Publication No. 91-e05

DIAGNOSTIC STUDY OF LAKE SAWYER

KING COUNTY, WASHINGTON

February 1989 through
March 1990

by

J. V. Carroll
G. J. Pelletier

Washington State Department of Ecology
Environmental Investigations and Laboratory Services Program
Surface Waters Investigations Section
Olympia, Washington 98504-6814

Water Body No. WA-09-9260
(Segment No. 04-09-06)

March 1991

ACKNOWLEDGEMENTS

Thanks to the Lake Sawyer Technical Advisory Committee (and Associated Members): especially Myron Saikewicz (Ecology), Joanne Davis (METRO), Bill Lee (City of Black Diamond), Cecil Carroll (EPA), Larry Kirchner (Seattle-King County Health Department), Don Finney (Muckleshoot Tribe) and Kim McKee (Ecology).

Special thanks to Mr. Bob Eaton and his wife Emma, Mr. Harold Sovie and Mr. Carl Falk who provided, through exceptional effort, invaluable lake stage data and weather data throughout the project.

Also, thanks to the following: Bill Kombol of Palmer Coking Coal Co., Keith Olson, John Lacross and John Stevens from the City of Black Diamond, and Carl Steiert of the Black Diamond Historical Museum.

Special thanks to the following for sample analyses:

Don Nichols and crew at Eastern Washington University's Turnbull Laboratory
Ecology's Manchester Laboratory
Jim Sweet of Aquatic Analysts
Arni Litt at Department of Zoology, University of Washington
Kathy Krugsland at Marine Chemistry Lab at University of Washington, Seattle
Eric Crecelius at Battelle Laboratory, Sequim, WA

Thanks to the following:

Ecology personnel in the field: Cyd Brower, Barb Carey, Laura Chern, Marcos Gorresen, Chipper Holler, Jory Oppenheimer, Rob Plotnikoff and Roger Willms.

Ecology personnel for data analysis and graphical presentation: Jim Cubbage, Chipper Holler and Roger Willms.

Ecology personnel who reviewed draft reports: Joe Joy and Dave Hallock.

Ecology personnel who typed and proofed manuscript: Kelly Carruth and Barbara Tovrea

TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

Page

TABLE OF CONTENTS (Continued)

This page is purposely left blank for duplex printing.

LIST OF TABLES

Page

LIST OF FIGURES

LIST OF FIGURES (Continued)

Page

This page is purposely left blank for duplex printing.

DIAGNOSTIC STUDY OF LAKE SAWYER
Executive Summary

Lake Sawyer is the fourth largest natural lake in King County, Washington. It is heavily used for recreational purposes and supports an active fishery. The shoreline is more than 85 percent developed with single-family residential homes. Public access and a boat launch is provided to the lake by Lake Sawyer Park, located on the central-western shore, operated by King County.

The city of Black Diamond in King County presently operates a wastewater treatment plant (WTP) which discharges to a natural wetland. Discharge of effluent to the marsh was designed to utilize natural processes in order to remove nutrients (nitrogen and phosphorus). The wetland drains into Rock Creek, which in turn discharges to Lake Sawyer. The WTP had been determined to have failed to meet design removals of phosphorus. Diversion of the WTP discharge from the Rock Creek system is currently planned.

Recent changes in both the Rock Creek watershed and shoreline development, and suspected increase in frequency of algae blooms have caused concern for the future water quality of Lake Sawyer. The main objective of this diagnostic study was to perform a comprehensive baseline water quality assessment, including an assessment of all nutrient inputs and their relative contributions to the trophic condition of Lake Sawyer. The relative magnitude of nutrient loading from various sources was quantified through development of a detailed nutrient budget for a study year and for a typical hydrologic year.

Ecology collected data for the present study from February 1989 through March 1990. Lake and watershed sampling occurred at three-week to monthly intervals during this period. Hydrologic monitoring of surface waters was continuously performed by water level probes and dataloggers. Chemical and biological samples were taken during each survey. Single surveys were used to collect and characterize lake sediments, benthic macroinvertebrates, and aquatic macrophytes. Additionally, the Department of Ecology retained Hart Crowser, Inc. to perform a hydrogeologic study of Lake Sawyer to estimate groundwater inflow/outflow and nutrient loadings, incorporating an assessment of the nutrient inputs to Lake Sawyer from nearshore septic systems. Their report, Lake Sawyer Hydrogeologic Study - Black Diamond, Washington, was finished in October 1990.

The water quality assessment revealed that algal biomass in Lake Sawyer is strongly correlated to both nitrogen and phosphorus. However, phosphorus is generally more limiting than nitrogen at the present time. Phosphorus is expected to be even more important as a controlling nutrient after WTP diversion. Hypolimnetic dissolved oxygen depletion occurs during stratification due to the oxidation of organic and inorganic matter in the water column and sediment oxygen demand. During these anoxic conditions within the hypolimnion of Lake Sawyer, phosphorus is released from the sediments.

Internal loads of total P and total N account for less than 20 percent of the total nutrient budgets. Internal recycling of nutrients probably is dominated by sediment feedback in the hypolimnion during anoxia. The present rate of hypolimnetic oxygen depletion is about the same as that observed prior to WTP installation in 1982. Therefore, hypolimnetic anoxia, and internal cycling of P and N, is not expected to change because of WTP diversion. The Black Diamond WTP presently contributes about half of the external total P load and 12 percent of the external total N load to Lake Sawyer. Septic systems treating wastewater from nearshore homes are estimated to contribute only about 1 percent of the total phosphorus and nitrogen loads to the lake. The volume-weighted whole-lake TP concentration for the study year (March through February) was 25.7 $\pm$ 3.1 μg P/L. Volume-weighted epilimnetic chlorophyll *a* concentration during the 1989 growing season (March through October) was 6.2 $\pm$ 1.5 μg/L. The present trophic state of Lake Sawyer is predominantly mesotrophic with approximately an 18 percent expectation of eutrophic conditions. The diversion of Black Diamond WTP effluent from Rock Creek and Lake Sawyer will result in decreased likelihood of eutrophic conditions within the lake. Without diversion, the lake would have experienced a significant increase in the likelihood of eutrophic characteristics and a predominance of undesirable conditions.

The lake is expected to respond fairly rapidly to changes in nutrient loading following diversion because of relatively rapid flushing and sedimentation rates. The lake is predicted to attain about 99 percent of the new steady state concentration within the first year after a change in the loading rate. Aquatic macrophytes are and will continue to be a nuisance in the shallower areas of Lake Sawyer and potentially could worsen in response to decreased algae and increased light penetration within the lake. The color of lake water due to naturally occurring dissolved humic substances, however, results in a maximum possible Secchi disk transparency of less than seven meters, which is relatively low compared to many oligotrophic lakes.

The equilibrium condition of the lake after WTP diversion is expected to include about a five percent likelihood of eutrophic conditions because of loading sources other than the WTP. This may represent the upper limit of acceptable risk. Any increases in nonpoint loading which may occur in the future with increased watershed development may result in an unacceptable likelihood of eutrophic characteristics.

The total P load to Lake Sawyer from Rock Creek prior to WTP installation was probably lower than the present load, but higher than the load expected following WTP diversion. The difference between loads before WTP installation and following WTP diversion may be explained by total P contributions from failing treatment systems in use before 1982. Rock Creek has experienced development within its watershed, responding with increased stormwater flow and deteriorating background water quality. Future development in the Rock Creek catchment could result in higher future nutrient loads than those projected following the Black Diamond WTP diversion. Ravensdale Creek supplies the majority (57%) of hydrologic input to Lake Sawyer and is relatively pristine. The water quality impact on Lake Sawyer, should Ravensdale Creek deteriorate (e.g. to the point that Rock Creek's background water quality has), would be detrimental, counteracting present efforts and expenditures to preserve Lake Sawyer (i.e., WTP diversion).

Any future development within the basin should incorporate management practices that minimize nonpoint phosphorus inputs to the lake. If possible, total P loading should be maintained at a level no higher than the total external and internal P load following WTP diversion (715 $\pm$ 128 kg P/year) or a steady state in-lake mean total P concentration of 16 μg P/L.

This diagnostic study does not meet all the requirements of a Phase I Lake Restoration Diagnostic/Feasibility Study. This report fulfills the diagnostic study portion; however, a timely feasibility study would still need to be performed to satisfy Phase I requirements.

This page is purposely left blank for duplex printing.

1.0 INTRODUCTION

The City of Black Diamond in King County presently operates a wastewater treatment plant (WTP) consisting of a two-cell, aerated lagoon which discharges to a natural wetland. Discharge of effluent to the marsh was designed to utilize natural processes in order to remove nutrients (nitrogen and phosphorus). The wetland drains into Rock Creek, which in turn discharges to Lake Sawyer (Figure 1.1). Lake Sawyer is currently classified as mesotrophic to eutrophic (Brenner and Davis, 1988).

The natural wetland component of the WTP was considered innovative by the Environmental Protection Agency (EPA) prior to grant funding and construction in the early 1980s, and was subsequently determined to have failed to meet design removals of phosphorus (R.W. Beck, 1985; Environmental Resources Management, 1986; Pelletier and Joy, 1989). Increased loading of phosphorus to Lake Sawyer resulting from WTP discharges to the wetland system has been postulated to result in increased intensity of algal blooms. The EPA recommended that a waste load allocation (WLA) study be performed in order to determine the amount of phosphorus that must be removed by the Black Diamond WTP system in order to protect the water quality of Lake Sawyer.

The Washington State Department of Ecology completed a WLA in September 1989 (Pelletier and Joy, 1989). The principal objective of the WLA was to evaluate the influence of the Black Diamond WTP on the present and potential eutrophication of Lake Sawyer and recommend the WTP discharge option which maintains acceptable Lake Sawyer water quality. The WLA consisted of a comprehensive review of existing hydrologic and water quality data for Lake Sawyer and its tributaries.

The WLA found that total phosphorus concentrations in Lake Sawyer increased following start-up of the Black Diamond WTP in 1982. The observed increase in whole-lake TP content corresponded closely to the estimated loading discharged from the WTP. The condition of Lake Sawyer was predicted to reach a eutrophic state in the future (2010) if discharges from the Black Diamond WTP continued at the existing or currently permitted levels of treatment. Diversion of the WTP discharge from the Rock Creek/Lake Sawyer system would probably return the condition of Lake Sawyer to the mesotrophic (threshold eutrophic) condition that existed prior to the WTP start-up. Complete diversion of the WTP discharge to the Rock Creek/Lake Sawyer system was recommended as well as an in-lake TP criterion of 25 μg P/L for protection of Lake Sawyer water quality. Based on the recommendations of the WLA and advanced waste treatment review by EPA (EPA, 1989), the complete diversion of the Black Diamond WTP was authorized and plans to construct a tie-in to the METRO sewer collection system were begun.

In conjunction with the WLA, the Department of Ecology began this diagnostic study in February 1989. The major objective of this study was to perform a comprehensive baseline water quality assessment of Lake Sawyer, which included an assessment of all nutrient inputs and their relative contributions to the trophic condition of Lake Sawyer. The Lake Sawyer WLA was developed using existing databases that were not developed with the intention of constructing

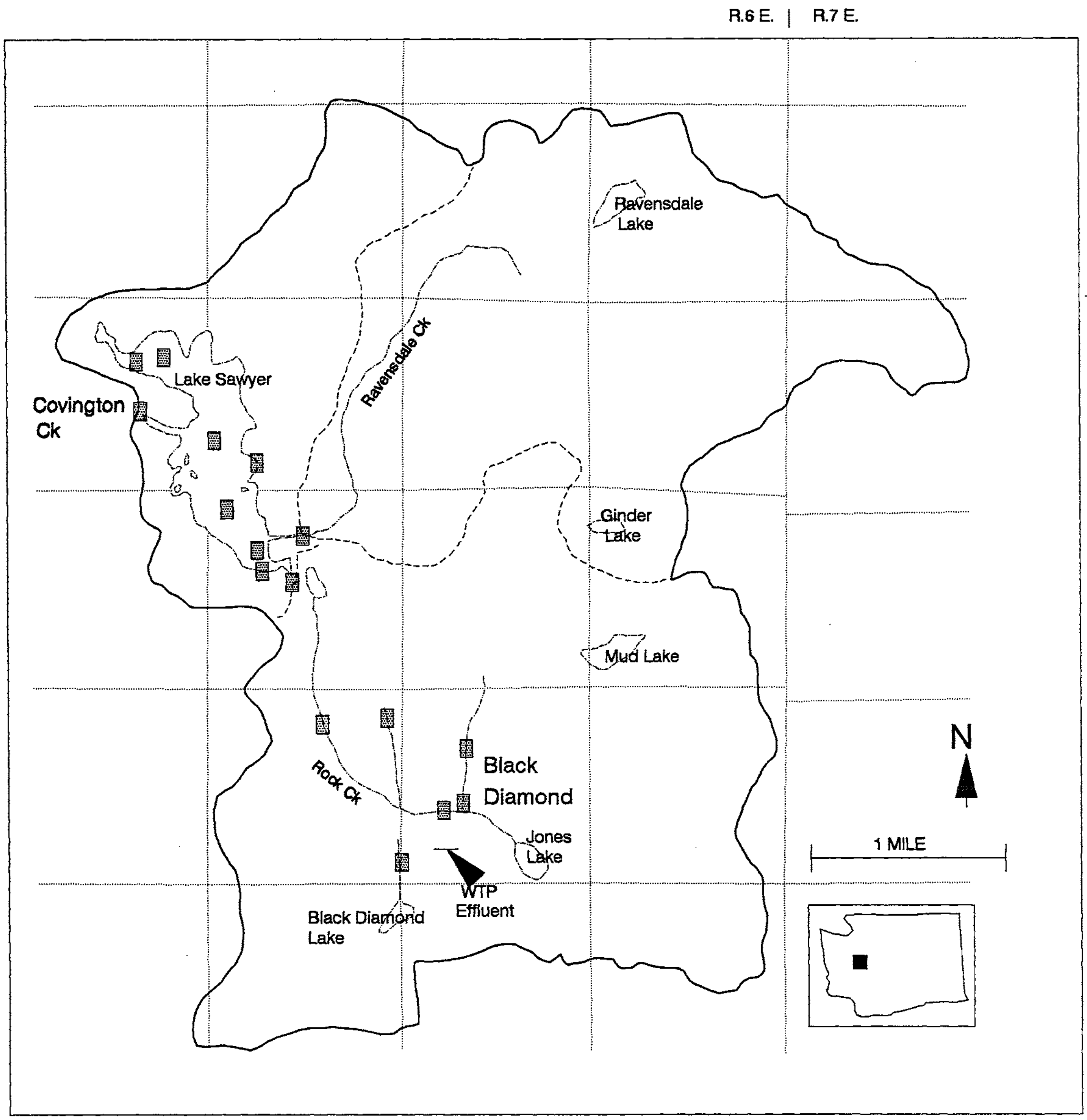


Figure 1.1. Vicinity map of Lake Sawyer showing major catchment boundaries for Ravensdale and Rock Creeks and Ecology monitoring stations.

nutrient mass balance estimates for the lake and wetland systems, therefore uncertainty surrounding the estimated loading of the WLA was rather high. This diagnostic study was designed to provide an extensive database for use in refining the WLA if it was decided that the Black Diamond WTP discharge represented an acceptable load to Lake Sawyer. Since the complete removal of the WTP discharge to Lake Sawyer is now imminent, predictions are specifically made in this report as to the response of Lake Sawyer to the WTP discharge diversion. Other sources of nutrient loading are also identified and evaluated.

Of importance in the Lake Sawyer nutrient mass balance was the determination of loading from nearshore septic systems and an evaluation of groundwater inflow and outflow components to Lake Sawyer. Hart Crowser, Inc. (with its principal subcontractors Entranco Engineers, Hokkaido Drilling and Developing, and Meredith Construction, Inc.) were retained by the Department of Ecology to conduct a hydrogeologic and on-site wastewater treatment study of Lake Sawyer, coinciding with Ecology's limnological investigation. Their report entitled Lake Sawyer Hydrogeologic Study - Black Diamond, Washington (Hart Crowser, 1990) was issued to the Department of Ecology in October 1990. Direct estimates of nutrient loading via the groundwater system were made with approximately 50 kg TN/year and 0.8 kg TP/year contributed from groundwater sources. In addition, nearshore septic systems were estimated to contribute 190 $\pm$ 90 kg TN/year and 9 $\pm$ 4 kg TP/year to Lake Sawyer.

This page is purposely left blank for duplex printing.

2.0 STUDY AREA AND BACKGROUND

2.1 Site Description

Lake Sawyer, the fourth largest natural lake in King County, lies within the Big Soos Creek sub-basin of the Green River drainage. Lake Sawyer's drainage basin covers about 34 km^2 (13 mi^2) at the outlet, Covington Creek (Figure 1.1). Various physical descriptions of the lake are listed in Table 2.1. The lake lies in an area of complex geology with low-permeable Vashon Till outcrops to the west, high-permeable recessional outwash drift to the east and north, and peat and muck areas (i.e., wetland areas) to the south (Hart Crowser, 1990; McConnell *et al.*, 1976). The 1973 bathymetric map shows the southern quarter of the lake to be shallower than the middle and northern areas (Figure 2.1). The two major inflow streams, Rock and Ravensdale Creeks, and an extensive wetland area enter at the southern end. The outlet, Covington Creek, leaves the lake from the central western shore. Lake level is controlled by a concrete dam at the outlet, which was constructed in 1952. Annual extremes of water level generally range about four feet from maximum to minimum level.

2.2 Public Access and Primary Uses

Besides having a heavily populated shoreline, public access is provided in one location on the central-western shore, north of the Covington Creek outlet canal. King County operates Lake Sawyer Park (2.7 acres) at this location. The park has picnic facilities, restrooms and a boat launch. Lake Sawyer is heavily used for recreational purposes. Recreational uses include fishing, boating, sailing, water skiing and swimming, as well as shoreside picnicing and aesthetic enjoyment. Angler use data is sketchy, but an estimated 20-30 anglers per day use Lake Sawyer from about April through October and 5-10 anglers per day use Lake Sawyer from November through March (Cropp, personal communication, 1990).[1] Most recreational activities were observed during the surveys by the investigators, particularly heavy boating activity during pleasant weather.

2.3 Population

Single-family residences and community open-space were estimated to occupy 85 percent of the Lake Sawyer shoreline in 1973 (McConnell *et al.*, 1976). Housing density along the shoreline has probably reached saturation as development has increased greatly. Aerial photographs from 1936 show less than 50 structures around the lake. The 1949 USGS topographical map of the area was drawn using 1943 aerial photos and shows approximately 160 structures around the lake. The 1973 photo-revised USGS map shows approximately 315 structures around Lake Sawyer. Tax assessment maps from 1987 show approximately 600 separate residential lots

---

[1] Cropp, T., 1990. Fish Biologist. WA Dept. of Wildlife.

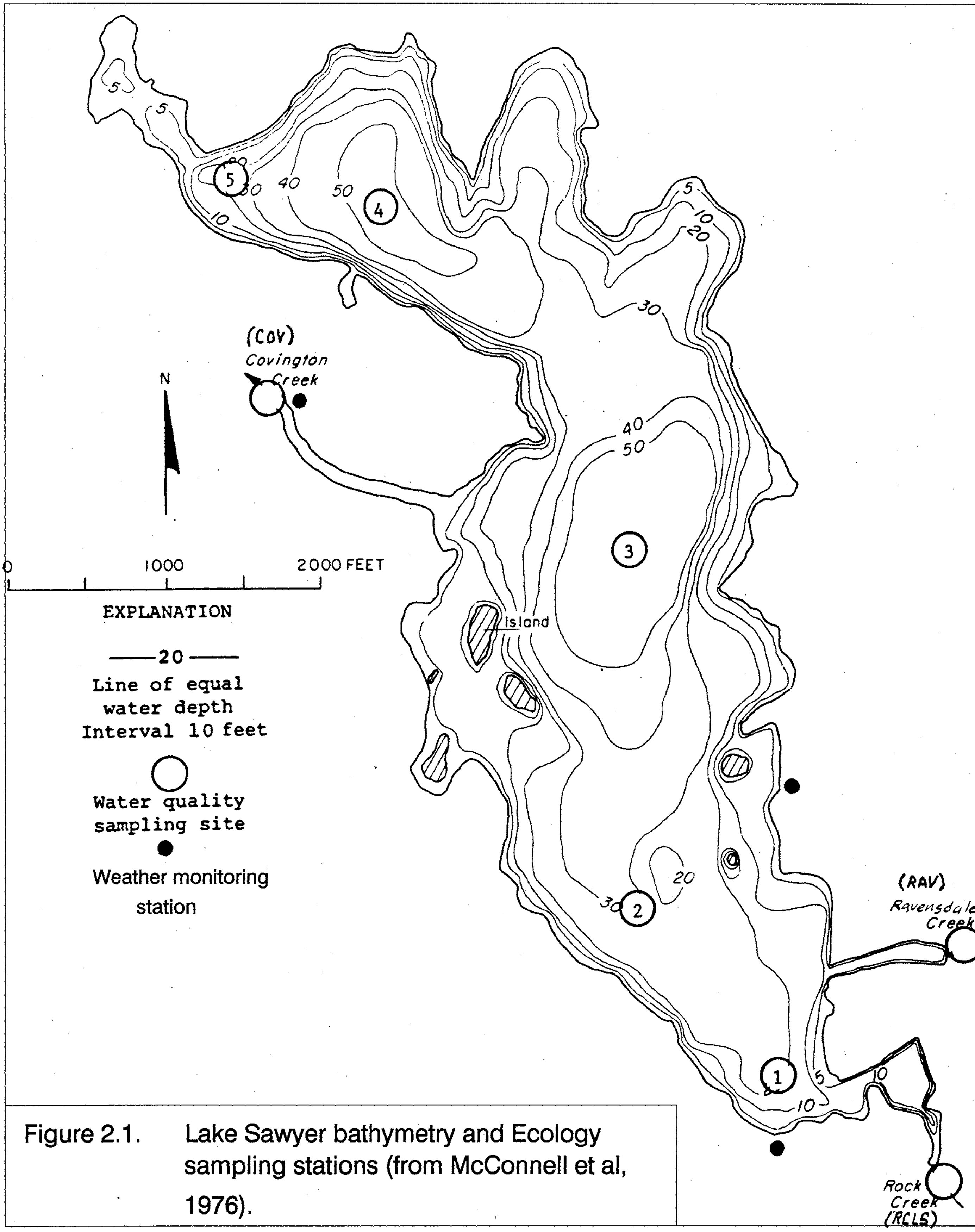


Figure 2.1. Lake Sawyer bathymetry and Ecology sampling stations (from McConnell et al, 1976).

Table 2.1. Physical characteristics of Lake Sawyer (after McConnell *et al.*, 1976, except as noted).

Parameter	English Units	Metric Units
Drainage area	13 sq.mi.	34 sq.km.
Altitude(1)	520 ft.	158 m.
Surface area(2)	280 acres	1.11 sq.km.
Lake volume(2)	7,000 ac.ft.	8.6 mill.cu.m.
Mean depth	25 ft.	7.62 m.
Maximum depth	58 ft.	18 m.
Shoreline length	36,000 ft.	11 km.

(1) Hart Crowser, 1990.
(2) Pelletier and Joy, 1989.

around Lake Sawyer.[2] Housing density and development are increasing in areas farther away from the lakeshore with areas east of the lake (i.e. the Ravensdale Creek drainage) showing the greatest potential for development.

The city of Black Diamond (population 1475; 1990 census) lies approximately one mile south of Lake Sawyer along Rock Creek (Figure 1.1). It is a residential community with a few commercial and institutional establishments and no major industrial development. However, the now inactive coal mines in and around the town were a major industry.

2.4 Land Use History

The land use history in the Lake Sawyer watershed is dominated by the once active coal mines in and around Black Diamond. The Town of Black Diamond was settled around 1884. A few early settlers pre-dated this settlement. In the early 1900s, the City of Black Diamond was a "boom town" supporting a large coal mining operation, with many high production coal shafts. The population in Black Diamond during this time period was larger than it is today. Coal mining has since declined, though the John Henry open-pit mine still operates up near Lake No. 12, and Ginder and Mud Lakes.

Coal mine shafts were structurally supported by wooden timbers, promoting a timber cutting industry in the area in the early 1900s. A sawmill located on the northwest end of Lake Sawyer operated from about the early 1900s until probably the 1930s. Most of the "old growth" timber in the watershed was probably cut during this early period. However, aerial photographs from 1936 show the majority of the Ravensdale Creek drainage devoid of vegetation, appearing to be a fairly recent clearcut. It is possible that fast-growing alder may have supplanted the "old growth" harvested in the early 1900s, and that the alder was cut in the mid 1930s. Alder, which is a softer hardwood, was used in the making of furniture.

2.5 Point and Nonpoint Sources of Pollution

Prior to 1983, wastewaters from the City of Black Diamond were treated by individual septic tank systems or by one of five community septic tank systems (KCM, 1979). All of the community septic systems and many individual systems had experienced failures during the 1970s, resulting in deterioration of Rock Creek and Ginder Creek water quality. Health concerns over sewage inputs to Rock Creek and Ginder Creek were one impetus for design and construction of an alternative wastewater treatment system. Ecology staff had documented water quality deterioration of Rock Creek and Ginder Creek (Thielen, 1978; Devitt, 1972). In 1982 a new WTP was put into service: a two-cell aerated lagoon using wetland dispersal for nutrient and solids removal.

[2] King County, Department of Assessments

The Black Diamond WTP discharges to a natural wetland that discharges to Rock Creek and then Lake Sawyer. The WTP effluent enters the wetland system approximately 1.8 miles upstream from Lake Sawyer. The wetland area surrounding the 0.9 mile of creek between the WTP line and the Morganville Bridge at river mile (RM) 0.9 is a complex convergence zone for several tributaries to Rock Creek: Black Diamond Lake Creek, Morganville marsh drainage, and Ginder Creek. The main trunk and its tributaries drain approximately 14.2 km^2 (5.5 mi^2) or 85 percent of the drainage basin by RM 0.9 (Walker and Veatch, 1964).

In addition to point source loading from the Black Diamond WTP, nonpoint sources also contribute to the total nutrient load to Lake Sawyer. The important nonpoint sources include background inputs from undeveloped lands (which can change depending on land-use activities), and diffuse inputs from developed land use, septic systems, and atmospheric deposition. The Lake Sawyer-Black Diamond WLA identified nonpoint sources as a possible significant component of total phosphorus TP load to Lake Sawyer (Pelletier and Joy, 1989). Any increases in nonpoint loading due to increased development in the watershed would be expected to result in corresponding increases in Lake Sawyer nutrient concentrations and acceleration of eutrophication.

2.6 Historical Data on Lake Sawyer Water Quality

A summary of the METRO, Ecology, private consultants, and USGS sampling locations, dates, parametric coverage, and depth intervals is given in the Lake Sawyer WLA (Pelletier and Joy, 1989). The data collected by METRO and USGS since the early 1970s indicate the lake usually undergoes stratification and hypolimnetic oxygen depletion in the central and northern basins in the late spring through mid-fall. The thermocline usually forms somewhere between 5 and 10 meters (16 to 33 feet). Dissolved oxygen concentrations in the hypolimnion fall rapidly to 1-2 mg/L between June and July. Hypolimnetic waters usually become anoxic by August. In 1973 the oxygen-depleted water included nearly half of the lake's volume.

METRO investigators have used chlorophyll *a*, total phosphorus, and transparency data to evaluate the trophic condition of lakes in King County (Brenner and Davis, 1988). Historically, Lake Sawyer has exhibited a primarily mesotrophic state according to these indicators. Summer chlorophyll *a* concentrations and transparency values have usually been in the oligotrophic and mesotrophic area of METRO's classification system. Winter phosphorus concentrations have usually indicated eutrophic or mesotrophic tendencies. A trend toward improving or declining water quality is not readily apparent from the METRO summaries.

Recent changes in both the Rock Creek watershed and shoreline development, and the increased frequency of resident complaints of late-fall blue-green algae blooms have caused concern for the future water quality of the lake (Brenner and Davis, 1988). The apparent failure of the Black Diamond WTP to remove phosphorus as designed is certainly one major element of concern. Another is the increase in housing units around the lake shore. All of the homes around the lake are on septic tank and drainfield systems. There is no on-site system maintenance program for the lake shore residents, so the age and condition of the systems are not well documented.

2.7 Tributary Water Quality

Some historical water quality data are available for Ravensdale and Rock Creeks. The data have been collected as part of the work with the Black Diamond WTP, therefore, far more data are available for the Rock Creek watershed than the Ravensdale Creek watershed. A summary of the data is given in the Lake Sawyer WLA (Pelletier and Joy, 1989). The Rock Creek data are from samples collected at least 0.9 mile upstream of the lake at the Morganville Bridge. Changes in water quality of Rock Creek between the Morganville Bridge and the inlet to Lake Sawyer were not known. Rock and Ravensdale Creeks, as tributaries to Lake Sawyer, must meet Class AA water quality standards (WAC 173-201-070), which are described in Table 2.2.

Pre-Black Diamond WTP tributary water quality samples were collected by KCM (KCM, 1982) from Rock, Ravensdale, and Ginder Creeks from 1980 to 1982. Additional water quality data are available for Rock Creek after the WTP start-up. Post-1982 data have been collected at the Morganville Bridge (RM 0.9) as part of the NPDES permit requirement. A gage was also installed to monitor flow. Unfortunately, flow data collected at the monitoring station became unreliable some time after 1983 (R.W. Beck, 1985). However, in-stream phosphorus concentrations showed a dramatic change in 1983 when the Black Diamond WTP went into operation.

