTB (high biovolume, high chlorophyll, low density).

b. Phytoplankton Community Composition

The dominant phytoplankton taxa vary widely in the lake depending on the season (Figure 34). In winter when biovolume is low, the dominant taxa are cryptomonids (*Cryptomonas erosa* and *Rhodomonas minuta*) followed by diatoms. No cyanobacteria were observed in February. In the spring, biovolume increases dramatically and diatoms represented at least 94 percent of the biovolume at each of the four lake sites. The dominant diatom was *Asterionella formosa*, a species often abundant in productive lakes (Reynolds 1984). Again, cyanobacteria were virtually absent in the spring. In late August, cyanobacteria is the dominant planktonic group representing over 40 percent of the biovolume in the south lake and over 85 percent of the biovolume in the north lake. The dominant cyanobacteria species was *Anabaena planctonica*. The cyanobacteria are considered a poor quality food for zooplankton, whereas the cryptomonids and, to a lesser extent, diatoms are high quality food sources.

The seasonal preferences of the dominant phytoplankton groups are better understood in terms of response to multiple factors. Reynolds (1984) used plots with three axes to indicate why different taxa have a competitive advantage under various circumstances. This same

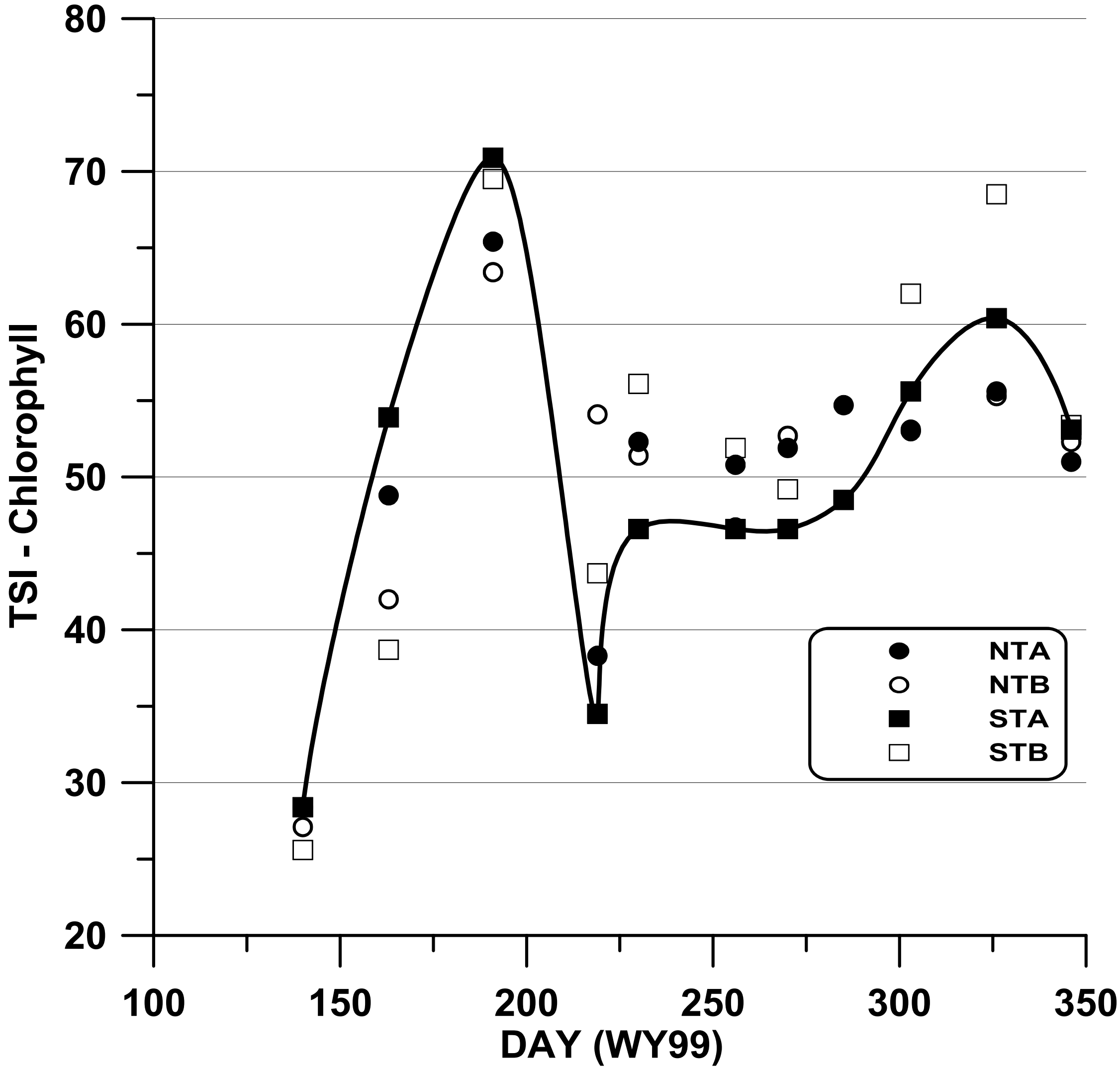


Figure 31 Trophic state index (TSI [Carlson [1977]) for the four lake sites computed on the basis of chlorophyll *a* concentrations. The curve represents a cubic spline fit for site STA.

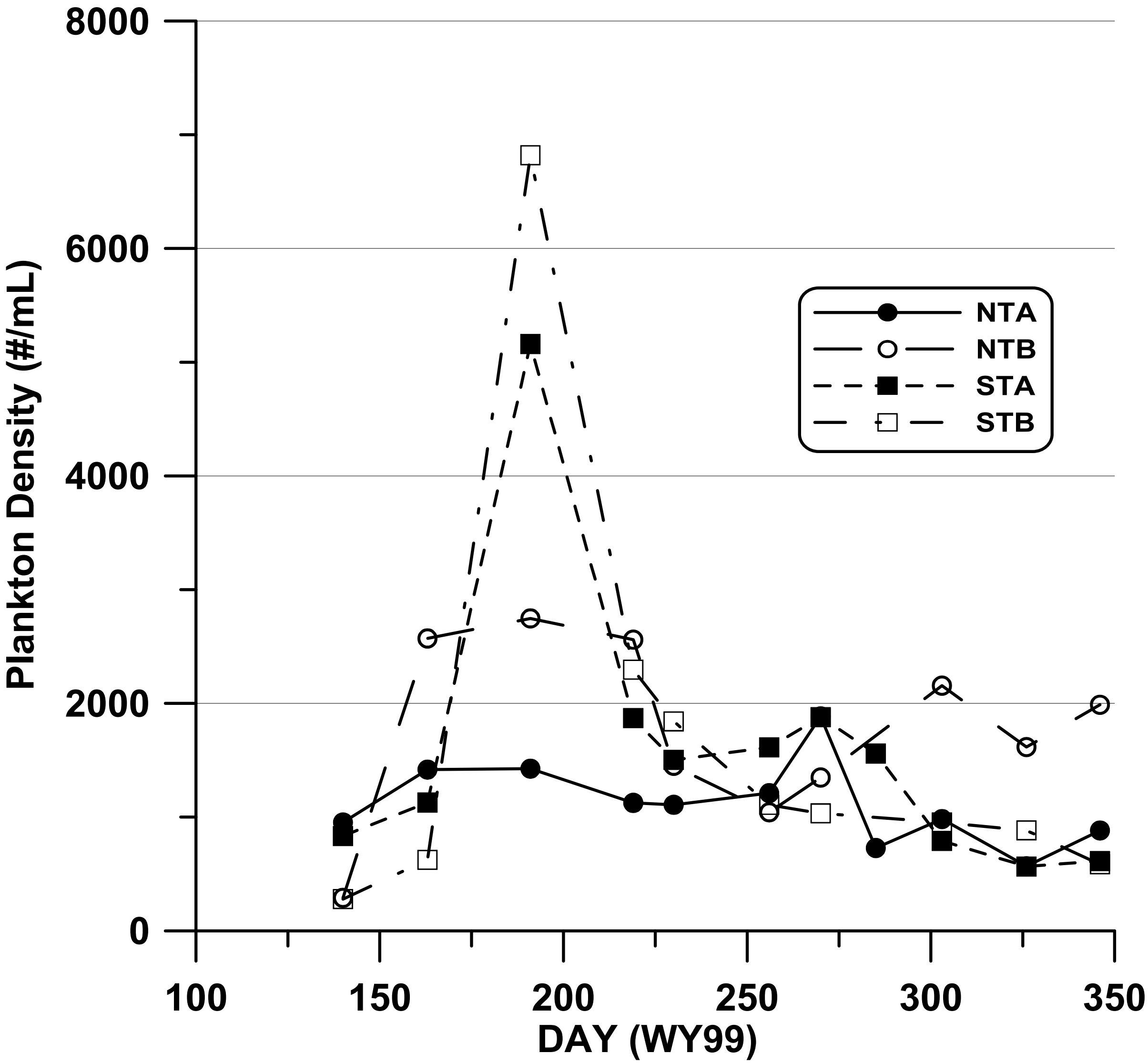


Figure 32. Phytoplankton density (#cells/ml) in Tenmile Lake.

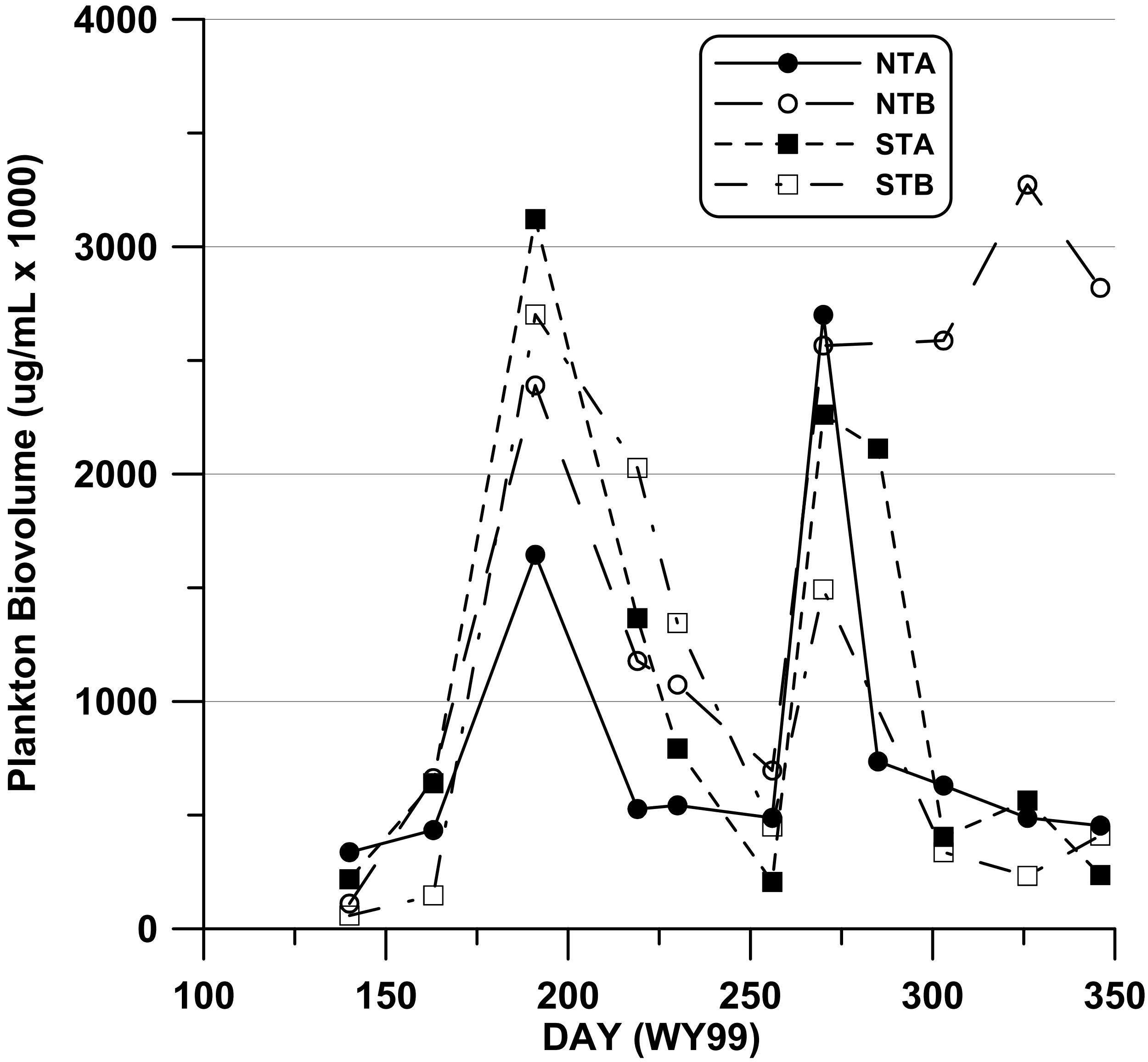


Figure 33. Phytoplankton biovolume in Tenmile Lake.

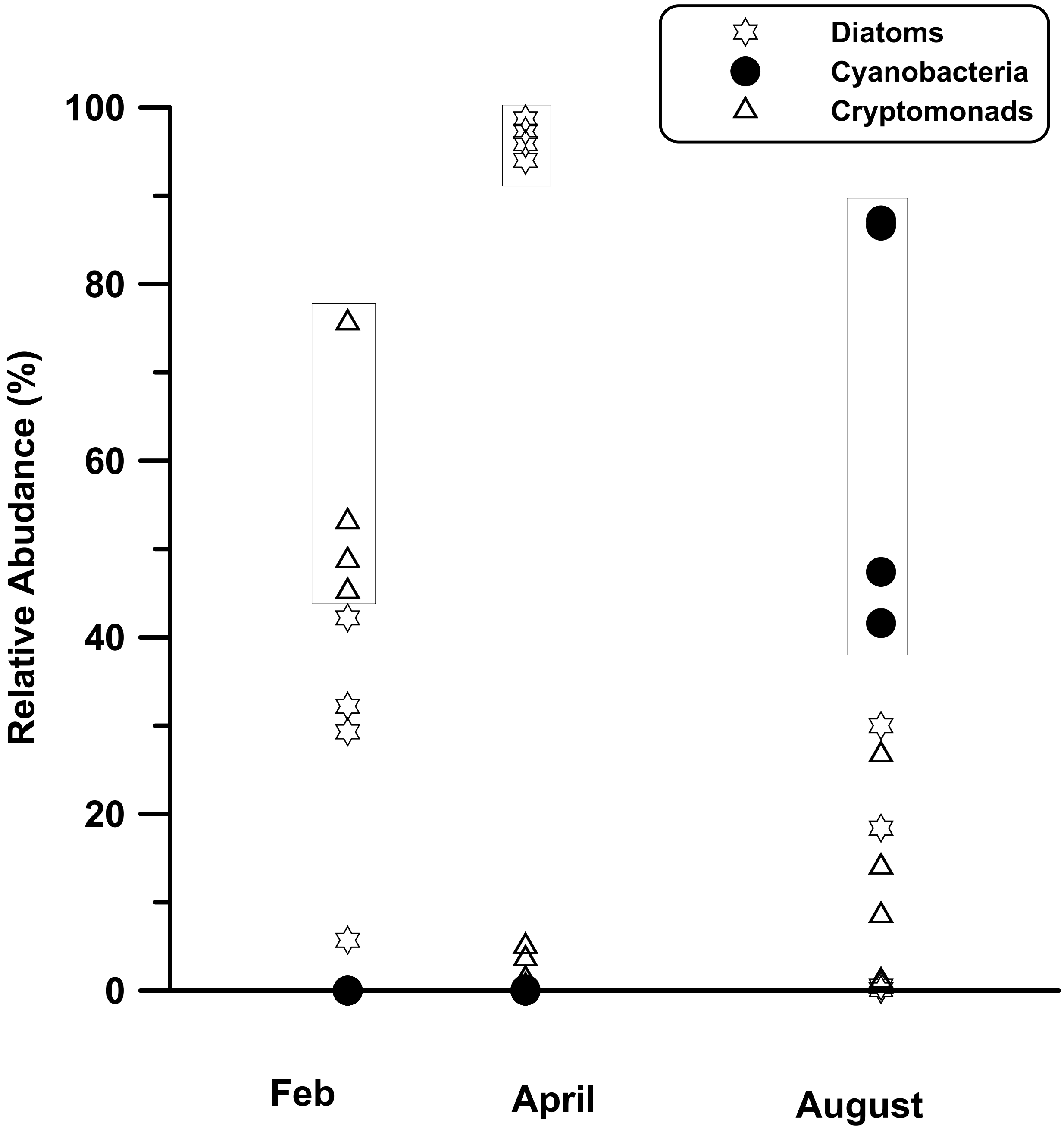


Figure 34. Relative abundance (based on present biovolume) of major taxa of phytoplankton in Tenmile Lake as a function of season.

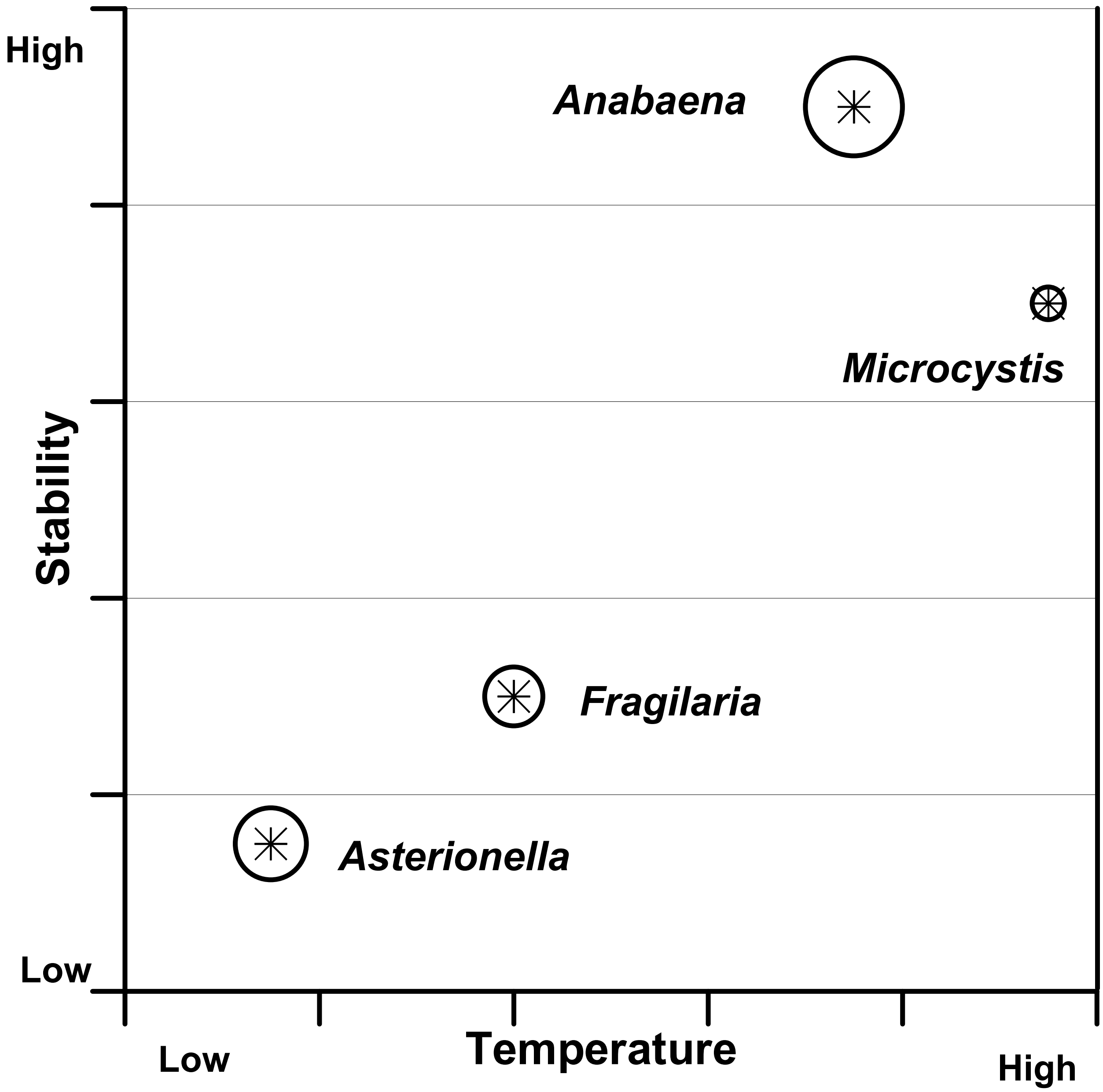


Figure 35. Environmental preferences for four common taxa observed in Tenmile Lake. The size of the circles corresponds to relative preferences for nutrients where large circles indicate preferences for greater concentrations of nutrients. (Modified from Reynolds 1984.)

information is conveyed in Figure 35 which illustrates the strong temperature and lake stability preferences of four important taxa found in Tenmile Lake. The diatoms, first *Asterionella* followed by *Fragilaria*, appear in the spring because of their preferences for well-mixed waters (to maintain their presence in the water column) and cool waters. Because diatoms rely on available inorganic N for a nitrogen source, the high concentrations of nitrate present in the lake during spring probably further aid their growth (Figure 36). Once the lake warms and becomes calmer, the cyanobacteria, *Anabaena, Aphanizomenon,* and *Microcystis,* have a competitive advantage. The calmer high nutrient conditions favor *Anabaena*, whereas *Mycrocystis* is more dependent on high temperatures.

Additional factors favoring the different taxa include the TN:TP ratio and the forms of N available to the algae. A high TN:TP ratio will favor cyanobacteria. Thus the very high TN:TP ratios from the altered catchments, such as Big Creek, deliver loads that increase spring blooms of diatoms, whereas the low TN:TP ratios in the summer (1:1) in Tenmile Lake result in the dominance by *Anabaena*, an N-fixing cyanobacteria (Figure 36). The presence of *Mycrocystis* in Tenmile Lake, which does not fix nitrogen, is probably made possible by the compatibility of *Mycrocystis* to acquire NH_3 from the lake when the lake becomes anaerobic at the bottom (Figure 17). *Mycrocystis* has tremendous capabilities to migrate vertically and acquire NH_3 from the bottom waters and return to the surface to photosynthesis (Reynolds 1984).

c. Mycrocystis

Supplemental to the overall watershed and lake study, additional lake sampling was conducted specifically to assess the dynamics of the potentially toxic blue-green alga, *Microcystis aeruginosa*. This species produces potent hepatotoxins (known as microcystins) that are capable of harmful effects to animals and humans (Chorus and Bartram 1999). A toxic bloom of *M. aeruginosa* was first documented in Tenmile Lakes in September of 1997, prompting the Oregon Department of Health to issue a health advisory recommending that the lake not be used for drinking water and that contact recreation be avoided (Kann and Gilroy 1998). The goal of this supplemental sampling, in partnership with the Tenmile Lakes Basin Partnership, was to specifically target *M. aeruginosa* to better understand conditions favorable for growth, bloom formation, and toxin production.

The *Microcystis* sampling and microcystin analysis indicated that the organism and its toxin were present at levels to be considered a human health concern (Table 6). *M. aeruginosa* was

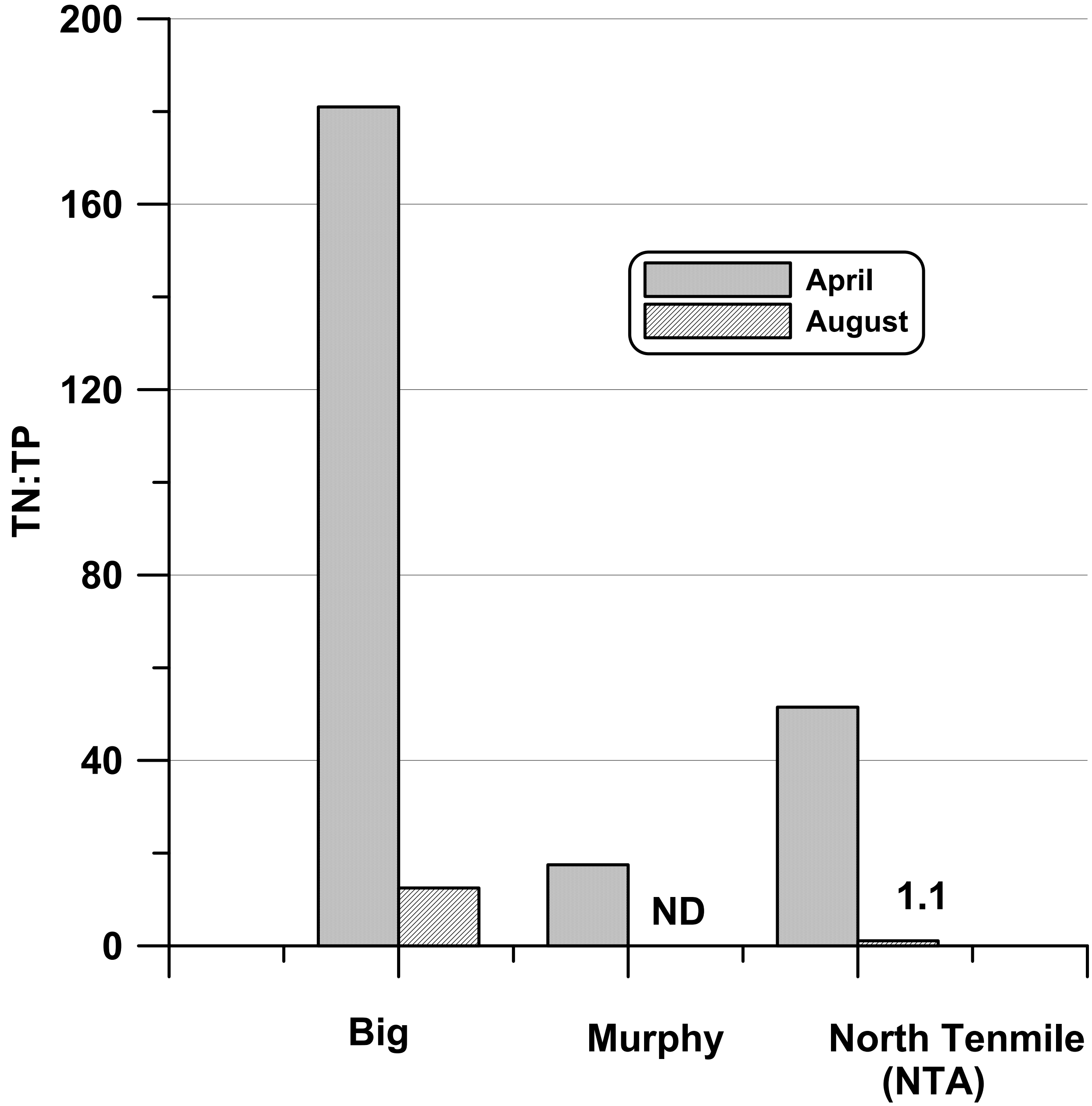


Figure 36. Ratio of total nitrogen to total phosphorus (TN:TP) for selected tributary and lake sites for the months of April and August.

Table 6. *Microcystis aeruginosa* colony abundance, biomass, and microcystin toxin concentration in North and South Tenmile Lakes, July 21-September 4, 2000.