Table 2.2. Class AA (extraordinary) and Lake Class freshwater quality standards and characteristic uses (WAC 173-201-045).

	CLASS AA	LAKE CLASS
Characteristic uses:	Shall include, but not be limited to, the following: domestic, industrial, and agricultural water supply; stock watering; salmonid and other fish migration, rearing, spawning, and harvesting; wildlife habitat; general recreation and aesthetic enjoyment; and commerce and navigation.	Same as AA
Water Quality Criteria		
Fecal Coliform:	Shall not exceed a geometric mean value of 50 organisms/100 mL, with not more than 10 percent of samples exceeding 100 organisms/100 mL.	Same as AA
Dissolved Oxygen:	Shall exceed 9.5 mg/L.	No measurable decrease from natural conditions.
Total Dissolved Gas:	Shall not exceed 110 percent saturation.	Same as AA
Temperature:	Shall not exceed 16.0 °C due to human activities. When natural conditions exceed 16 °C, no temperature increase will be allowed which will raise the receiving water temperature by greater than 0.3 °C. Increases from non-point sources shall not exceed 2.8 °C with a maximum of 16.3 °C.	No measurable change from natural conditions.
pH:	Shall be within the range of 6.5 to 8.5 with a man-caused variation within a range of less than 0.2 units.	No measurable change from natural conditions.
Turbidity:	Shall not exceed 5 NTU over background turbidity when the background turbidity is 50 NTU or less, or have more than a 10 percent increase in turbidity when the background turbidity is more than 50 NTU.	Shall not exceed 5 NTU over background turbidity.
Toxic, Radioactive, or Deleterious Material:	Shall be below concentrations which may adversely affect characteristic water uses, cause acute or chronic conditions to aquatic biota, or adversely affect public health.	Same as AA
Aesthetic Values:	Shall not be impaired by the presence of materials or their effects, excluding those of natural origin, which offend the senses of sight, smell, touch, or taste.	Same as AA

This page is purposely left blank for duplex printing.

3.0 STUDY METHODS

3.1 Hydrologic Monitoring

The water budget for Lake Sawyer can be described by inflows to the lake, outflows from the lake, and changes in lake storage. Rock Creek and Ravensdale Creek are the two surface water inflows at the south end of the lake. Other inflows include precipitation onto the lake surface and groundwater movement into the lake basin. Covington Creek is the surface water outflow at the central west side of Lake Sawyer. Other losses of water include evaporation from the lake surface and groundwater movement from the lake basin. The water budget for Lake Sawyer is described by:

$$\frac{dV}{dt} = P + S_{rcls} + S_{rav} \pm G_{ni} - S_{cov} - E \qquad \text{(eqn 3.1)}$$

where,

dV/dt	=	change in lake storage
P	=	precipitation
S_{rcls}	=	Rock Creek inflow
S_{rav}	=	Ravensdale Creek inflow
G_{ni}	=	net groundwater inflow/outflow
S_{cov}	=	Covington Creek outflow
E	=	evaporation

Surface Water Inflows and Outflows. Surface water inflows and outflows used in calculating the water budget for Lake Sawyer were primarily determined using stage-discharge relationships at near-lake locations. Rock Creek's contribution to Lake Sawyer was calculated using an upstream stage-discharge relationship (Rock Creek at Morganville Bridge) adjusted for changes enroute to lake (i.e. accounting for precipitation, evaporation, and storage change in the 9.3 acre wetland between Morganville Bridge and Lake Sawyer). Continuously monitoring electronic water level probes with dataloggers (capacitive water level probes and Starlog portable data loggers from Unidata America, Lake Oswego, OR) were placed instream at Covington Creek (just below the Lake Sawyer Dam), Ravensdale Creek (approximately 600 feet above the Lake Sawyer inflow) and Rock Creek wetland (just above the culvert to Lake Sawyer) (Figure 2.1). Water level probes and dataloggers were also installed at Rock Creek at Morganville Bridge gaging station, Rock Creek at Abrams Road, and the Black Diamond WTP effluent weir (Figure 3.1). These monitoring stations averaged and logged water levels at 15 minute intervals by scanning water levels every 5 seconds. Discreet discharge measurements were made every three to four weeks with extra observations made during several high flow events. Discharges were calculated using measured velocities from a Marsh-McBirney Model 201 velocity meter and measurements of the cross-sectional area of flowing water. Stage-discharge regression relationships were developed for each station using the measured flows and associated water levels to develop rating curves.

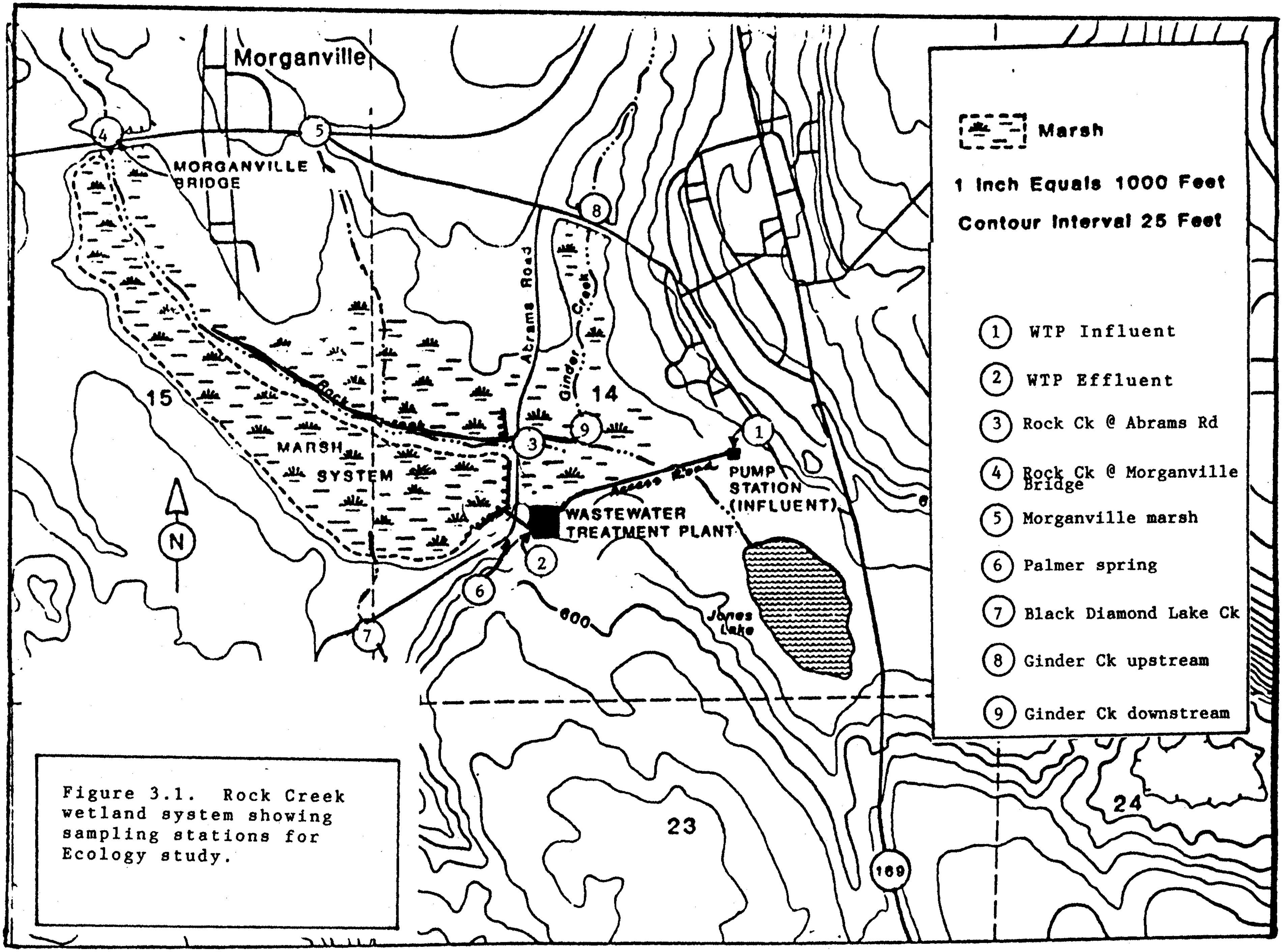


Figure 3.1. Rock Creek wetland system showing sampling stations for Ecology study.

The complete water level data file for each station was read into a BASIC computer program which used the regression equation from the applicable rating curve to calculate a discharge measurement for each 15 minute water level reading. Fifteen-minute flows were averaged to calculate average daily flows. Monthly flows were averaged from the daily flows.

The variability in the rating curves is composed of water level probe drift and resolution error, discharge measurement error, and stage-discharge hydraulic inconsistency. The standard error coefficient of variation was computed for each rating curve by comparing calculated and observed flows. The coefficient of variation was used to estimate the error of the mean flows. During the course of the study, but primarily during initial setup, equipment downtime and other problems resulted in incomplete water level records for some stations. In these cases, average daily flows were determined by either correlation to average daily flow of another station or by interpolation between flows when the missing period was short and when flows were relatively stable.

Precipitation. Precipitation was measured at three sites around the lake (Figure 2.1). Precipitation was recorded hourly at one site throughout the study period by a tipping-bucket rain gage with a datalogger. Two lakeshore resident volunteers also collected and measured daily precipitation. An average of the two lakeside volunteer stations and the tipping-bucket datalogger station was used to estimate rainfall.

Evaporation. Evaporation was calculated by using the modified mass transfer equation approach (Harbeck, 1962) outlined by Linsley *et al.*, (1975). A self contained weather station at the lakeshore recorded and logged air temperature, wind speed, and relative humidity every hour over the course of the study period. Calculated evaporation was compared to available pan evaporation measurements from a nearby Puyallup weather station (within 20 miles) and an average of the two was used in the lake evaporation determination. Pan evaporation was corrected using a pan coefficient of 0.70 (Dunne and Leopold, 1978) to estimate lake evaporation.

Storage Change. Lake stage on Lake Sawyer was recorded daily by two lakeside resident volunteers throughout the study period. Lake stage change was converted to storage volume change by multiplying the stage change by lake area (Table 2.1).

Groundwater Inflow and Outflow. Because of the difficulty of directly measuring groundwater flow through lakes, groundwater contribution and other minor contributions to the Lake Sawyer water budget were calculated as the residual in the water budget equation, as all other significant components were estimated directly. To confirm the water budget estimate of groundwater influence, direct calculations of groundwater flow were made from a flow net analysis of groundwater contours in the watershed and estimates of hydraulic conductivity and gradients near the lake. The hydrogeological influences to Lake Sawyer were assessed in an independent effort by Hart Crowser, Inc. (Hart Crowser, 1990).

3.2 Limnological and Water Quality Investigations

3.2.1 Limnological Monitoring

Lake Water Sampling. The water column of Lake Sawyer was sampled at five stations in the lake (see Figure 2.1), which were used to assess within lake distributions and variability. A systematic sampling design approach was used throughout the study period with a sampling frequency of every three weeks from February 1989 through November 1989 and every four weeks from December 1989 through April 1990.

Table 3.1 presents a summary of chemical parameters sampled in the lake. Vertical water column profiles of temperature, pH, dissolved oxygen, and conductivity (and occasionally oxidation reduction potential) were measured at one meter intervals from surface to bottom at all five stations using a HydroLab Surveyor II multiple sensor meter from a boat. Grab samples were taken using a four liter Van Dorn-type bottle sampler from a boat. Vertical profile characterizations of total phosphorus and total nitrogen were made at three meter intervals from the surface to bottom at all stations. Epilimnetic chlorophyll *a* samples were taken from the surface, three meter, and six meter depths at each station. Epilimnetic total soluble phosphorus, total soluble nitrogen and turbidity samples were taken at the surface, three meter, and six meter depths at Stations 3 and 4. Alkalinity, soluble reactive phosphorus, ammonia-nitrogen, and nitrate plus nitrite-nitrogen samples were also collected at three meter intervals from top to bottom at Stations 3 and 4.

Samples were placed into pre-cleaned bottles in accordance with cleaning protocols for the type of constituent to be analyzed (Huntamer, 1986). Meters used in the field were calibrated using reagent grade calibration standards. The dissolved oxygen probe was calibrated using modified-Winkler titration dissolved oxygen determinations.

Other lake water physical and chemical characteristics measured at each sampling event were Secchi disk transparencies at every station, total recoverable metals (aluminum and iron), and total organic carbon from Stations 3 and 4 at the six and 15 meter depth interval. Chloride was measured at inconsistent time intervals. Light transmission was measured at one meter intervals from surface to bottom at Station 3 using an underwater photometer.

Biological characteristics measured with regularity included chlorophyll *a*, fecal coliform, fecal streptococcus, phytoplankton, and zooplankton. Epilimnetic chlorophyll *a* was sampled during every lake sampling event at the 0, 3, and 6 meters depth intervals at each lake station. Fecal bacteria were sampled from the surface waters of Stations 3 and 4 during each sampling event. Phytoplankton were sampled every lake sampling at the 0, 3, and 6 meters depth intervals of Stations 3 and 4. Composite phytoplankton samples were made from each respective station and preserved in a dilute Lugol's solution (APHA *et al.*, 1985). Typically, zooplankton were collected in 6 meter (9 meter after October 23, 1989) vertical hauls at Stations 3 and 4 during

Table 3.1. Summary of chemical parameters sampled at in-lake water quality stations. [a]

Depth (m)	Sampling Station 1	2	3	4	5	Outlet
0.5	1	1	2	2	1	--
3	1	1	3	3	1	
6	1	1	4	4	1	
9			5	5		
12			5	5		
15			6	6		

Sampling parameter codes for laboratory analyses:

1) Total P, total N, and chlorophyll a

2) core nutrients (total P, soluble reactive P, total N, nitrate + nitrite N, ammonia N) plus total soluble P, total soluble N, chlorophyll a, phytoplankton biovolume (composite 0,3 and 6 m), alkalinity, turbidity, fecal coliform and strepticocci, and chloride.

3) core nutrients plus total soluble P, total soluble N, chlorophyll a, phytoplankton biovolume (composite 0,3 and 6 m), alkalinity, and turbidity.

4) core nutrients plus total soluble P, total soluble N, chlorophyll a, phytoplankton biovolume (composite 0,3 and 6 m), alkalinity, turbidity, metals (Al and Fe), and total organic carbon.

5) core nutrients plus alkalinity.

6) core nutrients, alkalinity, metals (Al and Fe), and total organic carbon.

[a] (DO, temperature, pH, and conductivity measured at 1 meter intervals, top to bottom, at all stations).

each sampling event using a Wisconsin plankton net with an 11.5 cm net opening and an 80 μm mesh size. The zooplankton samples were immediately preserved with a 70 percent ethanol solution.

Watershed Sampling. The Lake Sawyer watershed was sampled with the same sampling frequency as the lake (i.e. the day following the lake sampling). The watershed sampling sites included the Lake Sawyer surface water inflows (Ravensdale Creek and Rock Creek) and outflow (Covington Creek) at near-lake locations (Figure 2.1). The remaining watershed sampling sites focused on the extensive wetlands system and tributaries in the Rock Creek drainage including the Black Diamond WTP effluent discharge (see Figure 3.1). Grab samples were made from most of the watershed sites at least twice in one day to address field variability. Water quality determinations included total phosphorus, soluble reactive phosphorus, total nitrogen, nitrate plus nitrite-nitrogen, ammonia-nitrogen, total organic carbon, chloride, fecal coliform, and fecal streptococci. During each survey, a 24-hour composite sample was taken from the Black Diamond WTP lagoon effluent weir and on several occasions from the influent pump station tank resevoir. Temperature, pH, dissolved oxygen, and conductivity were also measured using field probes. Discharge measurements were made at each site as discussed in section 3.1.

3.2.2 Aquatic Macrophytes

Macrophyte samples from the littoral areas of Lake Sawyer were collected in a single survey on October 17, 1989, to characterize species composition and the total phosphorus and total nitrogen content in the standing crop. Macrophyte surveys by METRO (1976-1980) indicated light to moderate coverage to a depth of approximately three meters, which corresponded roughly to the Secchi disk visibility. Ten randomly selected sites were established along the length of the lake shore (Appendix E). At each site a sampling depth from zero to three meters was randomly determined. The established random depth at each site was sampled with a macrophyte rake as described by Gibbons (1986). The macrophyte rake used sampled a one square foot area. A net was used to collect dislodged, free-floating plant material. Samples were put into plastic bags and kept on ice until sorting. Macrophyte samples were rinsed and separated by species using the aquatic plant keys of Prescott (1980) and Hitchcock and Cronquist (1981). Each species at each station was analyzed for total biomass. Wet weights were determined after rinsing with distilled water and spinning in a lettuce rinser. Samples were dried at 105°C until constant weight was attained. Dried plant material from all stations was then ground and homogenized by species. Aliquots by species were analyzed for total nitrogen by the persulfate digestion and cadmium reduction method (D'Elia *et al.*, 1977) and also ashed in an acid-washed crucible at 550°C for 30 minutes for ash-free dry weight determinations. The crucibles were rinsed with deionized water and analyzed by species for total phosphorus by the same persulfate digestion, ascorbic acid method used for water samples. Duplicates, matrix spikes, and certified external reference material were run. Data for aquatic macrophytes are presented in Appendix E.

3.2.3 Benthic Macroinvertebrates

Benthic macroinvertebrates were sampled during a single survey on June 27, 1990. Weather for the survey was warm and sunny. A water column profile of temperature, pH, dissolved oxygen, and conductivity was made at mid-lake Station 3. Two transects were sampled for benthic macroinvertebrates, originating in the basins of Stations 3 and 4 and running eastward towards the lakeshore (Appendix D). Each transect was sampled at the deepest part of the basin, a moderate depth of 10.5 meters, and a more shallow depth of 4.5 meters. Replicate samples were taken along the Station 3 transect. Samples were taken with a Petite Ponar dredge which samples a 0.0232 square meter area. Samples were washed through a 600 μm mesh screened bucket on site to remove most of the fine sediments. The samples were immediately preserved in a 70 percent ethanol solution. The samples were sorted by first floating the organic component of the samples with a super-saturated sugar/water solution. The inorganic fraction was spot-checked before discarding. Aquatic insects were then hand-picked from the organic fraction under a low power dissecting scope. Insects were sorted by family and then keyed to sub-family and genus (sometimes species) using the keys of Pennak (1978), Merritt and Cummins (1984) and Mason (1968).

3.2.4 Sediment Analysis

Two methods were employed to collect sediments from Lake Sawyer. Sediment cores were taken from the bottom profundal sediments at Stations 3 and 4 during a single survey, and sediment dredge samples were taken from the profundal sediments of Stations 3 and 4 during a single survey.

Sediment Core Sampling. On April 10, 1990, four sediment cores were taken from the two deep basins of Lake Sawyer (approximately Stations 3 and 4). Cores were taken with a four foot length, two inch diameter piston corer device with a polycarbonate liner. Following collection, the cores were kept vertical within their liners and on ice until slicing which took place within forty-eight hours. Sixty 1 cm increment slices comprising the top sixty centimeters of lake sediment were extruded from each core. Sliced samples were freeze dried until constant weight was attained and saved for chemical analysis. Table 3.2 presents the summary of analyses which were performed on different intervals from the cores. Primarily, a vertical profile of radioisotope Pb-210 and phosphorus was used to examine the current and historical rates of phosphorus sedimentation. In addition, total nitrogen and total organic carbon were analyzed to derive their respective sedimentation accumulation rates. Stable lead (Pb) was analyzed in selected intervals of the core as was sediment diatom remains.

Sediment Core Laboratory Analyses. Sediment samples were sent to Battelle Northwest's Marine Sciences Laboratory in Sequim, Washington for freeze drying and Pb-210 determination. An aliquot of sample was analyzed for Pb-210 using the method described by Carpenter *et al.* (1981, 1982). The methodology consists of successive digestions of dried, homogenous sediments with concentrated acids which release the Pb-210 decay product, Po-210. The Po-210 is allowed to spontaneously plate onto silver discs and is then determined by alpha spectrometry.

Table 3.2. Summary of Lake Sawyer sediment core analyses.

Lake Sawyer sediment core - Station 3

core interval (cm)	total solids	Pb-210	stable Pb	CHN	TP
0-1	x	x		x	x
1-2	x	x		x	x
2-3	x	x		x	x
3-4	x	x		x	x
4-5	x	x	x	x	x
5-6	x	x	x	x	x
6-7	x	x	x	x	x
7-8	x	x	x	x	x
8-9	x	x	x	x	x
9-10	x	x	x	x	x
10-11	x	x	x	x	x
11-12	x	x	x	x	x
12-13	x	x	x	x	x
13-14	x	x	x	x	x
14-15	x	x	x	x	x
15-16	x	x	x	x	x
16-17	x	x	x	x	x
17-18	x	x	x	x	x
18-19	x	x	x	x	x
19-20	x	x	x	x	x
20-21	x		x	x	x
21-22	x	x	x		
22-23	x		x	x	x
23-24	x	x	x		
24-25	x		x	x	x
25-26	x	x	x		
26-27	x		x	x	x
27-28	x	x	x		
28-29	x		x	x	x
29-30	x	x	x		
30-31	lost in lab				
31-32	x	x	x		
32-33	x		x	x	x
33-34	x	x	x		
34-35	lost in lab				
35-36	x	x	x		
36-37	x		x	x	x
37-38	x	x	x		
38-39	x		x	x	x
39-40	x	x	x		
40-41	x		x	x	x
41-42	x	x	x		
42-43	x		x		
43-44	x	x	x		
44-45	x		x		
45-46	x	x	x		
46-47	x				
47-48	x	x			
48-49	x				
49-50	x				
50-51	x				
51-52	x				
52-53	x				
53-54	x				
54-55	x			x	x
55-56	x				
56-57	x				
57-58	x				
58-59	x				
59-60	x	x			

Lake Sawyer sediment core - Station 4

core interval (cm)	total solids	Pb-210	CHN	TP	DIATOMS
0-1	x	x	x	x	
1-2	x	x	x	x	x
2-3	x	x	x	x	
3-4	x	x	x	x	x
4-5	x	x	x	x	
5-6	x	x	x	x	x
6-7	x	x	x	x	
7-8	x	x	x	x	x
8-9	x	x	x	x	
9-10	x	x	x	x	x
10-11	x	x	x	x	
11-12	x	x	x	x	x
12-13	x	x	x	x	
13-14	x	x	x	x	x
14-15	x	x	x	x	
15-16	x	x	x	x	x
16-17	x	x	x	x	
17-18	x	x	x	x	x
18-19	x	x	x	x	
19-20	x	x	x	x	x
20-21	x				
21-22	x	x	x	x	x
22-23	x				
23-24	x	x	x	x	x
24-25	x				
25-26	x	x	x	x	x
26-27	x				
27-28	x	x	x	x	x
28-29	x				
29-30	x	x	x	x	x
30-31	x				
31-32	x	x	x	x	x
32-33	x				
33-34	x	x	x	x	x
34-35	x				
35-36	x	x	x	x	x
36-37	x				
37-38	x	x	x	x	x
38-39	x				
39-40	x	x	x	x	x
40-41	x				
41-42	x	x			x
42-43	x				
43-44	x	x			x
44-45	x				
45-46	x	x			x
46-47	x				
47-48	x	x			x
48-49	x				
49-50	x				x
50-51	x				
51-52	x				x
52-53	x				
53-54	x				x
54-55	x				
55-56	x				x
56-57	x				
57-58	x				x
58-59	x				
59-60	x	x			x

A Po-208 spike in each sample serves as a chemical yield tracer. Using this technique, a detection limit of 0.1 dpm/g was attained. Method blanks and duplicate samples were run in conjunction with the samples. All method blank results were below the detection limit of 0.1 dpm/g. All duplicate sample results were within ten percent.

Sediment samples were sent to Turnbull Laboratory of Eastern Washington University for analysis of sediment total phosphorus. Sediment samples were redried, weighed and ashed in a crucible. Deionized water was used to rinse ashes from the crucible and then dilute to a known volume. From here the samples were run as water samples for total phosphorus (i.e. persulfate digestion and ascorbic acid method). Method blank results ranged from 2.4 μg/L to 3.6 μg/L. Matrix spike recoveries ranged from 98% to 120% recovery with an average recovery of 113%. Sample duplicates had good precision levels with pooled squared root of the mean squared deviation (RMSE) of 0.017 mg P/g dry weight. Certified citrus leaves with a true value of 1.3 $\pm$ 0.2 mg P/g were run as a reference material with the sediment samples. Three separate runs of the citrus leaves resulted in analysis values of 1.2, 1.3, and 1.3 mg P/g, all within the true value range.

Sediment samples were sent to the University of Washington's Marine Chemistry Lab for total nitrogen (TN) and total organic carbon (TOC) determinations by vapor acidification and elemental analysis with a Carlo Erba Model 1106 CHN micro-analyzer. Sample preparation, instrument methodology, and an evaluation of performance is described by Hedges and Stern (1984). Duplicate samples were run with each CHN analyzer batch. The TOC and TN duplicate results had RMSEs of 0.066% and 0.043% of dry weight respectively. In addition, an in-house, Lake Washington sediment reference sample was run. The TOC and TN results for this reference material were 4.52 $\pm$ 0.24% and 0.36 $\pm$ 0.01% of dry weight respectively, compared with expected values of 4.62 $\pm$ 0.09% and 0.38 $\pm$ 0.01% of dry weight, respectively.

Sediment samples were sent through Ecology's Manchester Laboratory to an outside laboratory contractor for analysis of total recoverable stable lead (Pb). All samples were digested using microwave digestion. An aliquot of sample was put into a microwave digestion bomb and approximate 10 mLs of nitric acid was added to the bomb. After digestion, the samples were filtered and then analyzed for total recoverable lead with a graphite furnace using SW-846 Method 7421 (EPA, 1986b). One method blank, four duplicates and an external reference check standard were also analyzed. Ecology's Manchester Laboratory reviewed the QA/QC data and found the data acceptable for use.

Sediment samples were also sent to Aquatic Analysts of Portland, Oregon, for species enumeration and identification of sediment diatom remains. A total of 31 dried samples from the Station 4 core were analyzed. An aliquot of 0.020 grams of each sample was weighed and diluted to 250 mL, then 10 mL of these samples were filtered and prepared into microscope slides. The results were reported in the number of diatom units (i.e. discrete cells, colonies, or filiments) per milligram of dry sediment. Similarity indices were used to compare sediment diatom samples. The similarity index compares the relative abundance of each species present in two samples and yields an index value ranging from zero for total dissimilarity between samples to 100 for completely identical samples (Sweet, 1986). This index was used to compare each successive increment in the core. The formula for the similarity index is:

Similarity Index = 100 - (sum of Difference / 2)   (eqn. 3.2)

where "Difference" is the absolute value of the difference of the percent of each species in the two samples.

An examination of the relative ratios of planktonic pennate and centrate diatoms through the core was done to assess possible past ecological conditions in Lake Sawyer. In general, a higher abundance of centrate diatoms is associated with oligotrophic conditions while a higher abundance of pennate diatoms is associated with eutrophic conditions (Wetzel, 1983).

Sedimentation Rates From Pb-210 Profiles. The method for using Pb-210 for calculating sedimentation rates was first described by Goldberg (1963). Pb-210 is one of the decay products of naturally occurring Ra-226, which proceeds through a decay chain as follows:

Ra-226 >> Rn-222 >> Po-218 >> Pb-214 >> Bi-214 >> Po-214 >>

>> Pb-210 >> Bi-210 >> Po-210 >> Pb-206

Radon gas is emitted to the atmosphere primarily from the lithosphere and there decays rapidly (half-life = 3.8 days) to Pb-210 via several short-lived radionuclides. Pb-210 in the atmosphere rapidly becomes attached to aerosol particles which reside in the atmosphere usually for less than one month. The residence time of Pb-210 in the atmosphere is very short relative to its half-life of 22.3 years. Atmospheric Pb-210 flux varies with latitude and longitude because of variations in land masses (the principle source of Ra-226). Local rates of atmospheric deposition have been reported by Nevissi (1985) and Carpenter *et al.* (1981).

Sedimentation in lakes is a continuous process with both sediment and Pb-210 deposition rates being constant for relatively long periods of time. Since the half-life of Pb-210 is 22.3 years, the Pb-210 concentration in sediments deposited 22.3 years ago is one-half of the Pb-210 concentration in most recent top sediment layers. By measuring the radioactivity of Pb-210 at successive sediment depths, the age of each sediment depth can be calculated and sedimentation rates thereby determined. Because Pb-210 is also supplied to the sediments from terrigenous sources (e.g. watershed erosion), this relatively small amount of "supported" Pb-210 must be subtracted from the total Pb-210 to calculate "unsupported" Pb-210 concentrations from atmospheric flux. Unsupported Pb-210 is used to calculate sedimentation rates.

Two different theoretical models were used for estimating sedimentation rates from profiles of unsupported Pb-210: the constant initial concentration (CIC) and constant rate of supply (CRS) models. Both models begin with an assumption of constant atmospheric flux of Pb-210 and a constant residence time in the lake water column (i.e. constant rate of flux to the sediments).