STATION	DATE	*Microcystis* Abundance					Biomass (GALD X Colony #)				Microcystin	
		M. aeruginosa Colonies (no./L)	N & S Col. mean	North Col. Mean	South Col. mean	GALD* (um)	Biomass Estimate	N and S Bio. Mean	North Bio. Mean	South Bio. Mean	ELISA (µg/g)	ELISA (µg/L)
N11	21-Jul	333				540	180077				63.30	2.03
N16 (NTA)	21-Jul	133				264	35241				26.70	1.17
S3	21-Jul	133				358	47662				25.60	1.05
S8 (STA)	21-Jul	600	300	233	367	477	285905	137221	107659	166783	57.50	2.30
N16 (NTA)	10-Aug	333				451	150185				1.01	0.04
NTB	10-Aug	67				480	31997				0.80	0.03
S8 (STA)	10-Aug	333				538	179382				1.14	0.04
STB	10-Aug	133	217	200	233	318	42396	100990	91091	110889	1.11	0.04
N11	21-Aug	133				170	22598					
N16 (NTA)	21-Aug	0				.	0					
S3	21-Aug	133				244	32530					
S8 (STA)	21-Aug	800	267	67	467	220	175816	57736	11299	104173		
N11	4-Sep	1200				365	437490				6.72	0.52
N16 (NTA)	4-Sep	533				388	206646					
S3	4-Sep	200				309	61860				16.52	0.59
S8 (STA)	4-Sep	67	500	867	133	190	12665	179665	322068	37263	7.06	0.49

* GALD = greatest axial linear dimension.

present at low levels at the time of the first sample trip on July 21, 2000 (Figures 37 and 38), with the highest microcystin values (ELSIA µg L^{-1}) encountered at stations N11 and S8 (Figure 40). These values were ~2X the World Health Organization (WHO) guidance level for drinking water (1 ppb or 1 µg L^{-1}; Falconer et al. 1999). However the WHO guidance level was exceeded at all stations. Mean *M. aeruginosa* colony abundance and biomass remained fairly stable in both lakes at the time of the August 11th sampling date (Figures 38 and 39). However, microcystin toxin levels declined to well below the WHO guideline (Figure 39). A decrease in abundance and biomass occurred in North Lake, and a slight increase (for abundance) in South Lake on August 22nd. However, when overall counts of a species are low it is important to note that minor differences in colony counts (due to low numbers of colonies encountered) in the sub-samples can be extrapolated to seemingly larger differences when converting to numbers per

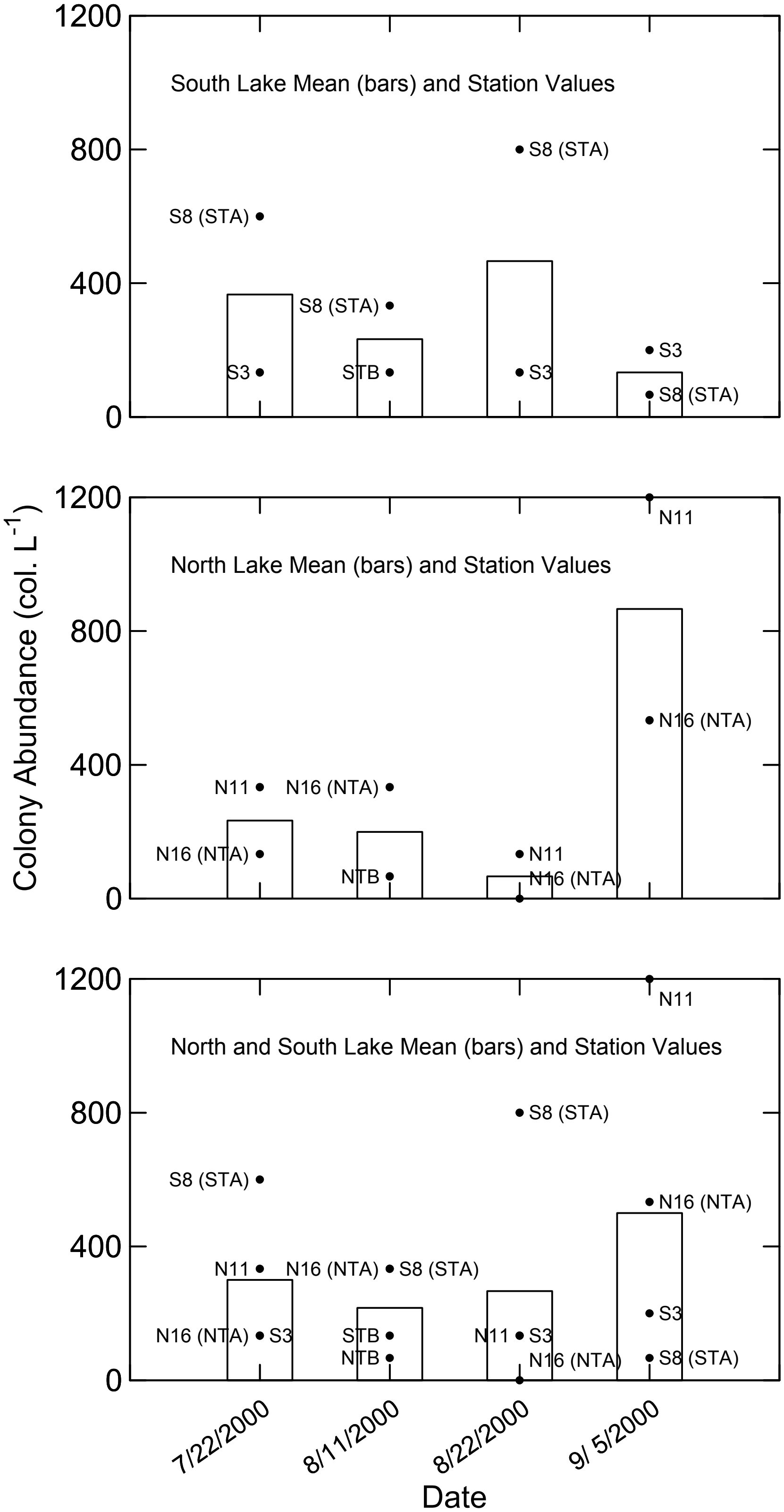


Figure 37. *Microcystis aeruginosa* colony abundance in North and South Tenmile Lakes, July 21-September 4, 2000.

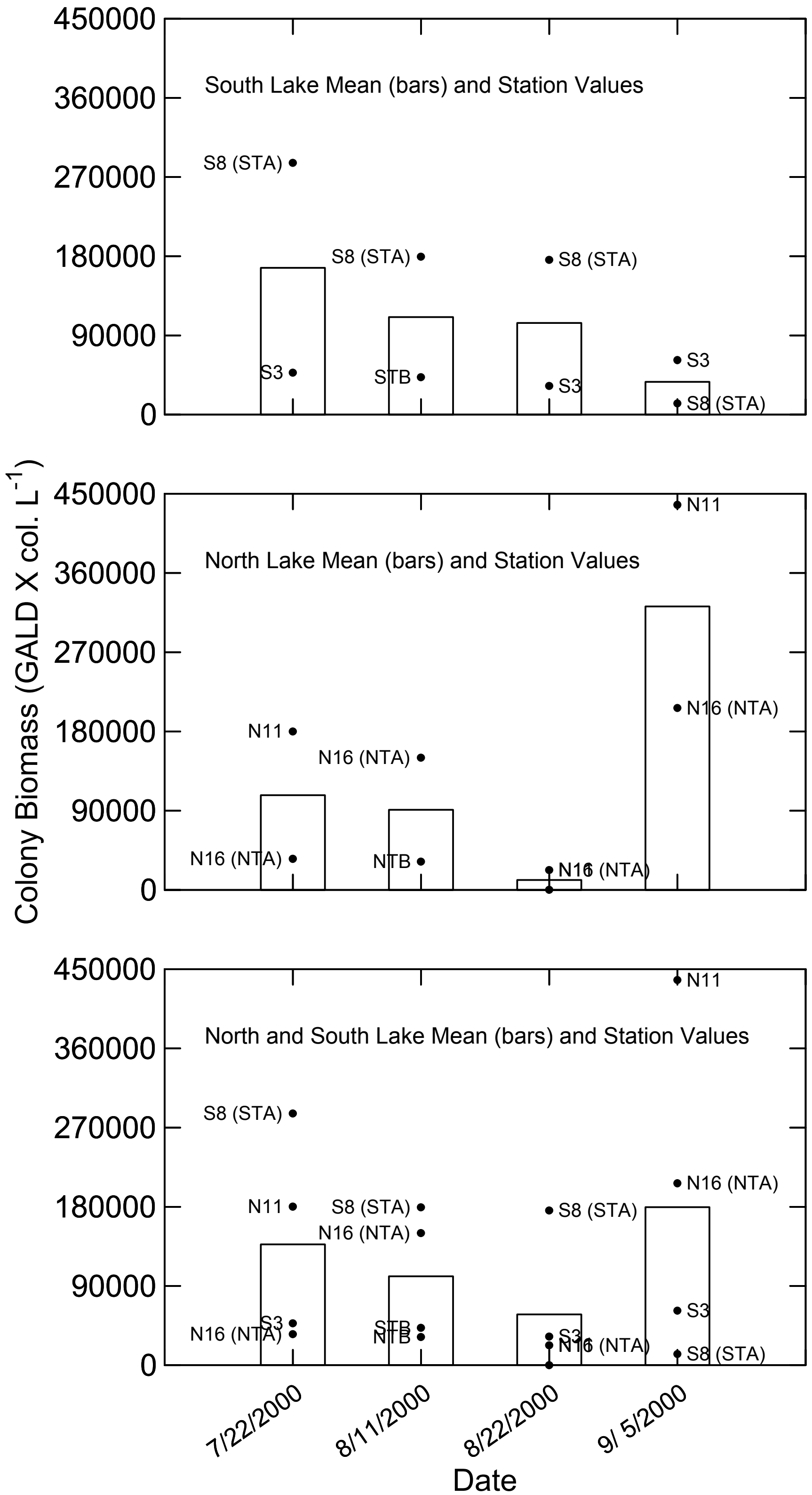


Figure 38. *Microcystis aeruginosa* biomass in North and South Tenmile Lakes, July 21-September 4, 2000.

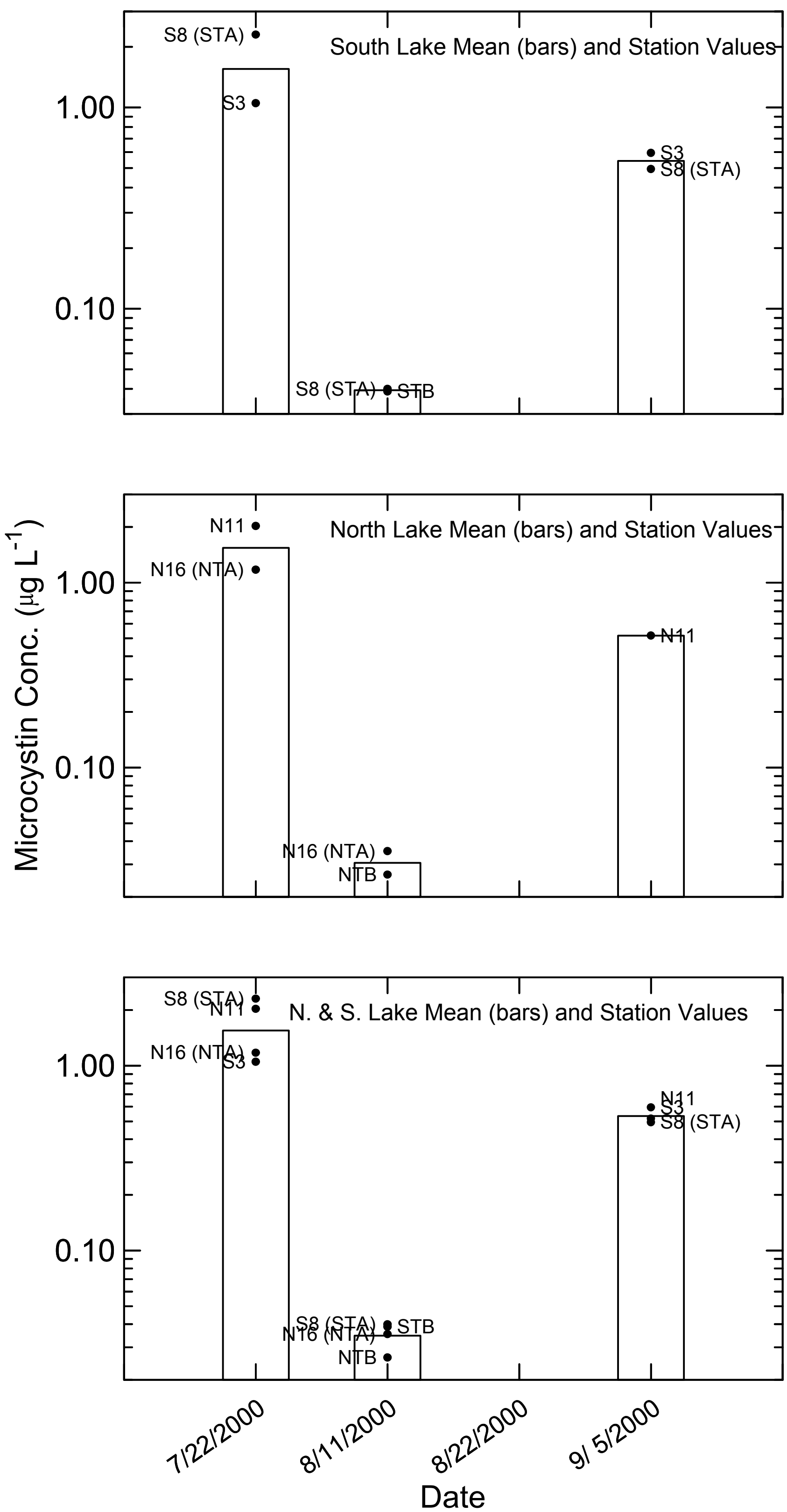


Figure 39. Microcystin toxin concentration in North and South Tenmile Lakes, July 21-September 4, 2000.

liter. Thus, individual sites that may seem to have increased (e.g., S8 which nearly doubled from 8/11 to 8/22) or decreased substantially, are in fact not significantly different. This is especially true when levels are low overall, and in this case *M. aeruginosa* is not the predominant species relative to the other phytoplankton present in the lakes. In fact, during the period encompassing these dates, *M. aeruginosa* represented only 0.2% to 13% of the total phytoplankton biomass, with other bloom-forming blue-green algae such as *Anabaena flos-aquae* and *Aphanizomenon flos-aquae* predominating.

South Lake biomass and abundance decreased substantially at both stations by the September 4th sample date, but increased in North Lake at both stations (Figures 37 and 38). Values were particularly high at station N11, located in Big Creek Arm. However, these higher biomass and abundance levels did not translate to higher microcystin toxin levels, which although higher than values on August 11th, were lower than those occurring on July 22nd (Figure 39). Microcystin toxin levels also remained below the WHO guidance level in both lakes. Moreover, ELISA (mg g^{-1} of dry weight algae) does not always correlate with ELISA (mg L^{-1}) due to the presence of other bloom forming algae present in the dry weight samples.

These survey results indicate that despite low levels of *M. aeruginosa* and the lack of a major *M. aeruginosa* bloom during the late July to early September period, that microcystin toxin concentration still exceeded WHO guideline levels for drinking water. This is a significant point given the use of Tenmile Lakes water as a potable water source in numerous lakeside homes. Due to the patchy nature of blue-green algal blooms, it is possible for higher *Microcystis* densities (and therefore higher microcystin toxin concentrations) to be present in areas not sampled in this survey, particularly along shorelines or during calm conditions of little to no wind. Given the lakes' demonstrated history of toxic *Microcystis* blooms, and the fact that all areas of the lake cannot be tested at all times, those utilizing the lake for drinking water should always follow Oregon Health Division recommendations for purification. In addition, recreational users should always avoid contact with water whenever noticeable surface concentrations of algae are evident or when the lake has an obvious green to blue-green appearance. Moreover, because pets or other domestic animals are the most likely to ingest contaminated water, these animals should not be allowed access to the lakeshore whenever either noticeable surface concentrations of algae or an obvious green to blue-green appearance is evident.

Even though *M. aeruginosa* was low relative to other phytoplankton, the lakes still experienced a substantial blue-green algal bloom. Such blooms are indicative of highly eutrophic (enriched) conditions, and further efforts should focus on controlling the causes of these blooms. The most effective way to reduce the frequency of both toxic and non-toxic blooms is through effective lake and watershed management. Yoo et al. (1995) conclude that prevention of cyanobacterial blooms is the key to the control of toxic blooms.

It is evident that under the current lake trophic state that the potential for large *Microcystis* blooms is present in the Tenmile Lake system. Whether or not a large bloom materializes, however, may be dependent upon year-to-year variability in climatic factors such as solar radiation, precipitation, wind, and temperature. It is clear that the current nutrient regime in Tenmile Lakes is sufficient to support toxic blooms of *Microcystis,* but that growth and dominance in any given year will be subject to the variability of climatic conditions. This is especially true for blue-greens such as *Microcystis,* which are generally favored by warmer and calmer in-lake conditions (Reynolds 1984). Previous sampling indicated that the seasonal pattern in *Microcystis* biomass and toxin concentration tended to follow the pattern in water surface temperature (Kann 1999).

5. Lake Sediment

The sediment core from the south lake (STA) was dated using ^{210}Pb and analyzed for fossil diatoms, ^{15}N, and cyanobacteria akinetes. The ^{210}Pb provides estimates of the age of the sediments and therefore allows one to compute the sediment accumulation rate (SAR). Diatom taxa preserve well in lake sediments because the cell wall is composed of silica. Diatoms have specific environmental requirements and the type of species in the sediments provide considerable information about the historical water quality in the lake. Nitrogen-15 is a naturally-occurring isotope of nitrogen (atomic weight 14). A shift in the $^{15}N/^{14}N$ ratio can provide insight into shifts in major sources of nitrogen to the lake. In particular, an elevated ratio can be associated with high inputs of marine-derived N from anadromous fish. It also can be caused by a shift in the proportion of N-fixing phytoplankton that might occur if the relative amount of cyanobacteria had increased. Akinetes are structures found on N-fixing cyanobacteria that can be preserved in the sediments. An increase in akinetes might signal an increase in cyanobacteria.

a. Sediment Accumulation Rates

The SAR measured in the main basin of South Tenmile Lake shows an increase from about 100 g/m^2/yr prior to settlement to a current value approaching 900 g/m^2/yr (Figure 40). The core from the main basin in North Tenmile Lake shows a baseline SAR near 400 g/m^2/yr increasing to a current rate of 800 g/m^2/yr (Figure 41). The core from Lindross Arm is comparable to the main basin core in the north lake site, but the core from site NTB, Coleman Arm, exhibits a current SAR of about 1300 g/m^2/yr (Figure 42). Thus, the minimum increase in SAR is about two-fold at NTA, increasing to maximum measured rates of four-fold baseline levels at NTB and possibly much greater at STA. The rates of phosphorus and nitrogen accumulation are greater than the increase in sediment because of the concomitant increase in the concentration of nutrients in the sediment (Eilers et al. 1996a). A comparison of the SAR in Tenmile Lake (STA) with Devils Lake, located further up the coast, shows some interesting parallels (Figure 43). The SAR in both lakes shows a pre-settlement SAR near 200 g/m^2/yr and a major increase between 1910 to 1920, possibly as a consequence of early logging combined with a major storm event (Eilers et al. 1996). However, Devils Lake appeared to show a rapid recovery following the event, whereas Tenmile Lake experienced high SAR through the early 1950s at which point it decreased to near pre-development levels. Both lakes have since shown a return to high SAR as lakeshore development and high rates of timber harvest contributed to renewed rates of erosion.

The sediment core samples collected under Phase II were processed by a different laboratory than in the previous sediment work by Eilers et al. (1996a). However, a comparison of the ^{210}Pb activity measured previously in Phase I shows reasonably close agreement in overall levels of activity with those measured in Phase II (Figures 44 and 45). The STA core from Phase I exhibits a higher degree of apparent disturbance in the upper 20 cm of the core, but the two cores from NTA exhibit virtually the same pattern.

b. Diatoms

The dominant diatom taxa in the lake sediments are illustrated in Figure 46. One hundred and ten diatom taxa were identified in the sediment core from site STA. The entire core is dominated by planktonic diatoms (78-98%). Three zones have been identified and are described below.

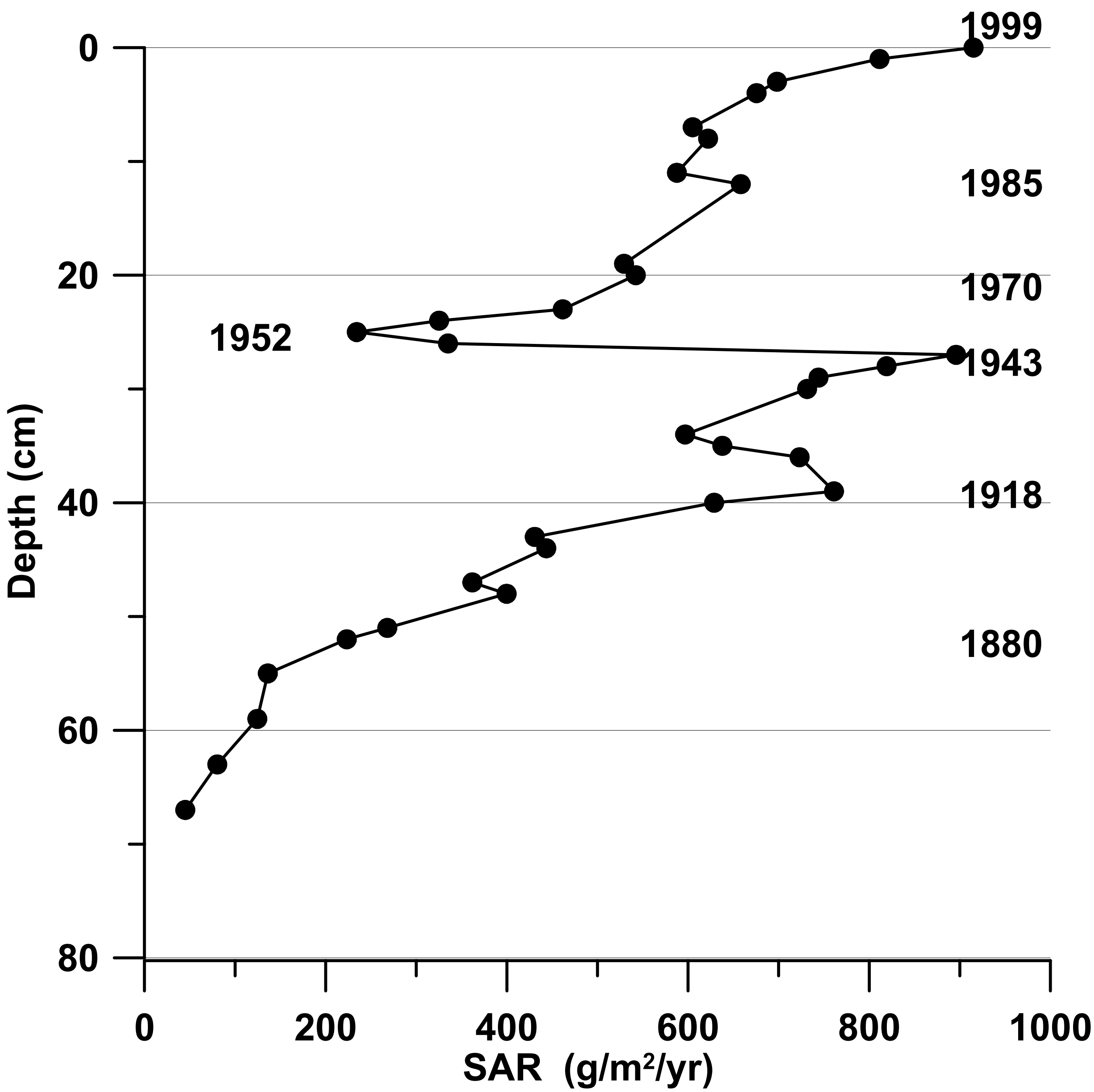


Figure 40. Sediment accumulation rates (SAR) for site STA, South Tenmile Lake.

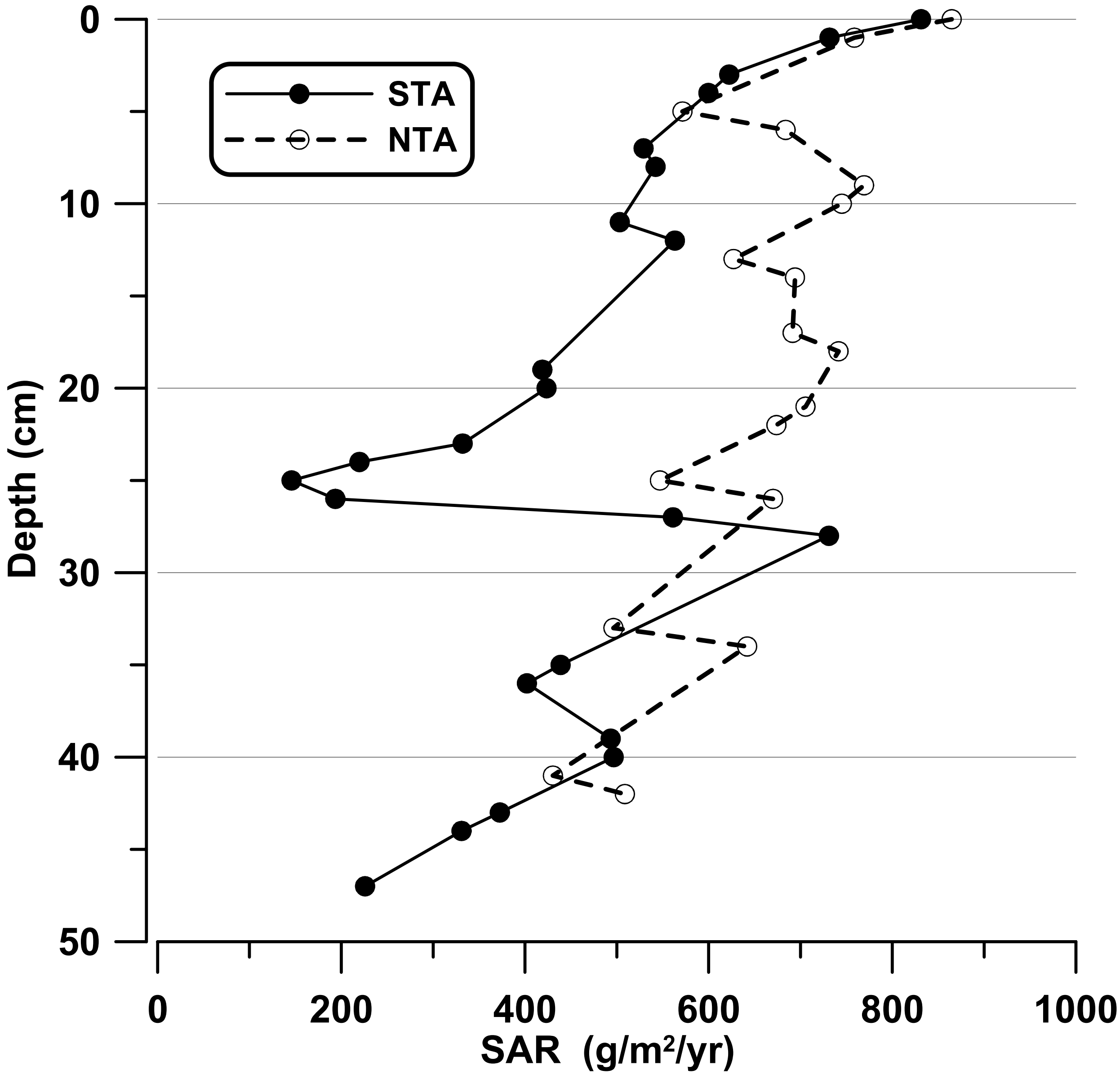


Figure 41. Sediment accumulation rates (SAR) for cores from sites STA (south lake) and NTA (north lake).

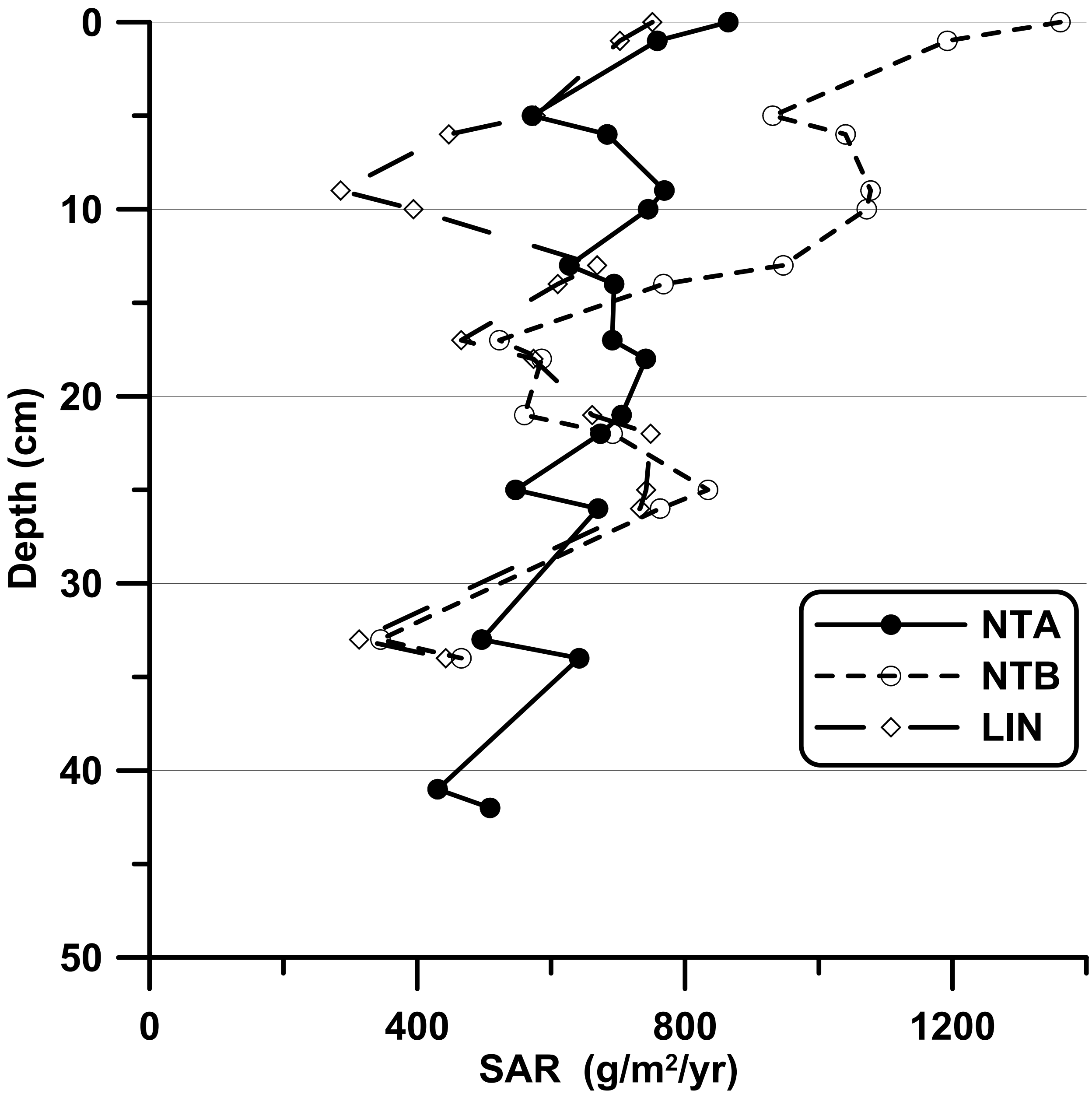


Figure 42. Sediment accumulation rates for the three cores collected from North Tenmile Lake where NTA and NTB are the primary sampling sites shown in Figure 5 and LIN is a core collected from the Lindross Arm of North Tenmile Lake.

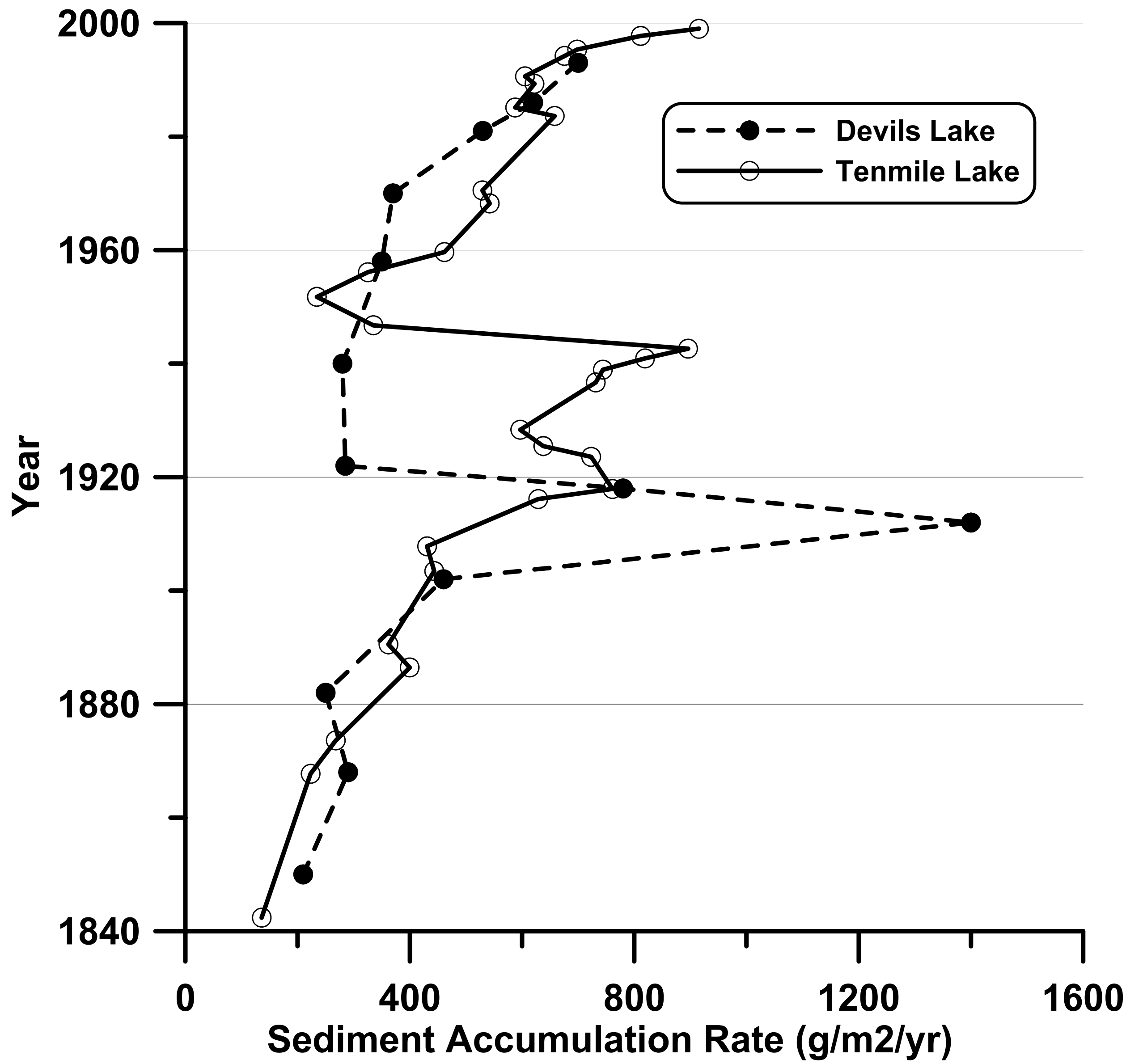


Figure 43. Sediment accumulation rates (SAR) from Tenmile Lake (core STA) and Devils Lake, OR. (Eilers et al. 1996b).

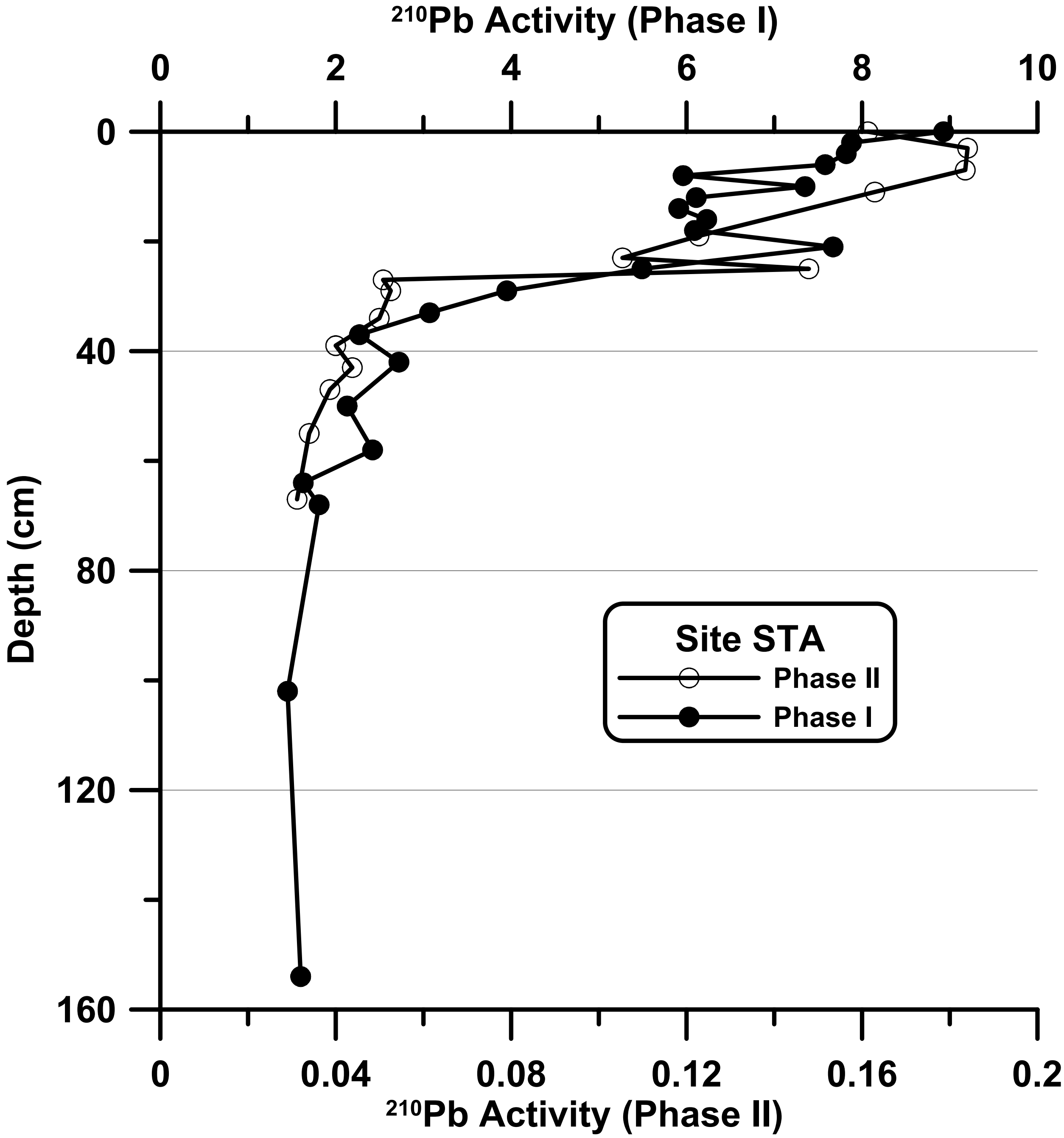


Figure 44. ^{210}Pb activity measured in sediment cores collected earlier by Eilers et al. (1996a) at site STA and again in Phase II. The Phase I cores were reported as disintegrations per minute (DPM), whereas the Phase II ^{210}PB activity is expressed as Bq/g.

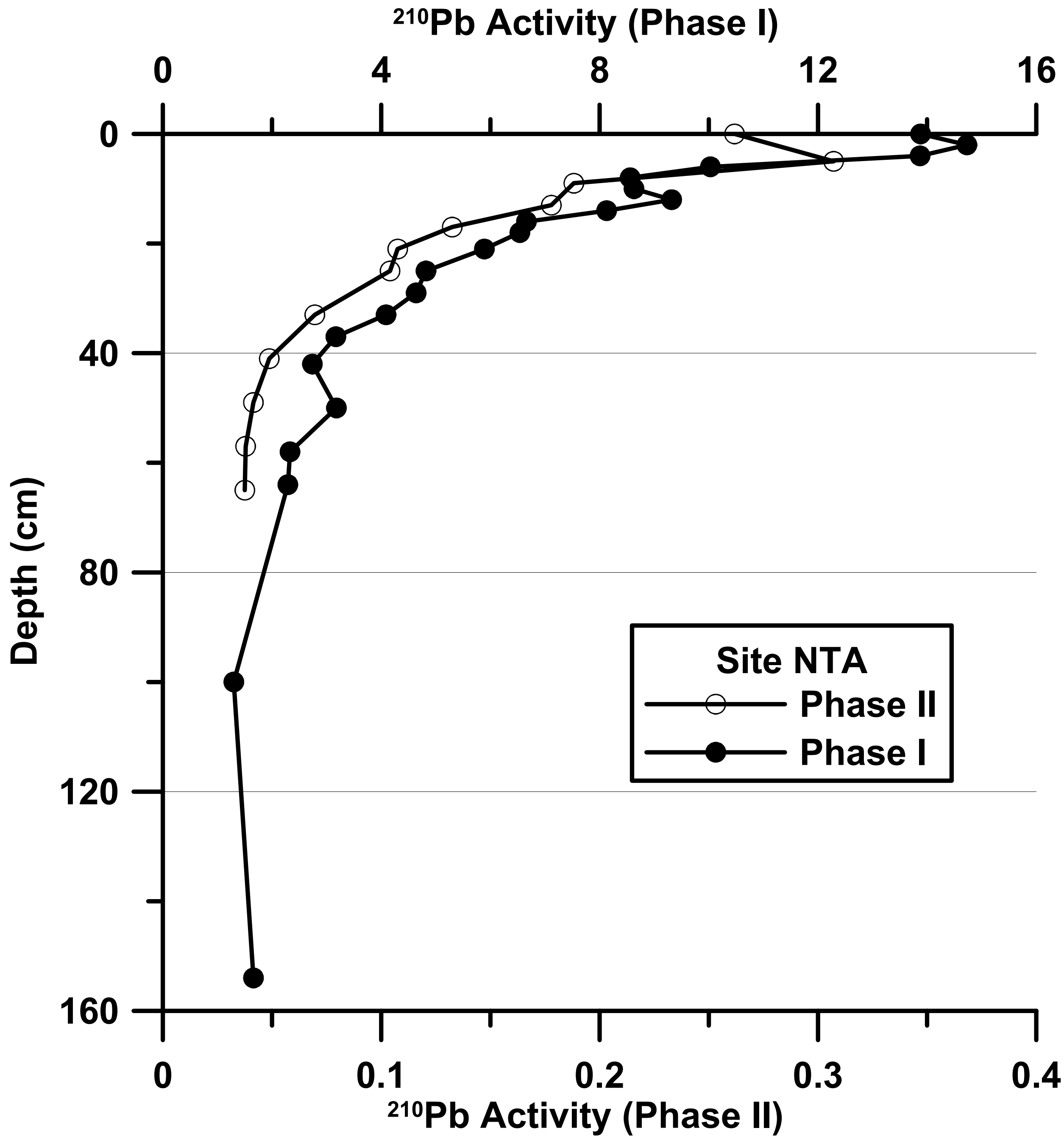


Figure 45. ^{210}Pb activity measured in sediment cores collected earlier by Eilers et al. (1996a) at site NTA and again in Phase II. The Phase I cores were reported as disintegrations per minute (DPM), whereas the Phase II ^{210}PB activity is expressed as Bq/g.

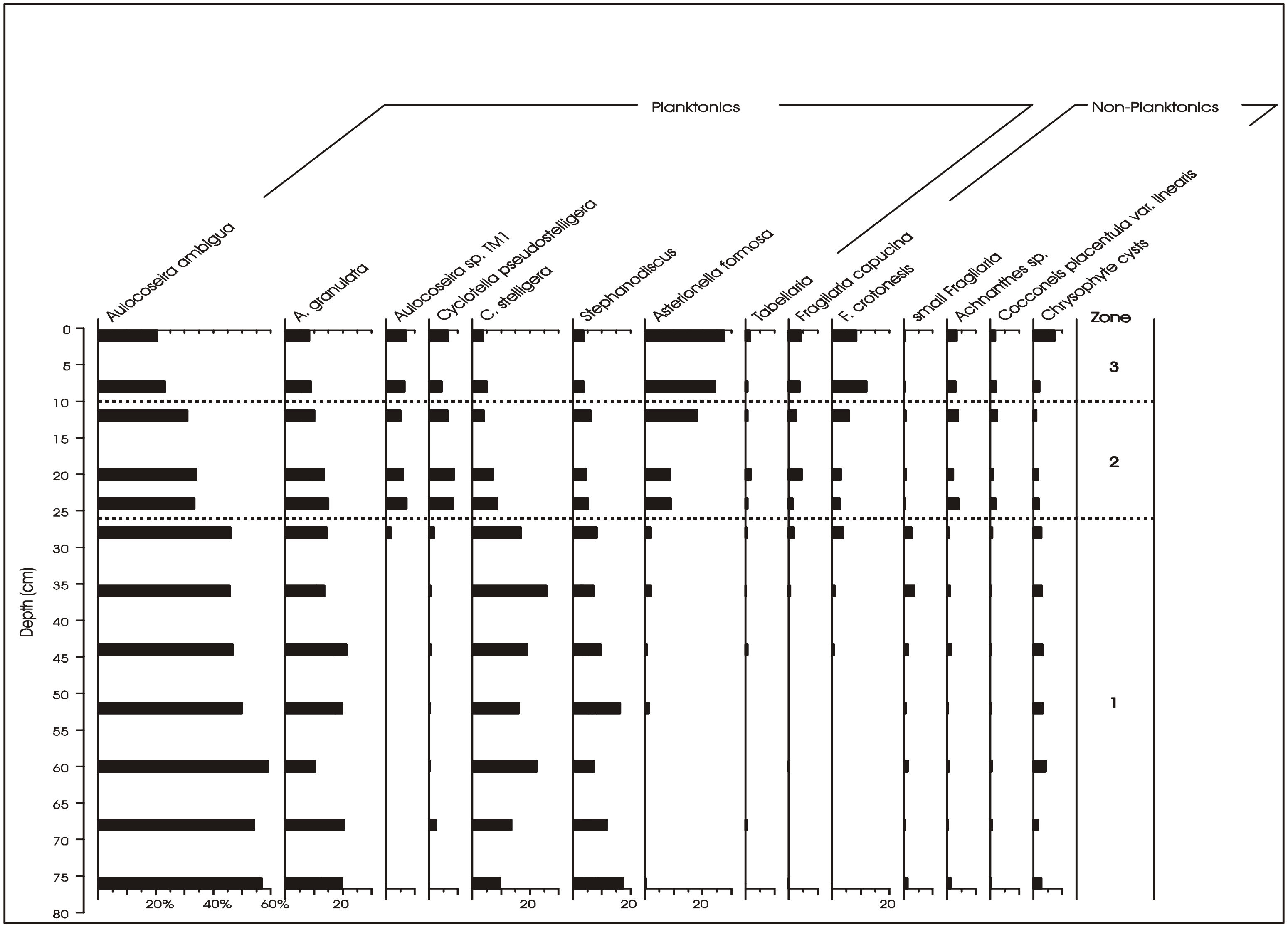


Figure 46. Dominant diatom taxa in the sediments of Tenmile Lake (core STA).

Zone I (76 cm-26 cm [<1000-1931 (± 14 yrs)])

Zone I is dominated by centric diatoms (86-96%), including *Aulacoseira, Melosira, Cyclotella, Stephanodiscus*, and *Thalassosira*. *Aulacoseira* cf. *ambigua* and *A. granulata* (59-77%) occur in the greatest abundances. *Cyclotella stelligera*, is also important (10-26%). *Stephanodiscus* (4-9%), mainly *S. medius, S. minutulus* and *S. Oregonicus*, and small benthic *Fragilaria* species (<1-4%) are present in small numbers. Araphideae diatoms occur in some samples, but only in small numbers. For example, *Asterionella formosa* occurs in five out of seven level (<1-2%), *F. crotonensis* occurs in three levels (<1-3%), *F. capucina* occurs in four levels (<1-2%) and *Tabellaria* occurs in four levels (<1%). Other minor changes are evident in Zone I. At 45 cm, the relative abundance of *Aulocoseira ambigua* declines below 50% and *Asterionella formosa* is present in the sediment beginning at 45 cm (circa 1903). Thus there is evidence that by the turn of the century, Tenmile Lake had already experienced some change in water quality.

Zone II (26 cm-10 cm [1931 (± 14 yrs) - 1984 (± 1 yr)])

Zone 2 is marked by an increase in Araphideae diatoms (15-28%) and a coincident decrease in centric diatoms (60-76%). Specifically, *Asterionella formosa* (9-18%), *F. crotonensis* (3-6%), *F. capucina* (1-5%), and *Tabellaria* (<1-2%) increase, whereas *Aulacoseira ambigua* and *A. granulata* (41-48%), *Cyclotella stelligera* (4-9%) and *Stephanodiscus* (2-3%) decrease. Notably, *Aulacoseira sp.* TM1 (5-7%) and *C. pseudostelligera* (6-8%) appear for the first time in significant numbers. Small benthic *Fragilaria* species (<1%) all but disappear, whereas two Monoraphideae diatoms, *Achnanthes* (2-4%) and *Cocconeis*, increase from <1.5% and <1%, respectively.

Zone III (10 cm-0 cm [1984 (± 1 yr) - 1999])

Zone III is delineated by a further increase in Araphideae diatoms (42%) and a further decrease in centric diatoms (49-50%). Specifically, *A. formosa* (24-28%), *F. crotonensis* (9-12%) and *F. capucina* (4%) increase, whereas *A. ambigua* and *A. granulata* (29-32%), *C. stelligera* (4-5%), and *Stephanodiscus* (2%) decrease. Several taxa, including *Tabellaria* (<1-1.5%), *Aulacoseira* sp TM1 and *C. pseudostelligera* (4-7%) decrease slightly and small benthic *Fragilaria* taxa almost disappear.

The shift in diatom community composition is perhaps best illustrated by closer examination of three taxonomic groups: (1) the centric diatoms (comprised largely of *Aulacoseira* with contributions from *Cyclotella* and *Stephanodiscus*), (2) *Fragilaria crotonensis*, and (3) *Asterionella formosa*. The centric diatoms are commonly found in less productive lakes. The latter two species are Araphideae taxa indicative of very productive waters. The decline of the centric diatoms (Figure 47) and increase of the *F. crotonensis* and *A. formosa* (Figure 48) are a clear indication that the lake has become a much more favorable habitat for diatoms that require high availability of nutrients. The shift in these taxa and the overall change in diatom community composition corresponds with the general zonation (I-III) in lake and watershed activities shown in Figure 49.

An examination of land use and lake changes shows that the high SAR (defined here as SAR > 600 $g/m^2/yr$) for South Tenmile Lake (site STA) corresponds with initial land use disturbance from timber harvest, agriculture, and stream channelization (Figure 49). The continued high SAR during the depression when timber harvest in the watershed was very low suggests that agricultural and stream channelization (or maintenance) was a major factor in sustaining the erosional inputs. The major deterioration in water quality, as indicated in Diatom Zone II, did not occur until after the SAR declined. Some of the noteworthy changes occurring in Diatom Zone II were the initiation of appreciable lakeshore development and the continuation and expansion of introduced fish species. In Diatom Zone III, timber harvest on the western portion of the watershed (largely private holdings) greatly increased, lakeshore development continued, and the introduced fish taxa continued to proliferate.

In summary, the diatom community composition in Tenmile Lake, as reflected in the sediments, has changed significantly. The direction of the change is towards taxa that are usually found in highly productive lakes.

c. Nitrogen

The sediment was analyzed for nitrogen, both to determine ambient concentrations and to examine the ratio of naturally occurring isotopes. The results show that the upper sediments have nearly twice as much nitrogen as the bottom sediments (Figure 50). In addition, the proportion of ^{15}N is increasing in the upper sediments. One of the concerns in evaluating nutrients in sediment is the extent to which they are labile and subject to processes that would alter the concentrations independent of their deposition history. Thus, it is possible that the

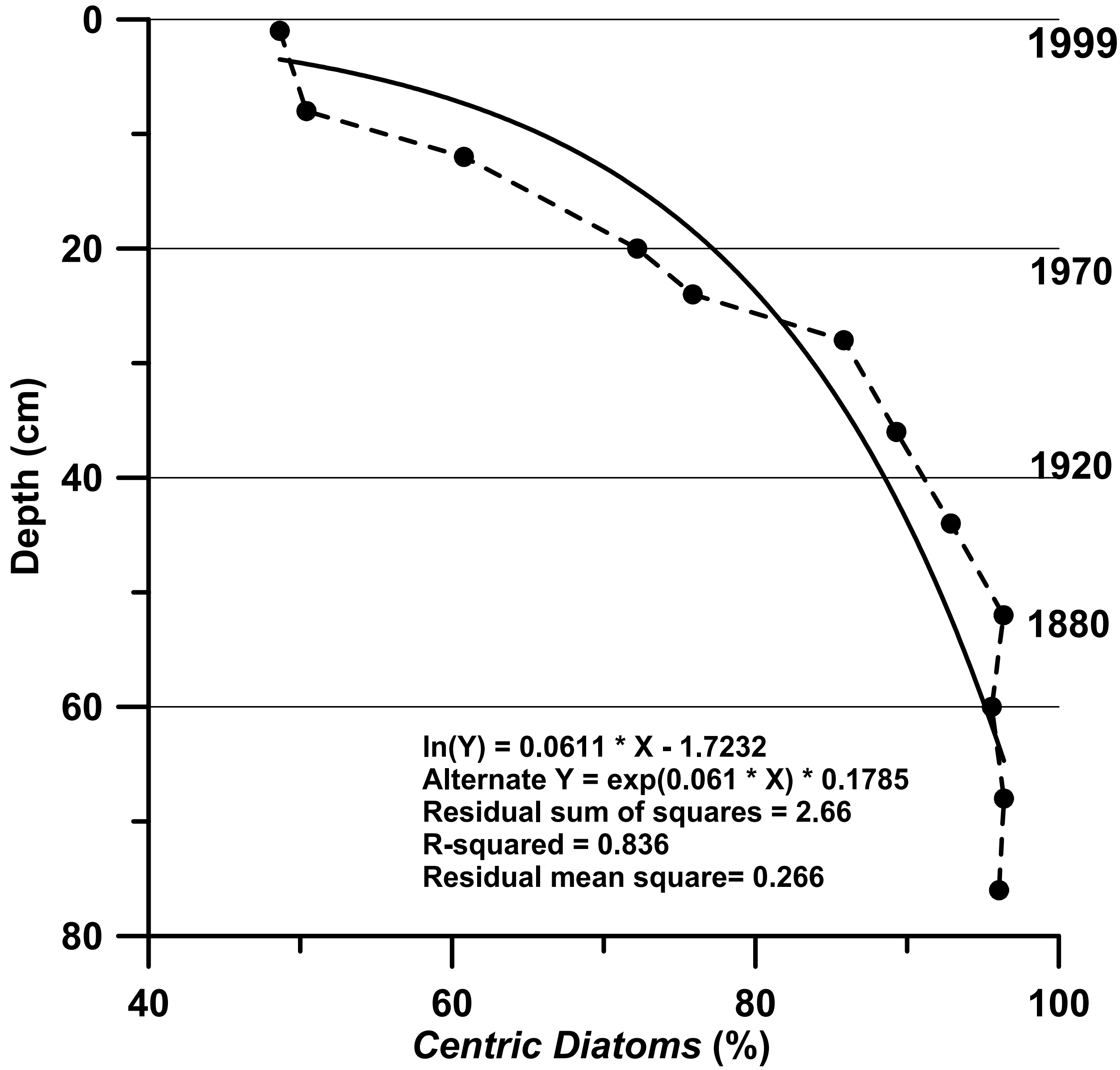


Figure 47. Relative abundance of centric diatoms present in the sediments (core STA).