The CIC model makes an additional assumption that the initial concentration of Pb-210 in the sediments is constant. Sedimentation rates for consistently decreasing Pb-210 concentration can therefore be estimated using linear regression of the logarithm of Pb-210 activity as a function of sediment depth or cumulative areal dry mass (Robbins and Edgington, 1975; Pennington *et al.*, 1976; Krishnaswamy *et al.*, 1971). The latter is preferable, since it automatically accounts for sediment compaction (Birch, 1976). The CIC model is not theoretically suited for estimating sedimentation when rates are changing (Appleby and Oldfield, 1978), which is often the case over the recent history of lakes due to changes in watershed land use and point source inputs (e.g. Black Diamond WTP discharge to Lake Sawyer).

The alternative CRS model does not make the assumption of constant initial Pb-210 concentration. Instead, the rate of Pb-210 flux to the sediment is assumed to be constant and initial Pb-210 concentrations in top sediment layers may vary depending on changes in total solids deposition (e.g. due to increases in primary productivity). The CRS model may be summarized as follows:

$$A(X_2) = A(X_1)\, e^{-kt} \qquad \text{(eqn. 3.3)}$$

therefore: $$t = (1/k)\, \mathrm{Ln}\, [\, A(X_1)/A(X_2)\,] \qquad \text{(eqn. 3.4)}$$

where: $A(X_2)$ = integrated areal unsupported Pb-210 below depth X_2 (dpm/cm^2)

$A(X_1)$ = integrated areal unsupported Pb-210 below depth X_1 (dpm/cm^2)

k = 0.03114 for Pb-210 (year^{-1})

t = time interval for sediment deposition between sediment depths X_1 and X_2 (years).

In applications where sedimentation rates have undergone prolonged periods of increase, the CIC model will always underestimate the true age of sediment below the beginning of the increase. Therefore, the CRS model is generally preferable for applications which involve changing sedimentation rates.

Sediment Dredge Samples. Profundal sediment samples were taken from the basins of Station 3 and 4 during the benthic macroinvertebrate survey on June 27, 1990. Sediments were collected using a Petite Ponar dredge sampling a 0.0232 square meter area. Intact dredge samples were sampled through top screens of the dredge with a pre-cleaned, stainless-steel spoon. The top centimeter of slurry at the sediment/water interface was removed and then the next four centimeters were collected and put into pre-cleaned glass jars with teflon lids. Duplicate samples were collected from each dredging. Samples were submitted to Ecology's Manchester Laboratory for total recoverable aluminum and iron determinations using inductively coupled plasma atomic emissions analysis (Method EPA-200.7; EPA, 1983). Internal QA/QC procedures were followed by the laboratory (Huntamer, 1986).

3.2.5 Lake Bathymetry

Accurate measurements of area, volume, and mean depth were developed from an existing USGS bathymetric map (Wolcott, 1965). The bathymetric map (Figure 2.1) was digitized and the morphometric parameters calculated. The volume of strata (layers) between contours was calculated using the formula for a truncated cone (Wetzel, 1982; Welch, 1948):

$$V = h\,[\,a_1 + a_2 + (a_1 * a_2)^{0.5}\,]/3 \qquad \text{(eqn. 3.5)}$$

where: V = the volume of stratum from a_1 to a_2
h = depth of stratum from a_1 to a_2
a_1 = area of upper surface of stratum
a_2 = area of lower surface of stratum

3.2.6 Laboratory Analyses

Most water quality analyses were conducted by the Turnbull Laboratory of Ecological Studies under the direction of Robert L. Carr at the Eastern Washington University at Cheney. The major exception was the lake water metal determinations and total organic carbon (TOC) determinations which were conducted by the Department of Ecology Manchester Laboratory. Samples were kept on ice in an ice chest in the field and then brought back to Ecology in Olympia, Washington for filtering (chlorophyll *a*, soluble reactive phosphorus, total soluble phosphorus, ammonia-nitrogen, nitrate plus nitrite-nitrogen, and total soluble nitrogen samples). Samples were filtered within ten hours of collection and then repacked in ice and shipped via bus to the Turnbull Laboratory of Ecological Studies in Cheney, Washington for constituent analysis. Receipt of samples at Turnbull Laboratory was within 24-four hours of sampling. Immediate processing or preservation was performed at the time of receipt. Table 3.3 presents the methodology and lower reporting limits for each parameter analyzed. All methods except those noted are recommended in Standard Methods (APHA *et al.*, 1985) and EPA laboratory manuals (EPA, 1983). Enumeration and taxonomic identification of the phytoplankton were performed by Jim Sweet of Aquatic Analysts in Portland, Oregon. Zooplankton enumeration and identification were performed by Arni Litt at the University of Washington, Department of Zoology, in Seattle. Raw data for Lake Sawyer chemical data plus chlorophyll *a* and fecal bacteria are presented in Appendix A. Phytoplankton data are presented in Appendix B. Zooplankton data are presented in Appendix C.

3.2.7 Quality Assurance/Quality Control

Quality Assurance/Quality Control (QA/QC) programs are important in water quality studies in order to control or estimate the uncertainty involved in analytical estimations. All analytical measurements and results are estimates of actual true values in the field and are subject to

Table 3.3. Summary of analytical methods for Lake Sawyer in-lake and watershed water quality study.

Parameter	Method	Reference [1]	LRL [2]
Temperature	Thermistor	--	--
pH	Field Probe	SM423	--
Dissolved Oxygen	Field Probe	SM421F	--
Turbidity	Nephelometer	SM214A	0.3 NTUs
Sp. Conductance	Field Probe	SM205	--
Alkalinity	Titration	SM403	--
Soluble Reactive P	Ascorbic Acid	EPA 365.3	5 µg/L
Total P and total particulate P	Persulfate Digest, Ascorbic Acid	EPA 365.3	5 µg/L
Ammonia N	Phenate	SM417C[3] Solarzano, 1969	15 µg/L
Nitrate + Nitrite N	Cadmium Reduction	SM418C	5 µg/L
Total N and total particulate N	Persulfate Digest, Cadmium Reduction	D'Elia et al., 1977	50 µg/L
Chlorophyll a	Trichromatic	SM1002G	2 µg/L
Biovolume	Invert Scope	SM1002F	--
Fecal Coliform	Membrane Filter	SM909C	--
Fecal Streptococci	Membrane Filter	SM910B	--
Chloride	Mercuric Nitrate	SM407B	0.3 mg/L

(1) SM: Standard Methods for the Examination of Water And Wastewater, 16th ed., APHA-AWWA-WPCF, 1985.

EPA: Methods for chemical analysis of water and wastes. EPA 600/4-79-020, 1983.

D'Elia, C.F., P.A. Steadler, and N. Corwin, 1977. Determination of total N in aqueous samples using persulfate digestion. Limn. Ocean. 22:760-764.

Solarzano, L., 1969. The determination of ammonia in natural water by the phenolhyypochlorite method. Limnol. Oceanogr. 14(5):799-801.

(2) Lower Reporting Limit (LRL) defined as 3 times the pooled standard deviation (RMSE) of low level replicates.

(3) SM417C used through June 12, 1989 samples and Solarzano (1969) used throughout the rest of the study.

various random and systematic errors. Errors can be introduced in either the sampling or analytical procedures. Usually, laboratory uncertainty is a smaller percentage than field variability, but uncertainty in analytical estimations is easiest to control and quantify. Sample collection and laboratory analyses were conducted using standard methods as described above. These methods provide the best analytical estimate possible.

Analytical precision was determined using laboratory duplicates and blind split field duplicates. Generally, laboratory duplicate determinations were achieved on five percent of the total numbers for each batch of nutrient, chlorophyll *a* and conductivity determinations. In addition, approximately another five percent were submitted to the laboratory as blind split field duplicates. Precision was defined as one standard deviation of low-level sample duplicates. The low-level cutoff range and the standard deviation of the sample duplicates within this range are reported below for each parameter. The reported standard deviation is a pooled square root of the mean squared deviation (RMSE) for both laboratory and blind duplicates. Without exception, the blind duplicates were more variable than the laboratory duplicates. This is to be expected since field duplicates results provide an estimate of overall precision, including both sampling and analytical error.

Laboratory blank and blind blank determinations were analyzed routinely. An evaluation of laboratory blank responses was done to check for laboratory analytical error. Blind blanks, which included field and filter blanks, were also evaluated for field or filter contamination. Detection limits were used as criteria to determine if a blank correction was necessary. If a blank response was below the detection limit then a blank correction was deemed unnecessary. All laboratory blank responses fell below the detection limits.

Bias due to an interference in the sample was evaluated with matrix spikes. Matrix spikes were performed on each batch of nutrient determinations. Five percent of the total samples submitted for a batch or minimum of two matrix spikes were performed for each batch. Spikes achieved a concentration increase of 2-10 times in spiked samples relative to unspiked samples with less than ten percent dilution of the sample. Pooled average recoveries within 5% of the 100% recovery standard were considered good. Pooled average recoveries within 10% of the 100% recovery standard were considered acceptable. All of the nutrient parameters had at least acceptable pooled average recoveries.

Bias due to systematic error was evaluated with internal check standards and external quality control reference samples obtained from the U.S. Environmental Protection Agency (EPA). Internal check standards consisted of internally prepared stock reagent reference materials analyzed at a level equal to at least five percent of the total samples submitted for a batch of nutrient determinations. Check standards were also analyzed for turbidity and chloride determinations. Similar criteria for ranges of acceptability of spike recoveries were used for check standard recoveries also. Within five percent was considered good, and within ten percent was considered acceptable. All pooled average check standard recoveries were at least within the ten percent acceptance range.

Quality control samples obtained from EPA had reported true values. The quality control samples were analyzed simultaneously with lake samples four times during the study period. Table 3.4 presents a summary of the EPA reference material true values (EPA mean) and laboratory results. The reported 95% confidence range around the EPA mean was used as the criteria range to determine if laboratory results contained a significant bias. If a laboratory value fell out of the EPA 95% confidence range then a significant bias was considered present and a correction was deemed necessary. If the laboratory value fell within the confidence range then no significant bias was considered present. For all parameters except ammonia-nitrogen, data were found to have acceptable accuracy (see below). Summarized below are the QA/QC data results reported by parameter.

Total Nitrogen. In general, the quality control results were found to be acceptable for total nitrogen (TN). The quality control checks using EPA control samples were all within the specified 95% confidence range, though one of the four was marginal as it fell on the lower border of that range. The pooled mean matrix spike recovery for total nitogen of 92.6 $\pm$ 8.2% was found to be acceptable. The pooled mean recovery for check standards was 99.3 $\pm$ 4.4%. No bias correction was deemed necessary. All laboratory and field/filter TN blank responses were below detection limit so blank correction of the data was not performed. TN duplicate analyses were quite variable with a RMSE of 77 μg/L for low level duplicates (laboratory and field split) of less than 500 μg/L. Laboratory duplicates had a RMSE of 29 μg/L at this cuttoff level. Low level duplicates of less than 200 μg/L had a RMSE of approximately 8 μg/L whereas the RMSE increased by approximately an order of magnitude if the low level cutoff concentration was increased to 400 μg/L and above.

Total Phosphorus. The quality control results for total phosphorus (TP) were very good. All laboratory means for the quality control evaluations fell well within the EPA 95% confidence level range. The pooled average matrix spike recovery for TP was 98.7 $\pm$ 3.8%. The pooled average recovery for internal check standards was 100.2 $\pm$ 5.0%. No bias correction was deemed necessary for TP data. TP laboratory blanks had a pooled mean of 1.3 $\pm$ 0.6 μg/L and field and filter blanks had a pooled mean of 1.3 $\pm$ 0.6 μg/L. Blank corrections were deemed unnecessary as blank responses were below detection limit. Low level concentrations of less than 100 μg/L TP had a RMSE of 1.8 μg/L.

Ammonia-Nitrogen. With the exception of a laboratory stock standard solution bias in the first half of the study, ammonia-nitrogen had acceptable quality control results. There was a significant positive bias in ammonia-nitrogen from the beginning of the study until the November 14, 1989 sampling (see Table 3.4). This bias was observed after the June 12, 1989 EPA QC sample run. Subsequent internal QA failed to expose the bias and the bias problem was thought to have been corrected. After the October 2, 1989 EPA sample run, however, the ammonia-nitrogen positive bias still persisted. It was noticed that the average ratio of reported/reference concentrations were consistent over time and concentration ranges. Based on the linearity of the average ratio of reported/reference ammonia-nitrogen concentrations, the stock standard solution was suspected as the cause of the bias. An experiment using old and new stock standard solutions for ammonia-nitrogen compared to the same EPA reference sample confirmed that the

Table 3.4. Summary of EPA reference material analyses.
(units are in ug/L unless otherwise noted)

PARAMETER	EPA MEAN	EPA SD	EPA 95% CONFID	LAB MEAN	STD DEV	
APRIL 10, 1989						
TN	528	---	438 - 598	438	20	
NH3-N	150	---	110 - 190	195	5	** (130%)
NO3+NO2-N	180	---	140 - 220	167	20	
SRP	40	---	20 - 60	40.2	0.4	
TP	115	---	105 - 129	119.4	0.8	
ALKALINITY (mg/L)	34.4	---	31.1 - 37.7	34.4	0.19	
CHLORIDE (mg/L)	80.8	---	75.5 - 86.3	80	0	
JUNE 12, 1989						
TN	545	97	351 - 739	459.6	62.9	
	250	81	88 - 412	165	30.0	
NH3-N	496	38	420 - 572	697	21.4	** (140.5%)
	200	26	148 - 252	275	3.6	** (137.5%)
NO3+NO2-N	495	29	437 - 553	474	36.3	
	197	18	161 - 233	212	9.4	
	24	12	0 - 48	26	4.9	
SRP	99.2	8.1	83 - 115	107	0.8	
	24.9	5.5	14 - 36	30	0.2	
TP	103	12	76 - 127	102	0.7	
	27	9	9 - 45	26	0.1	
OCTOBER 2,1989						
TN	545.0	97.0	351 - 739	495.0	8.9	
	299.4	83.6	132 - 467	235.3	6.6	
NH3-N	397.4	34.1	329 - 466	543.0	4.0	** (136.6%)
	200.0	26.0	148 - 252	274.5	4.1	** (137.2%)
NO3+NO2-N	395.5	25.7	344 - 447	411.8	2.7	
	18.9	11.7	-4.5 - 42	14.5	0.3	
SRP	99.2	8.1	83 - 115	105.4	0.5	
	19.0	5.3	9 - 30	22.9	0.3	
TP	153.8	14.5	124 - 183	152.6	0.2	
	32.3	9.5	13 - 51	31.3	1.2	
CHLOROPHYLL a (mg/m3)	3.76	---	3.15 - 4.37	3.55	---	
PHEOPHYTIN (mg/m3)	1.7	---	1.09 - 2.32	1.88	---	
NOVEMBER 14, 1989						
NH3-N (USING OLD STOCK	304	---	268 - 340	371	17.7	** (122%)
STANDARD SOLUTION)	152	---	134 - 170	197.5	5.9	** (129.9%)
NH3-N (USING NEW STOCK	304	---	268 - 340	292.9	14.5	
STANDARD SOLUTION)	152	---	134 - 170	144.9	4.4	
JANUARY 5, 1990						
TN	545	---	351 - 739	474	32	
NH3-N	395.5	---	344 - 447	411	22	
	395.5	---	344 - 447	416	8	
	200	---	148 - 252	215	4	
NO3+NO2-N	395.5	---	344 - 447	420	6	
SRP	99.2	---	83 - 115	102.5	2.6	
TP	153.8	---	124 - 183	155	1.8	
CHLOROPHYLL a (mg/m3)	3.76	---	3.15 - 4.37	3.52	---	
PHEOPHYTIN (mg/m3)	1.7	---	1.09 - 2.32	2.08	---	

** SIGNIFICANT DIFFERENCE FROM EPA MEAN CONCENTRATION
(LAB MEAN VALUE FALLS OUTSIDE EPA 95% CONFIDENCE BOUNDS)

EPA MEAN: CONCENTRATION OF DILUTED CHECK STANDARD ACCOUNTING FOR THE EFFECTS OF DILUTION old stock standard solution was positively biased. The average ratio of reported/reference ammonia-nitrogen concentrations formed the basis for a constant bias correction:

Corrected [ammonia-N] = 0.6686 * Reported [ammonia-N] (eqn. 3.6)

The old stock standard solution for ammonia-nitrogen had been used in the Turnbull Laboratory since the beginning of the Lake Sawyer study. Beginning with the November 13, 1989 samples, the new stock standard solution for ammonia-nitrogen was used. The above correction was made for all ammonia-nitrogen reported concentrations analyzed before November 13, 1989.

The pooled average recovery for ammonia-nitrogen matrix spikes was 92.1 ± 5.9%. Though the recoveries were rather low, they were within the ±10% acceptable range. The pooled average recovery for internal check standards was 93.5 ± 8.9%. No further bias correction was deemed necessary based on the matrix spike and check standard evaluations. Laboratory blank analyses for ammonia-nitrogen had a pooled mean of 3.1 ± 3.2 μg/L. Field and filter blanks had a pooled mean of 36.0 ± 2.7 μg/L. The ubiquitous nature of ammonia (e.g., ammonia present in the air) would explain the rather high field and filter contamination and therefore these values were disregarded. The laboratory used ammonia-free deionized water for their blank analyses. No correction was deemed necessary based on the blank data. The precision of ammonia-nitrogen duplicates was good with a RMSE of 4.6 μg/L for duplicate concentrations below 100 μg/L.

Nitrate Plus Nitrite-Nitrogen. The quality control results for nitrate plus nitrite-nitrogen were good. All laboratory mean values for the EPA control samples fell within the 95% confidence ranges of acceptability. The pooled average recovery for matrix spikes during the study period was 95.8 ± 5.4%. The pooled average recovery for internal check standards was 100.2 ± 3.9%. No bias correction was deemed necessary. Laboratory blanks and field/filter blanks for nitrate plus nitrite-N were all below detection limit and, therefore, no blank correction was deemed necessary. The precision of low level sample concentration duplicates (<100 μg/L) had a RMSE of 2 μg/L.

Soluble Reactive Phosphorus. The quality control results for soluble reactive phosphorus (SRP) were very good. All laboratory mean values for the EPA control samples fell well within the 95% confidence range of acceptability. The pooled average recovery for matrix spikes during the study period was 98.4 ± 3.1%. The pooled average recovery for internal check standards was 97.8 ± 6.1%. No bias correction was deemed necessary. The pooled mean laboratory blank for SRP was 2.2 ± 0.5 μg/L. The pooled mean field and filter blank was 2.3 ± 0.5 μg/L. No blank corrections were deemed necessary. The precision of low level sample concentration duplicates (<100 μg/L) had a RMSE of 1.2 μg/L.

Chloride. The quality control results for chloride were very good. The pooled average recovery for internal check standards was 100.9 ± 1.0%. The precision for laboratory duplicates was excellent. The RMSE was 0.09 mg/L between laboratory duplicates.

Chlorophyll *a*. The quality control results for chlorophyll *a* were very good. All laboratory mean values for EPA control samples fell within the 95% confidence range of acceptance, as did the inactive chlorophyll partition (phaeophytin) with which the chlorophyll *a* was corrected. The pooled mean blind blank for chlorophyll *a* was 0.15 ± 0.22 μg/L. The precision of blind split duplicates for chlorophyll *a* and phaeophytin were excellent with RMSEs of 0.59 μg/L and 0.54 μg/L, respectively.

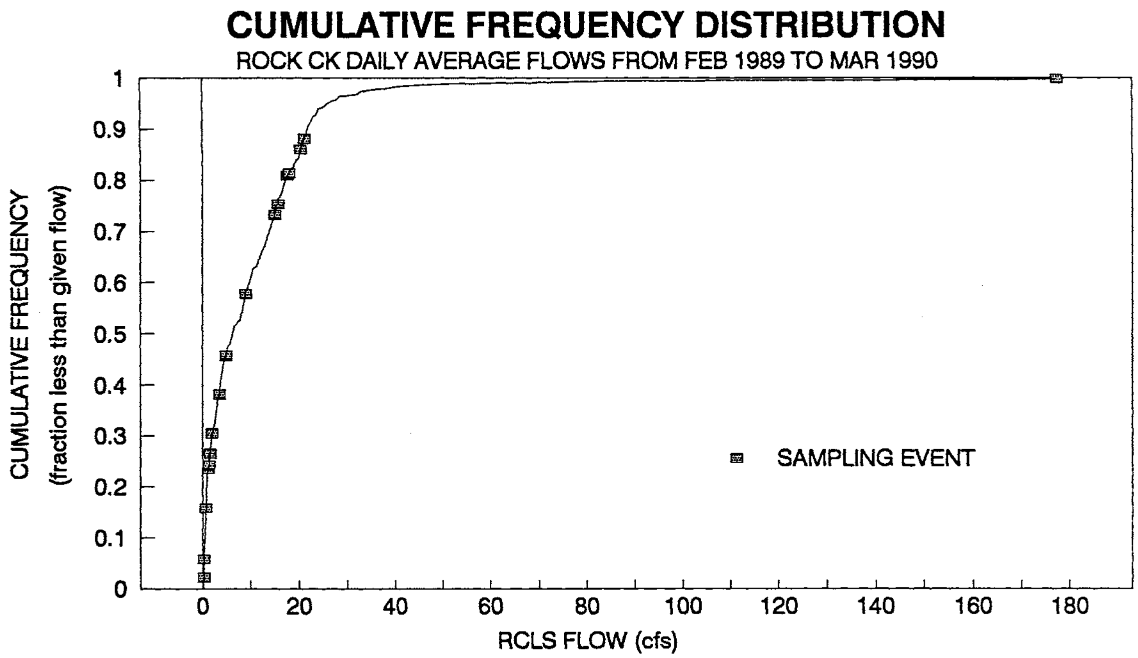
Alkalinity. Alkalinities had good laboratory duplicate precision results. The RMSE was 0.3 mg/L as $CaCO_3$.

Turbidity. The quality control results for turbidity were very good. The pooled average recovery for internal check standards was 98.7 ± 1.8%. The RMSE of laboratory duplicates was 0.11 NTUs.

3.3 Nutrient Loading Estimators

Nutrient loads for Lake Sawyer tributaries were calculated using regression estimators. Nutrient concentration data from the fixed-frequency sampling pattern (e.g. every 3-4 weeks; 18 total samplings) and daily flow records were used to model the nutrient loading. Figure 3.2 presents the cumulative frequency distribution of daily average flows at Ravensdale Creek and Rock Creek from February 1989 through March 1990, identifying the flows at which concentration data was collected during discreet sampling events. There was no attempt to sample specific flow events during the study period, however sampling events encompassed fairly complete ranges of flows from both Rock Creek and Ravensdale Creek including a large storm event on January 9, 1990. Sampling strategies that insure collection of data from a tributary's entire range of hydrologic conditions improve estimator performance (Preston *et al.*, 1989).

Strong increasing or decreasing relationships between nutrient concentration and flow were used as indications that a regression estimator would perform well (Johnson, 1979). In cases where strong relationships did not exist, performance comparisons of regression estimators versus the Beale ratio estimator (Beale, 1962) were made comparing the root mean squared error (RMSE) of predicted versus observed loadings. Though the Beale ratio estimator has been shown to successfully estimate P loading rates (Dolan *et al.*, 1981), regression estimators out-performed the ratio estimator in all cases. Regression estimators are more satisfactory for grab sample studies where solute concentration is correlated with stream discharge because regressions account for solute export at the highest flows better than other methods (Johnson, 1979). Numerous least-squares regression models fit the types of concentration-flow curves (Galat, 1990). The choice of data transformations and regression equation was made for each station on the basis of minimizing the RMSE of predicted and observed loads. Hyperbolic regressions were used for estimating N loading at Rock Creek and P loading at Ravensdale Creek. A linear regression was used to estimate N loading at Ravensdale Creek and a log-log cubic regression was used to estimate P loading at Rock Creek. Regressions were developed using WQHYDRO (Aroner, 1990), an environmental statistical data analysis graphics package.



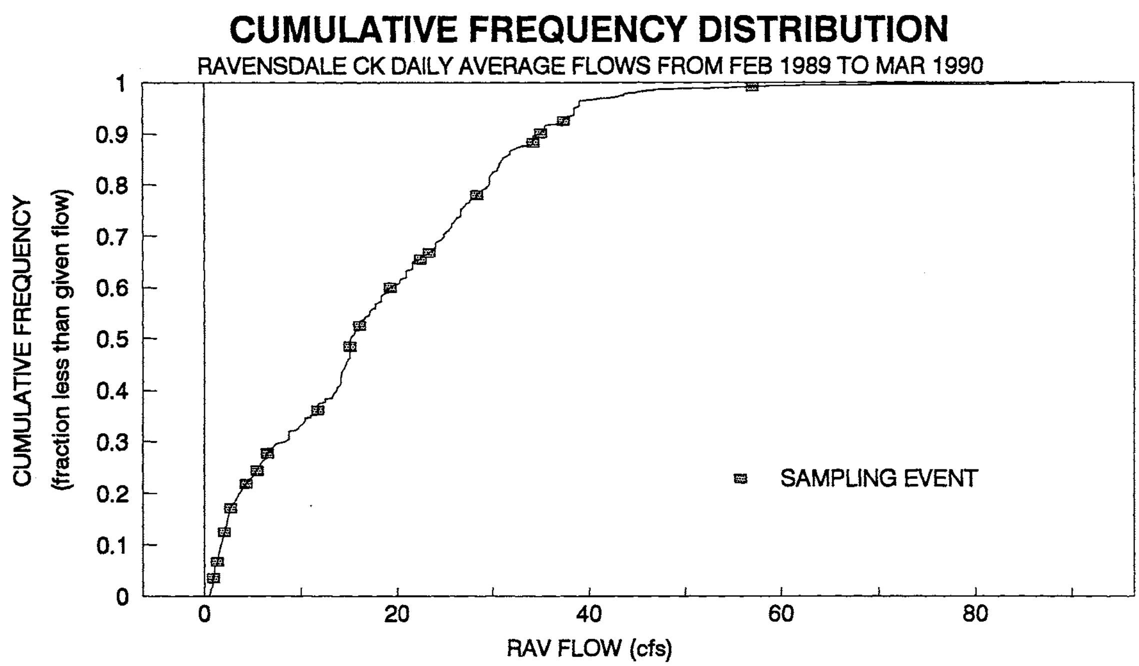


Figure 3.2. Cumulative frequency distributions of daily average flows from February 1989 to March 1990 at Rock Ck at Lake Sawyer and Ravensdale Ck, showing sampling event flows.

3.4 Uncertainty Analysis

The information value contained within a given estimated or predicted quantity is only as good as the confidence bounds which surround the estimate. Since the mass balance models used in this study are based on discharge and chemical measurements, a variety of potential measurement and modeling errors can contribute to the total uncertainty of a given quantity. Quantification and propagation of the uncertainty common to each term in a calculated value is necessary in order to determine the degree of confidence which can be placed on the prediction.

Statistical techniques which describe the effects of contributing uncertainties are broadly characterized as error propagation methods. For this report, we have utilized first-order and Monte Carlo simulation methodologies. The theory and application of uncertainty analysis techniques have been described by Reckhow and Chapra (1983), Cornell (1973), and Lettenmaier and Richey (1979).

First-Order Error Analysis. Briefly, the first-order technique is based upon the assumption that parameter variations can be propagated about the first derivative (i.e., first-order) of a function relative to those variables which make up the function. In general, for any calculated quantity Y which is derived from parameters denoted by X_i:

$$Y = f(X_1, X_2, \ldots, X_n) \qquad \text{(eqn. 3.7)}$$

the first-order variance of Y can be estimated as:

$$\mathrm{Var}(Y) = \sum_{i=1}^{n} \left[(\delta Y/\delta X_i)^2 \, \mathrm{Var}(X_i) \right] \qquad \text{(eqn. 3.8)}$$

The quantity $(\delta Y/\delta X_i)^2$ describes the sensitivity of the calculated value (Y) to changes in each parameter (X_i) of the function. Unless otherwise stated herein, parameter estimates are presented as the mean values plus or minus ($\pm$) the standard error (SE) of the mean (Zar, 1974).

Monte Carlo Simulation. Monte Carlo simulation involves assigning probability density functions to each variable or parameter reflecting the measured or estimated uncertainty of that parameter. Then, with "synthetic" sampling, values are randomly chosen from the distributions of each variable. These values are then inserted into the model or function which relates the variables, and a prediction is calculated. After this is repeated a large number of times (generally 500 iterations in this study), a distribution of predicted values results, which reflects the combined uncertainties of all input variables. Monte Carlo simulation was performed using the program @RISK (version 1.55; Palisades Corporation) with a Lotus 1-2-3 spreadsheet (version 2.2; Lotus Development Corporation).

4.0 HYDROLOGIC REGIME

4.1 Watershed Characteristics

The Lake Sawyer watershed drains approximately 34 square kilometers (13 mi^2) to the outlet dam at the start of Covington Creek. The lake's drainage basin lies in an area of complex geology with a mix of low-permeable Vashon Till outcrops and high-permeable Vashon Recessional Outwash drift with peat and muck areas (i.e. wetland areas) residing in the southern part of the drainage (Hart Crowser, 1990; McConnell *et al.*, 1976). Two major sub-basins, representing the two surface water inflows, Ravensdale Creek and Rock Creek, drain 90% of the total Lake Sawyer watershed basin. The remaining miscellaneous drainage is primarily from areas immediately adjacent to Lake Sawyer. Of the two major sub-basins, Ravensdale Creek (also known as Beaver Creek) drains 40% or approximately the northern half of the Lake Sawyer catchment area and Rock Creek drains the southern half of the Lake Sawyer watershed. The topography of the two major sub-basins is similar in that the altitude at the top of their drainages is about 360 meters (1200 feet) but the majority of their drainage lies between 160 meters (525 feet) and 183 meters (600 feet) mean sea level (MSL). Lake Sawyer surface elevation generally fluctuates between the 157 and 158.5 meters MSL (515 and 520 feet MSL). This indicates a moderately sloped topography for the majority of the Lake Sawyer watershed.

4.2 Lake Sawyer Water Budget (Study Year)

The water budget for Lake Sawyer can be described by the balance of inflows to the lake, outflows from the lake, and changes in lake storage (Figure 4.1). Rock Creek and Ravensdale Creek are the two surface water inflows at the south end of the lake. Other inflows include precipitation onto the lake surface and groundwater movement into the lake basin. Covington Creek is the surface water outflow at the central west side of Lake Sawyer. Other outflows include evaporation from the lake surface and groundwater movement from the lake basin. Groundwater movement through Lake Sawyer results in either a net inflow or net outflow over time. Similarly, the change in lake storage results in either a net increase or net decrease over time. The Lake Sawyer water budget for the study year is presented in Table 4.1. Below is a presentation of the findings for each of the water budget components.

4.2.1 Climate

The climate of the Lake Sawyer watershed area is typical for western Washington. Precipitation falls predominantly in the winter months. Moist air from the Pacific Ocean is brought inland by southerly winds produced by an Aleutian low pressure cell (Dexter *et al.*, 1981). Approximately 77% of the precipitation which fell at Lake Sawyer during the study period fell during the months of November through April (Figure 4.2). Direct precipitation onto the surface of Lake Sawyer averaged 1.6 ± 0.2 cfs during the entire study year, accounting for 5.8% of the total input to the lake (Table 4.2).

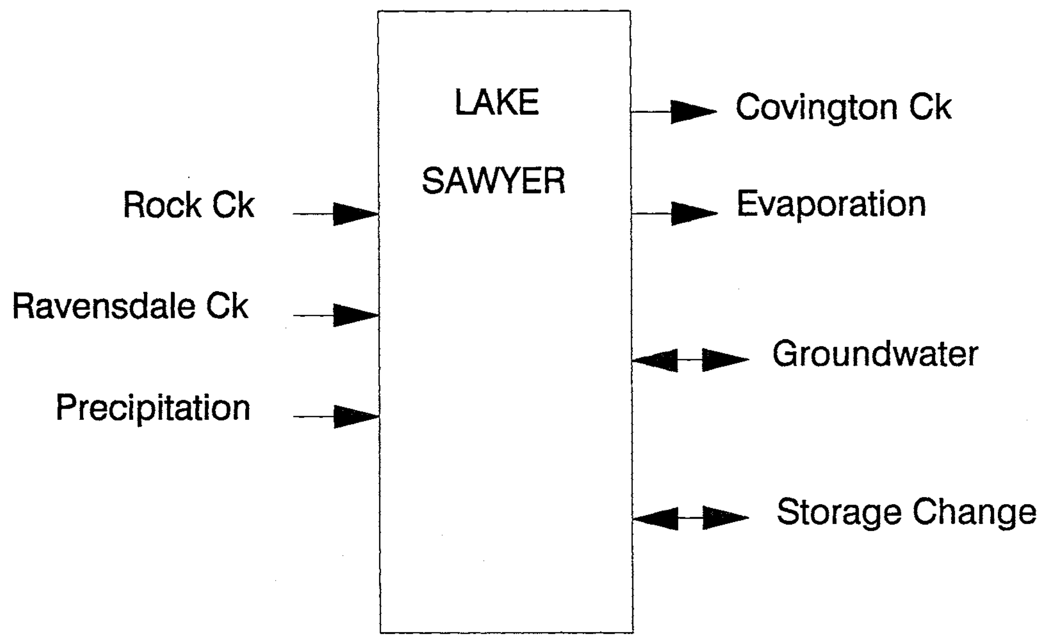


Figure 4.1. Schematic of the Lake Sawyer water budget.

Table 4.1 Study year and typical year monthly water budgets for Lake Sawyer: (all values are flow rates in cubic feet per second (cfs))

STUDY YEAR

EQUATION:	<------------OUTFLOWS	-------------> -	<--------------------INFLOWS	----------------	---------------------> +	STORAGE CHANGE(1) =	RESIDUAL(2)
COMPONENTS:	Covington Creek +	Evaporation -	Precipitation +	Ravensdale Creek +	Rock Creek +	Storage Change =	Groundwater
	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE
March	57.41 ± 1.66	0.12 ± 0.03	2.83 ± 0.29	33.24 ± 2.96	20.68 ± 0.99	-0.68 ± 0.07	0.10 ± 3.55
April	51.94 ± 1.50	0.35 ± 0.10	1.40 ± 0.14	30.73 ± 2.73	14.52 ± 0.69	-1.04 ± 0.10	4.59 ± 3.20
May	18.53 ± 0.54	0.62 ± 0.19	1.50 ± 0.15	15.59 ± 1.68	4.41 ± 0.21	-0.23 ± 0.02	-2.58 ± 1.80
June	11.16 ± 0.32	1.08 ± 0.32	0.77 ± 0.08	11.60 ± 1.25	2.61 ± 0.12	-0.85 ± 0.08	-3.59 ± 1.34
July	2.99 ± 0.09	0.94 ± 0.28	0.47 ± 0.05	6.28 ± 0.68	1.31 ± 0.06	-1.18 ± 0.12	-5.31 ± 0.75
August	0.61 ± 0.02	1.13 ± 0.34	0.35 ± 0.04	3.23 ± 0.35	0.55 ± 0.03	-3.78 ± 0.38	-6.16 ± 0.62
September	0.54 ± 0.02	0.96 ± 0.29	0.18 ± 0.02	1.95 ± 0.21	0.35 ± 0.02	-3.01 ± 0.30	-3.98 ± 0.47
October	0.74 ± 0.02	0.48 ± 0.14	1.13 ± 0.12	0.96 ± 0.10	1.13 ± 0.06	-2.37 ± 0.24	-4.36 ± 0.33
November	2.45 ± 0.07	0.58 ± 0.17	3.06 ± 0.32	5.41 ± 0.44	12.13 ± 0.58	9.46 ± 0.95	-8.12 ± 1.25
December	30.38 ± 0.88	0.19 ± 0.06	2.13 ± 0.22	21.16 ± 1.74	18.48 ± 0.88	1.91 ± 0.19	-9.29 ± 2.16
1990 January	59.96 ± 1.73	0.52 ± 0.16	4.43 ± 0.46	33.39 ± 2.74	29.87 ± 1.44	1.09 ± 0.11	-6.12 ± 3.58
February	57.68 ± 1.67	0.60 ± 0.18	1.98 ± 0.20	34.75 ± 2.85	18.93 ± 0.90	-0.61 ± 0.06	2.00 ± 3.43
March	39.54 ± 1.14	0.40 ± 0.12	1.70 ± 0.18	26.19 ± 2.15	12.55 ± 0.60	-0.27 ± 0.03	-0.78 ± 2.51
Annual (Mar-Feb):	24.53 ± 1.00	0.63 ± 0.21	1.69 ± 0.21	16.52 ± 1.83	10.42 ± 0.68	-0.11 ± 0.33	-3.57 ± 2.24
Annual (Apr-Mar):	23.04 ± 0.94	0.65 ± 0.22	1.59 ± 0.20	15.94 ± 1.73	9.74 ± 0.64	-0.07 ± 0.33	-3.64 ± 2.12

TYPICAL YEAR

EQUATION:	<------------OUTFLOWS	-------------> -	<--------------------INFLOWS	----------------	---------------------> +	STORAGE CHANGE(1) =	RESIDUAL(2)
COMPONENTS:	Covington Creek(3) +	Evaporation -	Precipitation +	Ravensdale Creek +	Rock Creek +	Storage Change =	Groundwater
	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE	Mean ± SE
March	45.23 ± 6.93	0.40 ± 0.12	1.91 ± 0.21	29.41 ± 5.11	17.97 ± 0.86	-0.56 ± 0.30	-4.21 ± 8.66
April	38.75 ± 2.79	0.35 ± 0.10	1.55 ± 0.17	25.20 ± 2.75	15.40 ± 0.74	-0.23 ± 0.13	-3.27 ± 3.99
May	21.83 ± 3.73	0.62 ± 0.19	1.15 ± 0.13	14.20 ± 2.69	8.67 ± 0.41	-0.50 ± 0.09	-2.07 ± 4.63
June	10.60 ± 1.97	1.08 ± 0.32	1.08 ± 0.12	6.89 ± 1.40	4.21 ± 0.20	-0.24 ± 0.12	-0.73 ± 2.45
July	4.80 ± 1.69	0.94 ± 0.28	0.54 ± 0.06	3.12 ± 1.13	1.91 ± 0.09	-1.41 ± 0.35	-1.23 ± 2.08
August	0.77 ± 0.44	1.13 ± 0.34	0.74 ± 0.08	0.50 ± 0.29	0.31 ± 0.01	-2.08 ± 0.64	-1.73 ± 0.90
September	1.45 ± 1.11	0.96 ± 0.29	1.19 ± 0.13	0.95 ± 0.73	0.58 ± 0.03	-1.02 ± 0.60	-1.31 ± 1.49
October	1.88 ± 1.71	0.48 ± 0.14	1.75 ± 0.20	1.22 ± 1.11	0.75 ± 0.04	-0.20 ± 0.36	-1.56 ± 2.09
November	19.21 ± 7.71	0.58 ± 0.17	2.69 ± 0.30	12.49 ± 5.11	7.63 ± 0.36	4.14 ± 1.04	1.11 ± 9.32
December	58.87 ± 15.45	0.19 ± 0.06	3.13 ± 0.35	38.28 ± 10.53	23.39 ± 1.12	2.96 ± 1.00	-2.79 ± 18.76
January	63.58 ± 9.98	0.52 ± 0.16	2.85 ± 0.32	41.35 ± 7.32	25.26 ± 1.21	0.53 ± 0.24	-4.83 ± 12.45
February	51.85 ± 3.00	0.60 ± 0.18	2.13 ± 0.24	33.72 ± 3.39	20.60 ± 0.98	-0.13 ± 0.41	-4.13 ± 4.66
Annual	26.57 ± 2.76	0.65 ± 0.22	1.73 ± 0.22	17.28 ± 1.87	10.56 ± 0.77	0.11 ± 0.54	-2.23 ± 7.88

1) Negative storage change indicates net decrease in lake storage, positive indicates net increase. Typical year storage changes from averages of 1964-75 USGS data.

2) Negative residual indicates net groundwater outflow, positive is net groundwater inflow.

3) Typical year outflow calculated from 1954-59 averages of USGS data for Covington Ck.

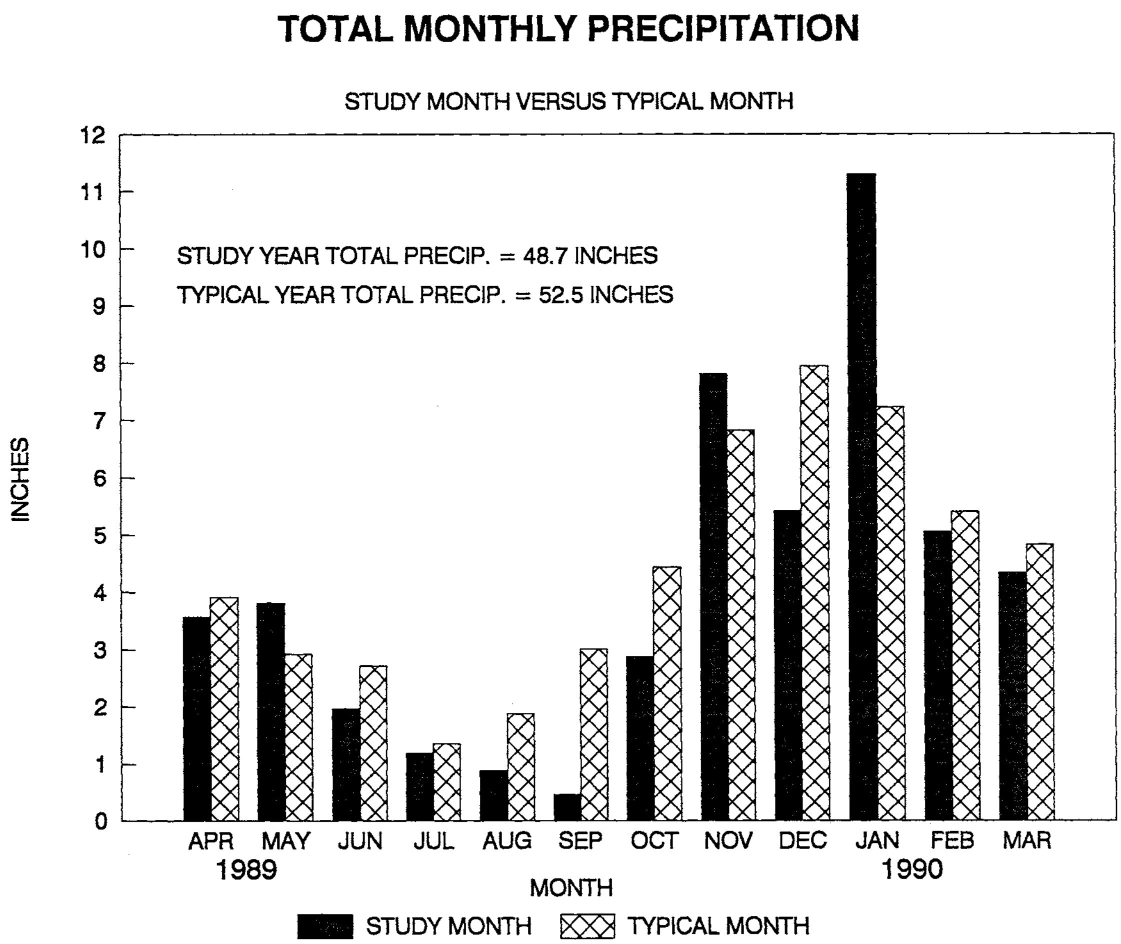


Figure 4.2. Comparison of study year precipitation and typical year precipitation at Lake Sawyer

Table 4.2. Lake Sawyer annual water budget summary - percent contribution of inflows and outflows.

STUDY YEAR (April 1989 - March 1990)

OUTFLOWS	cfs mean ± S.E.	% of total
Covington Creek	23.0 ± 0.9	84.3%
Evaporation	0.7 ± 0.2	2.4%
Groundwater	3.6 ± 2.1	13.3%
TOTAL OUTFLOWS =	27.3 ± 2.3	100.0%

INFLOWS	cfs mean ± S.E.	% of total
Precipitation	1.6 ± 0.2	5.8%
Ravensdale Creek	15.9 ± 1.7	58.5%
Rock Creek	9.7 ± 0.6	35.7%
TOTAL INFLOWS =	27.3 ± 1.9	100.0%

The summer months, May through October, are generally dry because of a semipermanent high pressure cell off the Washington coast which keeps the Aleutian low pressure cell to the north. Summer is also the time of the highest air temperatures and least sky cover. Approximately 74% of the evaporation which occurred from the Lake Sawyer water surface took place during the summer study period coincident with the higher temperatures. Direct evaporation from Lake Sawyer over the entire study period averaged 0.7 ± 0.2 cfs or only about 2.4% of the total hydrologic loss from the lake (Table 4.2).

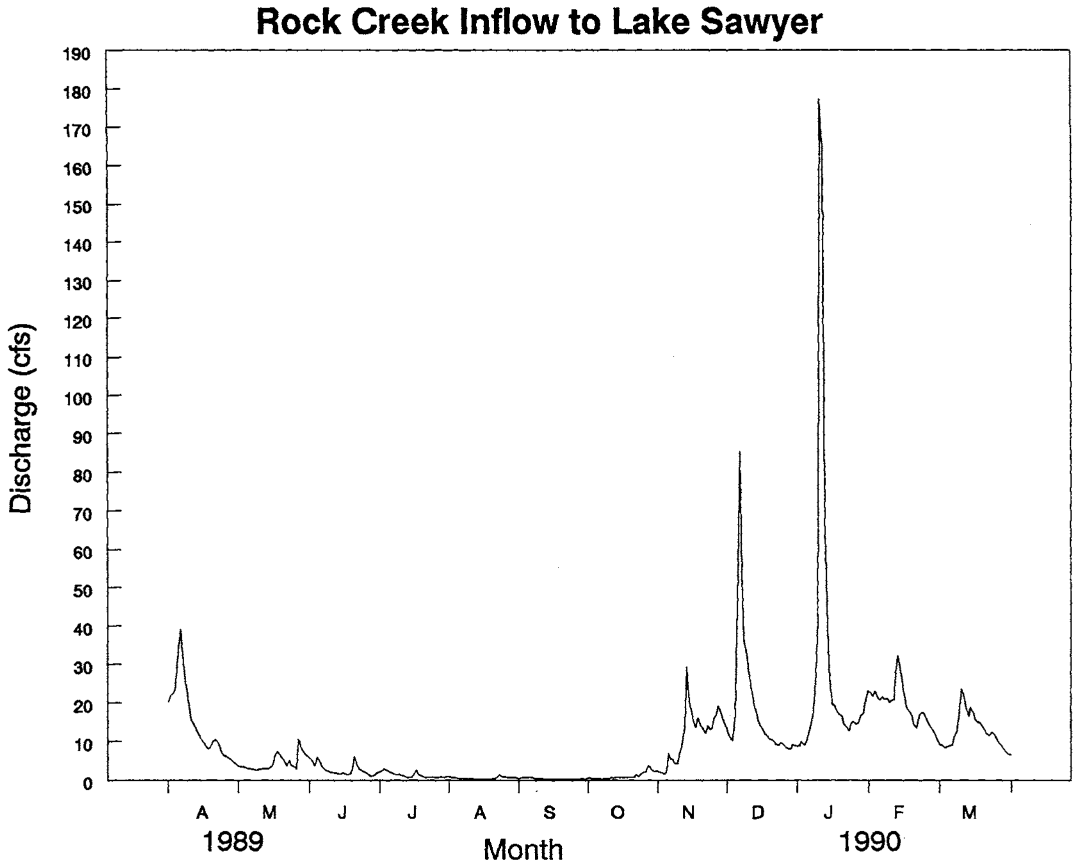
A long term precipitation record was not available for the Lake Sawyer vicinity. Comparisons between study year precipitation and average year precipitation were made using precipitation data and records from a nearby weather station at Landsburg, Washington. The Landsburg weather station reports to the National Climatic Center (NOAA) and has a 30-year (1951-1980) monthly average rainfall record available (National Climatic Center, 1982). Landsburg is approximately five miles northwest of Lake Sawyer and lies at approximately the same elevation (163 meters; 520 feet MSL). Landsburg precipitation correlated closely with measured Lake Sawyer study year precipitation with 8.27% higher annual precipitation.

Landsburg's annual precipitation during the Lake Sawyer study year was compared with Landsburg's 30-year annual average precipitation. The ratio between the two formed the basis for a proportional correction used to normalize the Lake Sawyer annual precipitation to a "typical year". Typical monthly precipitation at Lake Sawyer was apportioned from the typical annual precipitation based on the long-term monthly distributions at Landsburg. Figure 4.2 compares study year monthly precipitation and typical year monthly precipitation (calculated using the proportional correction described above) at Lake Sawyer. Though some study year months (particularly January, 1990) showed higher than normal rainfall, the overall Lake Sawyer study year annual average was a little lower than normal. The study year was fairly representative of a normal climatalogical year in general.

4.2.2 Surface Water Inflows

As mentioned above, Rock Creek and Ravensdale Creek are the two major surface water inflows to Lake Sawyer. Rock Creek has extensive natural wetland areas in its sub-basin. These areas of peat and muck are a complex convergence zone for several tributaries to Rock Creek, including Black Diamond Lake Creek, Morganville marsh drainage, Palmer Spring, and Ginder Creek. The City of Black Diamond and the smaller town of Morganville flank the north side of the Rock Creek sub-basin, consequently Rock Creek is influenced by development and urban activities, including the WTP that discharges through the wetland to Rock Creek. Ravensdale Creek has a relatively pristine watershed that drains primarily a managed forest area east of Lake Sawyer. A quarry, sawmill, Ravensdale Lake, and the small town of Ravensdale are located at the head of the Ravensdale Creek.

Of the total hydrologic input to Lake Sawyer during the study period, Ravensdale Creek had the largest contribution, accounting for 58.5% of the total (Table 4.2). Discharge from Ravensdale Creek averaged 15.9 ± 1.7 cfs over study period. Rock Creek accounted for 35.7% of the total



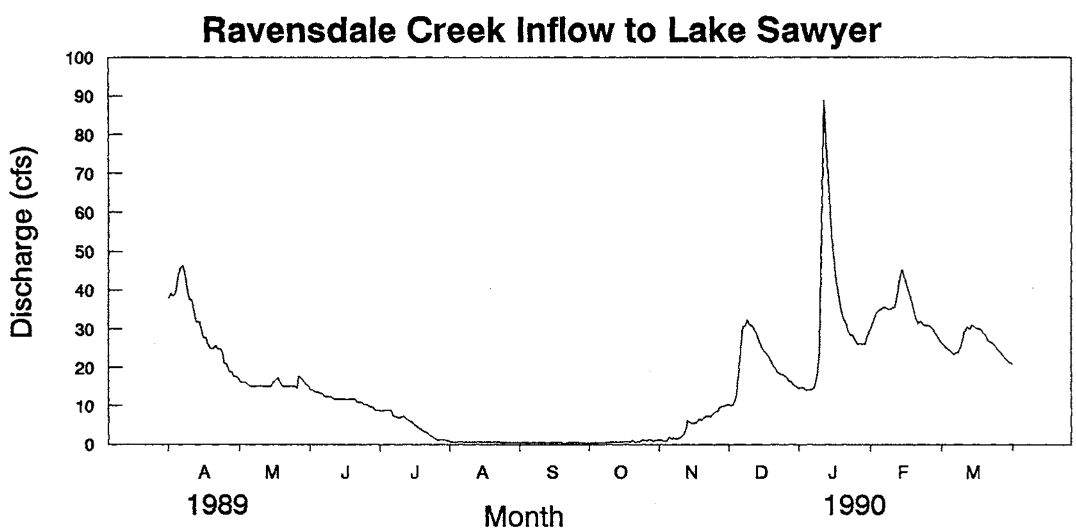


Figure 4.3. Hydrographs of surface water inflow tributaries to Lake Sawyer.

hydrologic input to Lake Sawyer, with an average discharge of 9.7 ± 0.6 cfs over the study period. Figure 4.3 presents the hydrographs for Rock Creek and Ravensdale Creek during the study period. Most of the runoff to the lake occurred in the wet season. Approximately 91% of Rock Creek's total flow to Lake Sawyer occurred between the months of November and April, coincident with the precipitation season. Nearly 79% of Ravensdale Creek's total discharge to Lake Sawyer occurred during the same period. Ravensdale Creek's wet season percent contribution was lower because of continuous baseflow during the summer dry period. Ravensdale Creek had year-round flow to Lake Sawyer due to this baseflow. Rock Creek's discharge to Lake Sawyer was almost zero from August through October and in one several week period from late September, 1989 to early October, 1989, no surface flow was observed.

Even though Rock Creek has extensive wetland areas in its basin which would seem to buffer high flow events (i.e. wetlands acting as a water storage battery), Rock Creek's hydrograph in Figure 4.3 shows many perturbations compared to the rather smooth hydrograph for Ravensdale Creek. Though Ravensdale Creek usually had more discharge, Rock Creek's maximum discharge exceeded Ravensdale Creek's discharge during two storm events (early December, 1989 and early January, 1990). Rock Creek's high response to precipitation events may be explained by development within the Rock Creek watershed. Stormflow runoff from the many impervious surfaces in Black Diamond and Morganville may assist in increasing the flow in Rock Creek during storm events. Ravensdale Creek's undeveloped watershed allows relatively slow infiltration and subsurface percolation of rainfall. The peaks in the hydrograph for Ravensdale Creek are lower and attenuated over a longer time period compared to Rock Creek's hydrograph peaks (Figure 4.3).

4.2.3 Groundwater Influences

Direct and accurate groundwater measurements are very difficult to make, therefore most lake water budgets resolve the groundwater component by treating it as a residual in the water budget (Cooke, *et al.*, 1986). The groundwater hydrologic influence on Lake Sawyer during the study year was calculated this way. Groundwater is continually entering and leaving Lake Sawyer. The water budget residual indicates the net effect of groundwater movement through Lake Sawyer, either a gain or loss. The uncertainty associated with the residual is rather high because it represents a first-order combination of all the other water budget component errors. Though the residual is made up of all unmeasured hydrologic gains and losses, the Lake Sawyer water budget residual was considered to singularly represent the groundwater component as all other significant components were measured or directly estimated.

Figure 4.4 presents a plot of the water budget residuals with 95% confidence intervals. There was a period of significant residual outflow from the lake from June until December, mainly encompassing the dry period of the year. The average net outflow of groundwater for the study year was 3.6 ± 2.1 cfs, representing 13.3% of the total hydrologic outflow from Lake Sawyer. The phenomena of net groundwater loss from Lake Sawyer was confirmed in an independent assessment of groundwater influence made by Hart Crowser, Inc., based on flownet analysis of

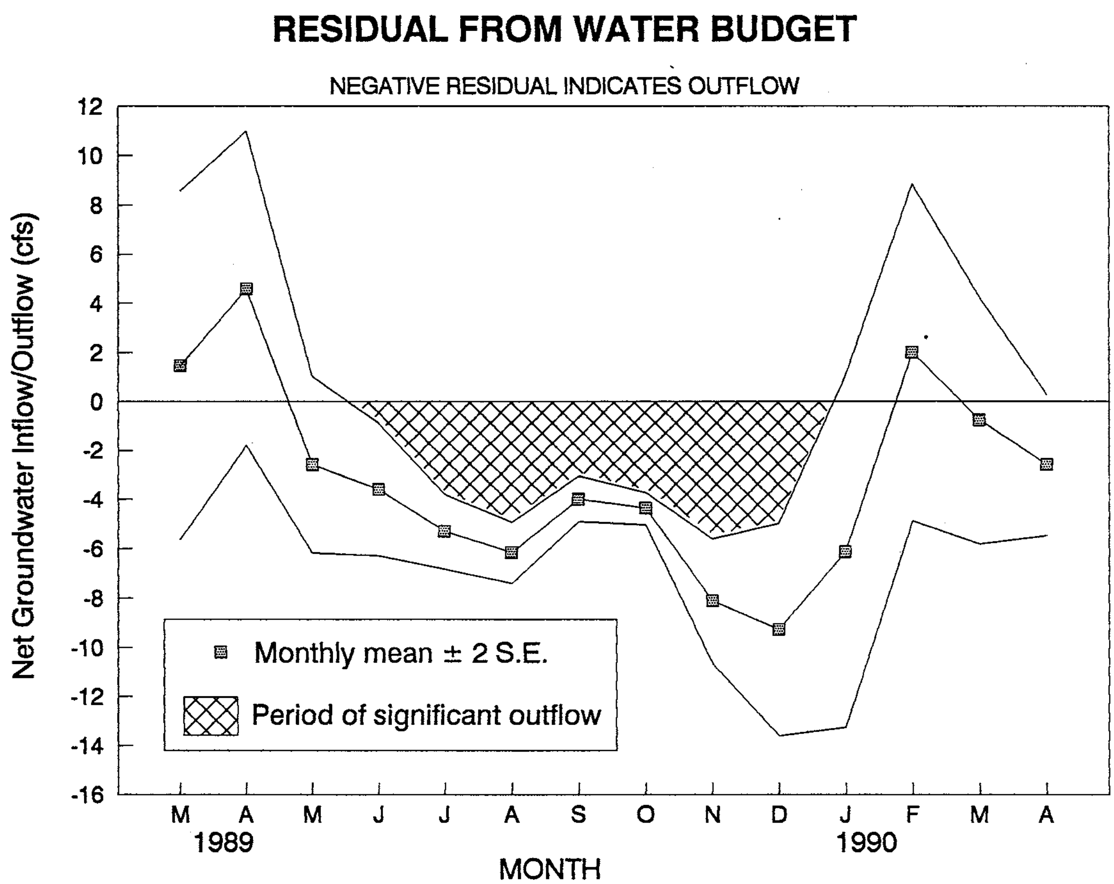


Figure 4.4. Lake Sawyer water budget residual, highlighting the period of significant outflow.

groundwater contours and estimates of hydraulic conductivity and gradients near Lake Sawyer to calculate estimated groundwater flows to and from the lake (Hart Crowser, 1990).

4.2.4 Outflow and Lake Storage

Covington Creek is the outlet of Lake Sawyer at the central western shore. Lake level and outflow from Lake Sawyer is controlled by a concrete dam at the outlet which was constructed in 1952. The concrete dam contains a fish ladder structure made up of "flashboards" which can be raised or lowered to adjust low lake levels, though there was no evidence of their disturbance during the study year. Figure 4.5 presents hydrographs of Covington Creek discharge and Lake Sawyer stage levels over the study period. Covington Creek outflow from the lake averaged 23.0 $\pm$ 0.9 cfs over the entire period, accounting for 84.3% of the total hydrologic output from Lake Sawyer (Table 4.2).