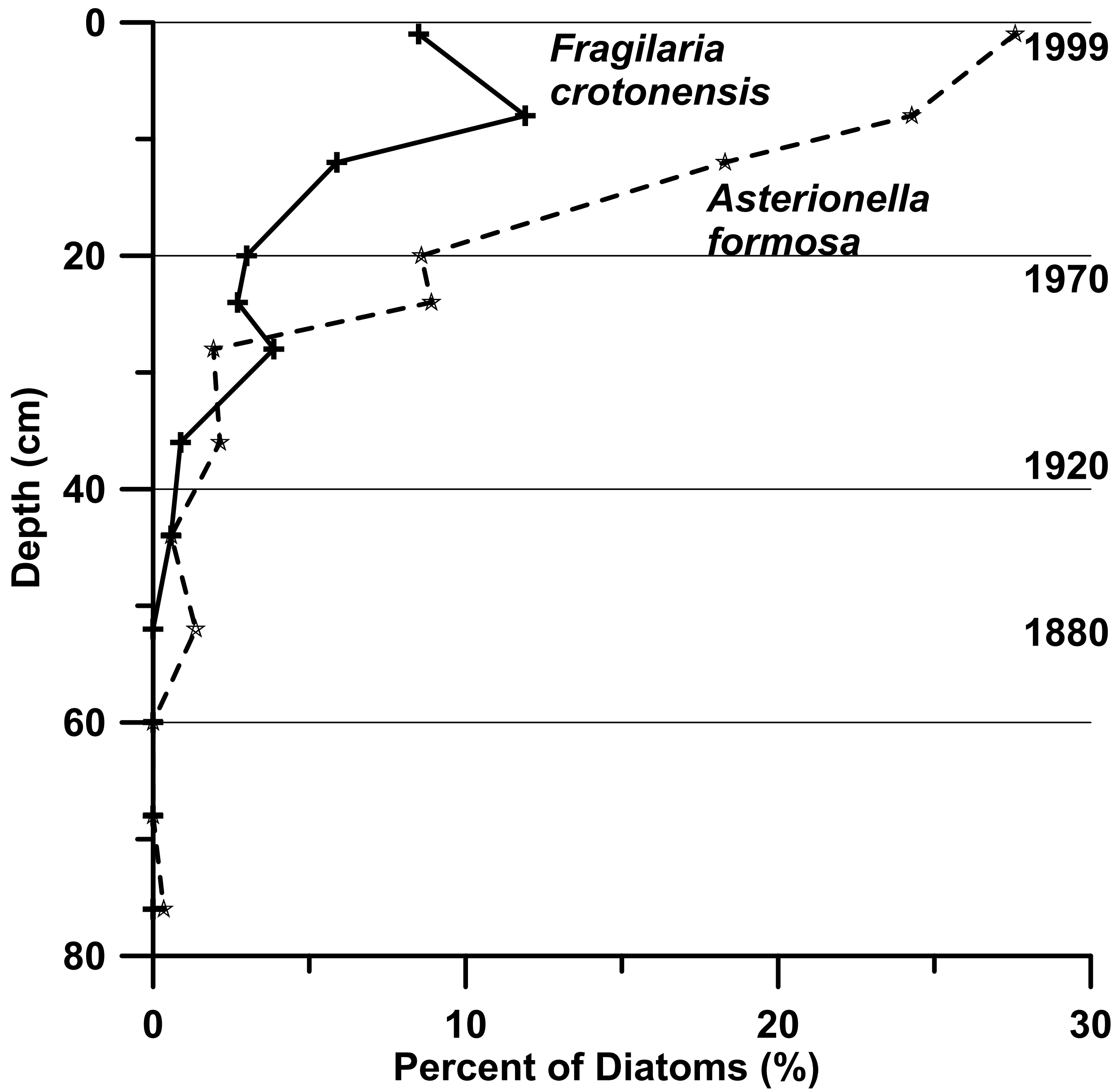


Figure 48. Relative percentage of *Fragilaria crotonensis* and *Asterionella formosa* present in the sediments of Tenmile Lake (core STA).

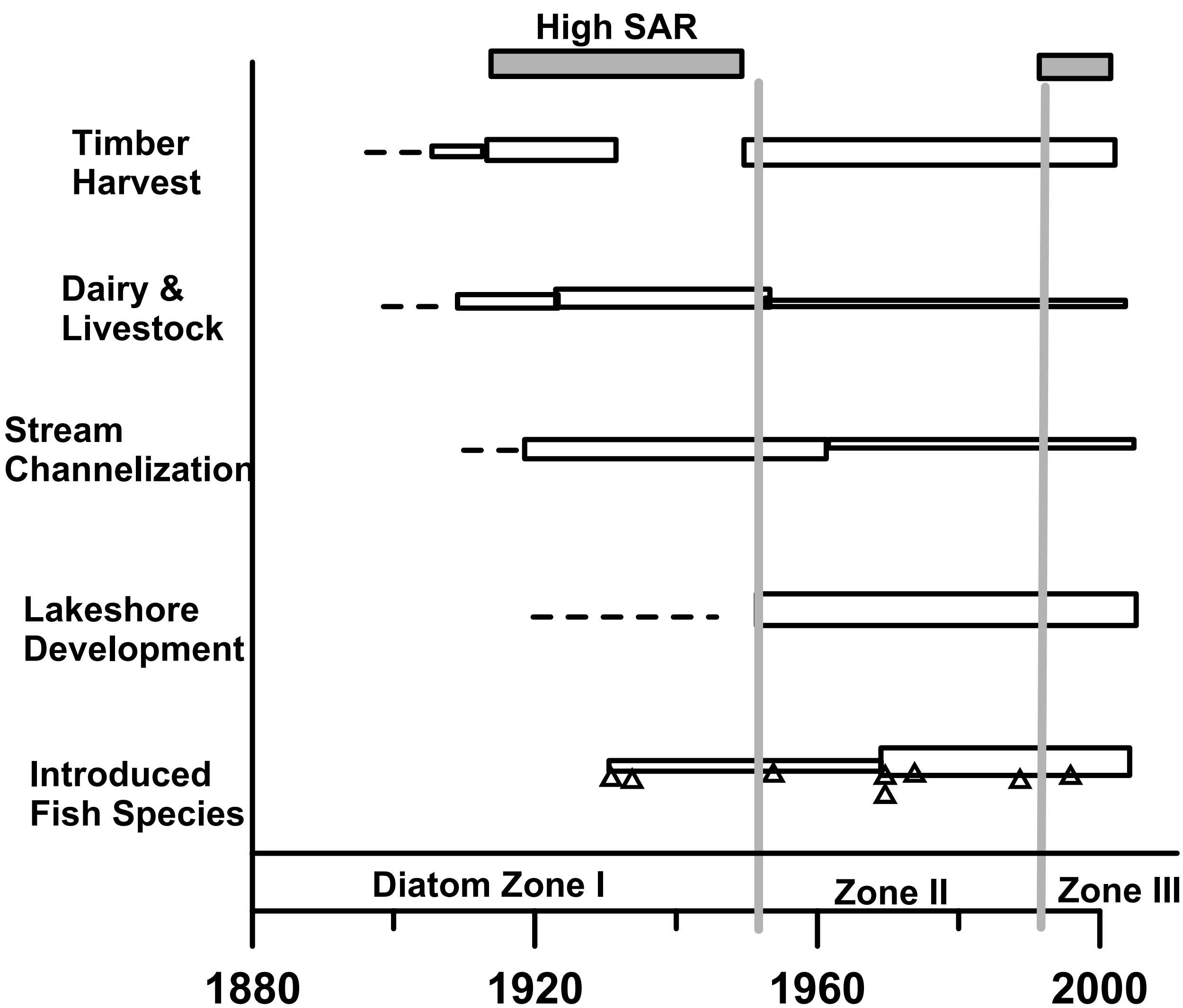


Figure 49. Zonation of the sediment diatoms (from Figure 47) compared to approximate timing of major watershed and in-lake changes. The width of the bars reflects perceived changes in the status of anthropogenic activities.

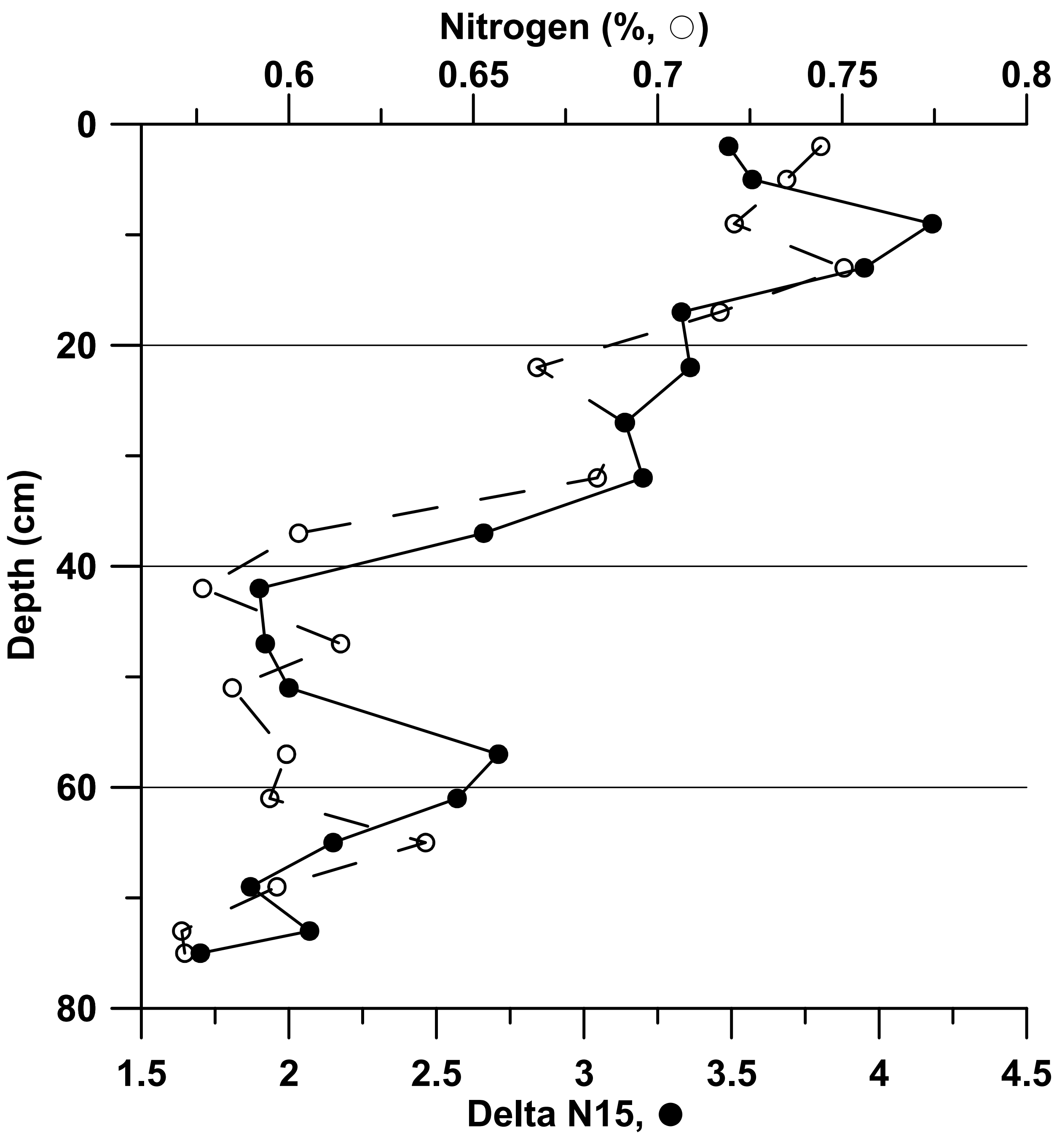


Figure 50. Nitrogen (N, % N by dry weight) and ^{15}N (as delta $^{15}N/^{14}N$) versus depth in sediments, Tenmile Lake.

sediment nitrogen profile is simply an artifact of in-lake processes. However, it is more difficult to attribute the increasing ratio of ^{15}N to diagenesis. The possibility exists that the $^{15}N/^{14}N$ ratio is also changing as benthic organisms differentially accumulated the heavier isotope (cf., Adams and Sterner 2000). However, taken in conjunction with results of the diatoms and the akinetes (discussed below), it is difficult to attribute both sets of sediment nitrogen results to sediment processes. If the nitrogen results do reflect their depositional history, it would indicate an increase in the productivity of the lake and a qualitative change in the organisms associated with that change. Specifically, the increase in ^{15}N would be expected if the population of N-fixing cyanobacteria (blue-green algae) had increased. Furthermore, the increase in ^{15}N would have had to be large to overcome the expected loss of marine-derived ^{15}N associated with the decline in the salmon runs into Tenmile Lake.

d. Cyanobacterial Akinetes

A brief examination was made of akinetes present in the sediments to determine if more detailed analyses would be warranted. The results of the selected sediment samples analyzed show that the rate of cyanobacterial akinetes accumulation is increasing and appears to be accelerating (Figure 51). The greater deposition of akinetes is indicative of an increase in cyanobacteria, particularly those taxa that fix nitrogen. These results are consistent with the results on the diatom stratigraphy which show an increase in lake productivity. The results are also consistent with the nitrogen data which would be expected to show a concomitant change with increasing biological productivity of any kind, but particularly if N-fixing cyanobacteria were involved in the change in productivity. Although a moderate population of cyanobacteria have been present in the lake for at least the last several hundred years, the rate of change in the 20th century is highly consistent with increased nutrient availability in the lake.

E. WATERSHED MODELING

1. Model Calibration

The SWAT model was calibrated using data primarily from the three principal tributary monitoring sites, Big, Benson, and Murphy Creeks. Supplemental information was derived from the unnamed tributary to Benson Creek, particularly with respect to nitrate loading from recent clearcuts. As noted earlier, the model was calibrated first for hydrology before proceeding with calibration of sediment, phosphorus, and nitrogen. The simulated versus measured discharge at

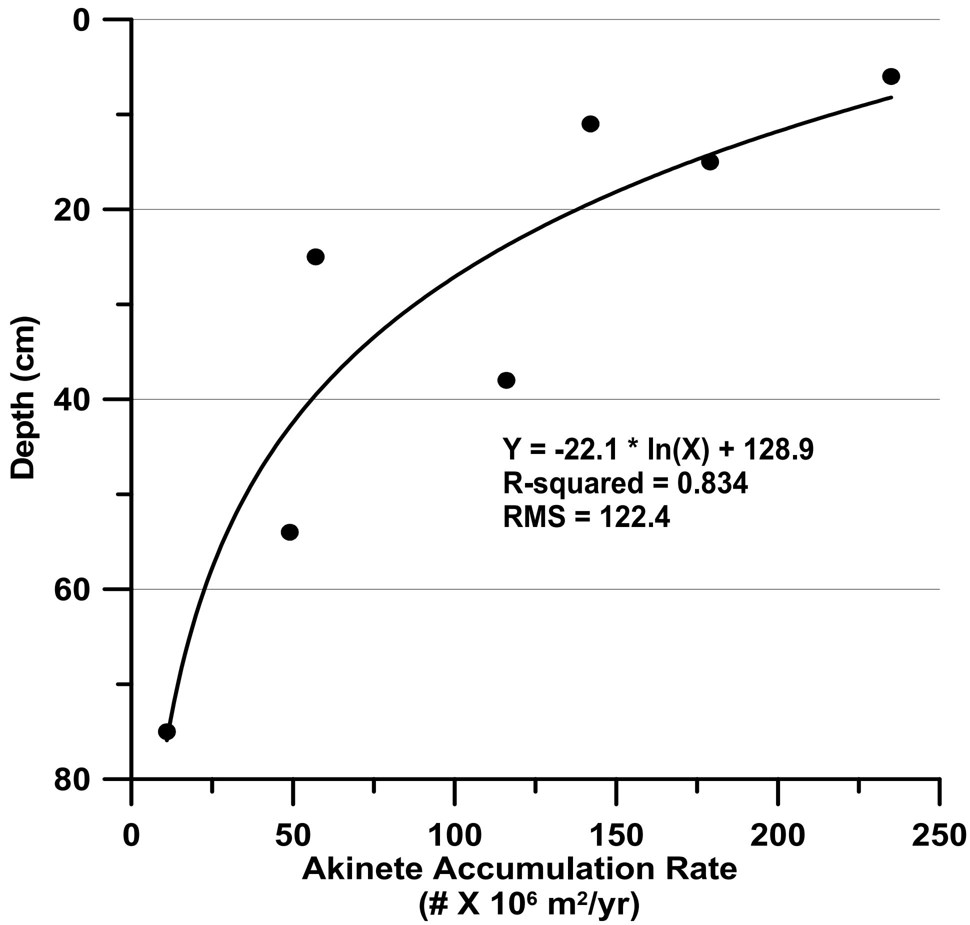


Figure 51. Accumulation rate of cyanobacterial akinetes in Tenmile Lake (core STA).

the three primary stations is shown in Figure 52. The model was able to simulate discharge at Big and Benson Creeks reasonably well. The hydrologic simulation for Murphy Creek was not as precise as the other two sites. We attribute this to the large proportion of unmeasured overbank discharge which occurred at the Murphy Creek site during high flow events. Traditional gaging techniques (stage-discharge rating curves) are problematic in these wetland-stream systems. Nevertheless, the difficulty in the hydrologic calibration for Murphy Creek, as we shall see in the following section, is minimized because of the low concentrations of the parameters of interest. Water quality simulations proceeded following the hydrology calibration. Results from calibration of TSS, TP, and NO_3-N for Murphy, Big, and Benson Creeks are presented in Figures 53-55. Results are presented as mean monthly loads for the period January 1999 to April 1999.

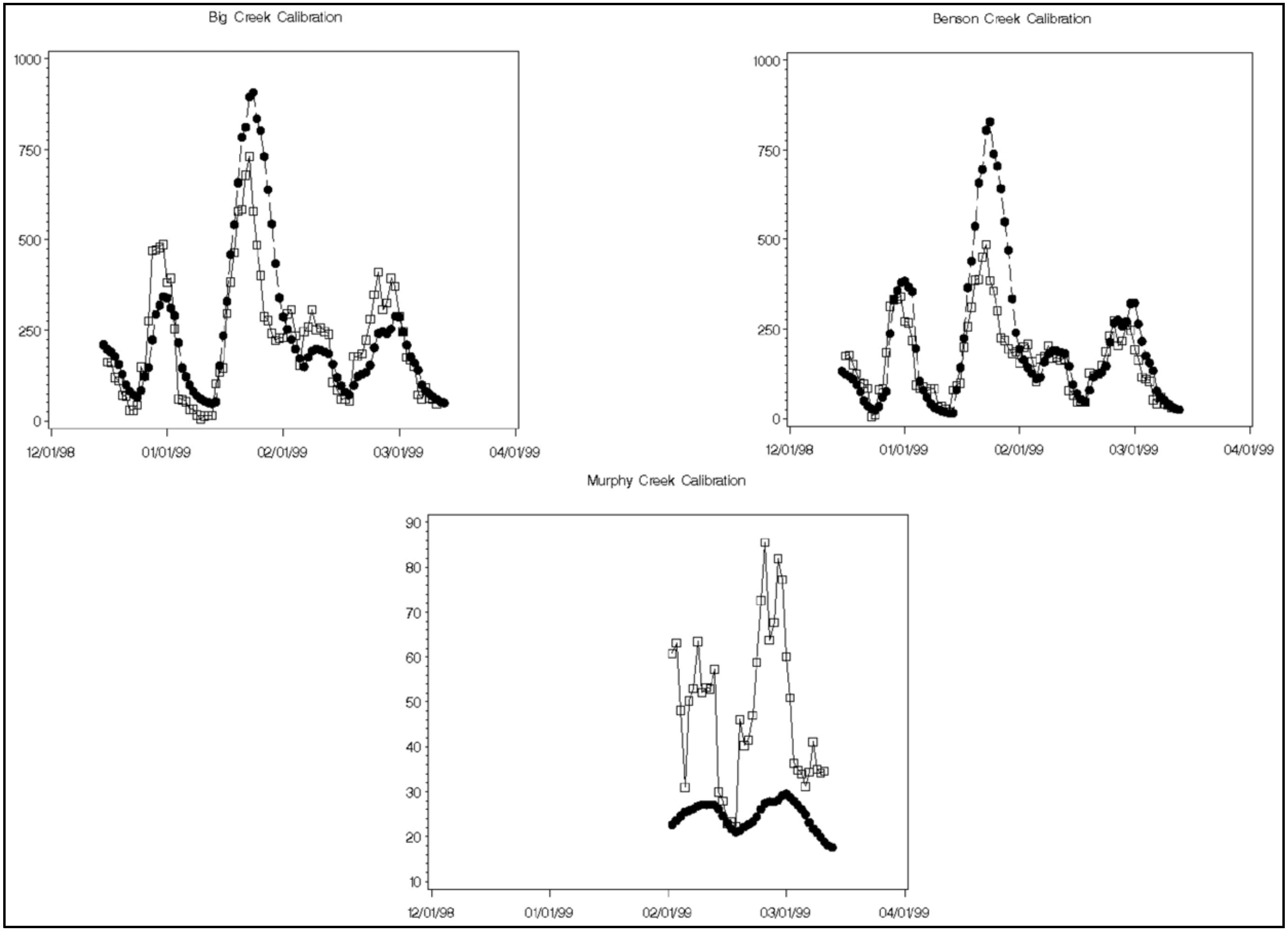


Figure 52. Simulated (□—□—□) versus measured (●—●—●) stream discharge (cfs) at Big, Benson, and Murphy Creeks.

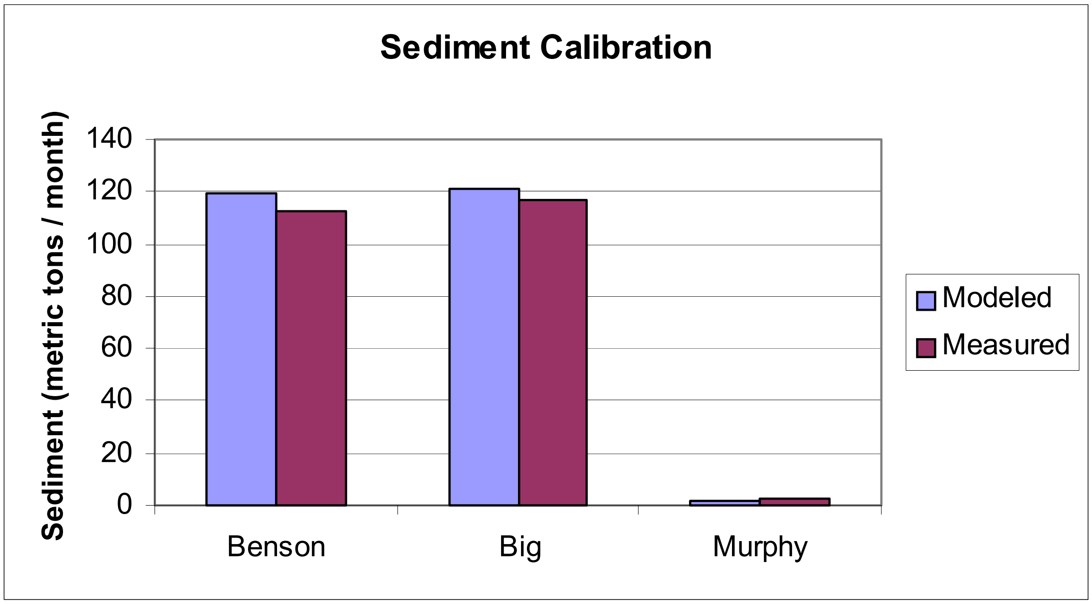


Figure 53. Simulated versus measured concentrations of TSS (mg/L) for Big, Benson, and Murphy Creeks.

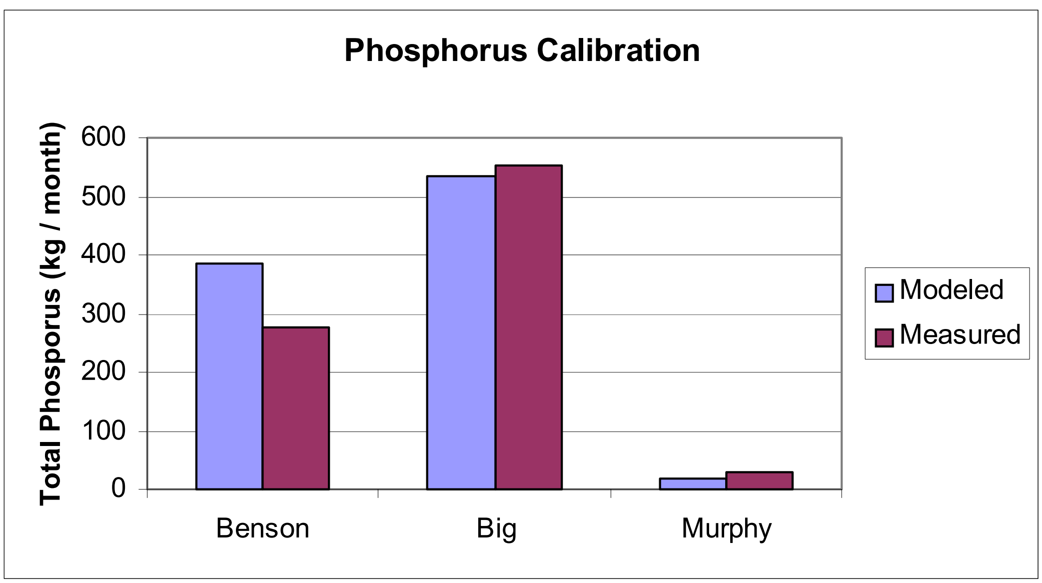


Figure 54. Simulated versus measured concentrations of TP (mg/L) for Big, Benson, and Murphy Creeks.

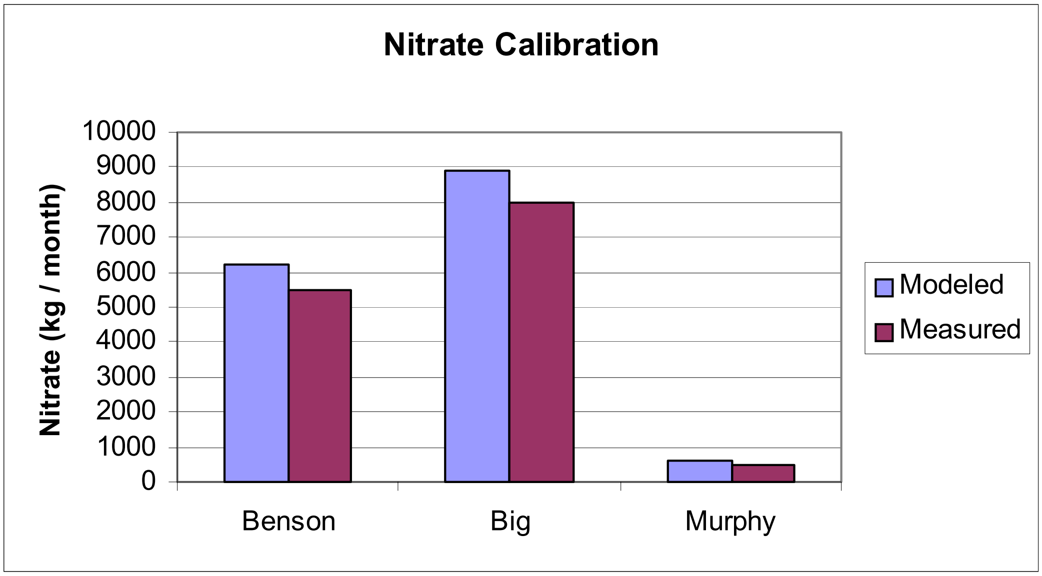


Figure 55. Simulated versus measured concentrations of nitrate (mg/L) for Big, Benson, and Murphy Creeks.

Once the SWAT model was calibrated to individual stormflow, the results were extrapolated to actual daily flows over a five-year period (1990-1995). This multiple-year period was used for all three model scenarios: current land use (1994), pre-development ("historical"), and current + wetlands ("wetlands"). The historical simulations were based on export coefficients used in the current scenario for mature timber in the uplands and wetlands (analogous to Murphy Creek) in the lowlands. The wetlands scenario was based on current land use, except tributary low lands currently used as grazed pasture were converted to wetlands from a distance of 1000 m from the lakeshore. The current analogue for this configuration is Adams Creek where the upper land use is grazed and the lowlands closest to the lake are maintained in wetlands.

The pre-development period represented by the model would probably be that present at circa 1850. This was prior to a major fire that destroyed the timber in a significant portion of the eastern watershed in the 1860s. It was also prior to any stream channelization and wetland drainage that occurred after 1890. A general description of Township 23S Range 12W by the surveyor in 1890 provides his impressions of the area in the Noble Creek drainage.

> *"Land very rough and mountainous. Soil principally clay bottom through which the different streams meander although not generally wide very rich; and covered with dense growth of timber principally Ash, Maple and Ader. The mountains between Streams very steep so much so could only measure with a one pole chain. Timber on almost all of them burnt and its place being taken with Thimbleberry and young Alders. This Twp. contains numerous lakes and considerable land that is overflowed with water which all can be drained at the present time."*
>
> 1890, Simon Cathcart, Surveyor

His assessment that the lowlands could be drained was accurate, judging from the current extent of channelized streams in the watershed.