As can be seen from Figure 4.5, even though outflow through Covington Creek almost ceased from August through November, lake level and storage continued to recede until mid-November. Inflows to Lake Sawyer were negligible during this period and the loss in lake storage was attributable to groundwater outflow and evaporation from the lake. The total hydrologic output from Lake Sawyer for the entire study year was 27.3 $\pm$ 2.4 cfs (Table 4.2), with an average flushing rate of about 2.8 lake volumes per year, corresponding to an average residence time of about 4.2 months.

Storage change over a year is made up of seasonal increases and decreases in lake level. It should be noted that annual storage change should approximate zero over a long-term average of years. During the study year the net annual storage change showed a slight but insignificant decrease over the 12 month period. Individual months in the study period usually showed significant increases or decreases in storage. Decreases in storage occurred from early spring through the summer, attaining a maximum storage loss rate in August. This was coincident with decreased inflow and evaporation/groundwater outflow maximas. An abrupt increase in storage took place in November as the lake filled up with the first rains of the winter.

4.3 Lake Sawyer Water Budget (Typical Year)

Table 4.1 shows the water budget for a typical year at Lake Sawyer. The study year water budget was normalized to a typical year in order to predict responses to various loadings to Lake Sawyer (e.g. WTP diversion) in the future. This was done by proportionally adjusting the precipitation and surface water inflow and outflow components of the study year water budget based on the ratio of study year to 30-year (1951-1980) average precipitation at Landsburg, Washington (see above). Typical monthly precipitation at Lake Sawyer was then apportioned from the typical annual average based on the long-term monthly precipitation distributions at Landsburg. Typical monthly outflow and inflows were apportioned using the 6-year (1953-60) monthly distributions of outflow rate at Covington Creek reported by USGS (Williams and Pearson, 1985).

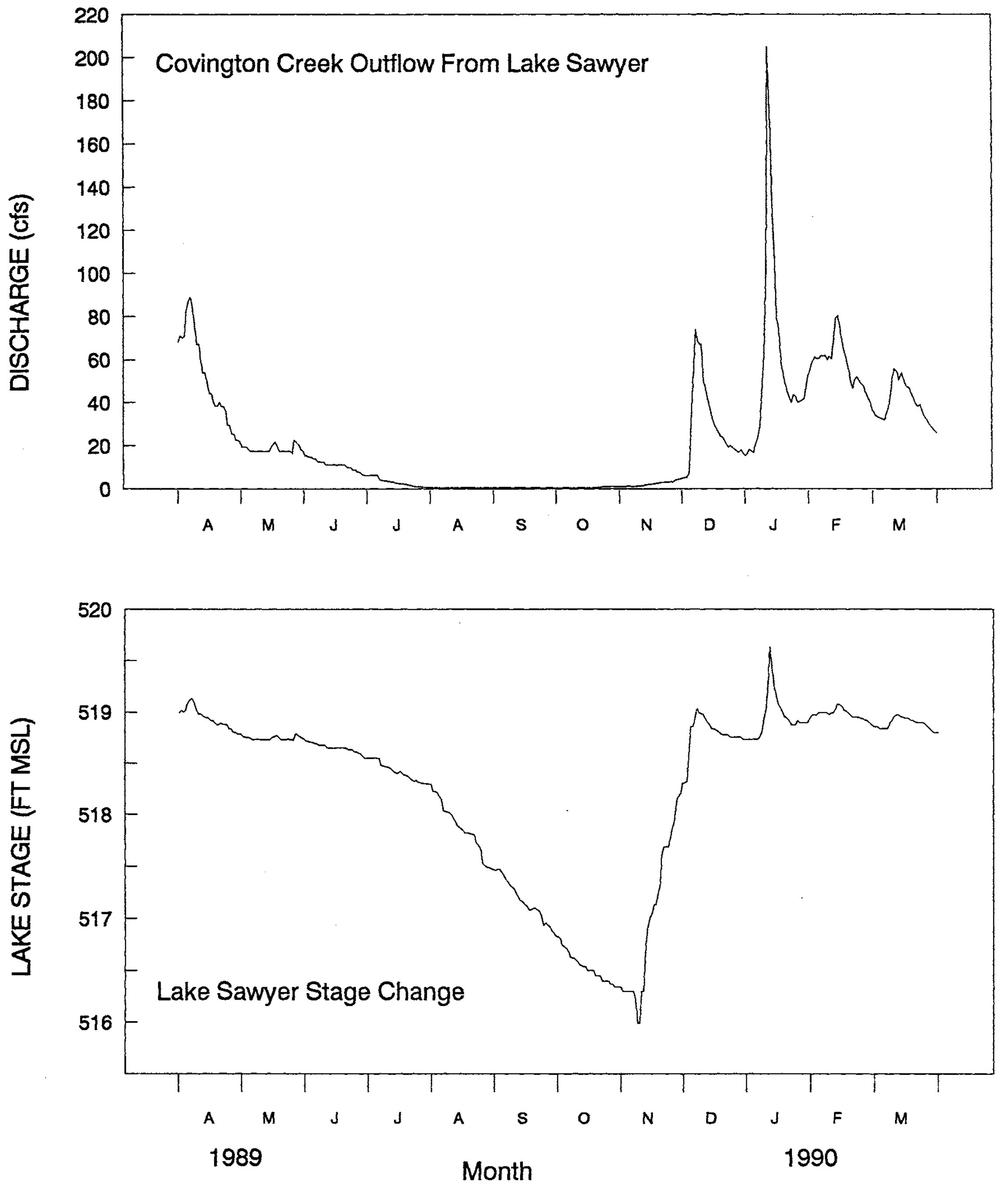


Figure 4.5. Outflow and storage components of the water budget during the study period (April 1989 - March 1990) at Lake Sawyer.

The 30-year record at Landsburg indicates that the precipitation recorded at Lake Sawyer over the twelve month study period (April 1989 through March 1990) was about 7.8% below normal. Proportionally increasing the study year's Covington Creek outflows results in a typical year outflow of 24.8 ± 0.8 cfs. This compares well to the 6-year gaging record (1953-1960) of Covington Creek (26.5 ± 2.8 cfs).

The evaporation component of the water budget was not adjusted as it was a minor component with a large relative uncertainty (±30%). Monthly storage change for a typical year was based on a 10-year record of monthly change in lake stage at Lake Sawyer (1964,1967-1975) available through USGS records. The groundwater component of the typical year water budget was calculated as the residual of the water balance, just as in the study year water budget. Overall, there was not a dramatic difference between the study year water budget and a typical (or average) year water budget. This was expected since precipitation during the study year did not deviate much from normal (Figure 4.2).

5.0 LAKE LIMNOLOGICAL RESULTS

5.1 Lake Physical and Chemical Characteristics

5.1.1 General Physical

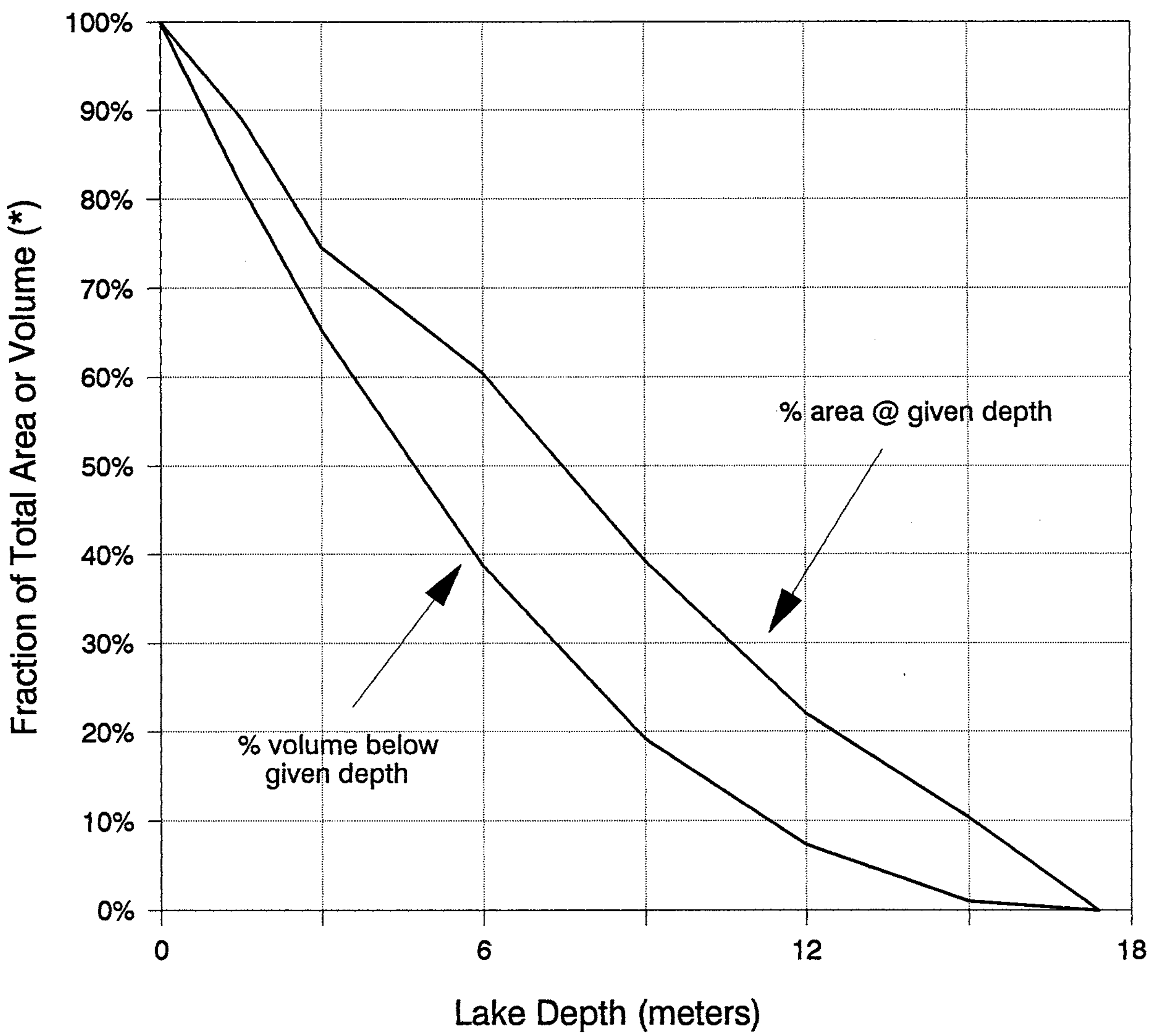
Along with watershed and climatic factors, the morphology of a lake basin affects lake dynamics and productivity. A description of the physical characteristics of Lake Sawyer was given in Section 2.2. Lake Sawyer is a natural lake of moderate depth. The lake basin is enlongated on a north/south axis and contains two distinct basins. The bathymetric map (Figure 2.1) shows the southern quarter of the lake to be shallower than the middle and northern areas. The extreme northwestern end of the lake was dredged some time prior to 1960 and the extreme southeastern end of the lake was dredged in the early 1970s. Mean depth is 7.6 meters (25 feet) and maximum depth is 18 meters (58 feet).

Figure 5.1 presents curves relating area and volume with depth for Lake Sawyer. The depth of Lake Sawyer allows the development of a hypolimnion during the summer months. The thermocline, or boundary between the hypolimnion and the epilimnion was approximately at the 7.5 meter depth during the study period. Approximately 30% of the lake volume is represented by the hypolimnion (7.5 meters to bottom) during lake stratification. The hypolimnetic lake volume is relatively small compared to the epilimnetic lake volume.

5.1.2 Light Transmission/Secchi Disk

Vertical penetration of light through the water column during the study period is depicted in Figure 5.2 as isopleths of percent of incident light. Secchi disk depths averaged the depth of 10.26% $\pm$ 0.88% (percent) of incident light during the growing season (March to October). Major variation in the vertical light penetration during the growing season (March through October) can be correlated to variation in chlorophyll *a* levels since the phytoplankton crop acts as a barrier to light (Figure 5.3). However, factors other than algae also limit light penetration in Lake Sawyer during the growing season as well as throughout the study period, as shown by the significant light extinction coefficient when chlorophyll *a* is absent (Figure 5.3). The non-algal light extinction in Lake Sawyer is probably caused by water color from naturally occuring dissolved humic substances derived from the tributary wetlands.

There was a strong reduction in light penetration during the unstratified months of November through the end of the study period. Increased coloring from humic substances and/or increased scattering from suspended particles (allochthonous and autochthonous) as within-lake mixing continued and streamflows commenced might have attributed to the light reduction. Epilimnetic turbidity values were at their highest ranging from about 1 to 1.5 NTUs during this period (Figure 5.2), while chloropyll *a* and phytoplankton biovolume (discussed below) were relatively low.



* Area @ 100% = 1.11e6 square meters
Volume @ 100% = 8.62e6 cubic meters

Figure 5.1. Hypsographic curves showing depth-volume and depth-area relationships in Lake Sawyer.

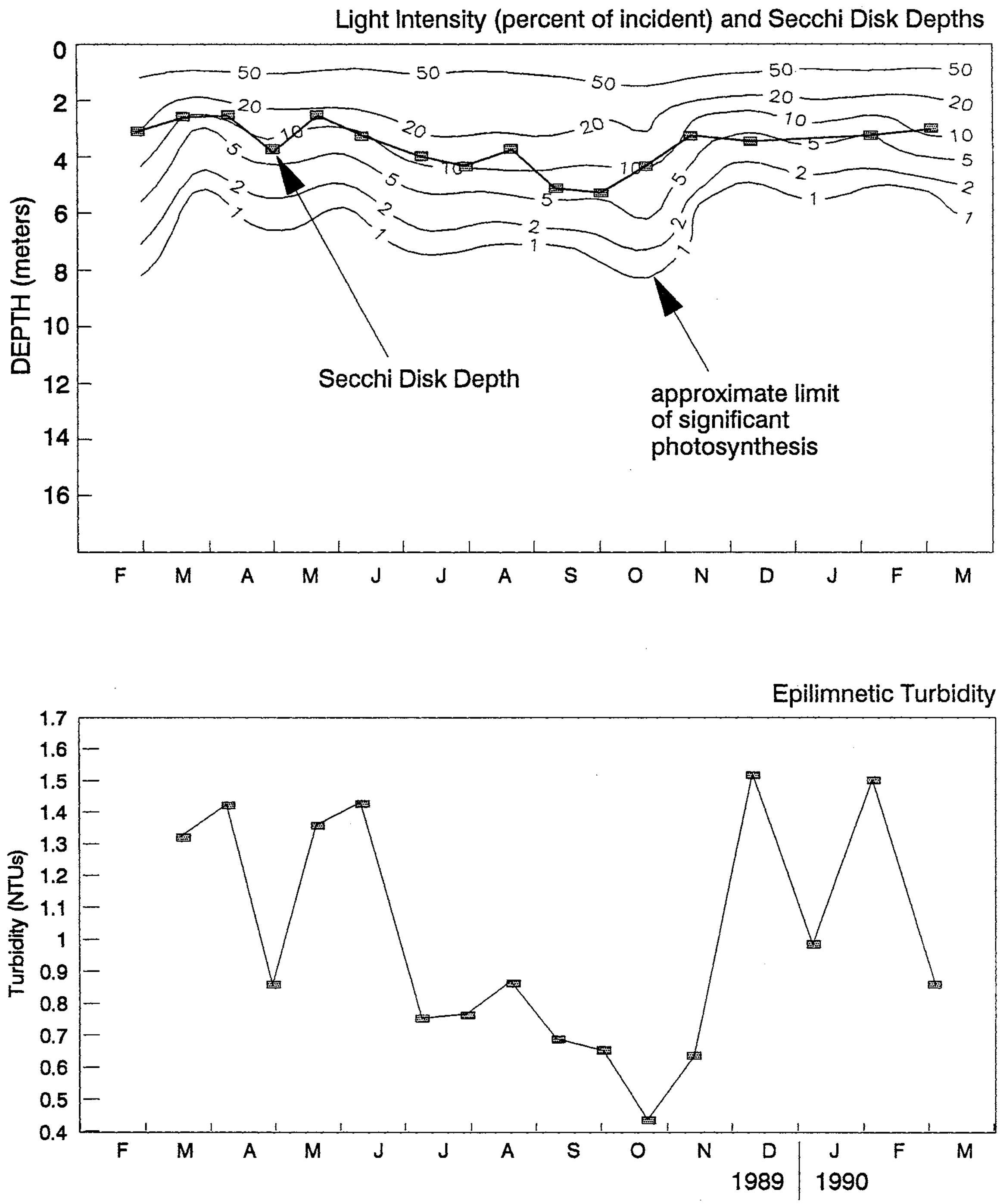


Figure 5.2. Percent incident light isopleths, Secchi disk depths and epilimnetic turbidity in Lake Sawyer.

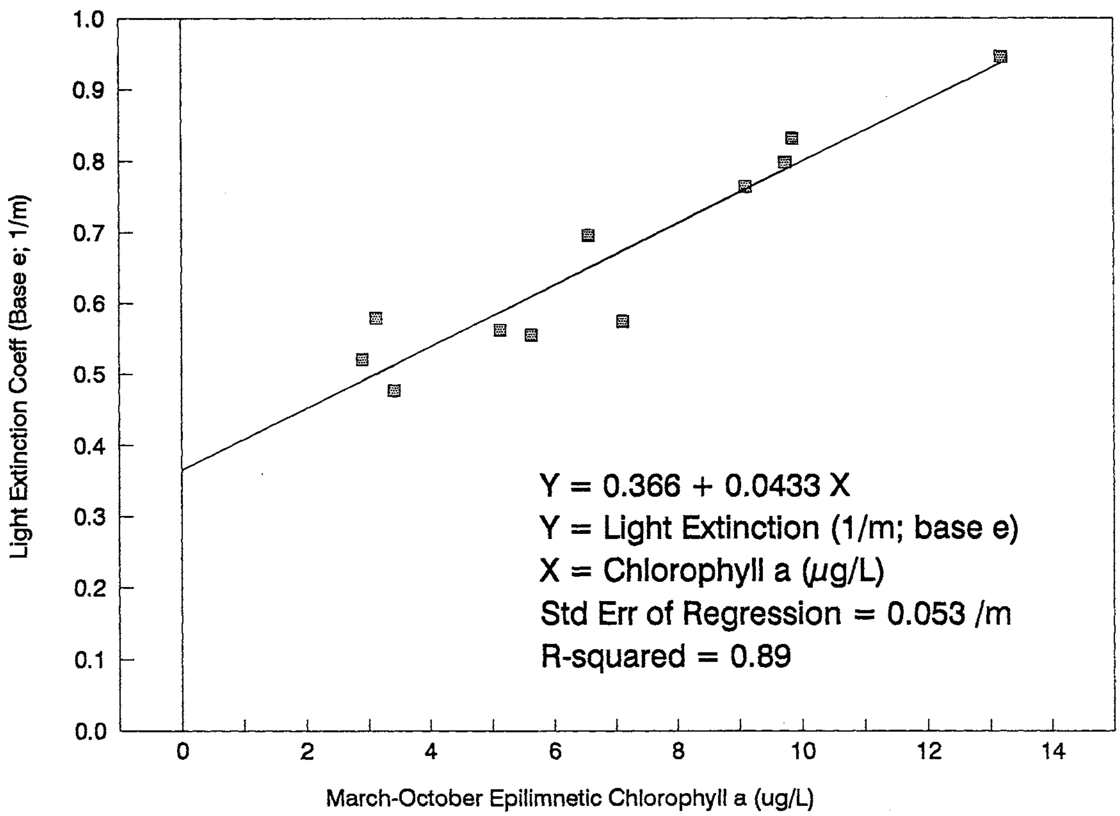


Figure 5.3. Relationship between light extinction and chlorophyll a during March through October, 1989.

The lower depth limit that permits photosynthesis (e.g. the euphotic zone) is approximated by the one percent incident light level (Cole, 1979). During the growing season (March to October), this lower limit of the euphotic zone usually extended to the seven meter depth (i.e., approximately the thermocline) which probably eliminates light as a limiting factor of growing season productivity.

5.1.3 Temperature

Lake Sawyer is classified as a warm, monomictic lake and therefore usually has only one circulation during the year. It stratifies directly in the spring and circulates freely in the winter at a temperature of four to six degrees Celsius. Thermal stratification occurs because of uneven heating of the water column from solar radiation which results in water density gradients. Mixing in the upper layers due to wind results in the uniformly warm upper epilimnion. Below lies the colder and stagnant hypolimnion with the metalimnion separating the two regions. For analysis, Lake Sawyer was treated as two compartments during stratification having an epilimnion beginning at the lake surface and extending to 7.5 meters and a hypolimnion starting at 7.5 meters and extending to the lake bottom.

Figure 5.4 presents temperature isotherms in a time-depth series in Lake Sawyer during the study period. Stratification developed within the lake beginning in April, 1989. Lake Sawyer experienced stable thermal stratification until almost November. Reduced solar insolation and a seasonal shift to stronger southerly winds began to erode the stratification in October and continued doing so until complete overturn had taken place by the beginning of December. There was no ice cover during the winter of 1989-90 and Lake Sawyer was observed to completely mix through the winter until re-establishment of thermal stratification began in March, 1990 (Figure 5.4).

The extreme temperatures observed in Lake Sawyer during the study period were 21.6°C on July 31, 1989 at the Station 3 water surface and 3.0°C on February 27, 1989, at the 16 meter depth of Station 3. Figure 5.5 presents the volume-weighted average epilimnetic and hypolimnetic water temperatures at Lake Sawyer during the study year. Describing the yearly cycle once again, the epilimnetic and hypolimnetic temperatures diverged rapidly beginning in March with the onset of stratification and then began converging in September until complete mixing was achieved in December.

A measure of lake stratification stability is the vertical diffusion coefficient at the thermocline, which represents the rate of vertical mixing. Vertical diffusion at the 7.5 meter depth in Lake Sawyer was calculated based on heat balance and temperature gradients (Thomann and Mueller, 1987) as:

$$V_h * \frac{\Delta T_h}{\Delta t} = \frac{E_t A_t}{Z_t} * (T_e - T_h) \qquad \text{(eqn 5.1)}$$

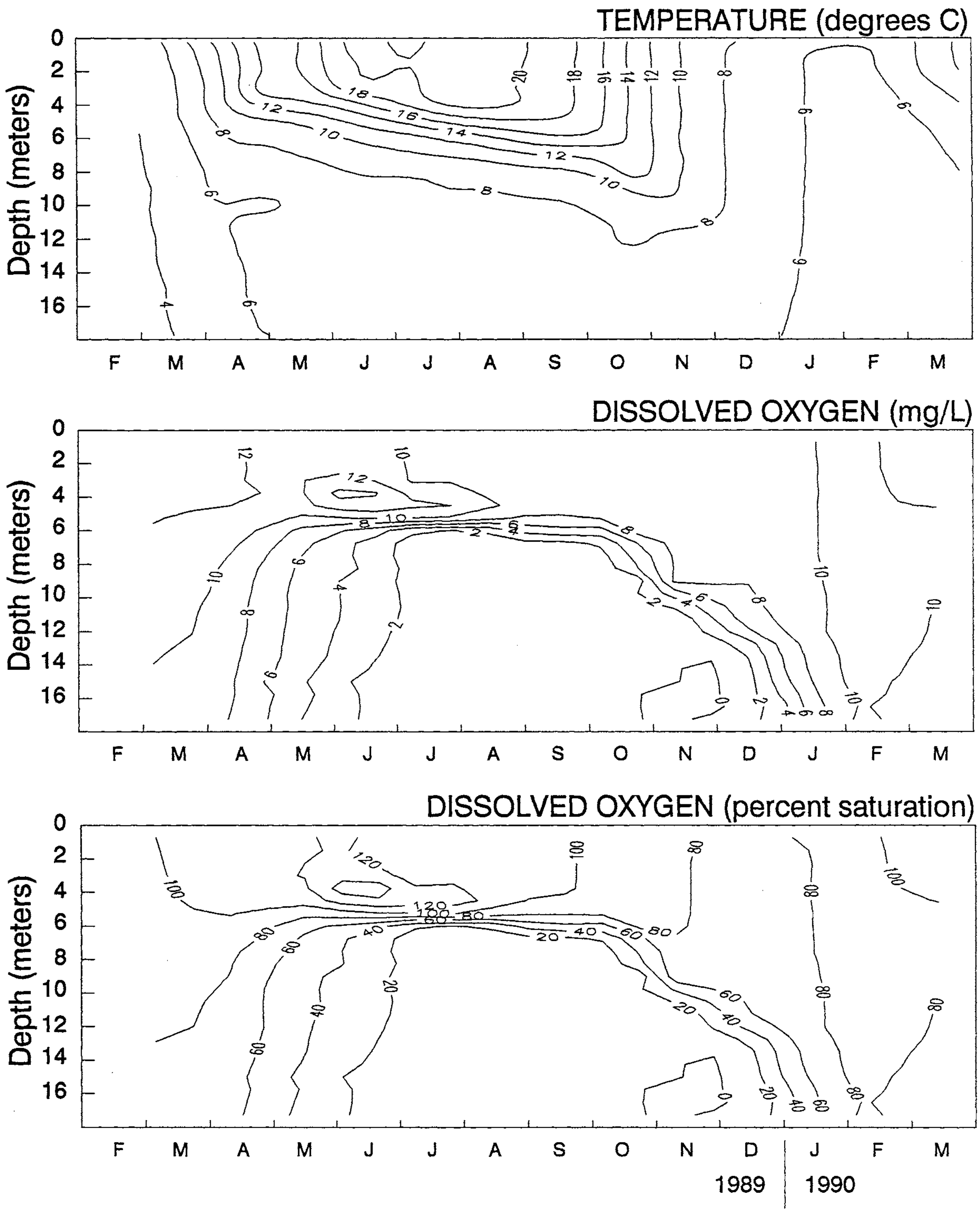


Figure 5.4. Temperature and dissolved oxygen isopleths in Lake Sawyer, February 1989 to March 1990.

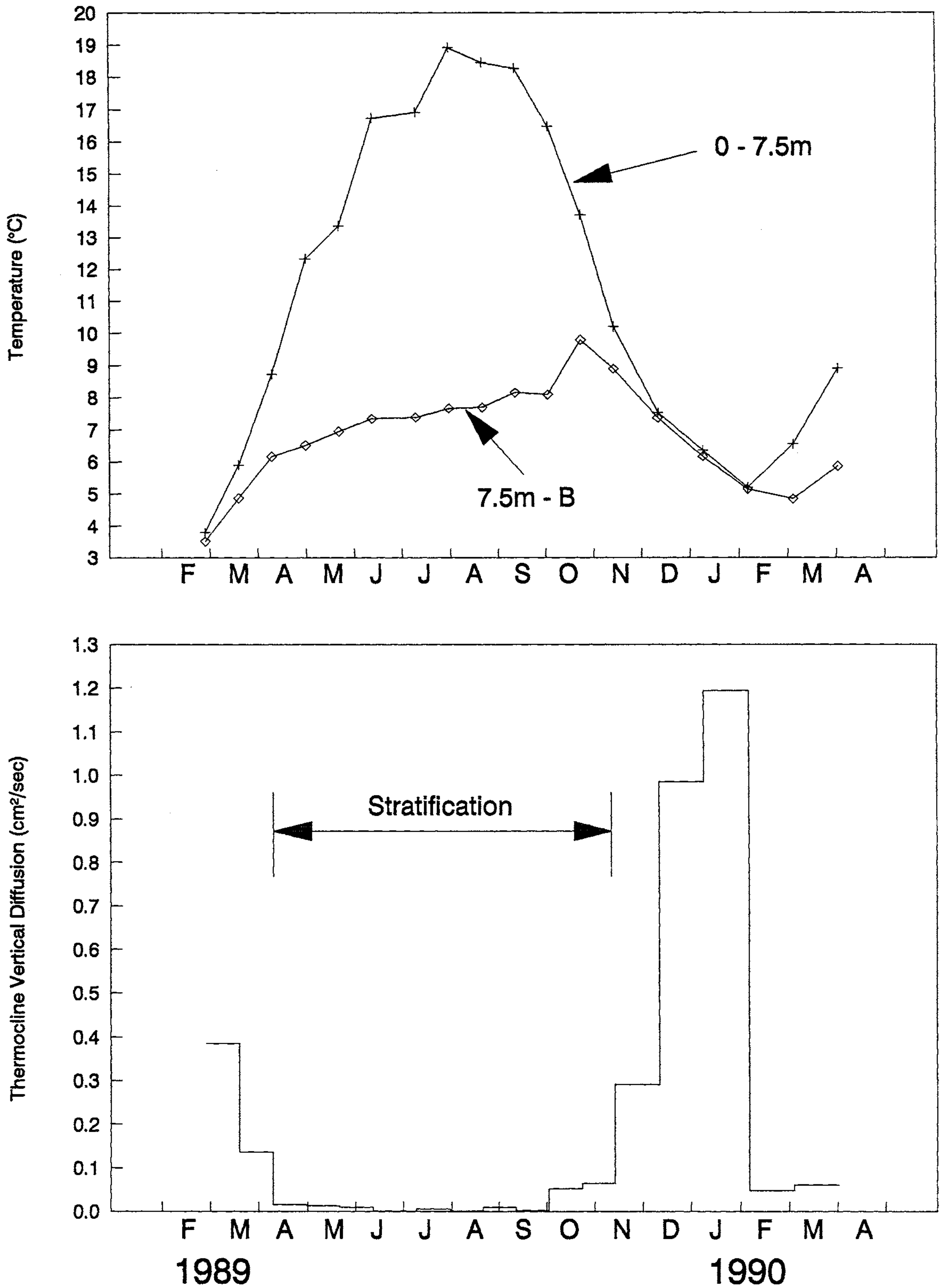


Figure 5.5. Temperature in the epilimnion (0-7.5m) and hypolimnion (7.5m-B), and calculated vertical diffusion coefficients across the thermocline (7.5m).

where V_h is the volume of the hypolimnion; T_h and T_e are the volume-weighted temperatures (°C) of the hypolimnion and epilimnion, respectively; A_t is the area of the 7.5 meter contour; and Z_t is the depth from midpoint of the epilimnion to midpoint of the hypolimnion. The equation above can be rearranged to solve for the bulk vertical mixing coefficient (E_t):

$$E_t = \frac{(T_h(t-1) - T_h(t))\, V_h * Z_t}{(T_e - T_h)\, \Delta t * A_t} \qquad \text{(eqn 5.2)}$$

A summary of calculated vertical diffusion coefficients at the thermocline is presented in Figure 5.5. Vertical diffusion across the thermocline abrubtly dropped in April, 1989, and remained stable at near zero until late October or November, 1989. This period of little vertical mixing defines the lake stratification period.