The wetland scenario is based on the vegetation-hydrology of the present Murphy Creek drainage. This area was formerly channelized, but for the last 20 years has had the cattle removed. The wetland "restoration" that has evolved at this site is probably unlike the historical wetlands which appeared to consist of lowland hardwoods (c.f., description by S. Cathcart). The current wetland is largely a monoculture stand of Reed Canary grass which actually may be even more effective than the native wetlands in filtering runoff. Murphy Creek is in the process of re-establishing a more natural meander, although some reaches remain unnaturally regular. However, without access to an undisturbed native wetland in the watershed we elected to assume that the Murphy Creek wetland is a reasonable approximation of historical conditions.

2. Model Results

Using the calibrations, we generated the following model estimates of water, sediment, nitrogen, and phosphorus yields for each of the 20 catchments (Figure 56) within the Tenmile Lake watershed (Tables 7-9). Model estimates for volume-weighted concentrations and annual loads are presented on a unit-area basis in Figures 57-59. The results illustrate a number of factors with respect to differential loading of nonpoint source pollutants to the lake. First, the model simulations indicate that water yield (the amount of total discharge from a catchment) varies by nearly a factor of two among catchments. Catchments such as Murphy Creek exhibit a low yield of water runoff. Other sites such as West Shutters and Devore also had similarly low runoff values. These three sites all have wetlands in the lower part of their respective watersheds which either causes an underestimation of the actual amount of measured runoff or the presence of the wetlands at the base of the watersheds actually alters the amount of runoff. Regardless of

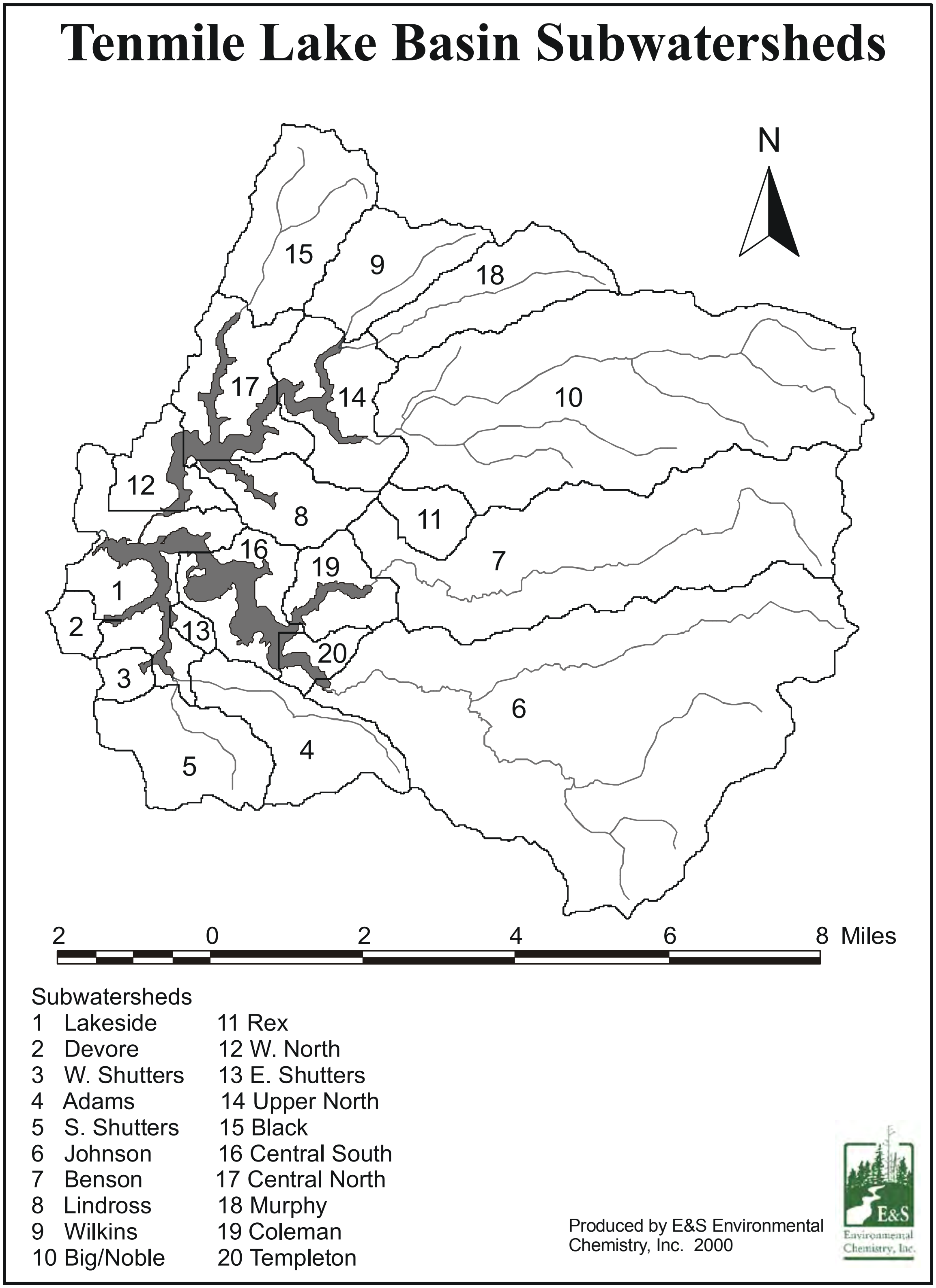


Figure 56. Partitioning of catchments in the Tenmile Lake watershed for purposes of SWAT model application.

Table 7. SWAT model estimates for the twenty catchments in the Tenmile Lake watershed under the *current* landuse.

			Annual Loads			Average Concentrations		
Name	Catch-ment #	Area (ha)	Sediment (T/ha)	Nitrate (Kg/ha)	Phosphorus (Kg/ha)	Sediment (mg/L)	Nitrate (mg/L)	Phosphorus (mg/L)
Lakeside	1	792	2.35	1.30	16.80	45.54	0.37	0.25
Devore	2	116	0.15	0.08	0.83	14.52	0.08	0.36
W. Shutters	3	109	0.08	0.10	0.05	8.05	0.08	0.02
Adams	4	693	0.09	1.32	2.89	9.04	0.56	0.12
S. Shutters	5	644	1.06	1.29	6.29	99.64	0.65	0.26
Johnson	6	4473	0.28	0.84	2.34	26.00	0.39	0.09
Benson	7	2411	0.26	0.69	1.82	28.46	0.37	0.10
Lindros	8	426	0.05	0.13	1.19	5.61	0.15	0.06
Wilkins	9	478	0.18	0.07	0.50	17.59	0.06	0.02
Big	10	3412	0.36	0.81	2.30	32.95	0.36	0.09
Rex	11	218	0.85	1.92	5.60	79.03	1.02	0.23
W. North	12	357	3.38	1.34	13.78	38.08	0.30	0.20
E. Shutters	13	105	0.44	0.22	1.80	39.25	0.03	0.07
Upper North	14	751	0.12	0.90	4.35	25.65	0.34	0.12
Black	15	819	0.26	0.06	0.66	25.30	0.06	0.03
Central South	16	553	1.35	0.74	8.59	33.67	0.39	0.19
Central North	17	678	0.07	0.39	3.29	23.46	0.30	0.15
Murphy	18	713	0.07	0.17	0.45	6.13	0.07	0.02
Coleman	19	343	0.13	0.62	4.19	27.17	0.40	0.14
Templeton	20	231	0.08	0.77	2.50	25.35	0.40	0.13

Table 8. SWAT model estimates for the twenty catchments in the Tenmile Lake watershed under the *pre-development* land cover.

			Annual Loads			Average Concentrations		
Name	Catch-ment #	Area (ha)	Sediment (T/ha)	Nitrate (Kg/ha)	Phosphorus (Kg/ha)	Sediment (mg/L)	Nitrate (mg/L)	Phosphorus (mg/L)
Lakeside	1	792	0.23	2.57	9.65	2.91	0.22	0.06
Devore	2	116	0.01	0.16	0.10	1.07	0.05	0.00
W. Shutters	3	109	0.03	0.15	0.23	2.75	0.05	0.01
Adams	4	693	0.01	0.20	0.10	1.01	0.07	0.00
S. Shutters	5	644	0.01	0.18	0.10	0.86	0.06	0.00
Johnson	6	4473	0.02	0.66	0.22	1.33	0.24	0.01
Benson	7	2411	0.01	0.41	0.13	0.99	0.14	0.01
Lindros	8	426	0.00	0.15	0.09	0.30	0.05	0.00
Wilkins	9	478	0.03	0.19	0.23	2.85	0.06	0.01
Big	10	3412	0.01	0.55	0.16	0.76	0.20	0.01
Rex	11	218	0.01	0.16	0.07	0.87	0.05	0.00
W. North	12	357	0.22	2.33	4.65	2.12	0.23	0.05
E. Shutters	13	105	0.01	0.03	0.07	1.03	0.00	0.00
Upper North	14	751	0.02	2.03	1.13	1.06	0.24	0.01
Black	15	819	0.03	0.23	0.25	2.75	0.08	0.01
Central South	16	553	0.18	1.45	4.28	2.24	0.24	0.03
Central North	17	678	0.01	1.36	0.69	1.23	0.23	0.03
Murphy	18	713	0.01	0.23	0.06	0.42	0.08	0.00
Coleman	19	343	0.02	1.90	1.08	1.12	0.20	0.02
Templeton	20	231	0.02	1.81	0.75	1.34	0.26	0.01

Table 9. SWAT model estimates for the twenty catchments in the Tenmile Lake watershed under the current landuse with wetlands restored from the lakeshore in areas currently being grazed.								
			Annual Loads			Average Concentrations		
Name	Catch-ment #	Area (ha)	Sediment (T/ha)	Nitrate (Kg/ha)	Phosphorus (Kg/ha)	Sediment (mg/L)	Nitrate (mg/L)	Phosphorus (mg/L)
Lakeside	1	792	2.35	1.30	16.80	29.45	0.23	0.19
Devore	2	116	0.15	0.08	0.83	7.80	0.04	0.02
W. Shutters	3	109	0.08	0.10	0.05	4.26	0.04	0.11
Adams	4	693	0.09	1.32	2.89	9.04	0.56	0.12
S. Shutters	5	644	1.06	1.29	6.29	10.74	0.06	0.01
Johnson	6	4473	0.28	0.84	2.34	6.89	0.22	0.05
Benson	7	2411	0.26	0.69	1.82	12.12	0.19	0.05
Lindros	8	426	0.05	0.13	1.19	2.60	0.05	0.02
Wilkins	9	478	0.18	0.07	0.50	11.08	0.06	0.02
Big	10	3412	0.36	0.81	2.30	8.72	0.18	0.04
Rex	11	218	0.85	1.92	5.60	79.03	1.02	0.23
W. North	12	357	3.38	1.34	13.78	25.15	0.20	0.15
E. Shutters	13	105	0.44	0.22	1.80	39.25	0.03	0.06
Upper North	14	751	0.12	0.90	4.35	9.12	0.22	0.07
Black	15	819	0.26	0.06	0.66	18.42	0.06	0.02
Central South	16	553	1.35	0.74	8.59	17.44	0.23	0.12
Central North	17	678	0.07	0.39	3.29	9.90	0.21	0.10
Murphy	18	713	0.07	0.17	0.45	6.23	0.07	0.02
Coleman	19	343	0.13	0.62	4.19	12.37	0.24	0.09
Templeton	20	231	0.08	0.77	2.50	6.99	0.23	0.07

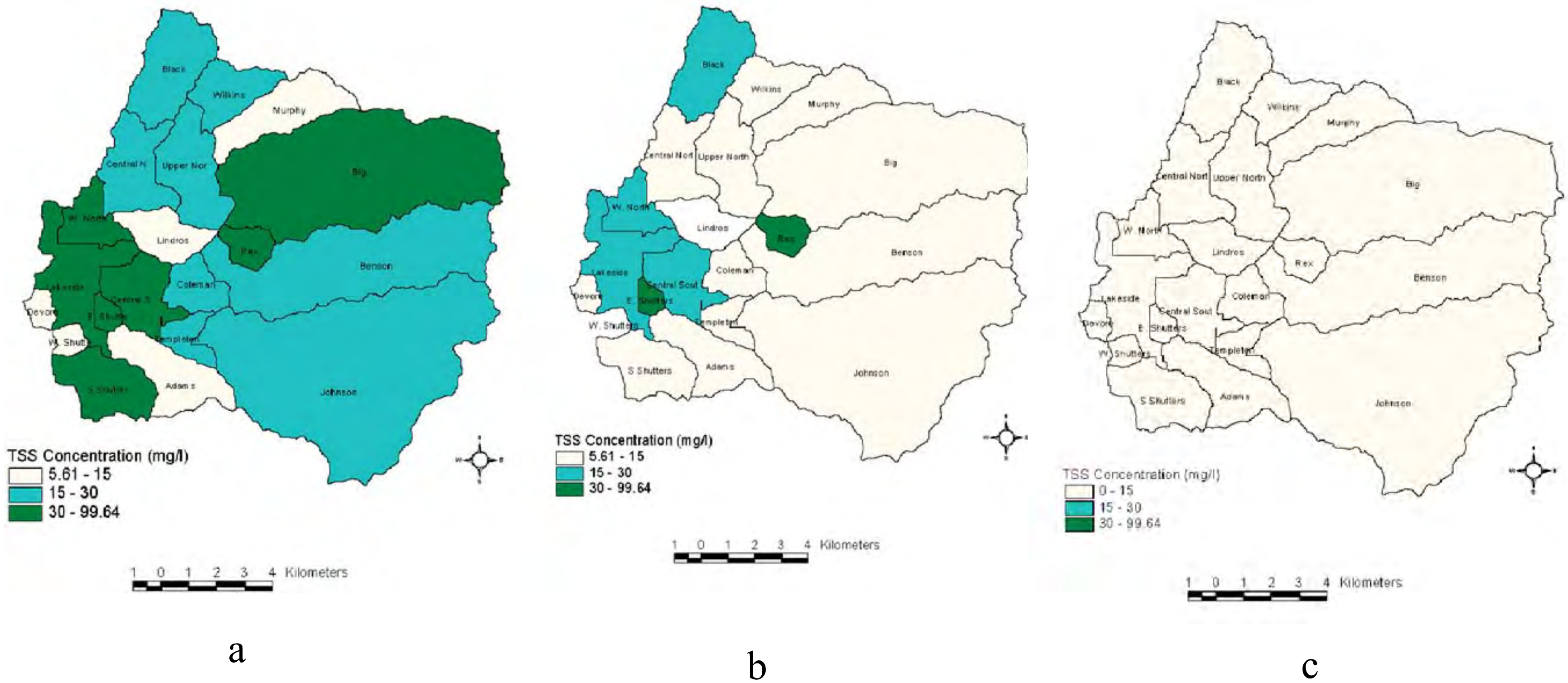


Figure 57. SWAT model simulations for concentration of TSS for (a) current wetland land use, (b) current land use with wetlands restored, and (c) pre-development.

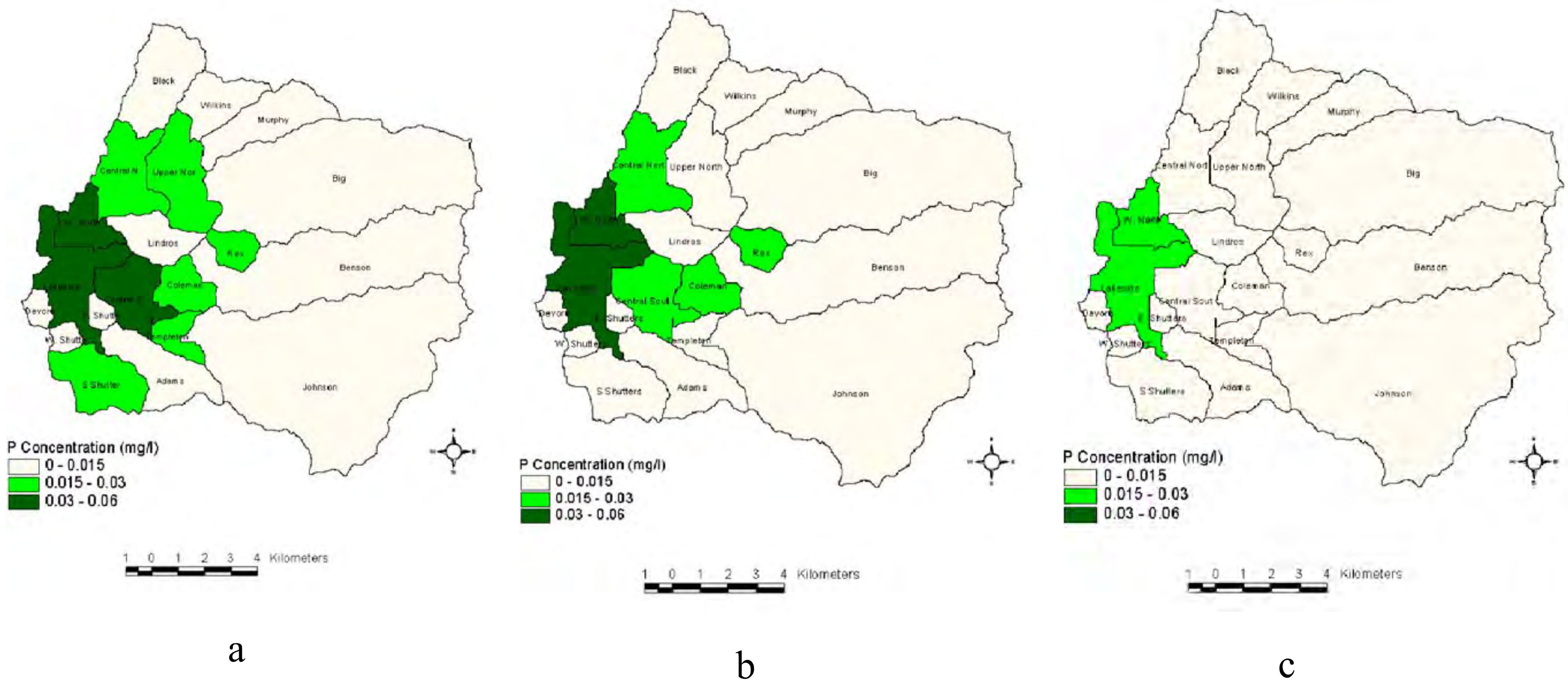


Figure 58. SWAT model simulations for concentration of phosphorus for (a) current wetland land use, (b) current land use with wetlands restored, and (c) pre-development.

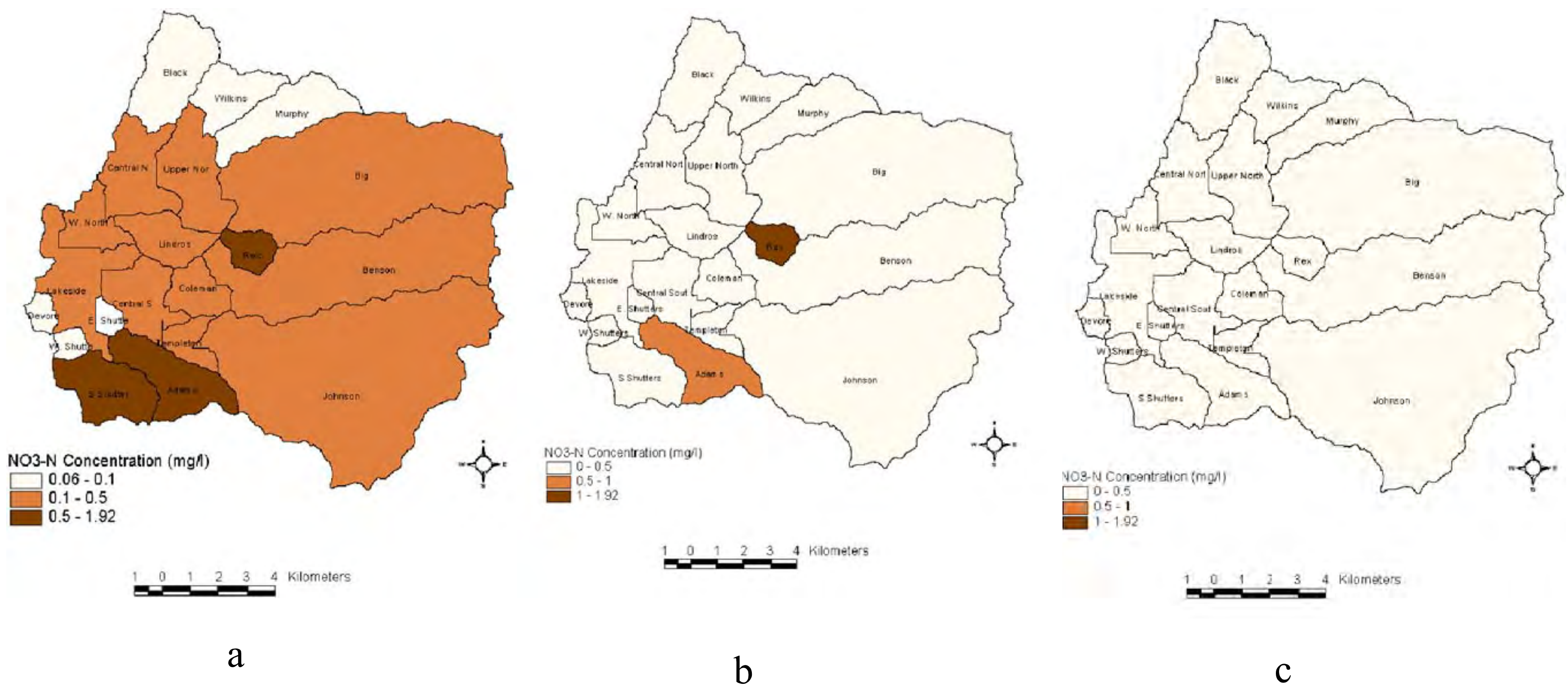


Figure 59. SWAT model simulations for concentration of nitrate for (a) current wetland land use, (b) current land use with wetlands restored, and (c) pre-development.

whether the amount of watershed discharge is altered or whether the flow becomes rerouted to the extent that it can't be easily measured, the effects appear to be similar. Watersheds with high water yields include Johnson Creek, South Shutters, and Adams Creek. Some of the elevated runoff for Johnson Creek may reflect the orographic influence from high precipitation in the upper (eastern) portion of the watershed. However, Johnson Creek also exhibits a high rate of timber harvest and grazed wetlands that contribute to the modeled response. South Shutters catchment is notable for the high percentage of grazed uplands which, combined with grazed wetlands, promote a high degree of runoff.

Simulated sediment yield varied by two orders of magnitude ranging from less than 0.1 T/ha/yr in catchments such as Templeton, Murphy, and West Shutters to values over 1 T/ha/yr in catchments such as Black Creek, Lindross Arm, and South Shutters. Murphy Creek and West Shutters both have wetlands at the base of the catchments which serve to filter upland sediment production. The low simulated sediment load in Templeton is, in part, an artifact of having to include part of the lake surface in partitioning the catchment (22 percent is listed as water surface). On the high end of sediment production are Black Creek, Lindross Arm, and South Shutters. All three of these catchments have a comparatively high percentage of clearcut land, but other factors are involved as well. The catchment with the greatest percentage of clearcut is Devore Arm (48 percent), yet the model simulation shows that sediment production is only a small fraction of that found in the high-sediment production catchments. The primary difference is the location of the land use disturbance with respect to the receiving water. The clearcutting in the Devore catchment is mostly upland with a buffer of established forest, whereas the timber harvest in the other catchments occurred on steep slopes closer to the lake. Additionally, the high sediment production catchments all have grazed wetlands, whereas the Devore catchment contains intact wetlands in the lowlands. One factor influencing the modeled sediment production in South Shutters is the high percentage of grazed uplands and wetlands.

Nitrogen export ranged by nearly two orders of magnitude from 0.07 kg N/ha/yr in Wilkins Creek to 1.29 kg N/ha/yr in South Shutters and 1.92 kg N/ha/yr in the clearcut tributary to Benson Creek (Table 7). The catchments with a large component of urban land use, West North Lake and Lakeside, also had a high yield of nitrogen (1.34 and 1.30 kg N/ha/yr, respectively). The general ranking of phosphorus loads from the catchments was similar to that of nitrogen, whereby high loads were associated with urban and grazing activities and low loads were associated with catchments with a high percentage of mature forest and intact wetlands at the base of the watershed.

Clearly, the greatest confidence in model results are associated with catchments for which detailed water quality data were collected. The model outputs for those catchments including the simulated results for the entire watershed are summarized in Figures 60-62. The results indicate that the larger catchments such as Big and Benson Creeks drive the results for the watershed as a whole. Higher exports of TSS, NO_3, and PO_4 from smaller catchments such as the Benson Tributary (referred to as "Rex" in Tables 7-9) are balanced by low exports from wetland-dominated sites such as Murphy Creek. The current loads of modeled parameters for the Tenmile Lake watershed exceed historical (pre-development) loads of TSS by at least an order of magnitude. The differential between historical and current loads of PO_4 are estimated to differ by a factor of three, whereas NO_3 appears to differ by a factor of two.

The SWAT model was used to simulate watershed response to a change in current land use. The simulation was based on the assumption that all lowland tributary sites were allowed to be re-established as mature wetlands (comparable to Murphy Creek) for the 1000 m extending from the lakeshore upstream.

The SWAT model represents loads as material moving from a specific hydrologic resource unit (HRU), hence under the Wetland scenario, the watershed loads are identical to the Current scenario loads. The Wetland scenario acts by filtering/denitrifying water passing through the wetland. The "wetland" model scenario attempts to simulate this process by superimposing a mathematical filter over the loads of TSS, NO_3, and PO_4 entering the wetland. Thus, the change between these scenarios must be made by comparing stream concentrations.

The results suggest that the wetland scenario would reduce concentrations of TSS in the average catchment to the lake from 30.5 mg/L to 16.3 mg/L and the corresponding reductions in concentration of NO_3 and PO_4 would be 0.32 mg/L to 0.21 mg/L and 19 µg/L to 10 µg/L, respectively. These changes represent a 47% decrease in TSS and PO_4 and a 34% decrease in NO_3. The reduction in watershed exports would be expected to further decrease by increasing the extent of the lowlands in restored wetlands. Note that restoring all lowlands to their original wetland condition will not return the watershed export rates to their historical levels because of expected continued activity in the uplands and the continued direct inputs to the lake from shoreline development.

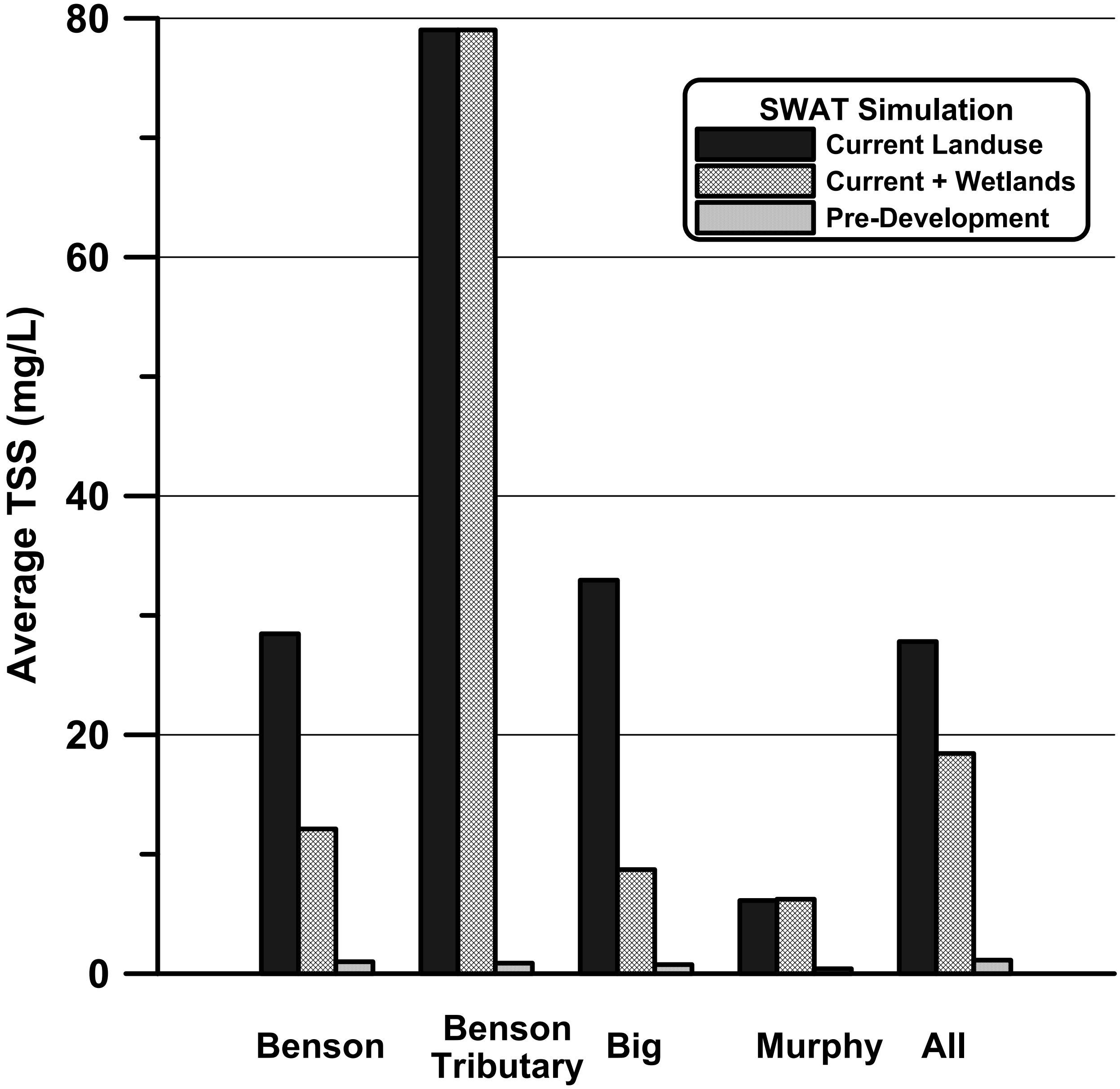


Figure 60. SWAT model simulations of sediment export (as flow-weighted concentrations) for monitored stream sites.