5.1.4 Dissolved Oxygen

The distribution of dissolved oxygen (D.O.) in lake waters has a substantial impact on biological interactions and the solubility of many nutrients (Wetzel, 1983). D.O. concentrations and percent saturation within Lake Sawyer are depicted in Figure 5.4 as concentration isopleths in relation to depth during the study period. Figure 5.6 presents volume-weighted epilimnetic and hypolimnetic D.O. over the course of the study period.

Volume-weighted epilimnetic D.O. concentrations ranged from approximately 7.5 to 12.5 mg/L over the course of the study (Figure 5.6). The highest D.O. concentration reached 17.1 mg/L at the four meter depth of Stations 4 and 5 during the June 12, 1989, survey. This high D.O. concentration coincided with a diatom bloom and probably reflects algal photosynthesis. The majority of the epilimnetic D.O. concentrations (down to the five meter depth) exceeded D.O. saturation from March through September due to algal productivity. Supersaturation was observed consistently from the end of May through July in conjunction with the large diatom bloom ascribed above. Values in excess of 140 percent saturation were observed during this time (Figure 5.4).

Oxygen was depleted in the hypolimnion during stratification. The minimal supply of oxygen in the hypolimnion was rapidly decreased due to oxidation of organic material and various inorganic compounds in the water column as well as sediment oxygen demand (Hutchinson, 1975). The volume-weighted hypolimnetic D.O. concentration was less than 2 mg/L for a four month period from July to October, attaining anoxia near the bottom during August and September (Figure 5.4 and 5.6). Following the 1989 fall overturn, the water column exhibited a D.O. gradient until mid-January, 1990 when the entire water column attained a D.O. concentration of approximately 10 mg/L.

The areal hypolimnetic oxygen deficit rate (HODR) is a measure of the depletion rate of hypolimnetic D.O. per unit area (Cooke *et al.*, 1986). The areal HODR has been shown to be

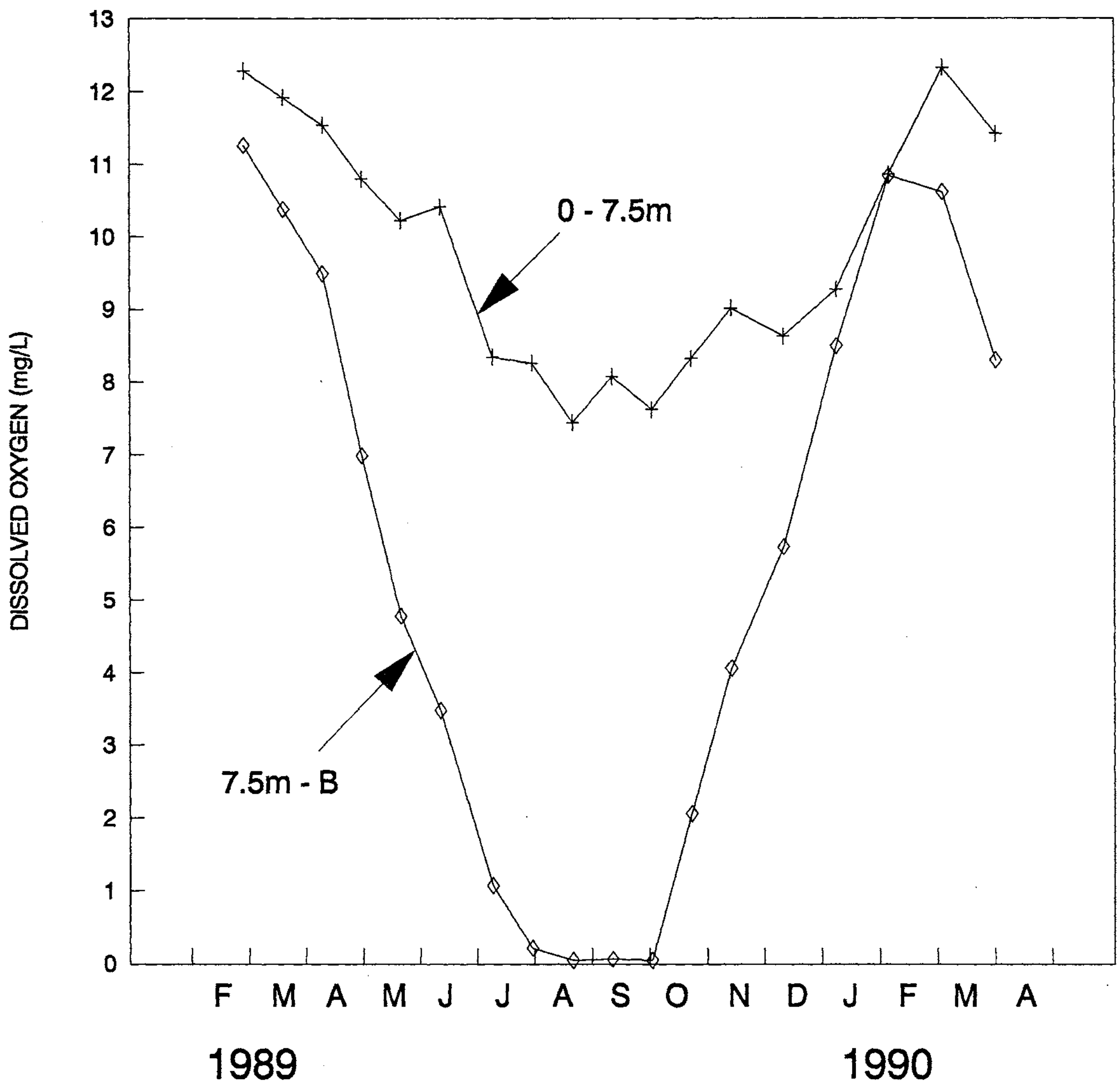


Figure 5.6. Dissolved oxygen in the epilimnion (0 - 7.5m) and hypolimnion (7.5m - B) of Lake Sawyer.

correlated to trophic status and phosphorus loading (Welch and Perkins, 1979). The HODR for Lake Sawyer during the period from March through June, 1989 when the D.O. concentration in the hypolimnion plummeted (Figure 5.6) was estimated to be 0.375 $\pm$ 0.142 g/m^2/day. Mortimer (1942) reported trophic state threshholds based on areal HODR of <0.250 g/m^2/day for oligotrophic conditions and >0.550 g/m^2/day for eutrophic conditions. The study year HODR at Lake Sawyer falls between the two, suggesting a mesotrophic condition.

A review of in-lake D.O. data from the USGS survey of Lake Sawyer on May 10 and June 25, 1973 (McConnell *et al.*, 1976), and METRO in-lake D.O. data from May 15 and July 18, 1979 (Brenner, personal communication, 1989)[3], was performed and HODRs calculated for each respective survey. Both surveys represent conditions prior to the Black Diamond WTP start-up. Results were similar to the 1989 study period with calculated HODRs of 0.322 g/m^2/day and 0.377 g/m^2/day for the 1973 and 1979 data, respectively. Both of these earlier HODRs would assign Lake Sawyer a mesotrophic condition using the classification described above. This stable deficit rate, even following increased nutrient loading (discussed below) is probably the result of the relatively high sediment oxygen demand.

The HODRs presented above have been measures of net hypolimnetic oxygen deficits and include both hypolimnetic D.O. loss due to hypolimnetic oxidation and hypolimnetic D.O. replenishment due to diffusion of dissolved oxygen from the epilimnion to the hypolimnion. Applying measured vertical diffusion coefficients (section 5.1.2), the HODR for the study period (March through June, 1989) was corrected for diffusion (Chapra and Reckhow, 1983). The resultant gross HODR was calculated to be 0.502 $\pm$ 0.096 g/m^2/day and is a direct estimate of the total hypolimnetic oxidation (i.e. hypolimnetic water column and sediment respiration). Gross HODR prior to WTP start-up was probably not significantly different from 1989 since diffusion processes are probably relatively consistent from year to year.

5.1.5 pH, Alkalinity, Conductivity

The pH in Lake Sawyer ranged from 6.0 to 8.4 during the study period (Figure 5.7). Periods of highest pH were associated with periods of high algal productivity (April through June). Lake Sawyer had low to moderate alkalinities ranging from 40.0 to 61.9 mg/L as $CaCO_3$ during the study period. This is typical for Western Washington lakes due to the lack of sedimentary carbonate geologic formations in the region. Conductivity in Lake Sawyer ranged from 116 to 184 μmhos at 25°C during the study period.

5.1.6 Iron

Total iron (Fe) in water was measured at the six meter depth and 15 meter depth at both Station 3 and 4 during each lake survey. Figure 5.8 presents a time-series plot of the average Fe concentration for each of the two depths. At the beginning of the study year both six meter and

---

[3]Bob Brenner, Municipality of Metropolitan Seattle, 1989.

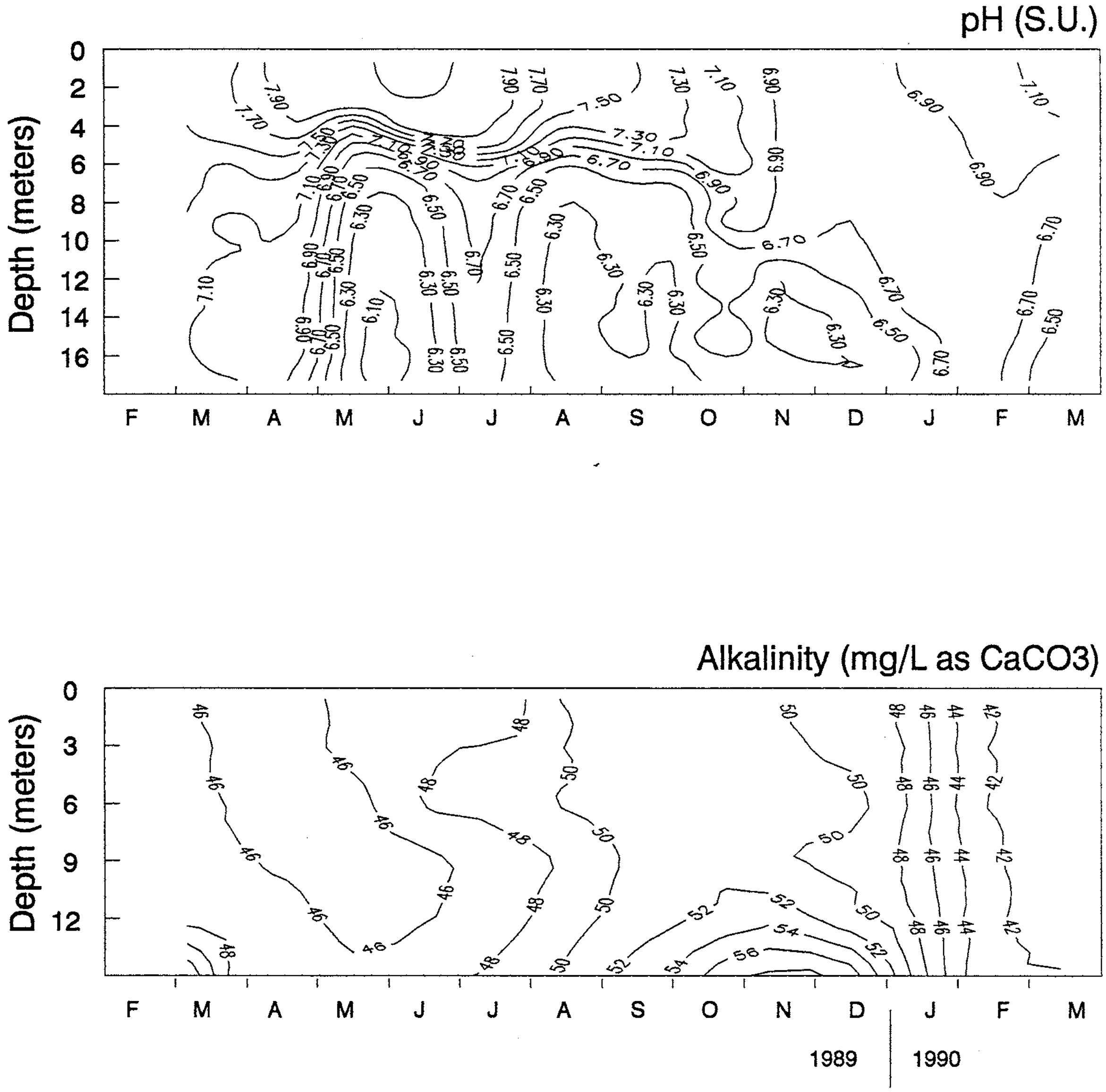


Figure 5.7. Isopleths of pH and alkalinity in Lake Sawyer, March 1989 through March 1990.

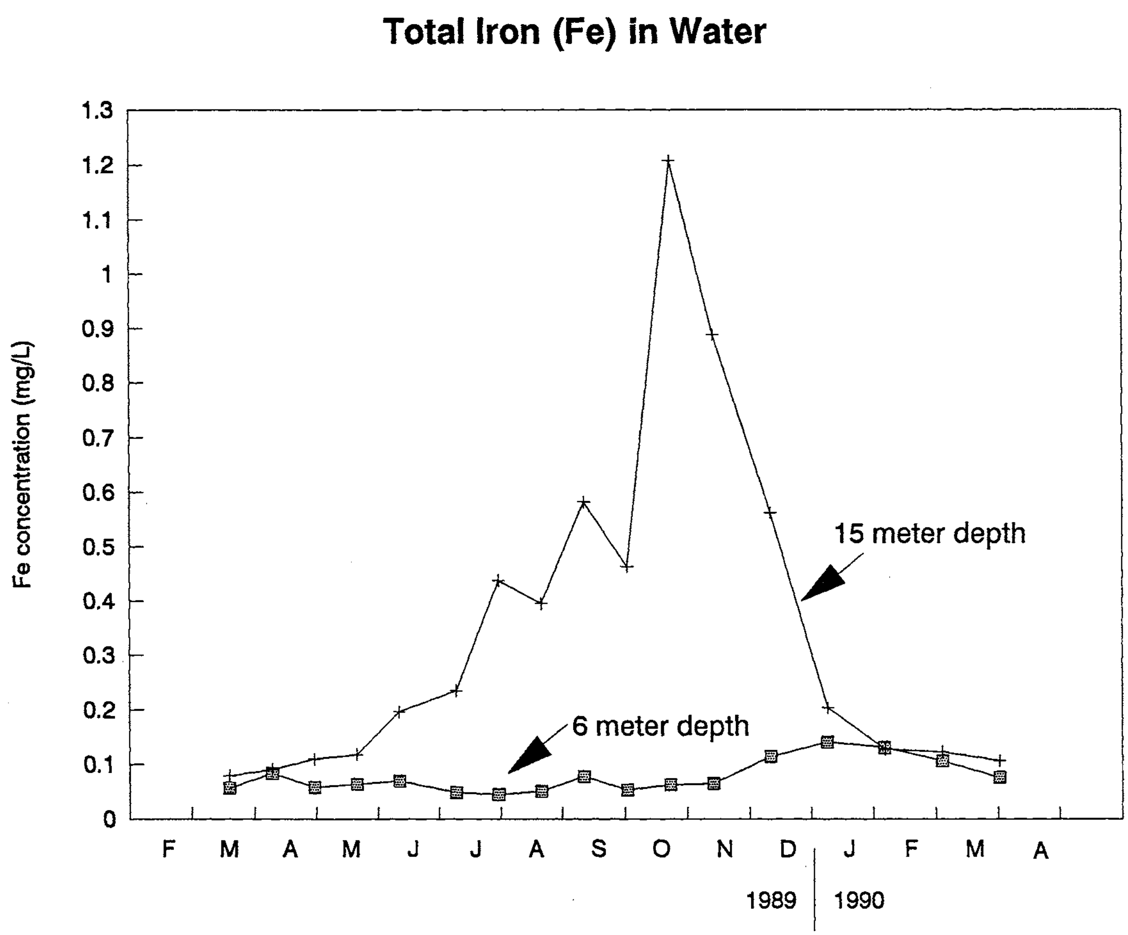


Figure 5.8. Epilimnetic and hypolimnetic total iron concentrations in Lake Sawyer.

15 meter concentrations averaged <0.1 mg Fe/L. Iron concentrations at the 15 meter depth (i.e. just above the sediments) showed a dramatic increase as the hypolimnion became anoxic during lake stratification. As D.O. concentrations dropped below 2 mg/L, iron in the sediments changed from an oxidized form (Fe^{+++}) to a reduced form (Fe^{++}). The oxidized form is insoluble and accumulates in the lake sediments, whereas the reduced form of iron is soluble and is released from the sediments into the overlying waters during anaerobic conditions (Wetzel, 1983).

The maximum iron concentration of almost 1.8 mg/L occurred in late October. Decreases in total iron concentrations in the lower hypolimnion in early October and in November during stratification (Figure 5.8) might be related to very low oxidation/reduction potentials of less than -50 mV measured at the 15 meter depth on those dates. When redox potentials extend this low, sulfate is reduced to sulfide. Much of the sulfide, in addition to forming hydrogen sulfide, combines with iron as ferrous sulfide (FeS), an insoluble precipitate in the water column. A strong sulfide malodor was present in the sample water drawn from the lower hypolimnion throughout this period also indicating the presence of the reduced sulfide compounds.

The concentration of total iron at the 15 meter depth continued to drop dramatically from November to January until it attained equilibrium with epilimnetic iron concentration near the 0.1 mg/L level. As aerobic waters began to circulate into the hypolimnion during this period (i.e. destratification), the reduced iron in the hypolimnion probably oxidized rapidly and precipitated out of the water column. The epilimnetic (6 meter) iron concentration averaged 0.076 ± 0.007 mg/L, remaining stable for the majority of the year with a only a slight increase (e.g. slightly >0.1 mg/L) noticeable during turnover.

The redox changes in the ionic state of iron play an important role in the internal cycling of phosphorus within Lake Sawyer and, to some degree, will buffer the effectiveness of any external restoration technique (e.g. diversion). The oxidized form of iron binds phosphorus and holds it in the lake sediments where it is not available for biological uptake. When the iron is reduced, phosphorus is released to the overlying waters making it available for uptake. Internal loading can be a significant part of a lake phosphorus budget.

5.1.7 Nutrients

Total phosphorus (TP) and total nitrogen (TN) were routinely measured in Lake Sawyer during the study period. Both are measures of all forms of each respective constituent (i.e. organic and inorganic, particulate and dissolved). In addition, the soluble fraction of phosphorus available for algal uptake was measured as soluble reactive phosphorus (SRP) and the soluble fractions of nitrogen available for algal uptake were measured as nitrate plus nitrite nitrogen and ammonia nitrogen. The dissolved inorganic fraction of nitrogen (DIN) is the summation of nitrate plus nitrite nitrogen and ammonia nitrogen. All concentrations of phosphorus and nitrogen forms in this report are expressed as P and N, respectively.

Table 5.1. Summary of volume-weighted and time-weighted epilimnetic and whole lake mean nutrient concentrations in Lake Sawyer during growing season and study year. (all values in ug/L as P for TP and SRP; as N for TN and DIN)

	TP	TN	SRP	DIN
GROWING SEASON WHOLE-LAKE CONCENTRATION(1)	20.4 ± 2.7	401 ± 17	8.7 ± 2.0	198 ± 13
ANNUAL WHOLE-LAKE CONCENTRATION(2)	25.7 ± 3.1	416 ± 20	13.0 ± 2.1	217 ± 13
GROWING SEASON EPILIMNETIC CONCENTRATION	17.7 ± 1.2	366 ± 14	5.8 ± 0.7	139 ± 13
ANNUAL EPILIMNETIC CONCENTRATION	22.2 ± 1.2	391 ± 17	9.8 ± 1.3	175 ± 15

(1) GROWING SEASON FROM MARCH 1989 THROUGH OCTOBER 1989

(2) ANNUAL PERIOD IS FROM MARCH 1989 THROUGH FEBRUARY 1990

Volume-weighted, whole-lake mean TP and SRP concentrations were 25.7 ± 3.1 μg/L and 13.0 ± 2.1 μg/L respectively during the study year (March 1989 through February 1990). Volume-weighted mean epilimnetic TP and SRP concentrations were 22.2 ± 1.2 μg/L and 9.8 ± 1.3 μg/L, respectively, during the study year (Table 5.1). Minimum concentrations of TP and SRP generally occurred in the epilimnion during the growing season (March through October), probably resulting from algal uptake and settling. The lowest growing season (March through October) volume-weighted epilimnetic TP concentratation observed (13.9 ± 1.2 μg/L) occurred on September 11, 1989.

Concentrations of TP and SRP were greatest in the hypolimnion during the latter part of stratification (October and November) when the hypolimnion became anoxic (Figure 5.9). Volume-weighted mean hypolimnetic concentrations of TP and SRP were 79.0 ± 25.1 μg/L and 64.6 ± 24.1 μg/L respectively on October 23, 1989. Maximum concentrations of TP (383 μg/L) and SRP (364 μg/L) occurred at 15 meter depth of Station 4 on this date. High values (>100 μg/L) for TP and SRP were also found in the basin of Station 3 during this time. This increase in concentration can be attributed to the reduction of ferric iron to ferrous iron in the anaerobic hypolimnion resulting in the release of bound phosphorus (section 5.1.7). Phosphorus was generally released from sediment as a biologically available form as indicated by the high SRP at the time.

Figure 5.10 presents the volume-weighted hypolimnetic and epilimnetic concentrations of TP and SRP. From the beginning of the study period (February 1989), the lake was well mixed with volume-weighted epilimnetic TP and SRP approximately equalling volume-weighted hypolimnetic TP and SRP, respectively. Volume-weighted hypolimnetic TP and SRP diverged from epilimnetic concentrations beginning in July until late October, correlating to the temporal trend of iron concentration (Figure 5.8). Epilimnetic TP and SRP concentrations remained stable during this same period (July through October). Beginning with destratification in October and November, hypolimnetic TP and SRP were entrained to the epilimnion. External P loads from the tributaries also began during this period. Volume-weighted epilimnetic TP and SRP concentrations quickly began to rise and volume-weighted hypolimnetic TP and SRP concentrations continued to decline until equilibrium whole-lake concentrations of 40.1 ± 2.3 μg/L of TP and 26.1 ± 1.9 μg/L of SRP were observed on January 8, 1990, when the entire water column was once again well-mixed.

TP and SRP concentrations achieved equilibrium in response to a complex combination of factors including changing internal and external loads (possibly dilution by lower concentration incoming waters), sedimentation and lake flushing. Phosphorus uptake and settling of a diatom bloom contributed to the phosphorus sedimentation at the beginning and end of the study period. Following whole-lake mixing, TP and SRP concentrations continued to decrease. By April 1990, whole-lake TP and SRP concentrations approached the previous growing season's epilimnetic averages.

The volume-weighted whole-lake mean annual concentration of TN and DIN (for study year March 1989 through February 1990) were respectively, 416 ± 20 μg/L and 217 ± 15 μg/L

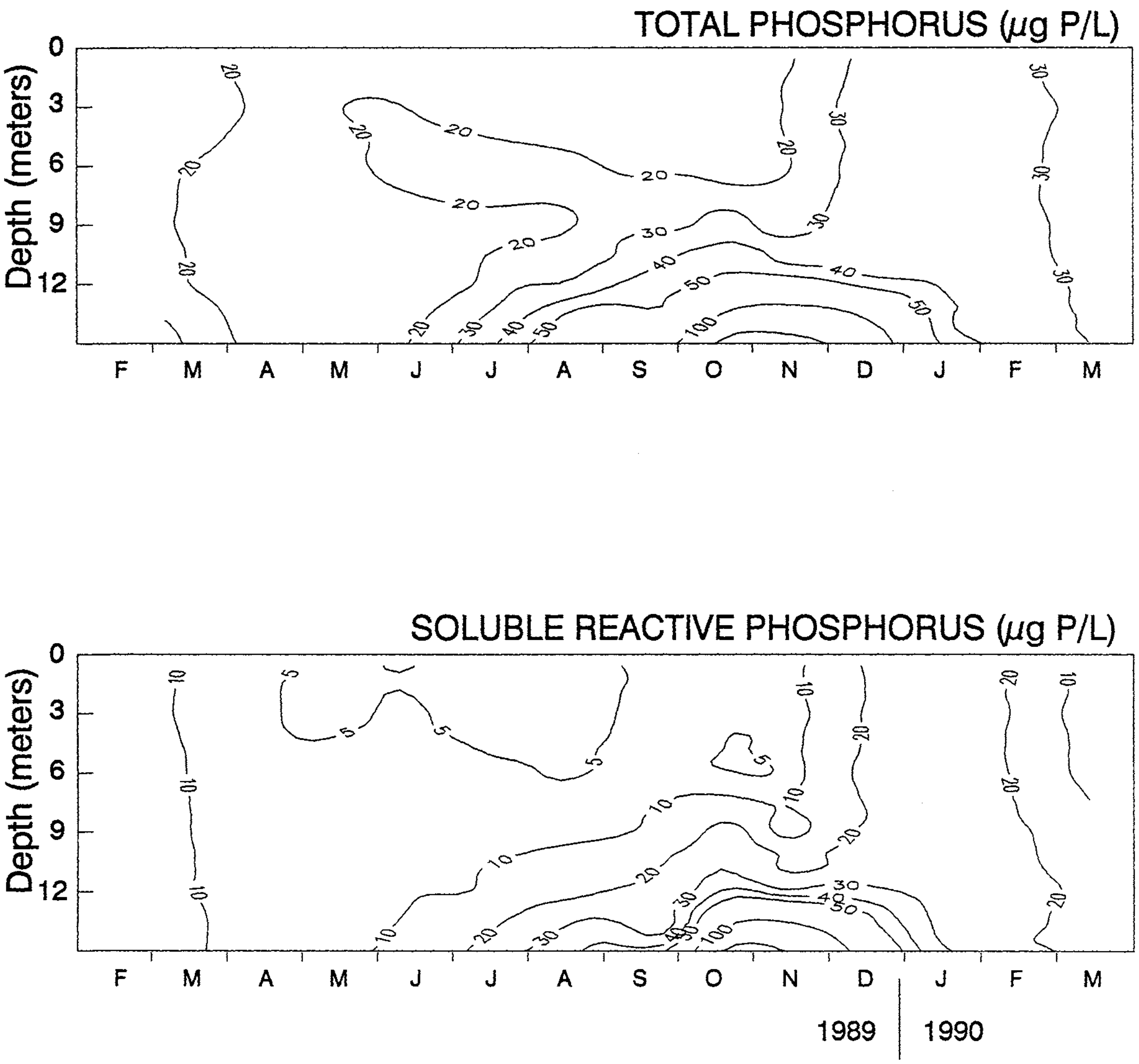


Figure 5.9. Total phosphorus and soluble reactive phosphorus isopleths in Lake Sawyer, February 1989 to March 1990.