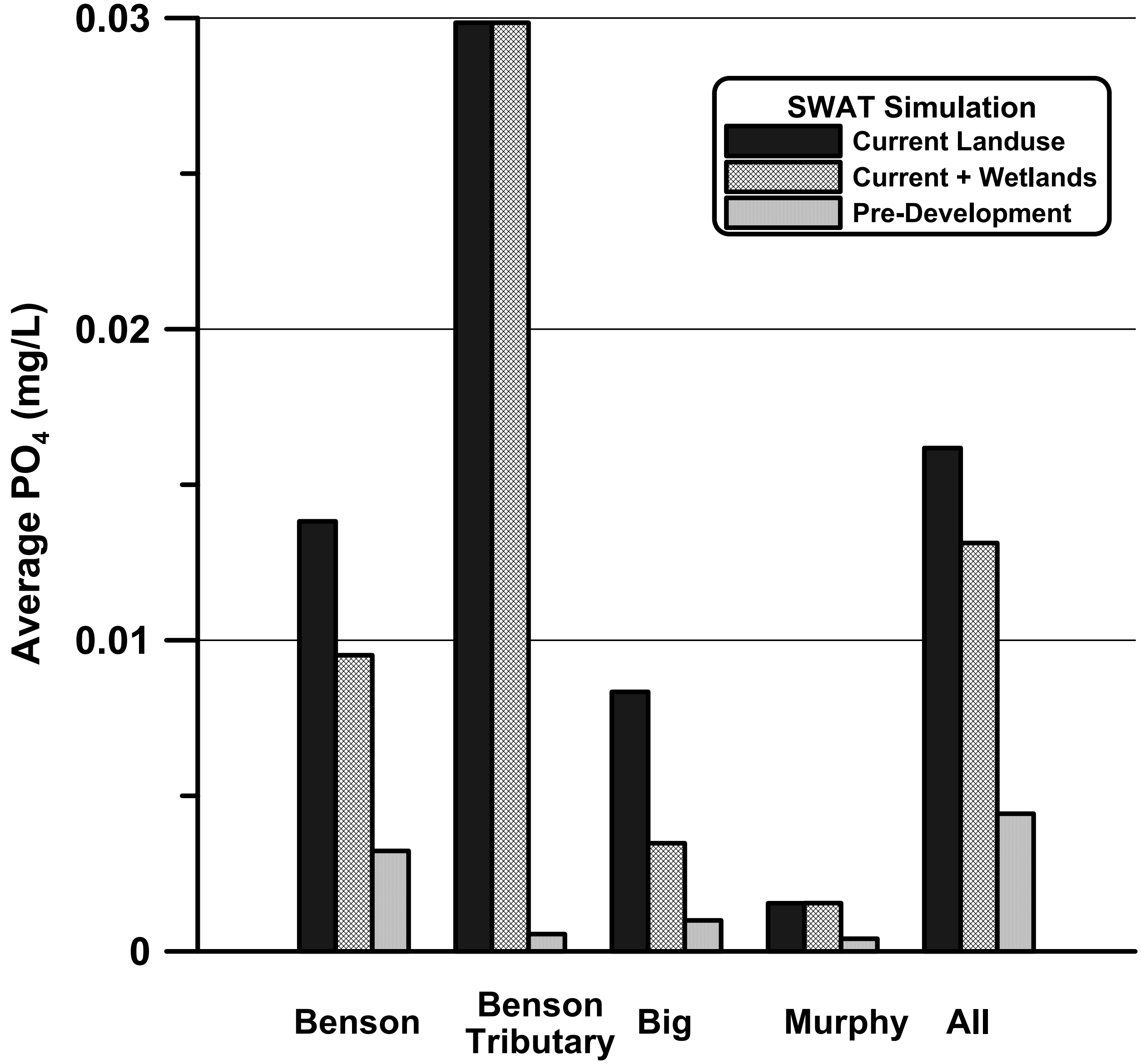


Figure 61. SWAT model simulations of phosphorus export (as flow-weighted concentrations) for monitored stream sites.

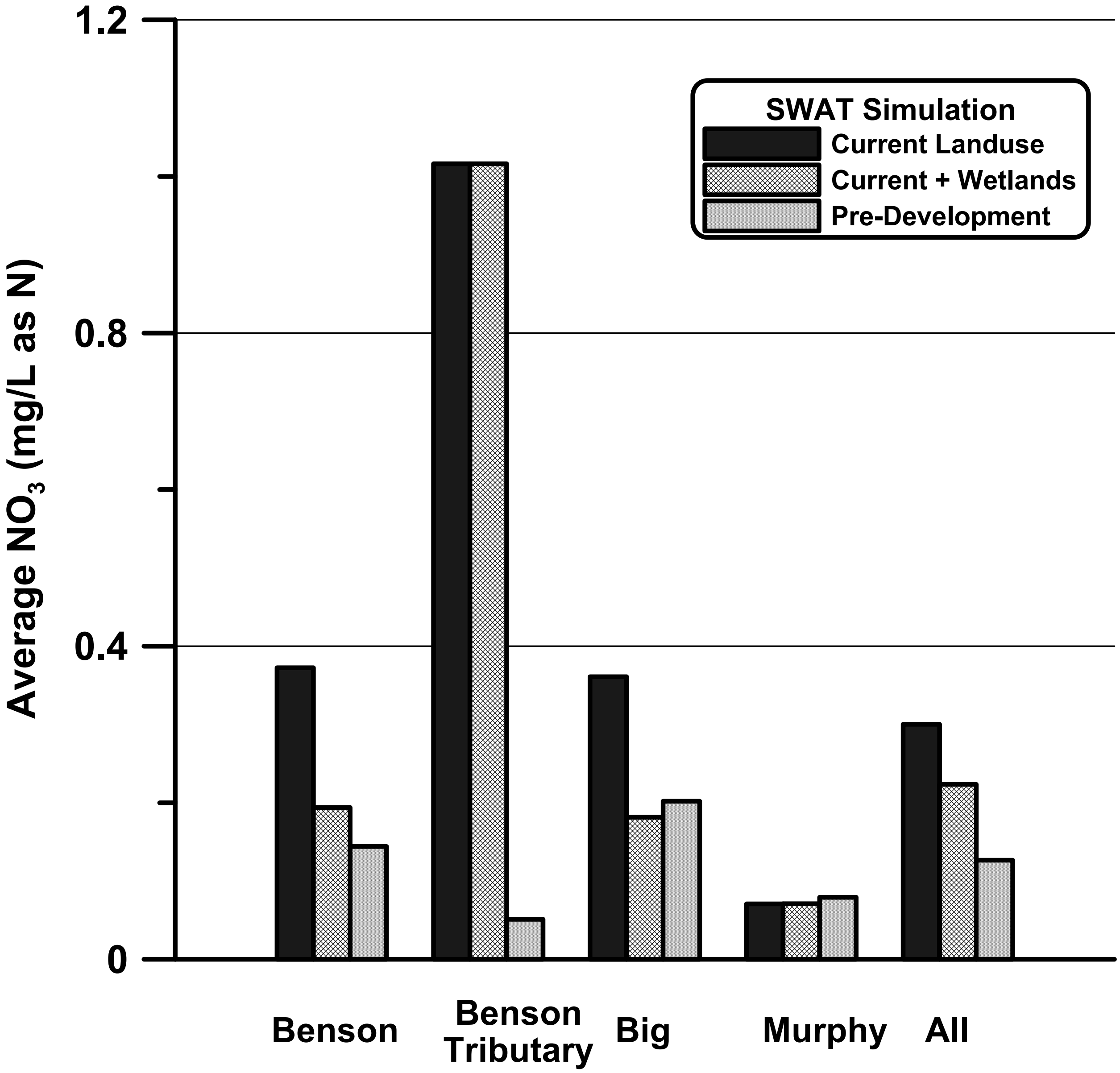


Figure 62. SWAT model simulations of nitrate export (as flow-weighted concentrations) for monitored stream sites.

The model results illustrate the importance of the wetlands for those areas upstream of these sites. The wetland scenario has no impact on the export of material from the upland sites such as the Benson Creek tributary, but the value of the wetland is that it still functions sufficiently well to export high water quality despite timber harvest in the uplands (as represented by the Benson tributary catchment).

An examination of aerial photographs of the mouths of Murphy and Big Creeks taken in 1967 and 1994 illustrates some dramatic differences in land use changes (Figures 63 and 64). In 1967 Murphy Creek lowlands were being used for cattle grazing, much as other lowland areas in the watershed are currently being used. At that time, the uplands was fully forested in Douglas fir. By 1994, both the lowlands and uplands had changed. The cattle were removed from the lowlands circa 1980 (Sally Thomas, pers. comm. 2002) and have remained off the land through the present. However, the northern uplands portion of the catchment was clearcut in 1994 (or possibly in 1993). However, despite this intensive timber harvest there was little change in the shape or extent of wetlands at the mouth of Murphy Creek.

At Big Creek we observe a dramatic change in land use. In 1967, the lakeshore in the embayment was largely undeveloped, with only two docks visible in the aerial photograph. By 1994, approximately 18 private docks are visible and a commercial marina is present. Although some recent timber harvest was evident on the south side of the embayment in 1967, a much more extensive clearcut was present on the north side in 1994. Whereas, the mouth of Murphy Creek showed little change in the intervening 27 years, an extensive delta had formed at the mouth of Big Creek. Furthermore, the stream channel had changed locations to the north side of the embayment either because of direct human intervention or as a consequence of the elevated sediment load. The docks on the east end of the embayment (from Sun Lake Marina eastward) are now experiencing rapid shoaling and difficulty in accessing the lake.

The model simulates export of PO_4 rather than TP. The simulated change in PO_4 export is relatively modest for the wetland scenario. However, we would expect a greater reduction in TP because of the relationship between TSS and TP (Figure 65). The ratio between PO_4 and TP is about 3:1 in this watershed (cf., Figure 12) which provides an indication of the expected difference in simulated TP versus PO_4 in Tenmile Lake watersheds.

The spatial distribution of the catchment model scenarios illustrates some noteworthy distinctions in pollutant sources. The SWAT model output for volume-weighted concentrations of TSS indicates that Big Creek, the tributary to Benson and catchments immediately adjacent to

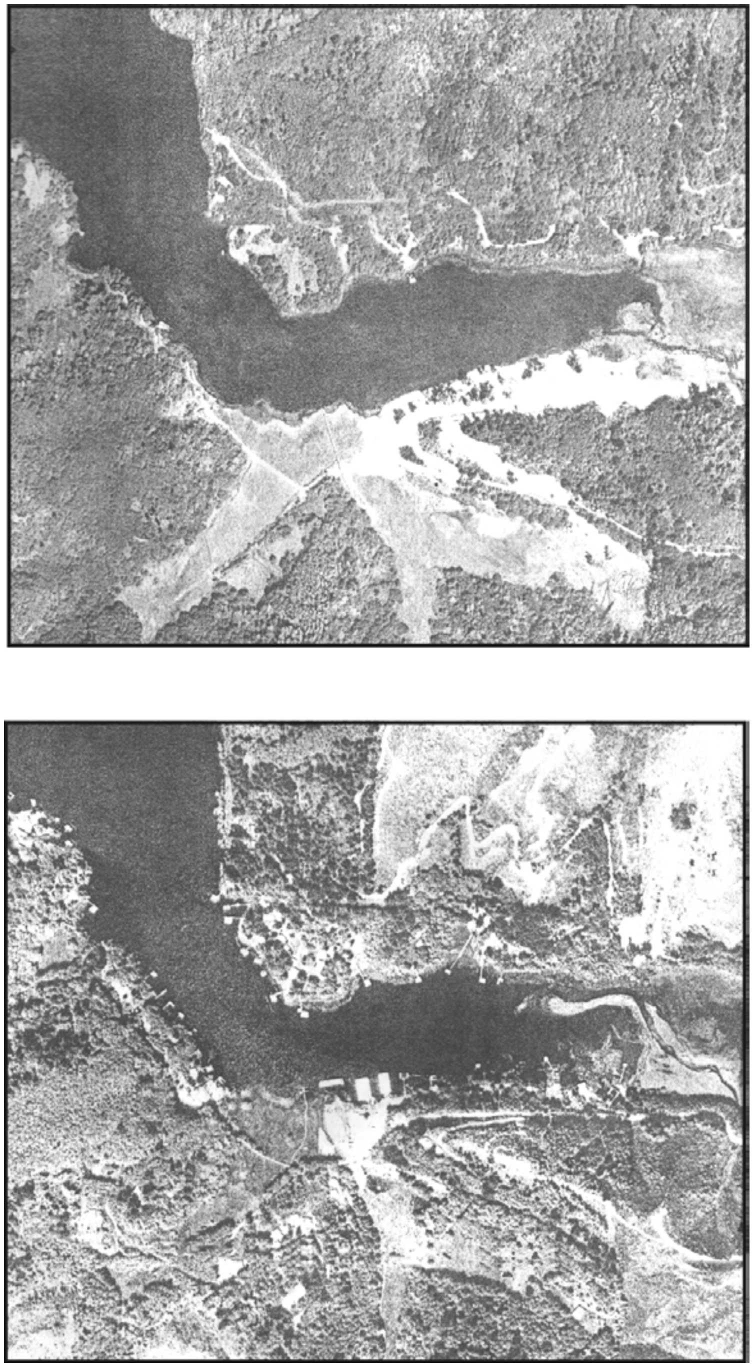


Figure 63. Aerial photographs of Big Arm (North Tenmile Lake) in 1967 (above) and 1994 (below) illustrating the encroachment of the Big Arm Creek delta into the embayment.

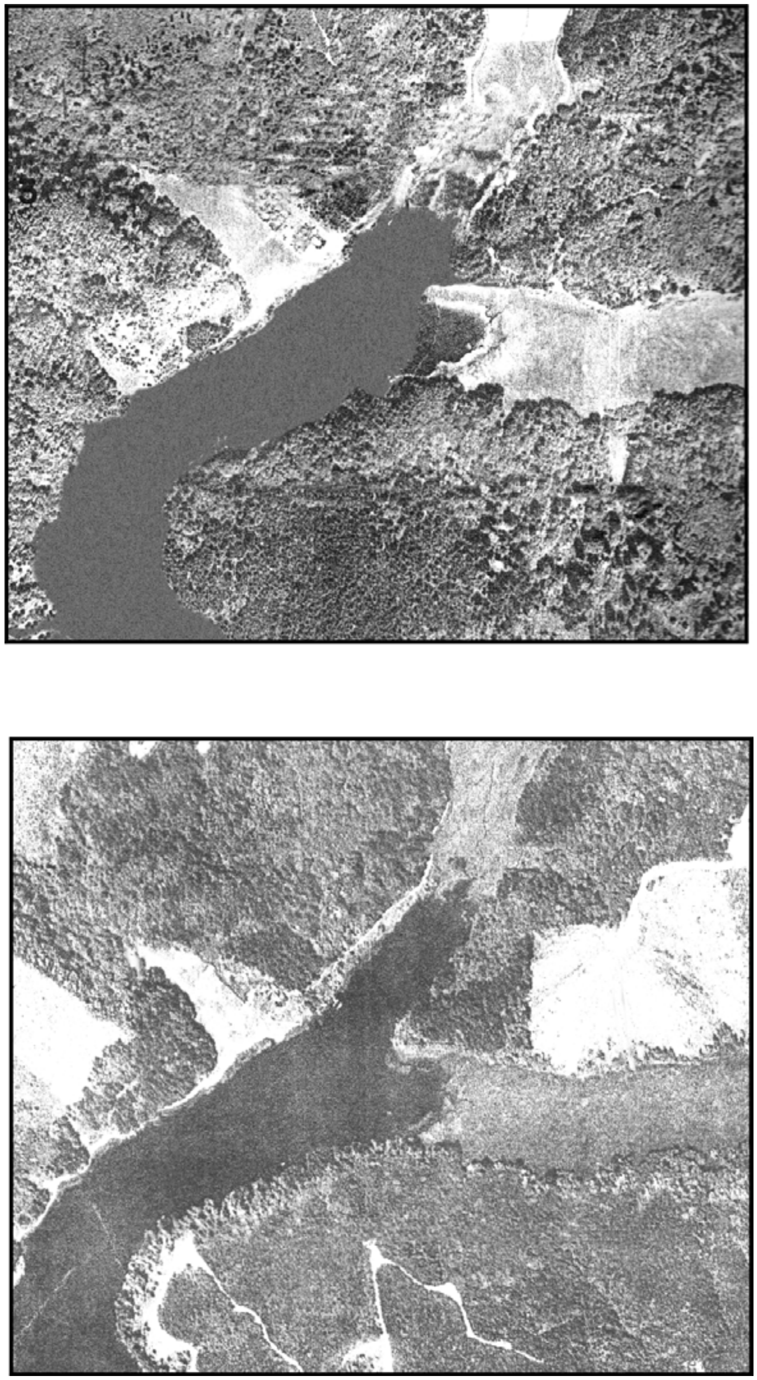


Figure 64. Aerial photographs of Carlson Arm (North Tenmile Lake) in 1967 and 1994 showing the inlets for Willkins Creek (north) and Murphy Creek (east).

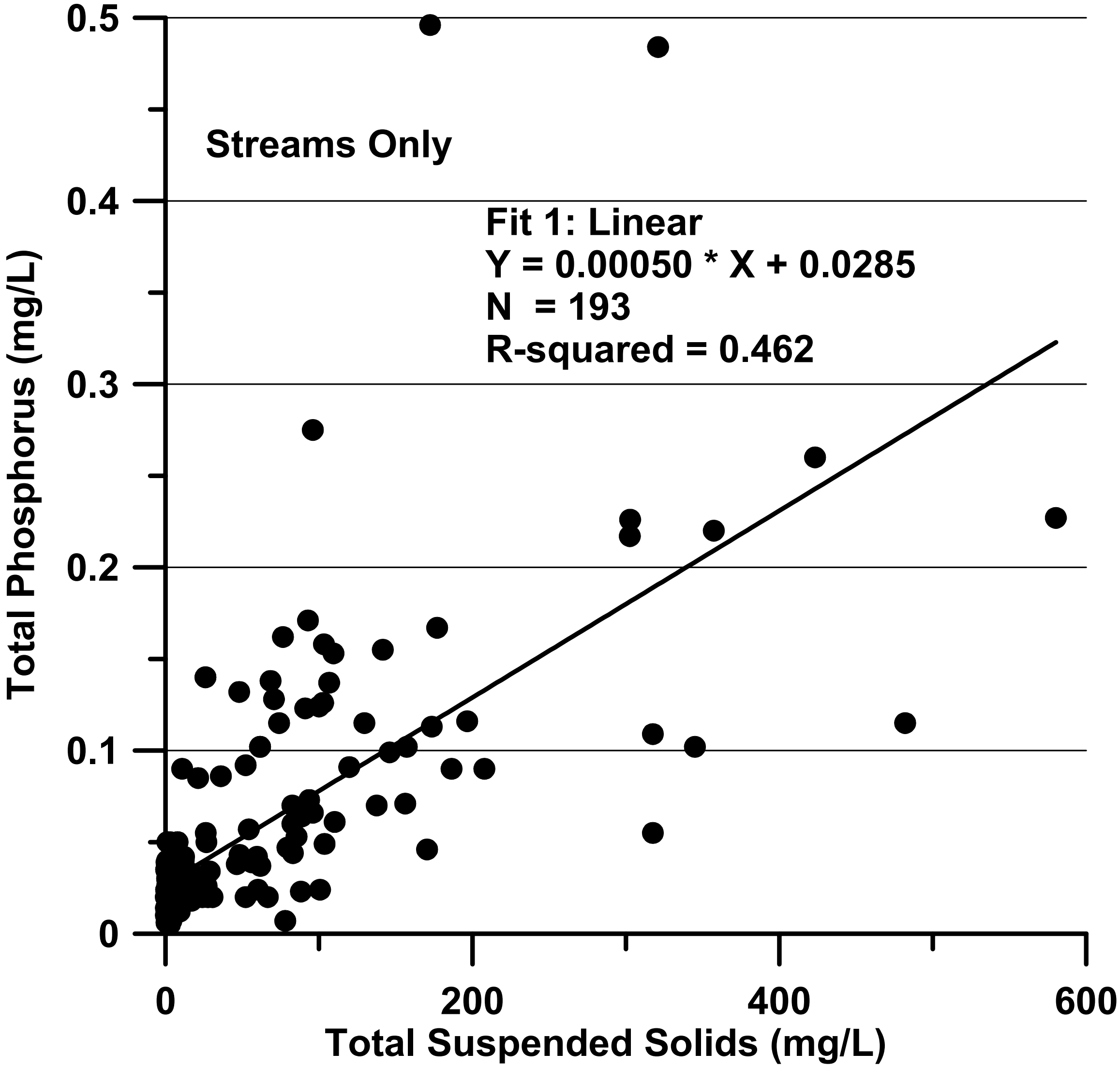


Figure 65. Relationship between total phosphorus and total suspended solids for tributaries to Tenmile Lake.

Tenmile Lake in and near the city of Lakeside, contribute a disproportionately greater amount of TSS to the lake. Catchments with significant wetlands such as Murphy Creek, Adams Creek, West Shutters, and Devore contribute runoff with low TSS concentrations. The Lindross catchment presumably contributes relatively low sediment loads because of low stream development. The Wetlands scenario shows vast improvement in the export concentrations of TSS compared to current land use. Two catchments, “Rex” and East Shutters, are forecasted to remain high because the model simulation is based on these sites remaining as clearcut.

Obviously, the vegetation is returning to these sites, but other locations in the watershed will still be subject to timber harvest and the locations of high export areas will change with time.

For phosphorus (PO_4), the patterns among the three model scenarios is more strongly related to soil properties. The soils in the far western portion of the watershed will continue to leach at rates greater than the eastern portion of the watershed (Figure 65). The wetlands scenario is not highly successful in reducing PO_4 concentrations in the western portion because this land will remain urbanized and will continue to export high concentrations of PO_4 to Tenmile Lake. In addition, export of PO_4 from wetlands can remain quite high because of solubilization of P under reducing conditions in the wetland soils. Total phosphorus loads, on the other hand, would be greatly reduced under a Wetland scenario because of the strong association of TSS and TP (Figure 65). However, because flows from the western portion of the watershed dominate the hydrology of the lake, it is the western portion of the watershed that must continue to receive the greatest effort at nutrient reduction.

The model indicates that the current export of NO_3 from the uplands is much greater than under historical conditions (Figure 59). However, the implementation of the Wetlands scenario would greatly reduce NO_3 concentrations in waters flowing to Tenmile Lake. Nitrate concentrations would remain high in waters draining clearcuts (i.e., "Rex" catchment).

3. Septic System Inputs

The SWAT model provides estimates of sediment and nutrient inputs to Tenmile Lake based on watershed processes and land use patterns. There are at least three other factors that influence nutrient concentrations in Tenmile Lake besides watershed inputs from land uses that are not dealt with in the SWAT model: (1) septic tank inputs, (2) internal nutrient cycling in the lake, and (3) biomanipulation of trophic structures. Although none of these three factors were formally addressed in this study, we attempted to assess the relative contributions of N and P from septic system loads using a spreadsheet analysis (Table 10). Using published estimates of nutrient generation per capita and estimating occupancy per month yields a range of monthly phosphorus loads. The upper and lower estimates are derived from assumptions regarding the degree of attenuation of N and P in the soils. Although it is not possible to verify these estimates of septic tank loads, they do provide some basis for assessing the potential role of septic systems to the total nutrient load.

Table 10. Tenmile Lake Septic Tank Loading Estimates

										High Estimate[b]		Low Estimate[c]	
Month	Month	# Days	Percent Occupancy	Density	Homes/ lake	TN[a] g/cap/d	TP[a] g/cap/d	TN Load g	TP Load g	TN Load kg	TP Load kg	TN Load kg	TP Load kg
Jan	1	31	20	2	250	23	2.3	71300	7130	71.3	7.1	30.3	3.03
Feb	2	28	30	2	250	23	2.3	96600	9660	96.6	9.7	41.1	4.11
Mar	3	31	40	2	250	23	2.3	142600	14260	142.6	14.3	60.6	6.06
Apr	4	30	50	2	250	23	2.3	172500	17250	172.5	17.2	73.3	7.33
May	5	31	60	2	250	23	2.3	213900	21390	213.9	21.4	90.9	9.09
Jun	6	30	70	3	250	23	2.3	362250	36225	362.25	36.2	154.0	15.40
Jul	7	31	80	4	250	23	2.3	570400	57040	570.4	57.0	242.4	24.24
Aug	8	31	80	4	250	23	2.3	570400	57040	570.4	57.0	242.4	24.24
Sep	9	30	60	3	250	23	2.3	310500	31050	310.5	31.0	132.0	13.20
Oct	10	31	40	2	250	23	2.3	142600	14260	142.6	14.3	60.6	6.06
Nov	11	30	30	2	250	23	2.3	103500	10350	103.5	10.4	44.0	4.40
Dec	12	31	20	2	250	23	2.3	71300	7130	71.3	7.13	30.3	3.03
Per Lake		365						2827850	282785	2828	283	1161	116
Per watershed								5655700	565570	5656	566	2322	232

[a] Derived from Chapra (1997)
[b] Assumes no attenuation of nutrients in the soils and that all nutrients are delivered to the lake
[c] Assumes a retention/reduction of septic loads based on soil interactions

The estimated lake-wide annual septic inputs of TP range from 120 to 283 kg compared to the SWAT model estimate of 384 kg/yr of PO_4. If we assume that the ratio of TP to PO_4 is 3:1 for the Tenmile watershed, then the annual load of TP based on the SWAT model is about 1150 kg/yr. This would place the annual septic system contribution of TP at about 10-22% of the total watershed inputs. However, because septic system contributions are expected to be greatest in the summer and early fall (Figure 66), it is likely that septic inputs constitute an important component supplying nutrients to support blooms of cyanobacteria from August to October.