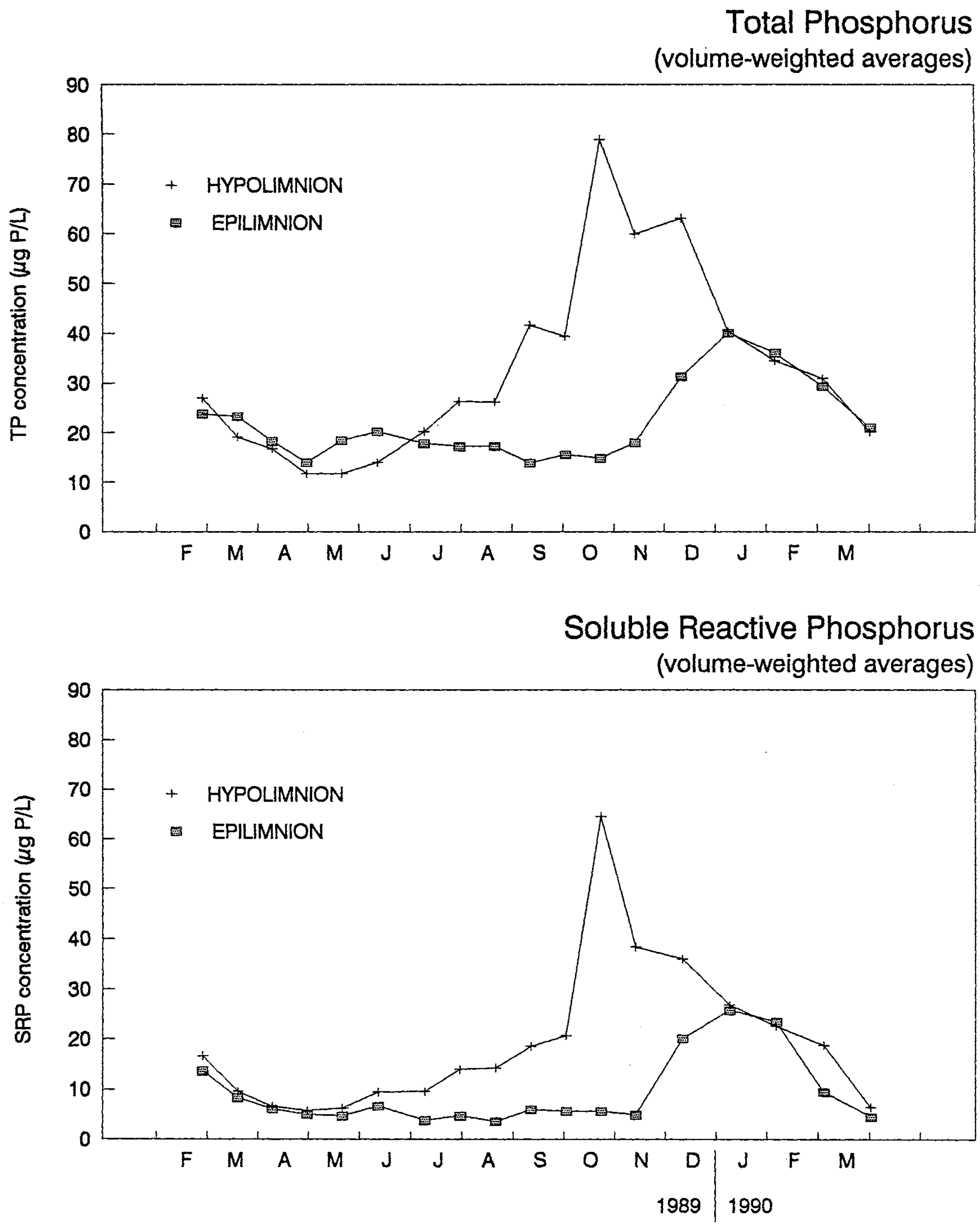


Figure 5.10. Epilimnetic and hypolimnetic total P and SRP in Lake Sawyer from February 1989 through March 1990.

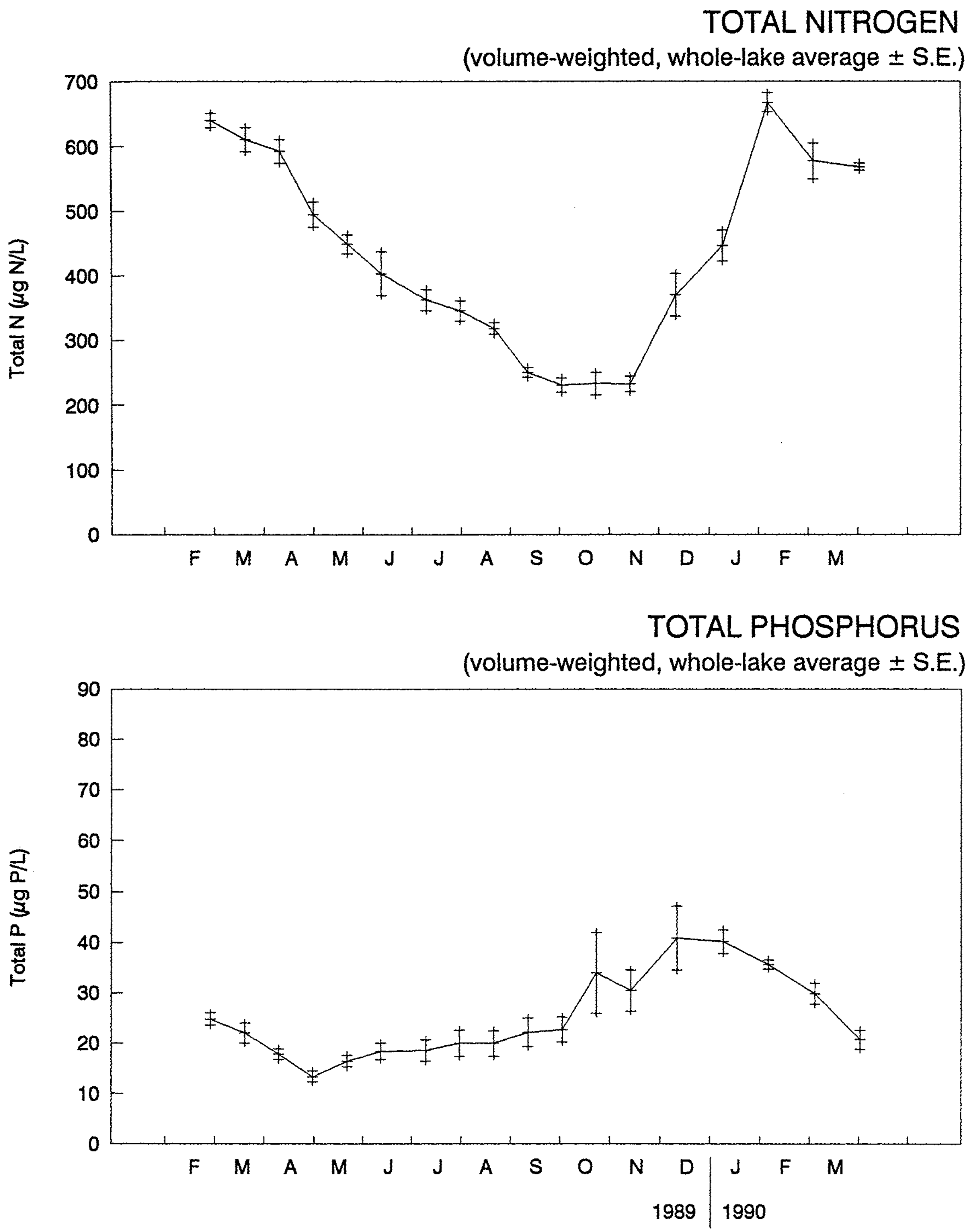


Figure 5.11. Whole-lake total P and total N in Lake Sawyer from February 1989 through March 1990.

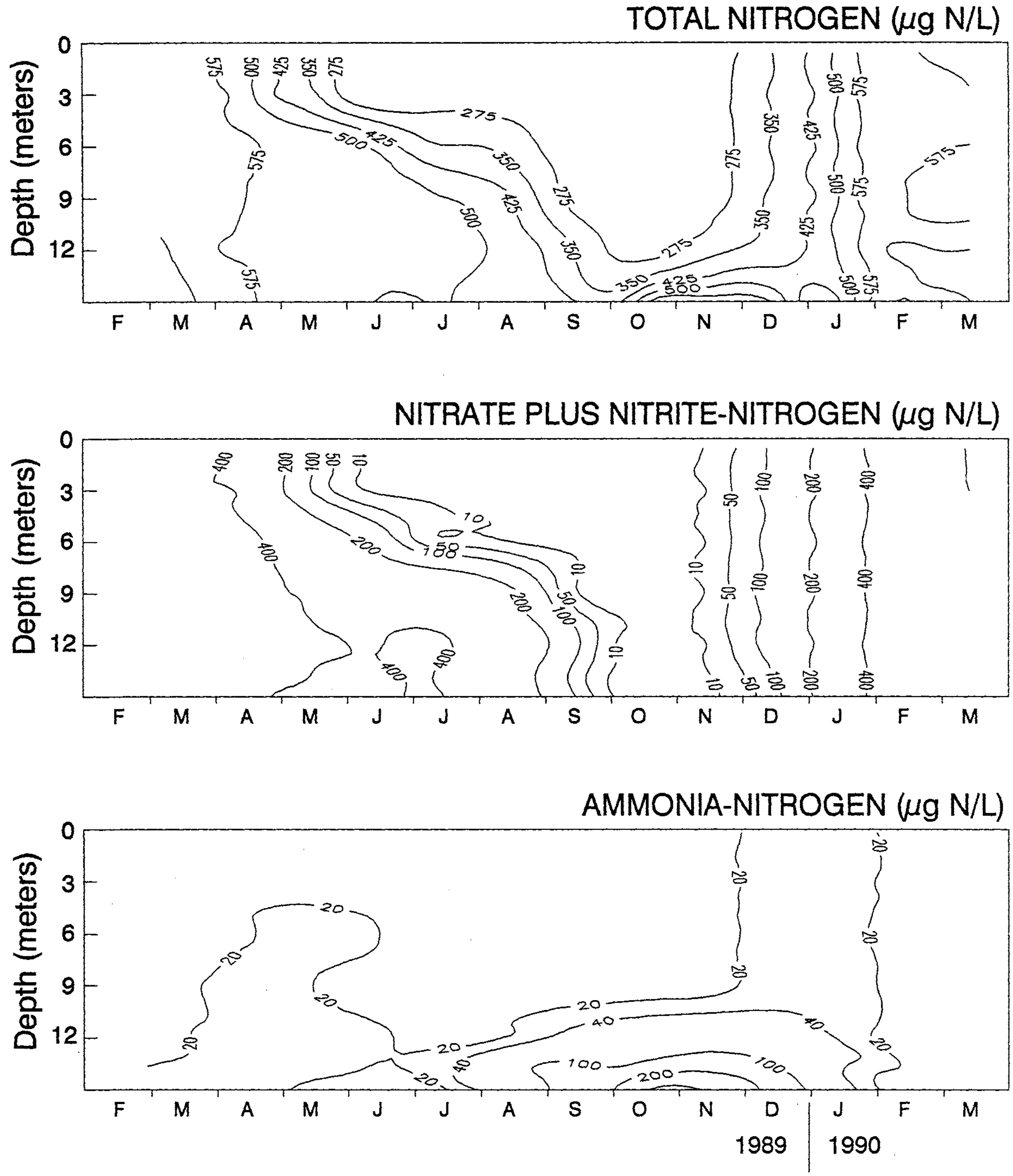


Figure 5.12. Total nitrogen, nitrate plus nitrite as nitrogen and ammonia as nitrogen isopleths in Lake Sawyer from February 1989 to March 1990.

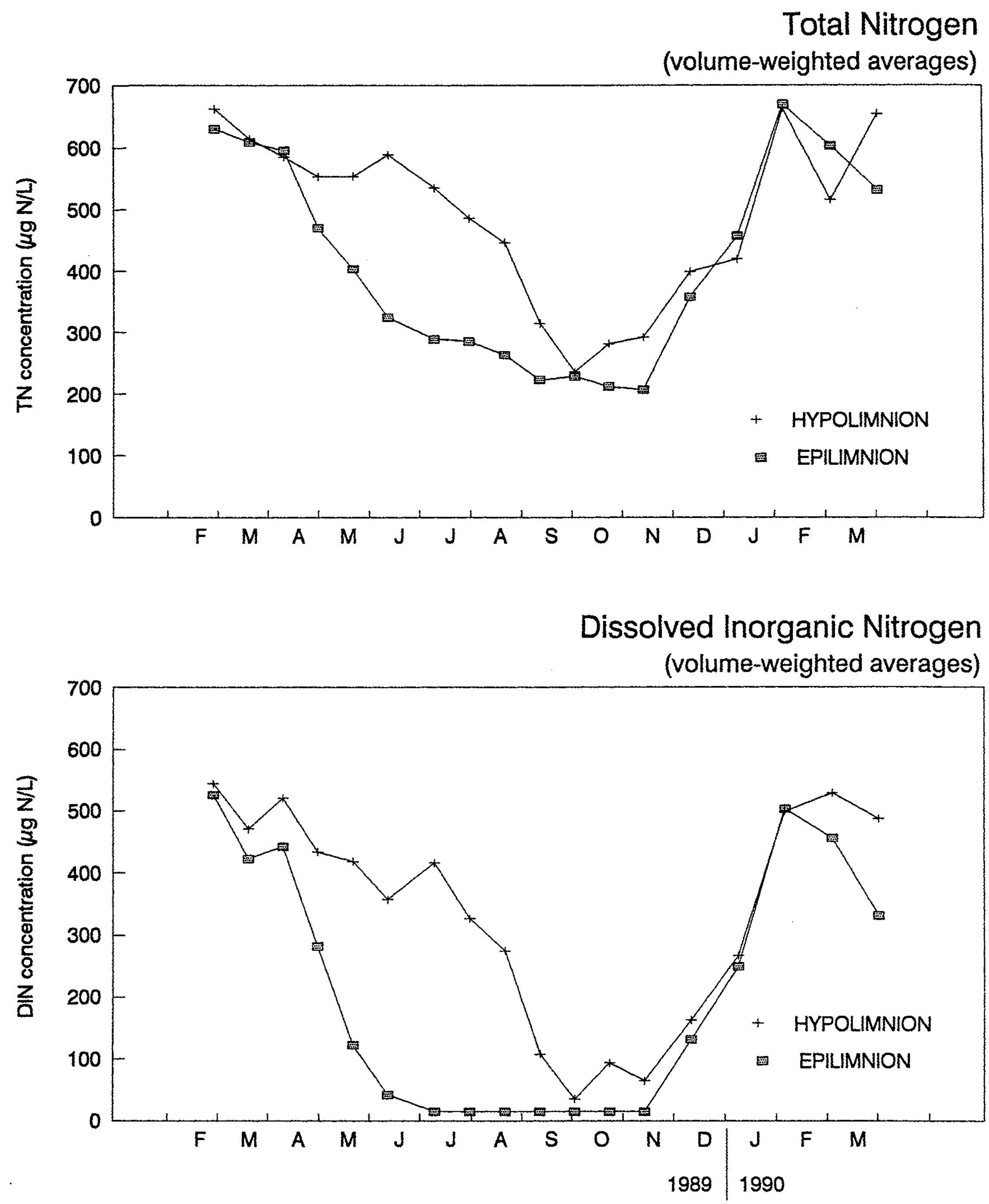


Figure 5.13. Epilimnetic and hypolimnetic total N and DIN in Lake Sawyer from February 1989 through March 1990.

(Table 5.1). TN lake concentrations exhibited a wide range from a volume-weighted whole-lake average of 231 ± 11 μg/L on October 2, 1989 to 667 ± 14 μg/L on February 5, 1990. TN concentrations were typically greatest in the winter during the highest inflows and following turnover (Figure 5.11). The lowest TN concentrations were observed in September through October during stratification, probably resulting from algal uptake and sedimentation.

The volume-weighted epilimnetic mean DIN concentrations during the study year (March 1989 through February 1990) and the growing season (March 1989 through October 1989) were 175 ± 15 μg/L and 139 ± 13 μg/L, respectively (Table 5.1). The entire epilimnion was practically devoid of DIN from July through October (Figure 5.13) as volume-weighted epilimnetic concentrations fell below the lower reporting limit (<15 μg/L). Both nitrate plus nitrite-nitrogen and ammonia-nitogen were depleted below their respective lower reporting limits during this time. This depletion probably represents algal uptake and could possibly have indicated a short-term nitrogen limitation. Nutrient limitation and N:P ratios will be discussed in section 6.1.

In the hypolimnion, DIN concentrations decreased markedly from July through September, though an accumulation of ammonia-nitrogen was noticed in the lower hypolimnion during this same period which is typical of a stratified lake with an anoxic hypolimnion (Wetzel, 1983). In October there seemed to be a small net increase of DIN in the hypolimnion (Figure 5.13) composed mainly of ammonia-nitrogen (Figure 5.12) which would probably be explained by sediment release of nitrogen (discussed in section 7.2). Low redox potentials at the anoxic sediment/water interface can reduce the adsorptive capacity of the sediments for ammonia-nitrogen (Kamiyama, *et al.*, 1977) resulting in a marked release of ammonia-nitrogen from the sediments (Wetzel, 1983; Austin and Lee, 1973). Resuspension from mixing may also have contributed to the sediment release as destratification commenced during October (Wetzel et al, 1972; Davis, 1973).

5.2 Lake Water Biological Characteristics

5.2.1 Chlorophyll *a*

Epilimnetic chlorophyll *a* concentrations are presented in Figure 5.14 as isopleths in a time-depth series and as volume-weighted averages for each sample date. The highest volume-weighted average chlorophyll *a* concentration observed was 13.2 ± 3.3 μg/L occurring on March 20, 1989. The lowest volume-weighted average chlorophyll *a* concentration observed was 2.9 ± 0.6 μg/L occurring near the end of the growing season on September 11, 1989. The lowest concentration correlated with a minimum phytoplankton biovolume measurement, however the highest chlorophyll *a* concentration did not complement the greatest phytoplankton biovolume measurement. In general, though, there was a good relationship between chlorophyll *a* and phytoplankton biovolume (Figure 5.15). The volume-weighted and time-weighted average chlorophyll *a* concentration during the growing season (March through October) was 6.9 ± 1.5 μg/L which did not differ much from the annual volume-weighted average of 6.2 ± 1.5 μg/L.

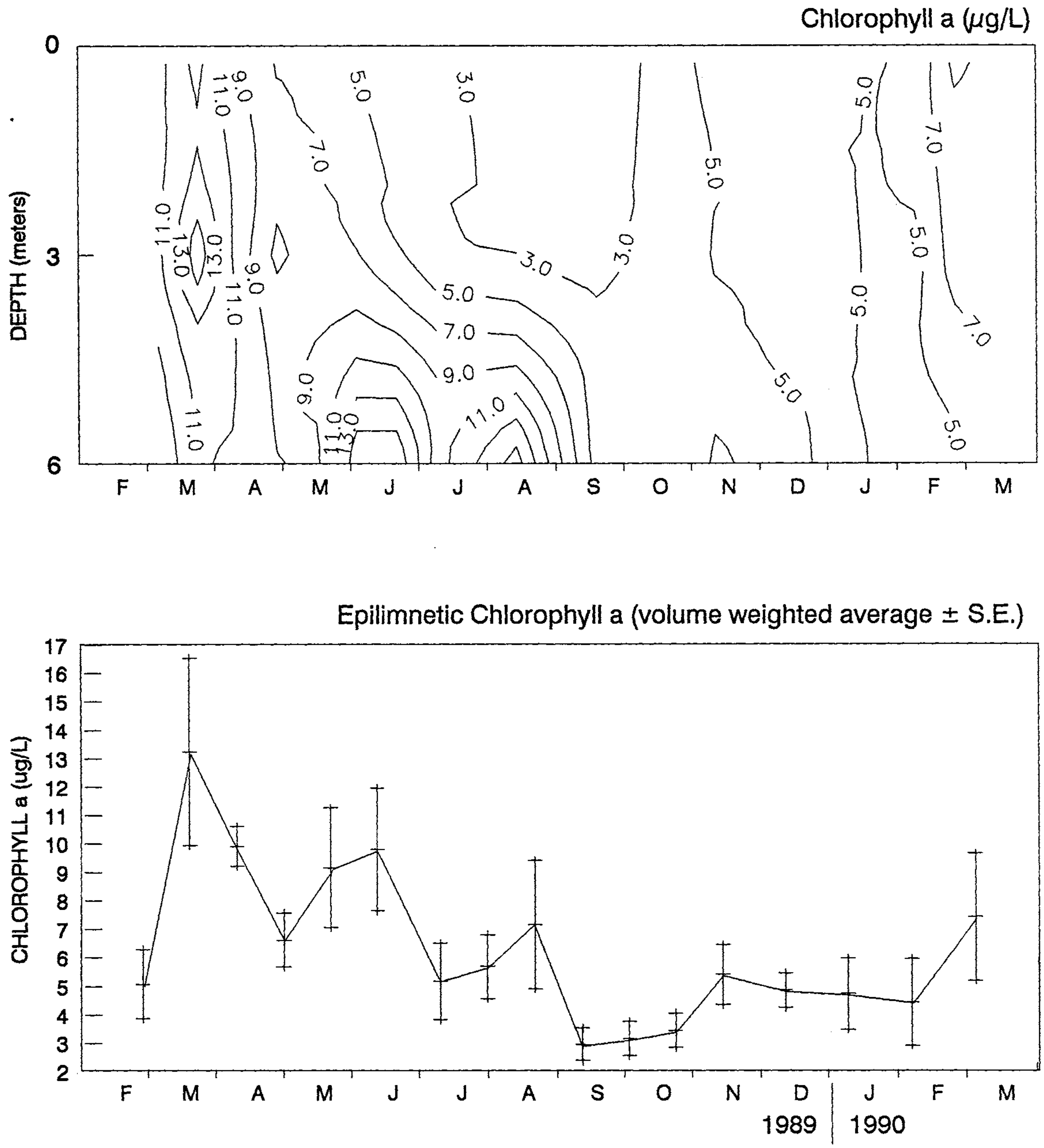


Figure 5.14. Isopleths of chlorophyll a in the epilimnion and volume-weighted average epilimnetic chlorophyll a in Lake Sawyer.

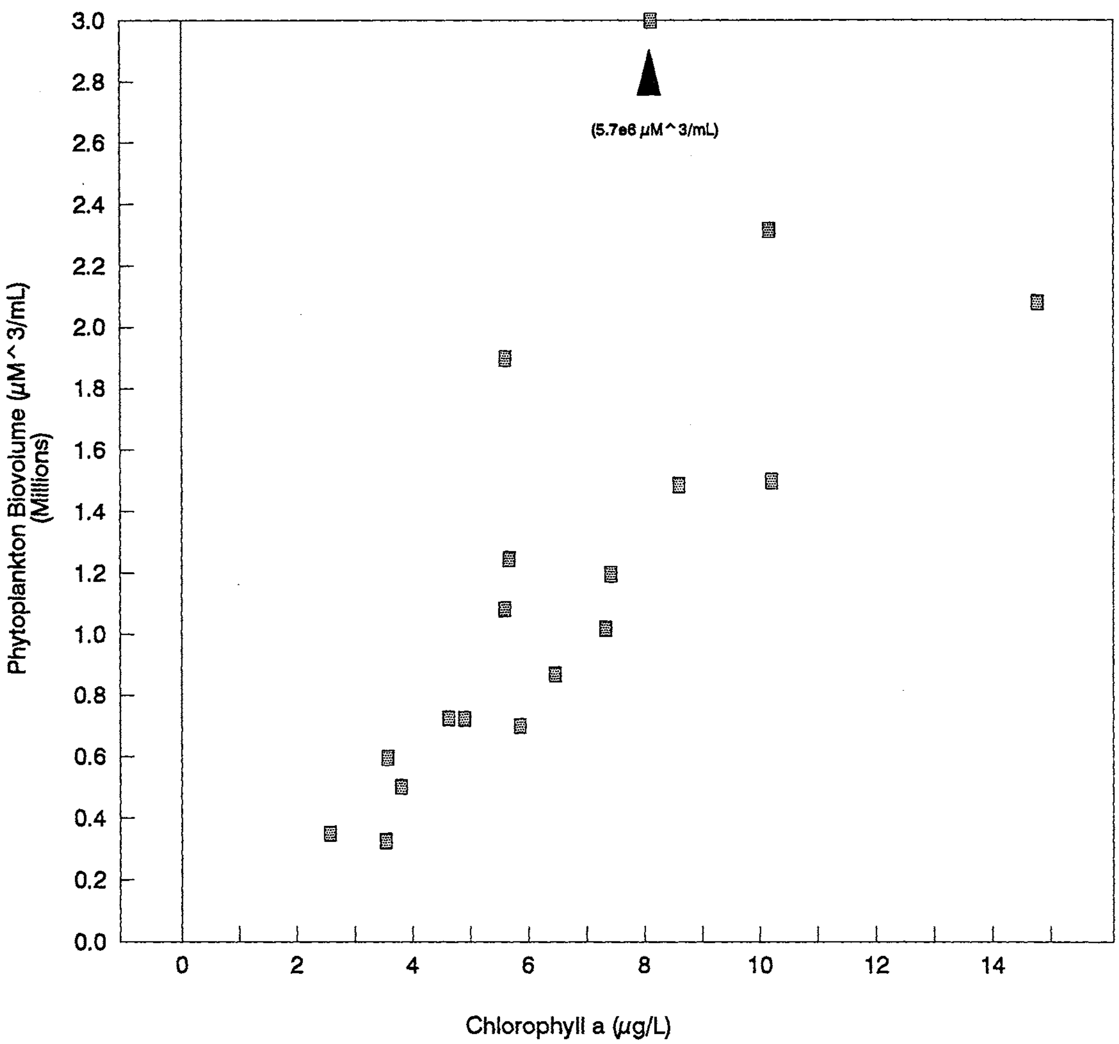


Figure 5.15. Relationship between chlorophyll a and phytoplankton biovolume. in Lake Sawyer from March 1989 to April 1990.

5.2.2 Phytoplankton

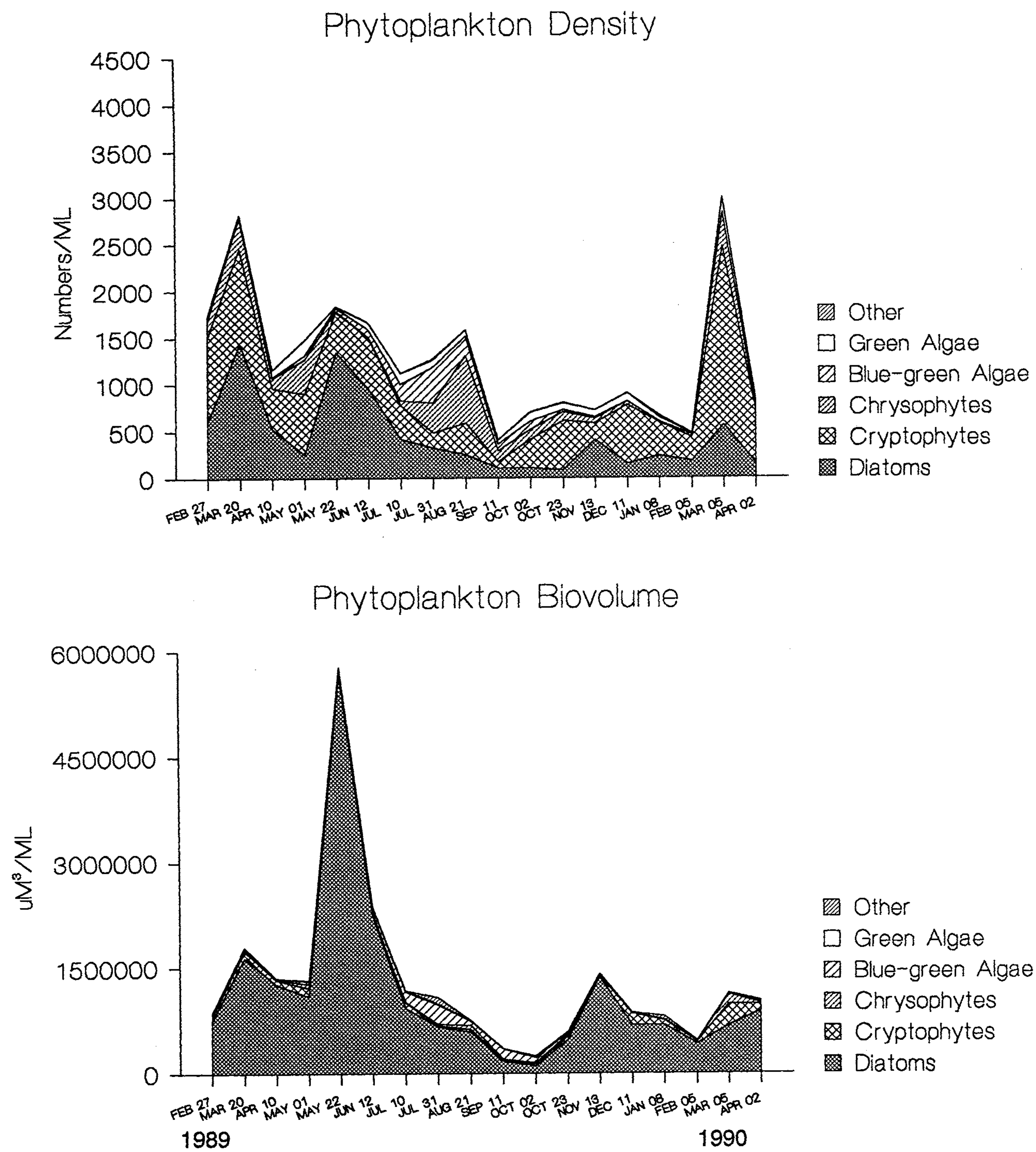
Cumulative densities and biomass of major algal groups at Station 3 and 4 during the study period are presented in Figures 5.16 and 5.17. In general, the two stations exhibited similar trends throughout the study period with exception of significantly larger blue-green algae biomass at Station 4 on August 21, 1989. Most algal groups were represented throughout the study period, but algal biomass during almost the entire period was dominated by diatoms.