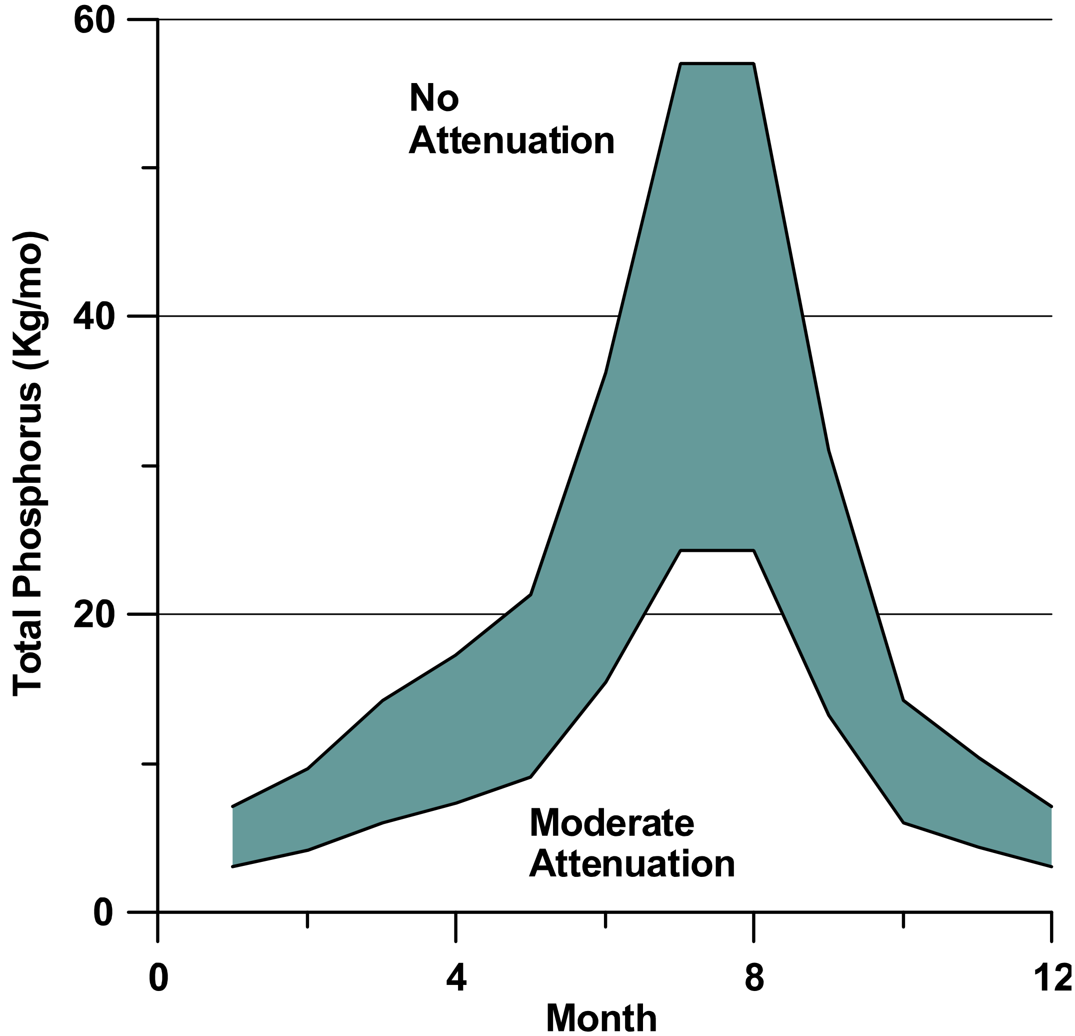


Figure 66. Estimates of monthly loading of TP to Tenmile Lake from septic system inputs. The upper and lower estimates reflect some of the uncertainty in attenuation of nutrients in the soil.

F. DISCUSSION

1. Current Conditions

Water quality in Tenmile Lake can be assessed from direct measurement of nutrients, Secchi disk transparency, dissolved oxygen depletion, phytoplankton community composition, chlorophyll *a*, and phytoplankton abundance. Mean annual values for typical parameters used to assess conditions in lakes indicate that Tenmile Lake is mesotrophic to eutrophic, with most indicators present in the "eutrophic" ranges (Table 11). This assessment of water quality varies depending on the parameter used to assess the conditions and the site in the lake. The site most consistently exhibiting the lowest water quality is NTB, at the intersection of Big Creek Arm and Carlson Arm. Secchi disk transparency is low, and total phosphorus, phytoplankton density, and phytoplankton biovolume generally are greatest at this site, which is consistent with visual observations of turbid inputs from Big Creek and significant algal blooms in the summer. However, the peak chlorophyll *a* concentrations were measured in the south lake. Other metrics of lake trophic status such as ^{14}C primary production rates and macrophyte biomass were not available in this study, however reconnaissance of the lake showed high abundance of macrophytes throughout the lake at depths less than 5 m. An examination of the macrophyte community showed that the dominant species is *Egeria densa* (Systma 1995), an exotic species morphologically similar to the native genus, *Elodea*. Despite its relatively

Table 11. OECD boundary values for fixed trophic classification system (modified from OECD 1982). Observed values for Tenmile Lake are highlighted.

Trophic Category	Mean TP[1]	Mean Chlorophyll *a* [2]	Maximum Chlorophyll *a* [3]	Mean Secchi[4]	Minimum Secchi[5]
Ultra-oligotrophic	<4.0	<1.0	<2.5	>12.0	>6.0
Oligotrophic	<10.0	<2.5	<8.0	>6.0	>3.0
Mesotrophic	**10-35**	2.5-8	8-25	6-3	3-1.5
Eutrophic	35-100	**8-25**	**25-75**	**3-1.5**	**1.5-0.7**
Hypertrophic	>100	>25	>75	<1.5	<0.7

[1] mean annual in-lake total phosphorus concentration (µg/l)
[2] mean annual chlorophyll *a* concentration in surface waters (µg/l)
[3] peak annual chlorophyll *a* concentration in surface waters (µg/l)
[4] mean annual Secchi depth transparency (m)
[5] minimum annual Secchi depth transparency (m)

shallow morphometry and polymictic nature, Tenmile Lake also exhibits rapid uptake oxygen in the bottom waters during brief periods of stratification. These all suggest that Tenmile Lake is meso-eutrophic to eutrophic. An assessment of water quality in Tenmile Lake conducted in July and August, 1994 showed similar results for TP, chlorophyll *a*, transparency, and dissolved oxygen depletion (Systma 1995). Earlier investigations by Phinney and McLachlan (1956), McHugh (1972), and Johnson et al. (1985) all cite the presence of algae or cyanobacteria taxa indicative of eutrophic lakes. The following sections discuss whether the current conditions represent a change from historical conditions and if so, the nature of the change.

An additional space-for-time substitution available is to compare water quality in neighboring Eel Lake with Tenmile Lake. The comparison of total phosphorus, chlorophyll *a*, and Secchi disk transparency shows that Eel Lake exhibits much better water quality than Tenmile Lake (Figure 67). Although Eel Lake is considerably deeper than Tenmile Lake (Z_{max} = 19.8 m versus 8.2 m, respectively), both lakes are highly dendritic and share the same physiographic features (Figure 2). The striking differences in water quality between the two lakes suggests that water quality in Tenmile Lake could be improved by dealing with some of the watershed and in-lake sources of nutrients.

2. Historical Conditions Based on Space-for-Time Substitutions

Three tributaries to Tenmile Lake were monitored on an intensive basis from November 1998 through May 1999. Streamflow declined after May and tributary inputs became relatively insignificant (on a lake-wide basis). During this period, Big and Benson Creeks, which appear to be representative of most tributaries to the lake, delivered sediment and nutrients at a rate far greater than occurred in Murphy Creek. Loads, expressed on a per-hectare basis, show that sediment yields from Big and Benson Creeks are at least ten times greater than the yield from Murphy Creek (Tables 7-9). The loads of nitrogen and phosphorus from Big and Benson Creeks are at least three times those from Murphy Creek. Although Big and Benson Creeks are degraded relative to Murphy Creek, these two catchments may not represent worse-case conditions in the watershed.

The modeling results indicate that nearly one-half of the remaining catchments may have higher yields (on a per-hectare basis) than Big and Benson Creeks. The modeling results are based on aerial photography collected in 1994 and updated with an aerial survey in 1999. Consequently, the actual spatial information on land use does not correspond precisely with

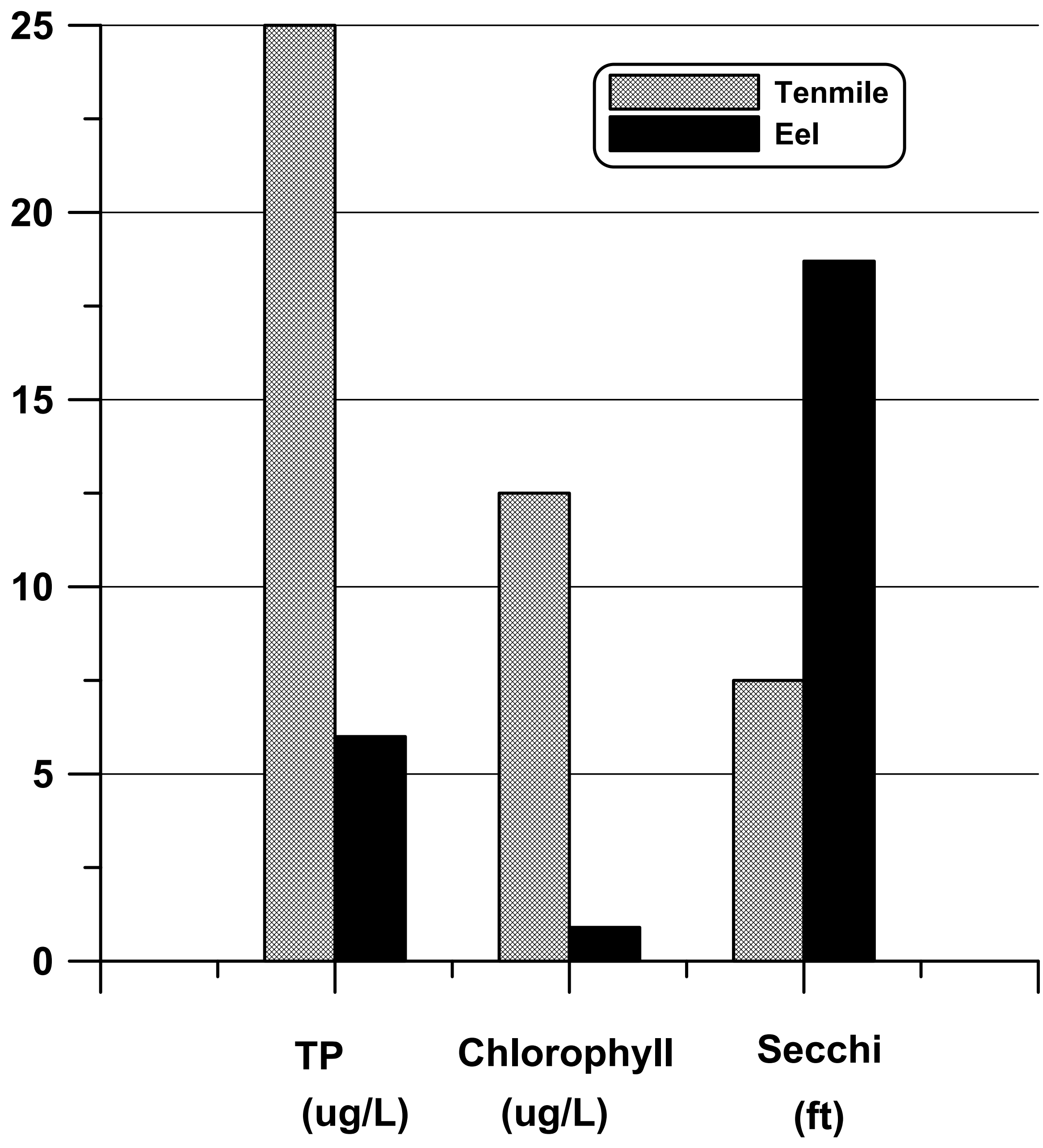


Figure 67. Comparison of total phosphorus, chlorophyll *a*, and Secchi disk transparency in Eel Lake (Johnson et al. 1985) and Tenmile Lake (this study).

current distributions of nonpoint source loads. One of the most variable factors in the watershed land use is timber harvest. Areas which were harvested in the early 1990s have since experienced appreciable timber regrowth, causing the sediment and nutrient loads to decline. Conversely, some areas which were mature timber in 1994 have since been harvested, causing loads to increase in those areas. The location of wetlands and recent clearcuts affects all phases of the modeling. Not only are sediment and total phosphorus production increased during logging, but nitrogen export is also affected. For example, a comparison of nitrate concentrations in Murphy Creek and a tributary to Benson Creek show a dramatic change in nitrogen yield (Figure 68). About 80% of the Benson Creek tributary catchment had been logged the previous year.

The comparatively high water quality delivered from Murphy Creek is attributed largely to the restored wetland on the lower 2.5 km of the stream, which provides a high degree of protection from timber harvest in the uplands. Using Murphy Creek as an indication of historical stream water quality shows that current water quality in most of the tributaries has been severely degraded. A modeling analysis of sediment and nutrient loads under pre-development conditions (i.e., native mature forest in the uplands and intact, unchanneled wetlands in the lowlands) indicates that historical water quality in Tenmile Lake would have been much better relative to current conditions (Tables 7-9). This analysis does not account for the loss of marine-derived nutrients to the watershed that has occurred with the decline in the anadromous fisheries. Schmidt et al. (1998), Kline et al. (1993), and Bilby et al. (1996) have shown that marine-derived nutrients from anadromous fish can be important components of nutrient cycling in Alaska and Washington. The run of sockeye salmon in Karluk Lake, AK provides an estimated 40 percent of the total phosphorus in the lake (Schmidt et al. 1998). In a small Washington stream, spawning coho salmon provided more than 30 percent of the nitrogen found in juvenile coho salmon (Bilby et al. 1996). However, even a 50 percent increase in the estimated historical nutrient load from Murphy Creek results in the undeveloped nutrient load being at least double its present rate from Big and Benson Creeks. Recall that the model estimates show most of the tributaries having considerably greater loads than Big and Benson Creeks. Furthermore, consideration of the effect of historical anadromous runs would not alter the estimates for sediment production from the watershed. In summary, the tributary monitoring and the modeling derived from the monitoring show that the watershed loads of sediment and nutrients are substantially greater than historical values.

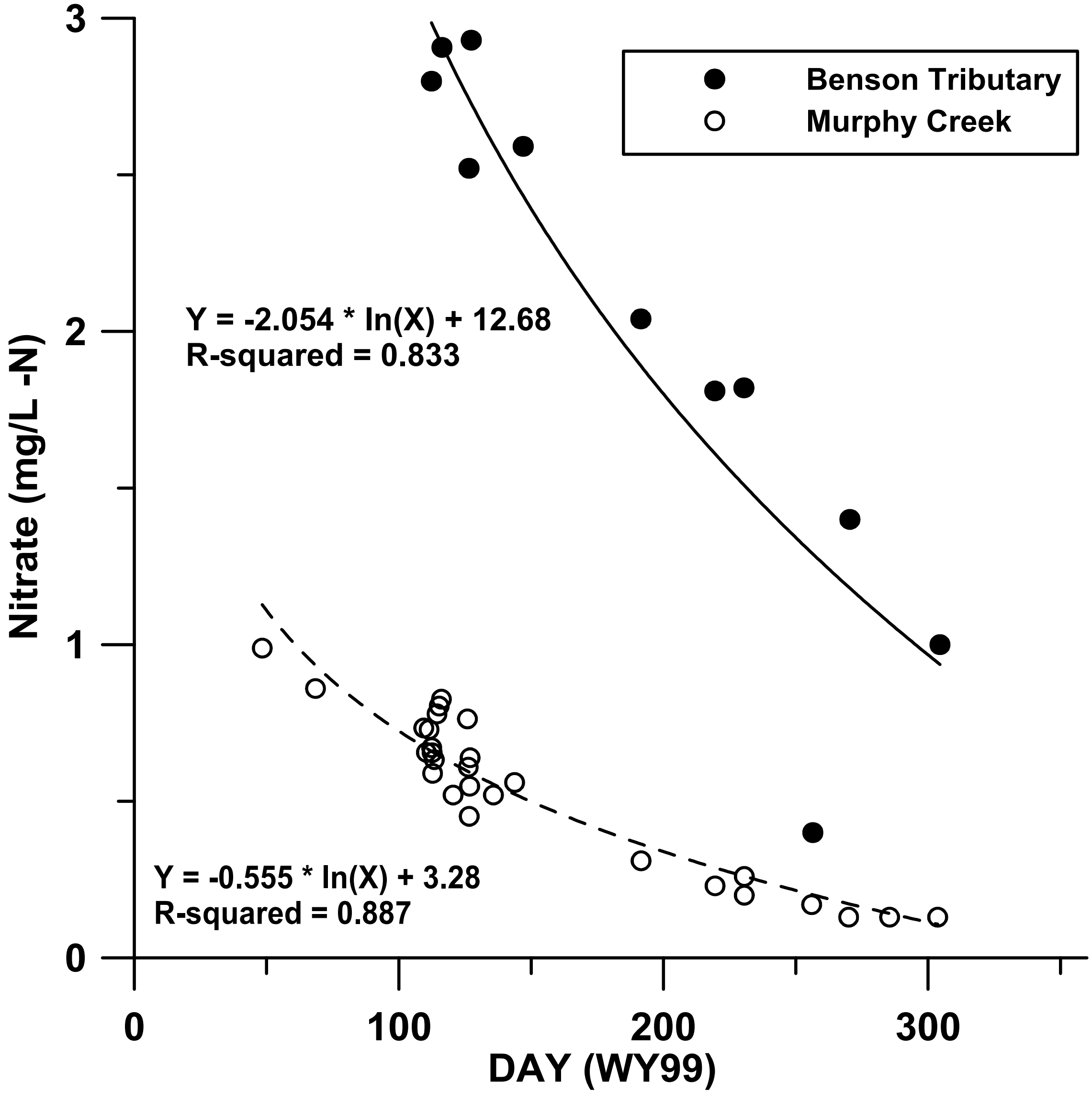


Figure 68. Nitrate (NO_3-N, mg/L) concentrations in Murphy Creek and an unnamed, recently logged tributary to Benson Creek.

3. Historical Conditions Inferred from Analysis of the Sediments

The sediments were analyzed for sediment accumulation rate (SAR), nitrogen, ^{15}N, cyanobacterial akinetes, and diatoms. The SAR has increased two to four-fold over pre-development conditions. An initial increase was noted in the late 19th century, corresponding with the early land clearing and settlement in the area. A major increase in SAR occurred in the period 1910-1920, at a time when commercial logging activity was high. The SAR declined circa 1950, but has shown a continuous increase since then. The general shape of the SAR record (i.e., increase in SAR, a mid-century partial recovery followed by an accelerated increase in SAR) is similar to that observed in Devils Lake, located on the Oregon central coast (Eilers et al. 1996b). The increased SAR is consistent with both the monitoring and watershed modeling results which point to high rates of sediment yield from much of the watershed.

Nitrogen and the ratio of ^{15}N/^{14}N show increases in the upper sediments. As noted earlier, the nitrogen results are subject to a moderate degree of uncertainty because of diagenesis within the sediments, but the findings are consistent with an increase in lake productivity and an increase in the biomass of nitrogen-fixing cyanobacteria. The sediment akinete results also suggest that there has been a several-fold increase in the biomass of cyanobacteria in Tenmile Lake in recent decades.

The diatom results show a significant increase in the relative abundance of taxa that favor eutrophic waters. There has been a major increase in *Asterionella formosa* and a smaller increase in *Fragilaria crotonensis*, taxa commonly associated with eutrophic lakes. The diatom bloom observed in April, 1999 was comprised largely of *A. formosa*. Benthic (bottom-dwelling) diatoms have decreased in the lake and the diatom community is now largely planktonic. These changes are consistent with a decrease in transparency in the lake, most likely caused by an increase in planktonic algae and cyanobacteria.

4. Watershed Versus In-Lake and Near-Shore Factors

The data consistently show that water quality in the lake is impaired, that the tributary water quality is in most cases degraded, and that the lake water quality has declined during the last century. Because the focus of this study has been largely on watershed processes and stream monitoring, it is tempting to infer that the degradation of the lake has been caused by practices within the watershed such as timber harvest, livestock grazing, and stream channelization. Indeed, it is clear that watershed activities have altered the loads of sediment and nutrients to the lake. However, it does not necessarily follow that a lake improvement program based solely on improving land use practices in the watershed will yield proportional improvements in lake water quality. Although decreasing the sediment and nutrient loads from the watershed is a necessary element of improving Tenmile Lake, there are at least three other factors that have not been fully addressed in this study that may have a major influence on the lake.

One factor is the issue of land use development along the lakeshore. The SWAT model in its present formulation has limited capability for including nutrient input associated with housing development. Housing development increases sediment production during construction and continues to supply the lake with increased loading of nitrogen and phosphorus for the duration of occupancy and beyond. A recent survey of septic systems around Tenmile Lake found a number of them had no records of permits or were deficient in a number of respects (P. Blake, DEQ, pers. comm., 2001). In many cases, the development associated with shorelines resembles development elsewhere, including installation of extensive lawns and landscaping that require fertilizer input. This is often accompanied by removal of natural vegetation.

A second factor not addressed in this study is the presence and abundance of aquatic macrophytes. The current information indicates that Tenmile Lake is heavily infested with the exotic *Egeria densa*. It is unknown if the *E. densa* has simply replaced the native *Elodea*, or whether the current spatial distribution and density of *Egeria* exceeds the historical condition of *Elodea*. Macrophytes have the capacity to alter internal processes in the lake through increased primary production (macrophytes have a greater primary production rate per unit than algae [cf., Chapra 1997]) and can extract nutrients from the sediments. Upon senescing in the fall these accumulated nutrients are made available through mineralization and the decaying macrophytes exert a biochemical oxygen demand. If the macrophyte biomass has increased, it would be expected to have a negative effect on the lake.

Lastly, we did not collect data regarding the possible influence of the fisheries on water quality in Tenmile Lake. It is now understood that lake productivity is affected not only by inputs from the watershed, but also by the biological activity within the lake. Altering the fisheries can promote major changes in the zooplankton community which in turn can alter the grazing rate of phytoplankton (Sarnelle 1992, 1993). Tenmile Lake historically was dominated by anadromous fisheries of coho, steelhead, and sea run cutthroat trout. The coho spawning used to be so intense in the tributaries that redds were systematically destroyed by incoming fish intent on locating suitable sites (Schwartz 1977). The lake served as a nursery for rearing smolts to a size that would guaranty a successful return of adult salmonids.

The current fisheries is vastly different than the historical condition. The dominant fish are now exotic species, endemic to the Midwest. Largemouth bass is the primary game fish and there are abundant populations of bluegill, yellow perch, and crappie (Abrams et al. 1991). The fisheries is currently dominated by highly planktivorous fish. According to our present understanding of biomanipulation effects, these taxa (e.g. bluegill, yellow perch) are very efficient at consuming the larger zooplankton species. The reduction of large zooplankton, in turn, reduces grazing pressure on the phytoplankton which allows phytoplankton biomass to increase. Phytoplankton biomass is able to take advantage of the reduced grazing pressure because more nutrients are available from watershed inputs. If the biomanipulation theory is correct, as it applies to Tenmile Lake, it suggests that water quality problems in the lake are the product of changes in both the watershed and fisheries. Therefore, a water quality improvement program for Tenmile Lake may need to incorporate both watershed restoration and some modification of the fish community composition.

5. Lake Restoration Issues

a. *A Review of Water Quality Problems in Tenmile Lake*

Most of the models relating lake productivity to nutrient concentrations are predicated on the assumption that phosphorus is the limiting nutrient (cf. Ryding and Rast 1989). However, focusing the rehabilitation efforts for Tenmile Lake exclusively on phosphorus may not be sufficient. There is reason to believe that loading of N to Tenmile Lake has increased at a greater rate than the increase in P loading. Firstly, nitrogen loading greatly increases as a consequence of timber harvest. Removal of trees and other vegetation reduced interception of atmospheric deposition of nitrogen and the increased decay of plant litter promotes nitrification, causing a substantial increase in N "leakage" (Waring and Schlesinger 1985; Schlesinger 1991). We observed this during monitoring of a small tributary to Benson Creek, which had been clear-cut (Figure 67). Nitrogen loading to the watershed has also been increased as a consequence of livestock operations and septic system effluent. In addition to increased sources of N loading, there have been major hydrologic modifications that enhanced export of N to the lake. These activities include the channelization of all major tributaries to the lake and the elimination of wetlands along the stream flow-paths. The comparison of nitrate concentrations in Murphy

Creek, with a restored wetland, compared to the other major tributaries (Figure 13) illustrates the combined effect of livestock removal and wetland restoration in controlling nitrogen export.

The effect of increased nitrogen loading may be to allow for increased algal production by chlorophytes, which are incapable of fixing atmospheric N. For example, the high biomass of *Asterionella formosa*, a diatom, measured in the spring may be aided by both high phosphorus and available nitrate (Figures 11 and 13).

One of the most serious public health-related issues affecting Tenmile Lake is the presence of *Microcystis aeruginosa*, a cyanobacteria that produces the Hepatoxin microcystis. Unlike other cyanobacteria found in Tenmile Lake, such as *Anabaena flos-aquae*, that are capable of fixation of atmospheric N, *Microcystis* requires a source of nitrogen, usually as nitrate or ammonium (Chorus and Bartram 1999). Inorganic nitrogen concentrations were generally undetected (at 5 µg/L-N), suggesting that *Microcystis* is acquiring N immediately as it becomes available from sources such as septic inputs, uptake of ammonia from anaerobic bottom waters, excretion from fish, or lysing of decomposing phytoplankton cells.

The analysis of cyanobacterial akinetes showed that blue-green algae were present in Tenmile Lake prior to settlement, but the populations of at least some taxa have increased several fold during the 20th century. Cyanobacteria, particularly taxa such as *Anabaena flos-aquae* and *Microcystis aeruginosa*, achieve a competitive advantage in phosphorus-enriched lakes (Reynolds 1984) and thus any lake rehabilitation strategy needs to focus on reducing nutrient inputs.

The depletion of oxygen in Tenmile Lake, even though it is a transitory occurrence, is a symptom of excessively high primary production. Decaying organic matter consumes oxygen and in the process makes additional nutrients from the sediment available to the water column. Because Tenmile Lake is comparatively shallow, it is a poor candidate for mechanical interventions such as aeration to solve the depletion of dissolved oxygen. Consequently, the only approach to address this problem is to reduce primary production through reduction of nutrient loading.

Tenmile Lake is currently on the Oregon 303(d) list because of “aquatic weeds or algae.” Although there is a demonstrated problem associated with the quantity and species composition of phytoplankton, it is unclear whether there is a comparable problem associated with aquatic macrophytes. We know that one of the dominant macrophytes in Tenmile Lake is *Egeria densa*, an invasive plant introduced from South America. This species has most likely replaced the native *Elodea*, which grows in similar habitats and is morphologically similar to *Egeria*. However, we have no information whether the current distribution and abundance of *Egeria* in Tenmile Lake differs from pre-settlement condition of *Elodea*. There are two arguments to suggest that macrophytes extent or density may have increased. One is that increased nutrient input has stimulated macrophyte growth. Secondly, increased sediment accumulation may have created more favorable habitat for macrophytes. One argument to suggest that macrophyte abundance has not increased is that increased phytoplankton biomass has decreased the available light, thus decreasing available macrophyte habitat. Alternatively, both sets of forces may be acting to affect the macrophytes. Increased nutrient supply could be increasing macrophyte density in shallow areas and light limitation may be decreasing macrophyte density or extent in deeper portions of the lake. Until methods can be developed to reconstruct changes in macrophyte populations (emergent species of macrophytes can be studied quite easily because they produce seeds which can be identified in the sediments [cf. Whitlock et al. 2000]; however *Elergia* and *Elodea* are submergent forms which are not so easily evaluated), we are unable to accurately reconstruct the changes to aquatic weeds in Tenmile Lake.

Another problem in Tenmile Lake is the increase in exotic fish taxa and the concomitant decrease in anadromous fish (Abrams et al. 1991). The alteration of fish populations in Tenmile Lake, although not originally caused by nutrient inputs from the watershed, may play a role in the recovery of the lake. The pre-development fisheries of Tenmile Lake was dominated by runs of coho salmon. The returning adults spawned and their carcasses provided for nutrient cycling back to the watershed and lake. However, in small tributary streams to Tenmile Lake many of the carcasses were consumed by aquatic insects and foragers (birds, bears, racoons) rather than by decomposing and leaching minerals back to the water. The effect of anadromous fisheries on nutrient cycling can be determined to some degree by measuring changes in marine-derived nitrogen as reflected in the ratio of $^{15}N/^{14}N$. A decrease in the $^{15}N/^{14}N$ ratio often reflects a decrease in returning salmon (cf., Bilby et al. 1996). However, despite the dramatic reduction in the salmon population, the $^{15}N/^{14}N$ ratio in the sediments increased (Figure 50). This indicates that increased nutrient loading from the watershed has overwhelmed whatever loss in marine-derived nutrients occurred.

A second nutrient-related change may have occurred in Tenmile Lake associated with the alteration in fisheries. The fisheries in Tenmile Lake is now dominated by centrarchids (crappie, bluegill, largemouth bass), species which are highly effective consumers of zooplankton. High predation rates on zooplankton can reduce grazing pressure on phytoplankton, and allow for greater biomass of algae and cyanobacteria. Alternatively, increased populations of zooplanktivorous fish and benthivores can stimulate phytoplankton growth by making more nutrients available in the water column. Thus, even in the absence of elevated watershed loading of nutrients, it is unclear whether phytoplankton abundance in Tenmile Lake would not remain a nuisance. Excessive zooplankton grazing following introduction of the exotic tui chub is proposed as a mechanism to explain deterioration of water quality in Diamond Lake (Eilers et al. 2001).

To meet the objective of a decrease in phytoplankton abundance, particularly in cyanobacteria, will require a reduction in watershed loading of nutrients. However, because of the alteration in the fisheries, it is likely that a reduction in nutrient loading will not be sufficient. A substantial reduction in nutrient loading can be achieved by elimination of stream channelization, wetland restoration, and septic system rejuvenation. A restoration of the anadromous fisheries may be far more difficult to achieve. An attempt to eliminate the perch and centrarchids with a rotenone treatment in 1968 failed to destroy the target fish. The second attempt to reduce the target species by introducing largemouth bass (*Micropterus salmoides*) resulted in a decline of the salmonids, rather than reduction in the target species (Figure 4).

Another important element of restoring Tenmile Lake concerns the issue of human health. The study documented the presence of the cyanobacteria, *Microcystis* and its toxin microcystin. This toxin has long-term effects on liver functioning and efforts should be made to reduce conditions favorable to the growth of *Microcystis*. Like many cyanobacteria, *Microcystis* is present under high nutrient conditions. Ammonia, which is made available in the lake through decomposition reactions under anoxic conditions, may provide a critical source of nitrogen to this taxa. The study, however, illustrated that *Anabaena flos-aquae* was the dominant cyanobacteria present in the lake. Funds were not available in this study to determine whether the toxin sometimes produced by this taxa, anatoxin-a, was present in Tenmile Lake. We encourage the organizations responsible for checking and maintaining the safety of public drinking water supplies (a designated beneficial use of Tenmile Lake) evaluate the lake for anatoxin-a.

b. Evidence of Nutrient Enrichment

There are five lines of evidence for concluding that nutrient loading to Tenmile Lake has increased. Each of these lines of evidence has uncertainty associated with it, but the consistency of both the direction and general magnitude of the increase is striking. The lines of evidence, listed in increasing level of uncertainty are:

(i) comparative stream monitoring
(ii) sediment record
(iii) watershed modeling
(iv) comparative lake analysis
(v) biological community composition

(i) Comparative Stream Monitoring

The monitoring of stream water quality from Murphy Creek, a system with a restored wetland, compared to the major stream systems with grazing and channelization (Big, Benson, Johnson) shows a dramatic difference in the water quality of inlets to Tenmile Lake. Murphy Creek, although still subject to active timber harvest in the uplands, discharges water that is two orders of magnitude lower in total suspended solids and at least 50% lower in concentrations of nitrogen and phosphorus. The direct monitoring of these tributaries provides unequivocal evidence that the present nutrient loading from the watershed has been elevated by 50% (N) to 90% (P) above pre-development levels.

(ii) Sediment Record

All four sediment cores collected in Phase II and the two cores collected in Phase I show increases in sediment accumulation rates (SAR) of 100 to 400% above pre-development levels. The rates are greatest adjacent to tributaries with high disturbance and channelization. All other signals in the sediment record including sediment nutrients, diatoms, and cyanobacterial akinetes, show a major enrichment of the lake. The internal consistency of all sediment information suggests that nutrient loading has increased greatly - perhaps a two- to four-fold increase for total phosphorus and comparable increases in nitrogen.

(iii) Watershed Modeling

Application of the SWAT watershed model to Tenmile Lakes basin shows widespread increases in sediment and nutrient loading to the lake. Although the stream monitoring and paleolimnological data were used to refine some of the model output, use of default coefficients for the different land use cover types would still yield dramatic increases in sediment and nutrient loading to the lake.

(iv) Comparative Lake Analysis

Examination of nearby lakes in a similar geomorphic setting with different land use histories provides some indication of how Tenmile Lake has changed. Eel Lake located immediately north of Tenmile Lake has experienced some timber harvest, but no stream channelization or grazing. Eel Lake has one-third the level of total phosphorus, has twice the lake transparency, and one-sixth the concentration of chlorophyll *a* compared to Tenmile Lake. Although Eel Lake is deeper than Tenmile Lake, it is striking how different the water quality is in these neighboring lakes. Comparison with Devils Lake, located further north, exhibits a watershed history similar to Tenmile Lake. Both Devils and Tenmile lakes show similar paleolimnological responses.

(v) Biological Community Composition

The phytoplankton community composition in Tenmile Lake is indicative of a lake with high nutrient loading. Chlorophyll *a* levels up to 60 µg/L have been measured in the open lake twice and cyanobacteria are dominant in the summer and fall. An extreme value of 150 µg/L of chlorophyll *a* was measured in Carlson Arm adjacent to Big Creek. Toxic algae create conditions unfavorable for human health. The fisheries community has been radically altered with the introduction of numerous non-native fish, including centrarchids, yellow perch, and bullhead. The centrarchids and yellow perch, in particular, have been documented to degrade water quality in numerous lakes by consuming the larger zooplankton taxa, leaving a smaller zooplankton assemblage with lower phytoplankton grazing efficiency. The non-native fisheries community thus directly affects the water quality by altering the zooplankton/phytoplankton community structure, but may also increase internal cycling of nutrients within the lake.

In summary, five lines of evidence are offered to support the conclusion that the nutrient loading to Tenmile Lake has dramatically increased in the 20th century. Two of the lines of evidence, stream monitoring and the sediment record, are based on data collected in the watershed and lake, respectively. The watershed modeling, although refined using data collected here, would still yield similar conclusions using default information for land use - water quality relationships. The last two lines of evidence, comparative lake analysis and evaluation of the current biological lake community could have alternative explanations. However, they are consistent with the previous lines of evidence and provide additional support for the enrichment of Tenmile Lake.

c. Uncertainty in Estimates of Nutrient Enrichment and Prognosis for Lake Recovery

No formal analysis of uncertainty was conducted for this study, but an appreciation for the level of uncertainty can be derived by examining the variability of the measured and inferred lines of evidence. The level of uncertainty varies, depending upon which of the three primary variables is considered. For total suspended solids (TSS) in the streams and sediment accumulation rate (SAR) in the lake, the evidence is clear. Sediment export increased as much as two orders of magnitude based on the stream monitoring data, but the increase in lakewide SAR was between two- to four-fold. The sediment record probably provides a much better representation of the overall watershed response because it integrates inputs over time and space rather than the instantaneous selected observations from the stream monitoring. The watershed modeling process is to some degree circular, since the monitoring and sediment data are used to adjust model coefficients. However, even using the most conservative of export coefficients for the altered land use would result in substantial increases in sediment transport. No information for evaluating sediment inputs is available from the comparative lake analysis or biological assessment of Tenmile Lake. Nevertheless, the first three lines of evidence provide irrefutable evidence that sediment export from the watershed and accumulation within the lake has increased greatly in the 20th century.

The input of phosphorus, excluding any additional watershed sources, would likely match that of the sediment input because of the strong binding affinity between P and soil particles. Watershed activities such as livestock grazing and septic inputs would only serve to increase the P load above the rate of sediment input. Thus, the uncertainty that the rate of P loading to the watershed has increased greatly is even less than that for TSS. In addition, the P concentrations in Tenmile Lake can be evaluated with respect to those found in Eel Lake which indicate a 3 to 4 fold difference with Tenmile Lake. Finally, the phytoplankton community composition is highly indicative of a nutrient-enriched system, a feature that would be unlikely in an undisturbed lake in this geomorphic setting.

The input of NO_3 and other forms of nitrogen have also increased as shown in the stream monitoring, sediment accumulation of N, and watershed modeling. Clearcut watersheds show a three-fold increase in NO_3 export, uncorrected for increased runoff associated with vegetation removal, over a more natural stream system. Apparent nitrogen accumulation in the lake sediments suggests a striking increase in the rate of input superimposed on the already high rate of sediment accumulation. Watershed modeling shows elevated NO_3 export over pre-settlement conditions. The uncertainty in N loadings to Tenmile Lake is greater than for both TSS and TP. TSS is comprised largely of inorganic soil particles and except for acting as an exchange site for P, is largely inert. Nitrogen, however, is highly mobile in the inorganic forms (NO_3, NH_3) and can evolve as a gas (N_2, NH_3). Organic N is subject to mineralization reactions into highly reactive inorganic forms. Consequently, although all evidence indicates that N loading to Tenmile Lake has increased on a magnitude similar to or greater than P, the uncertainty regarding this conclusion is greater than for TSS and P.

The discussion regarding uncertainty in the analyses is germane for this application only if the level of uncertainty would impact the decision-making process for lake and watershed rehabilitation efforts. In our opinion, the level of uncertainty is so small that there is no technical basis for not undertaking a restoration effort to greatly reduce watershed export of sediment and nutrients. The uncertainty is not in the level of magnitude required to address the water quality problems in the lake – the uncertainty is associated with the likely response of the lake to a reduction in watershed loads. The uncertainty in lake response is caused primarily by: (1) hysteresis, and (2) biological mediation. Hysteresis is a phenomena in which the reverse path in a process differs from the forward path (Figure 69). In this application, the lake response to a given phosphorus concentration during the eutrophication process may differ from the same lake response during the recovery process. The process was noted in recovery of lakes to reduction in P loading because the accumulation of P in the sediments required an extended period to re-equilibrate with the overlying water column (cf., Chapra 1997). Even in the absence of nutrient enrichment of the sediments, lake response may be altered because of the permanent loss of lake volume from sediment input, expansion of macrophyte beds, and other long-term changes to the

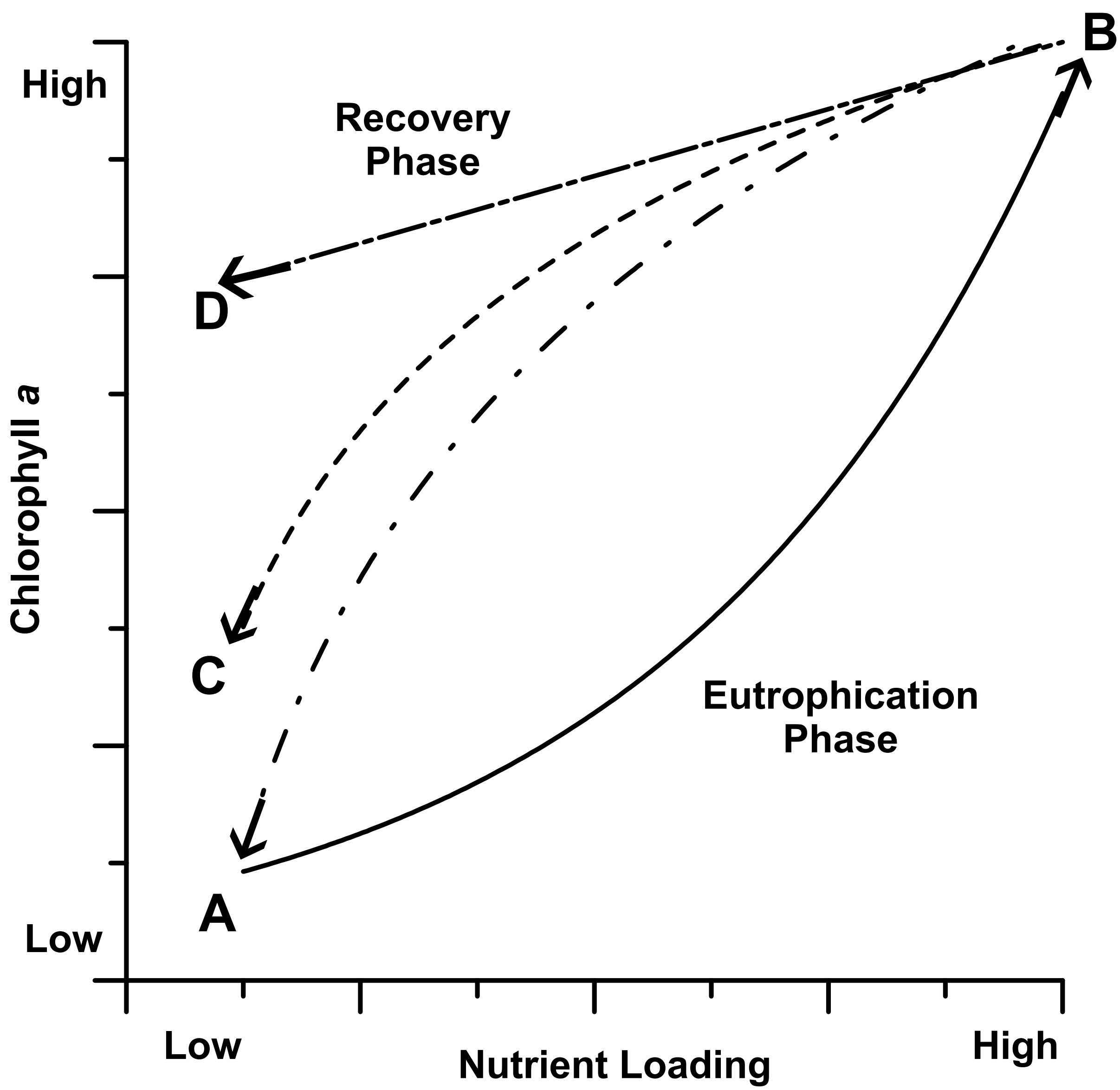


Figure 69. A schematic representation of lake response to increased nutrient loading (curve A → B) and multiple scenarios for lake response to decreased nutrient loading (B → A; B → C; and B → D) depending on factors within the lake.

lake. Thus, there is no guarantee that if watershed loads were reduced to pre-settlement rates, that the lake would return to its former condition.

The second major area of uncertainty relates to the fact that the water quality in a lake is the product of both the watershed loading and internal cycling of nutrients. Nutrient cycling within a lake can be profoundly altered by changing biota. This is particularly the case for fisheries in which the current fisheries community affects the lake metabolism very differently than the original fisheries. In Tenmile Lake, historical predation on zooplankton was likely quite modest under a salmonid-dominated regime. Consequently, grazing pressure by zooplankton on the phytoplankton was probably quite high. Under the current fisheries, pressure on the zooplankton community is high because of the highly effective predation by centrarchids and yellow perch. Consequently, grazing pressure on the phytoplankton is reduced, thus allowing higher rates of algal biomass and a different relationship between TP and chlorophyll *a* (Figure 69; curve B → D).

d. Moving Towards Lake Rehabilitation

The restoration of Tenmile Lake to its pre-development condition is probably not technically feasible. Even if all land uses were converted back to the native vegetation and all structures and human activity were removed from the watershed, other changes already imposed on the lake would likely inhibit complete restoration. Nevertheless, water quality conditions are poor and some intervention must take place to prevent further deterioration. There are a number of management actions that can be taken to improve water quality conditions in the lake and thus achieve a reasonable degree of rehabilitation. Ideally, a comprehensive lake and watershed management plan should be developed which synthesizes the scientific findings, public involvement, and economic realities. Such a plan is beyond the scope of this nutrient study, but we expect that such a plan, at a minimum, would address nutrient loads, wetland restoration, exotic fish species, septic loads, and monitoring requirements.

The recovery of Tenmile Lake is linked not only to reduction of nutrient sources, such as septic tank loads, but also requires a restoration of more natural hydrologic flow paths. The generation of nutrients in the watershed is only harmful to the lake if the hydrologic flow paths exist to transport the sediment and nutrients. The development of agriculture in the Tenmile Lake watershed was made possible by channelization of virtually all major lowland valleys. Unlike many applications elsewhere, the channelization of this watershed was conducted by channelizing both sides of a valley. The result was a very efficient means of transporting runoff to allow the central portion of the valleys to remain as pasture. An illustration of the consequence of channelization in the watershed on stream morphometry is provided in Figure 70. Murphy Creek has a small channel that is easily exceeded during high precipitation events. As a consequence, much of the high flows are distributed across a large wetland at very low velocity. In contrast, flows in Benson Creek (and also Big and Johnson) are distributed in an enlarged channel designed to carry much greater stream discharge prior to exceeding capacity (Figure 70). Benson Creek at this location carries nearly three times the discharge of Murphy Creek on a unit-area basis.

The stream geometry of these streams at these sites is very different. Benson Creek has a width:depth (w:d.) ratio of about 2.5 compared to a value of about 1.2 for Murphy Creek. However, when the bank full capacity of Murphy Creek is exceeded, the w:d ratio greatly increases ($\geq$ 300), which results in a huge increase in the wetted perimeter, a large increase in channel resistence, and very low effective stream velocities.

The higher discharges in the channelized streams such as Benson Creek lead to greater stream velocities because of shortening of stream lengths and the positive relationship between discharge and stream velocity for a given slope. The increase in the proportion of flow in the confined channels increases the stream power - hence its ability to do work. The greater stream power is indicated by the erosive capacity of a stream or its ability to transport an already entrained sediment load. This concept is illustrated in Figure 71, which shows that for a given particle size found in silt loam soils typical of some soils in the Tenmile watershed, the flows in the channel of Benson Creek (A) are actively eroding soils, whereas velocities in the overbank area of Murphy Creek (B) are only capable of transporting the small size material.

This study has indicated that to achieve a substantial improvement in water quality will require major reductions in sediment and nutrient loads from the major tributaries, combined with reductions in loads from septic systems. Based on the high water quality measured in Murphy Creek, it appears feasible to achieve major reductions in tributary inputs by returning some of the land that is currently grazed to its original wetland status. The effectiveness of this treatment will depend on the extent of wetlands restored and the degree to which stream channels are allowed to convert from their artificially-maintained channels to sinuous vegetated channels.

The reduction of nutrient loads from septic systems is currently being addressed by DEQ through a program of septic system permit review. This requires that land owners comply with existing statutes, upgrading or replacing septic systems as needed. Of the nearly 500 dwellings

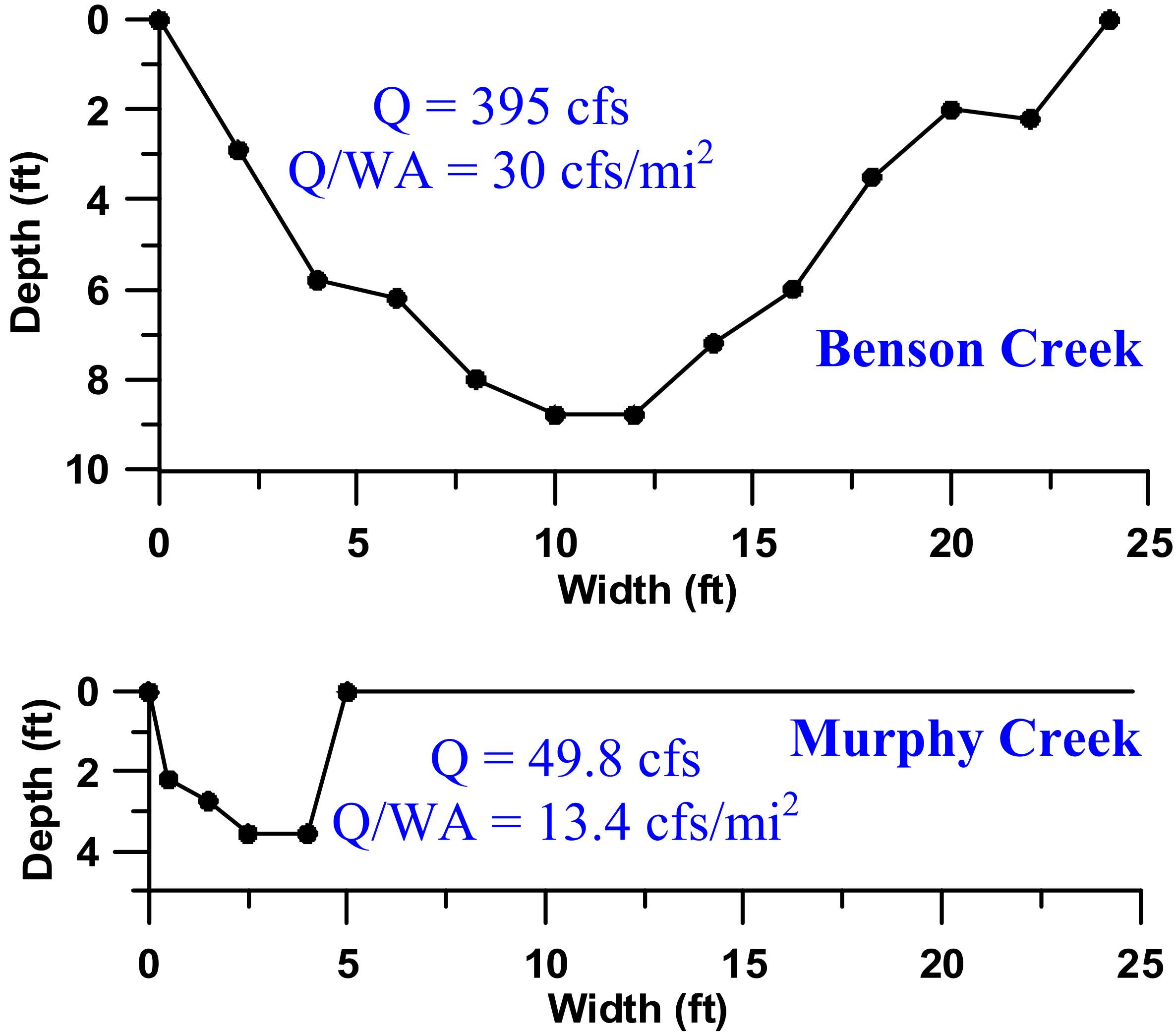


Figure 70. Stream channel cross sections for Benson Creek (top) and Murphy Creek (bottom) at the water quality monitoring sites. The stream discharge (Q, cfs) and discharge normalized per unit area (Q/WA, cfs/mi^2) is shown for both sites at bank-full flows.

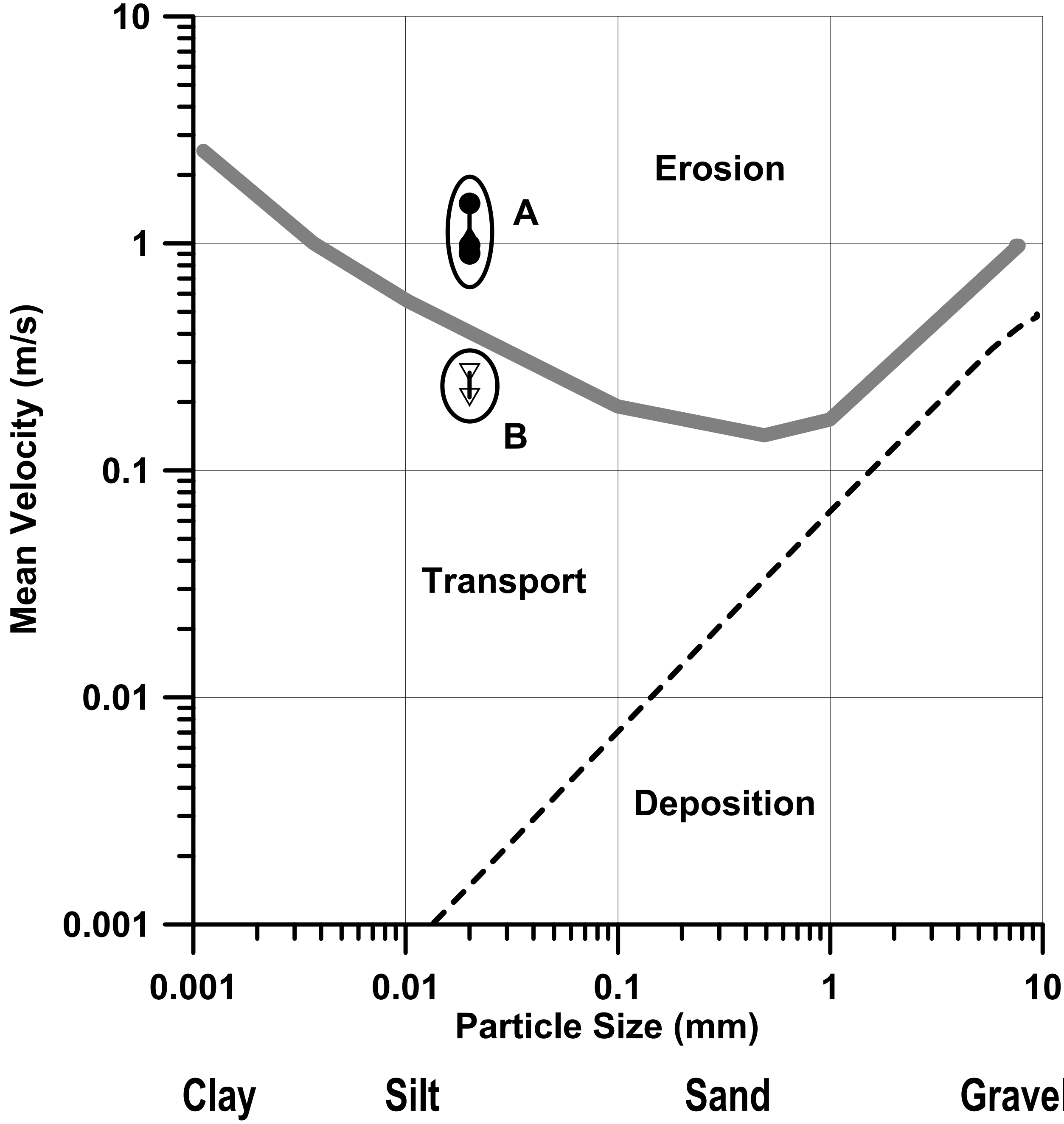


Figure 71. A Hulstrom plot showing the relationship to particle size transport as a function of stream velocity. The stream velocities measured during a storm event in the channel of Murphy Creek (A) are contrasted with the measured velocity in the over-bank region (B).

on the lake, approximately one-half have deficient septic systems. There is a presumption that improving the septic systems adjacent to Tenmile Lake will reduce nutrient loads to the lake. If approximately 50% of the septic systems are either inadequately designed, constructed, or maintained, and addressing these systems will reduce the nutrient output by half, then we could expect a 25% overall reduction in nutrient inputs from these sources upon full implementation of the program. These issues of nutrient loading need to be integrated with other lake and watershed concerns to address the following topics.

1. Reduction of Watershed Nutrient Sources
   a. Elimination of livestock grazing adjacent to lake
   b. Modification of shoreline development to reduce erosion
   c. Enforcement of timber harvest practices
   d. Implementation of improved septic systems and septic system maintenance

2. Reduction of Nutrient and Sediment Transport
   a. Restoration of original stream channel morphometry to minimize channelization
   b. Restoration of wetlands, particularly in areas adjacent to lake
   c. Restoration of shoreline vegetation

3. In-Lake Nutrient Management
   a. Continue activities to restore salmonid spawning and rearing habitat in watershed
   b. Evaluate methods for reducing populations of exotic fish species

4. Management and Monitoring
   a. Implement hydrologic monitoring network
      (i) Precipitation gages
      (ii) Streamflow gages (major inlets + Tenmile Lake and Tenmile Creek)
      (iii) Lake stage(s)
   b. Implement a long-term water quality monitoring program
      (i) Streams
         - Nutrients
      (ii) Lakes
         - Nutrients
         - Phytoplankton/Zooplankton
         - In situ profile chemistry

Implementation of such a plan would institutionalize activities needed for lake recovery and increase the level of accountability in meeting the long-term objectives of lake rehabilitation. A lake management plan will eventually be developed as part of the TMDL process for Tenmile Lake, although we are unaware of any legal barriers preventing local units of government from proceeding with such a plan immediately.

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