Early spring algal outbursts occurred in March of 1989 and 1990 exhibiting the highest observed algal densities (>3000/mL) for both years and the highest volume-weighted chlorophyll *a* concentration (13.2 $\pm$ 3.3 μg/L) in 1989. These blooms were dominated in numbers by *Rhodomonas minuta*, *Stephanodiscus astraea minutula*, and *Mallomonas* sp. though the majority of the algal biomass was attributed to the diatoms *Stephanodiscus astraea*, *Melosira granulata*, *Melosira italica*, and *Tabellaria fenestrata*. All of these diatom species are associated with lakes having high phosphorus content (Sweet, 1986). Increasing light exposure during the onset of stratification was probably the dominant factor to the development of these early populations as water temperatures were still low (Wetzel, 1983). Diatoms are frequently dominant in early spring populations (Pechlaner, 1970).

The greatest algal biomass observed in Lake Sawyer during the study period occurred on May 22, 1989 with biovolumes approximating 5.7 million μm^3/ml at both Stations 3 and 4. This bloom was composed almost entirely of the diatoms *Cyclotella comta* and *Fragilaria crotonensis* and corresponded to a volume-weighted chlorophyll *a* concentration of 9.1 $\pm$ 2.1 μg/L. Both diatom species are associated with mesotrophic to eutrophic lakes (Sweet, 1986). Chlorophyll *a* content within algal cells varies with light intensity and species composition throughout the year. However, chlorophyll *a* is commonly used as a measure of algal biomass (OECD, 1982; Welch and Lindell, 1980).

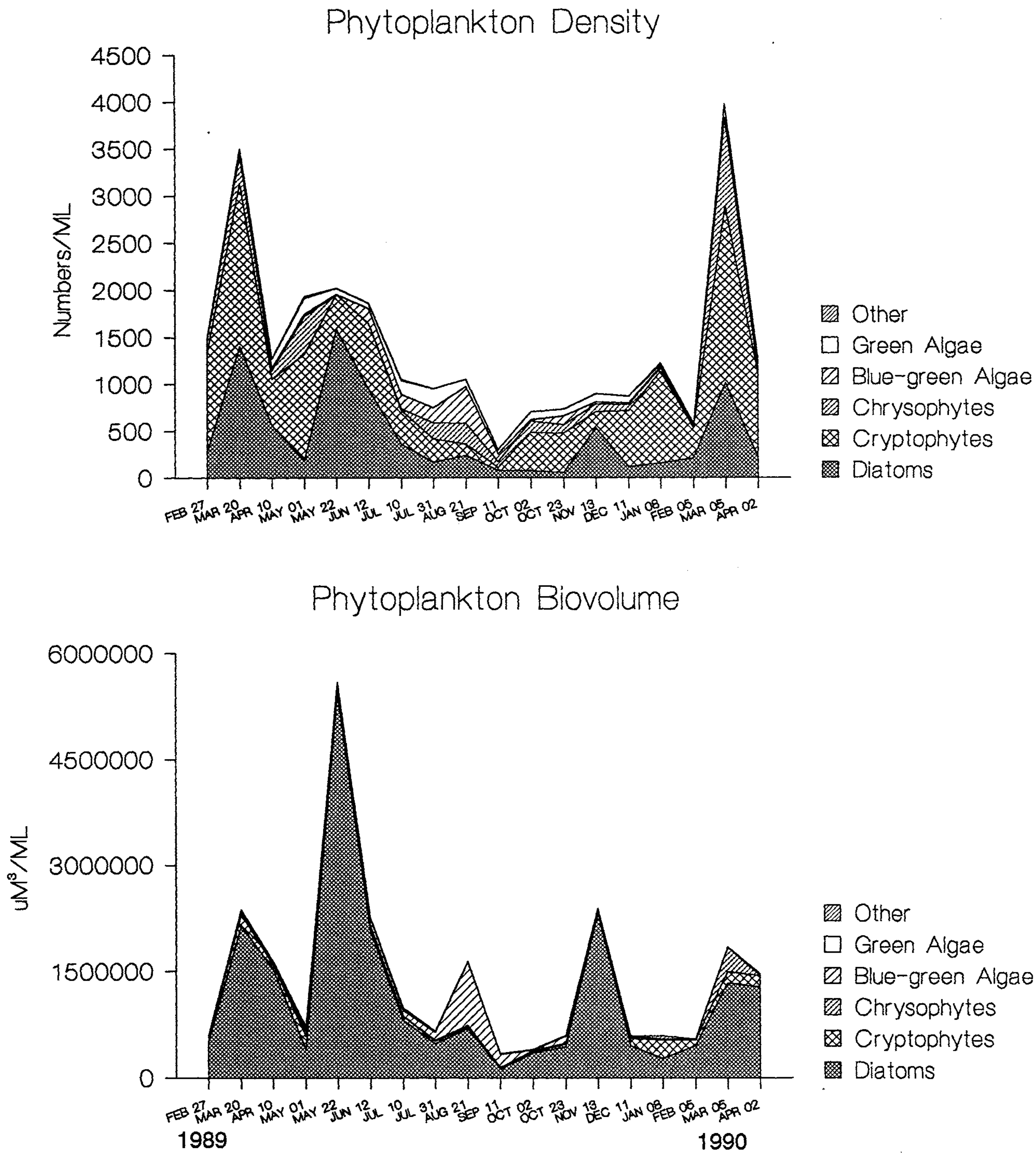
Summer algae populations were more diversified in numbers, though the cumulative biomass (and chlorophyll *a*) was generally at its lowest during the latter part of the growing season. The diatom, *Cyclotella comta*, almost exclusively dominated the algal biomass through September, though a nuisance blue-green algae, *Anabaena planctonica*, proliferated for a short time in August and September, even dominating the biomass at Station 4 on August 21 and September 11. Another blue-green algae, *Aphanizomenon flos-aquae*, contributed significantly to the biomass at Station 3 on October 2. Beginning in October, a succession of diatoms associated with mesotrophic to eutrophic lakes (Sweet, 1986) dominated the biomass including *Fragilaria crotonensis*, *Asterionella formosa*, *Tabellaria fenestrata* (dominating the large biomass on November 13), and *Stephanodiscus astraea*. The cryptomonad, *Cryptomonas erosa*, was prevalent in Lake Sawyer during the winter months of 1989/90.

The rather low algal biovolumes during the latter part of the growing season were most likely due to nutrient limitation (though zooplankton grazing probably contributed) as available forms of phosphorus and nitrogen were either near or below detection limits in the epilimnion. The more limiting nutrient is not clear, though a combination of N and P limitation is probable



(3 week intervals between sample dates, last 4 dates at 4 week intervals)

Figure 5.16. Densities and biovolumes of major algal groups in Lake Sawyer at station 3. Graphs are stacked line plots where algal groups are cumulatively stacked for each sampling date of the study period.



(3 week intervals between sample dates, last 4 dates at 4 week intervals)

Figure 5.17. Densities and biovolumes of major algal groups in Lake Sawyer at station 4. Graphs are stacked line plots where algal groups are cumulatively stacked for each sampling date of the study period.

(discussed in Section 7.1). The appearance of the nuisance algae, *Anabaena planctonica*, during this time might suggest more eutrophic conditions in Lake Sawyer, however this algae could have also succeeded due to its ability to fix nitrogen which may have been limiting growth during the latter part of the growing season.

5.2.3 Zooplankton

Zooplankton densities were quite low in Lake Sawyer relative to other lakes of Western Washington. Zooplankton densities, exclusive of rotifers, ranged from near zero to 64 organisms/L. Low abundances may be attributable to large planktivorous fish populations which seem predominant in Lake Sawyer (section 5.2.6). Fish predation may have been enhanced by the anoxic hypolimnion in Lake Sawyer which precluded the zooplankton grazers from using the hypolimnion as a daytime refuge from the sight-feeding planktivorous fish (Cooke *et al.*, 1986). Elimination of zooplankton species and subsequent low grazing mortality of algae may have resulted in higher algae populations.

Cumulative densities of the major zooplankton groups at Station 3 and 4 during the study period are presented in Figures 5.18 and 5.19. Zooplankton abundance was dominated by rotifers though these small animals did not account for any substantial biovolume. Also presented in Figures 5.18 and 5.19 are the relative biomass of the major zooplankton groups in Lake Sawyer. Relative biomass was calculated from density data using mean standard dry weight measurements of adult zooplankton species from several lakes in the Puget Sound region (Pederson, 1974; Litt, personal communication[4]). Dominant species present in Lake Sawyer in terms of biomass were *Daphnia pulicaria*, *Cyclops biscuspidatus*, *Diaptomus oregonensis* and *Bosmina longirostris*. Percent of the total biomass was estimated assuming weights of 26.7 μg/*Daphnia*, 4.05 μg/*Cyclops*, 11.25 μg/*Calanoid* and 2.00 μg/*Bosmina*. The majority of the rotifer species generally weigh less than 0.1 μg/*Rotifera*. These figures were used with the sole intention of approximating the relative biomass composition of species present and are not necessarily reflective of actual biomass in Lake Sawyer. The biomass of zooplankton was probably dominated by the herbivore, *Daphnia pulicaria*, given these assumptions. *Cyclops biscuspidatus*, a predator of smaller animals, usually not inclusive of *D. pulicaria* (Edmondson and Litt, 1982), contributed to the zooplankton biomass in Lake Sawyer also.

Increased zooplankton abundance followed shortly after the phytoplankton biovolume maxima on May 22, 1989. The dominant zooplankton species present, *D. pulicaria*, appears to be a more indiscriminant grazer than other species and has been shown to readily ingest the phytoplankton species (e.g. *Cyclotella comta* and *Fragilaria crotonensis*) dominating the large phytoplankton biomass in Lake Sawyer during that time period (Infante and Edmondson, 1985). Another period of high zooplankton abundance dominated by *Cyclops* in late October 1989 was observed at Station 3 and not at Station 4.

---

[4]Arnie Litt, Dept. of Zoology, University of Washington, 1990.

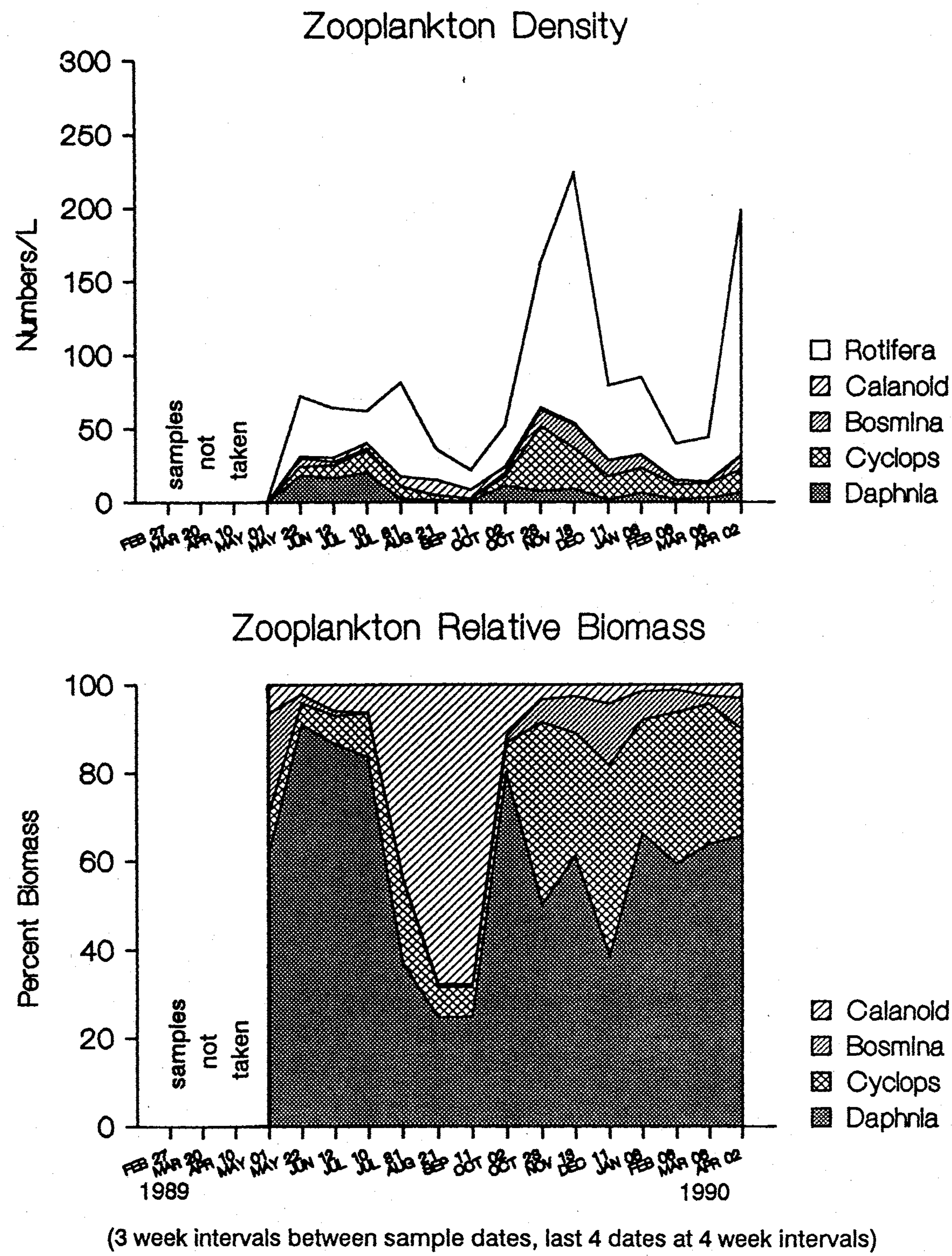
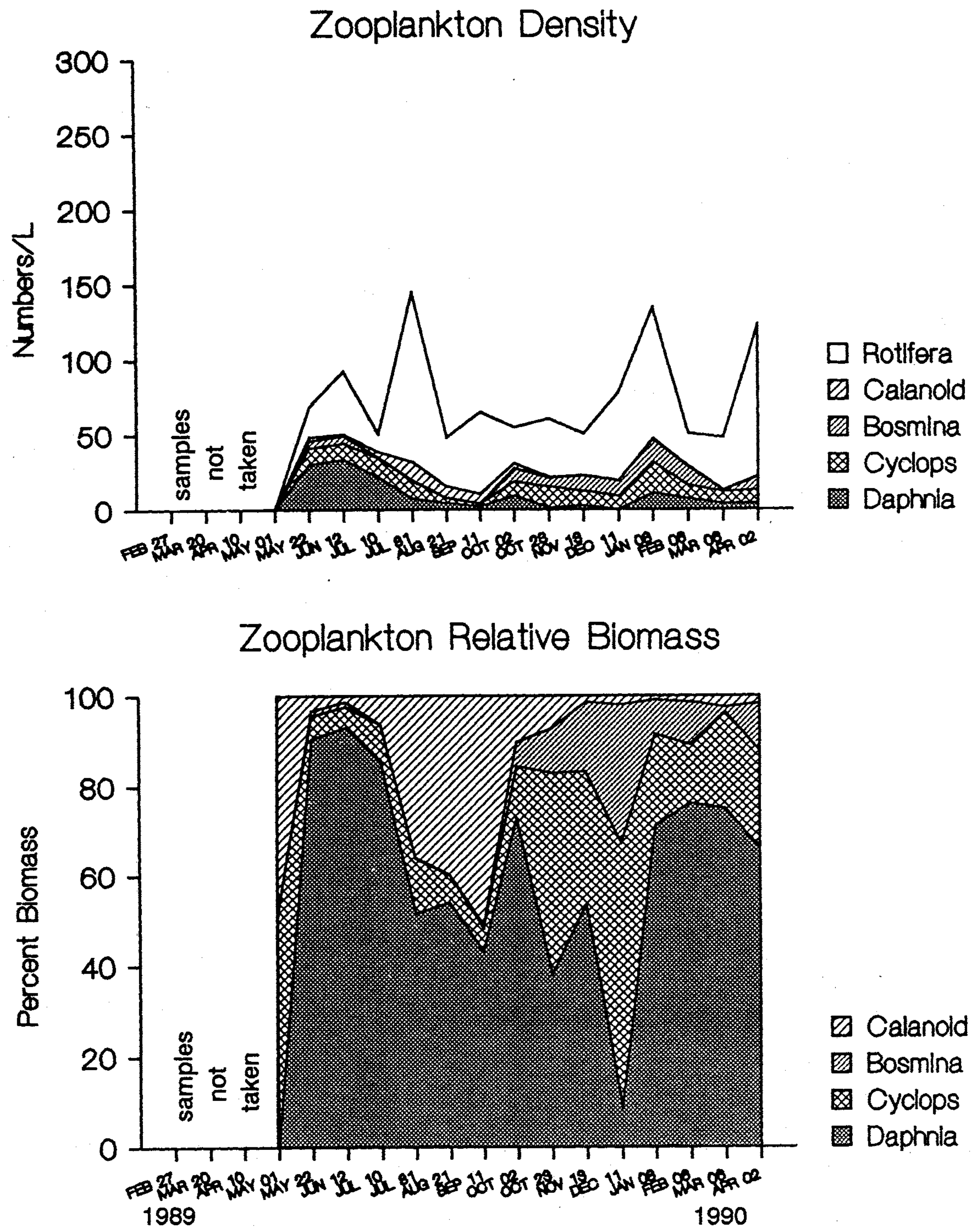


Figure 5.18. Densities and relative biomass (see text) of major zooplankton groups in Lake Sawyer at station 3. Graphs are stacked line plots where zooplankton groups are cumulatively stacked for each sampling date of the study period.



(3 week intervals between sample dates, last 4 dates at 4 week intervals)

Figure 5.19. Densities and relative biomass (see text) of major zooplankton groups in Lake Sawyer at station 4. Graphs are stacked line plots where zooplankton groups are cumulatively stacked for each sampling date of the study period.

5.2.4 Benthic Macroinvertebrates

The abundance and taxa of benthic macroinvertebrates found at the six stations sampled June 27, 1990 are presented in Appendix D. A location map of sample sites and a water column profile of temperature, D.O., pH, and conductivity in Lake Sawyer on the sampling day are also presented. Macroinvertebrates can serve as good indicators of deterioration of water quality because they are relatively sedentary and thus changes in species composition and biomass offer reasonable insight to long-term and short-term water quality degradation (Welch, 1980).

In general, Lake Sawyer exhibited poor diversity and low numbers of benthic macroinvertebrates. The two primary taxonomic groups present in Lake Sawyer included *Diptera* (midges) and *Oligochaeta* (aquatic earthworms) which are noted for their hardiness in polluted, eutrophic waters. Midges were most prevalent with a predominance of *Chironomidae* larvae (particularly *Chironomus chironomus*) occuring at all sampled depth strata (e.g. 4.5, 10.5 and 18 meters). The phantom midge, *Chaoborus* sp., also appeared in larvae and pupae form at the lower strata of Station 3. The second most prevalent form, *Oligochaetes*, consisted of the family *Naididae* and was present at all sampled depths, though not at all stations. Total densities at each sampling station ranged from 258 to 1030 individuals/m^2.

Probably the most important factor affecting the benthic macroinvertebrate community structure is the anoxic hypolimnion which develops during stratification in Lake Sawyer. Severe oxygen deficits limit species diversity, as only a few groups such as the *Diptera* and the *Oligochaeta* have been shown to survive in low oxygen environments (Cole, 1979). Fairly high organic enrichment probably also served to limit the numbers and diversity of benthic macroinvertebrates. Both deep station samples consisted of organically rich profundal sediments. The shallow stations consisted of high amounts of organic detritus and one consisted of large amounts of decaying woody debris.

5.2.5 Aquatic Macrophytes

Aquatic macrophyte surveys showed light to moderate coverage of macrophytes to a depth of approximately three meters with areal coverages of 86 to 103 acres (Municipality of Metropolitan Seattle (METRO), 1976-1980). Maximum potential macrophyte coverage was reported as 155 acres. Maps of macrophyte species coverage during the METRO surveys (1976-1980) with species lists and brief survey descriptions are listed in Appendix E. In summary of the METRO data, *Myriophyllum spicatum* and *Potamogeton richardsonii* dominated the macrophyte community during the 1976-1980 survey years, with a relative areal increase of *Potamogeton spp.* throughout the years. These species were reported present in Lake Sawyer since at least the late 1960s. Various herbicides and homemade cutting bars/cultivators were reported to be used by lakeside residents for macrophyte control, especially *Myriophyllum*, possibly resulting in preferential recolonization of *Potamogeton spp.*

A macrophyte survey of Lake Sawyer was performed in October 1989 (described in Section 3.3.2). The standing crop had achieved its potential maximum biomass by this time of

Table 5.2. Population densities and maximum nutrient contribution of submerged macrophytes in Lake Sawyer, sampled on October 17, 1989, 10 samples.

SPECIES	Potamogeton spp.	Myriophyllum spicatum	Nymphaea odorato	Other species	Total species
DRY MASS (g/m^2)	8.75	24.4	29.35	4.68	67.18
PERCENT OF TOTAL BIOMASS (%)	13%	36%	44%	7%	100%
TOTAL (1) POTENTIAL BIOMASS (kg)	5480	15200	18400	2930	42000
TP CONCENTR. (mg P/g - dry weight)	2.51	1.85	0.97	1.96	7.29
TN CONCENTR. (mg N/g - dry weight)	6.99	3.87	8.24	7.20	26.30
TP CONTENT IN BIOMASS (kg)	14	28	18	6	65
TN CONTENT IN BIOMASS (kg)	38	59	152	21	270

(1) THE MAXIMUM POTENTIAL MACROPHYTE COVERAGE = 155 ACRES (AS REPORTED BY METRO, WATER QUALITY DIVISION (1980); 155 ACRES = ABOUT 627,000 SQUARE METERS.

year. Areal coverage and densities appeared similar to those reported by METRO in 1980. Areas of greatest densities and coverage were the northwest end (particularly the dredged area), the western shore (near the outlet channel and islands) and the southeastern shore in and around the dredged inlets, though a band of macrophytes could be found around most of the lake. Internal nutrient loading from rooted macrophytes is probably primarily due to winter senescence and subsequent remineralization of P and N in biomass (Carpenter, 1983). The biomass of the standing crop of macrophytes in Lake Sawyer was estimated in terms of dry mass, P content and N content based on the ten random samples collected during the October macrophyte survey. Appendix G presents raw data results.

Myriophyllum spicatum and *Potamogeton spp.* dominated the macrophyte community at Lake Sawyer during the 1989 growing season based on visual inspection during the survey. Based on the ten random samplings, the macrophyte biomass in Lake Sawyer was dominated by *Nymphaea odorata* (44%), *Myriophyllum spicatum* (36%), and *Potamogeton spp.* (13%) (Table 5.2). There were only several areas of *Nymphaea* on Lake Sawyer, however, two of the ten random sampling locations were located in *Nymphaea* patches. The percent biomass estimate of *Nymphaea* may be an overestimate. Assuming a maximum potential macrophyte coverage of 155 acres at Lake Sawyer, the mass of phosphorus and nitrogen in the macrophyte standing crop was estimated as approximately 66 kg P and 271 kg N (Table 5.2).

Aquatic macrophytes will probably continue to pose a nuisance to the shallow areas of Lake Sawyer, despite the Black Diamond WTP diversion. In fact, if phytoplankton blooms are reduced, water clarity increases may cause greater macrophytes areal coverage and biomass as light penetration into the water column may increase.

5.2.6 Fish

Recreational fishing is an important use of Lake Sawyer. Public access, made available by the King County public park and boat launch facility, attracts many outside fishermen in addition to the lakeside residents. According to the Department of Wildlife, resident species present include rainbow trout, cutthroat trout, large-mouth and small-mouth bass, black crappie, yellow perch, pumpkinseed, sunfish, and brown bullhead (Cropp, 1990; personal communication)[5]. Rainbow trout are the principal species that the Department of Wildlife plants in Lake Sawyer, although kokanee have been stocked occasionally when available. The most recent rainbow plants were March 14, 1990 (7,555 individuals at 20.7/lb.), June 22, 1989 (28,280 at 70/lb.), May 20, 1988 (3,822 at 6/lb.), and December 4, 1987 (105,025 at 164/lb.). In addition to resident species, Lake Sawyer appears to be seasonally inhabited by steelhead, chinook, and coho salmon. A few of these fish were reported by local residents to be crossing the fish ladder at the Lake Sawyer dam.

[5]Cropp, T., Fish Biologist, WA Dept. of Wildlife, 1990.

Electrofishing by the Department of Wildlife in August of 1986 showed largemouth bass from 84 to 400 mm (probably seven year classes) and smallmouth bass from 76 to 400 mm (probably eight year classes). Several conversations between fishermen and the investigators during the Ecology study period indicated catches of yellow perch of unknown size classes.

In general, Lake Sawyer seems to support a warm-water fishery, probably due to an anoxic hypolimnion during stratification which excludes predatory game fish that prefer cold water (Cooke *et al.*, 1986). Overpopulation of yellow perch, black crappies, and/or brown bullheads and subsequent food competition between these non-native species and the trout populations is common in lowland Puget Sound lakes and have served to reduce trout populations (Wydoski and Whitney, 1979).

On the weekend of May 13-14, 1989, a fishkill of approximately 1000 fish, mainly perch, was reported in Lake Sawyer. Investigations on the following days by METRO and Ecology failed to reveal the cause of death. Dissolved oxygen was found to be adequate, un-ionized ammonia was well below chronic levels, and metal contamination was not suspected (see memoranda in Appendix F). Fish were reported dying continually, a few at a time for the next few weeks. A pathologist for the Washington State Department of Wildlife observed fish dying on June 24, 1989, and determined that the fish had died from a natural disease caused by the bacteria, *Chondrococcus columnaris* with possibly secondary systemic bacterial infections belonging to either the *Aermonas* or *Pseudomonas* groups (Appendix F). It is thought that the large fish kill which took place on May 13 and 14 was attributable to this bacterial disease. Outbreaks of this disease had been reported in catfish at Green Lake in Seattle and in yellow perch in Lake Osoyoos during the same period.

The fish kill in May seemed to be localized in the shallower southwestern part of Lake Sawyer. Yellow perch are known to spawn approximately this time of year in the shallow waters (Wydoski and Whitney, 1979). A normal spawning mortality is expected as the spawning effort produces a high amount of stress in the fish. It is not known, but this stress may have left the perch population in a weakened condition and more susceptable to a disease outbreak. Additional stresses such as higher water temperatures and high daytime oxygen saturation of the water in excess of 120% in the epilimnion during this period may have contributed.

5.2.7 Coliform Bacteria

Fecal coliform and fecal streptococcus densities were measured from the surface of Stations 3 and 4 during each lake sampling survey. Fecal coliform densities ranged from zero to seven organisms/100 mL and fecal streptococci ranged from zero to nine organisms/100 mL in Lake Sawyer during the study period, well below the Washington State "lake class" water quality criteria of a geometric mean <50 organisms/100 mL and fewer than 10% of samples >100 organisms/100 mL (WAC 173-201-045).

5.3 Lake Sediment Analysis Results

5.3.1 Pb-210 Activities and Sedimentation Rates

Sedimentation rates were determined using Pb-210, as described above in Section 3.2.4. The depth profile of unsupported Pb-210 is shown in Figure 5.20. The average rate of sedimentation, using the CIC model, is 0.32 $\pm$ 0.01 cm/year in the upper 50 cm of the core, which represents about 150 years of deposition. The age of various sediment depths was calculated using the CRS model because of expected changes in recent sedimentation rates. The CRS model dates (Figure 5.20) correspond closely with the expected date of the inflection point of increasing stable Pb concentration at approximately 25 cm. Stable lead increases in sediment are associated with the introduction of leaded gasoline between 1910 and 1920, and are commonly used in studies of regional lakes to estimate long-term average sedimentation rates (Welch and Smayda, 1986; Spyridakis and Barnes, 1977; and Barnes *et al.*, 1978). The stable lead profile in Lake Sawyer provides independent confirmation of the Pb-210 derived dates.

The total amount of unsupported Pb-210 present in the cores from Stations 3 and 4 were 29.5 and 43.7 dpm/cm^2, which corresponds to steady-state profundal flux of 0.92 and 1.36 dpm/cm^2-yr, respectively. The profundal Pb-210 flux can be compared with atmospheric flux to indicate the degree of focusing of sediment towards the deepest point in the lake (Von Damm et al., 1979). Local atmospheric Pb-210 flux has been reported by Nevissi (1985) at 0.44 $\pm$ 0.24 dpm/cm^2-yr, and by Carpenter *et al.* (1981) at 0.73 $\pm$ 0.12 dpm/cm^2-yr. The results of these two studies were pooled to estimate average atmospheric flux at Lake Sawyer of approximately 0.59 $\pm$ 0.27 dpm/cm^2-yr, which is significantly lower than measured profundal flux.

Sedimentation at the deepest point in a lake is usually much greater than the whole-lake average rate because of focusing effects of several in-lake processes, including redistribution with intermittent complete mixing, and slope slumping and sliding (Hilton *et al.*, 1986). The rates of sediment deposition at the profundal core stations were estimated using Pb-210 data to be 1.6 and 2.3 times greater than average whole-lake deposition rates at Station 3 and 4, based on the ratio of profundal to atmospheric flux to each core. The higher degree of sediment focusing at Station 4 may be due to the steeper bottom slopes in the northern basin (Figure 2.1). The factors measured using Pb-210 compare reasonably well with the ratio of maximum to mean lake water depth (2.3 for Lake Sawyer), which is sometimes used to estimate focusing (Lehman, 1975).

The sediment concentrations of TP, TN, and TOC are shown in Figure 5.21 along with profundal total solids sedimentation rates. Sedimentation of TP, TN, and TOC are commonly used as indicators of historical loading and algal productivity (Shapiro et al., 1971; Birch et al., 1980). However, sediment concentrations of carbon and nitrogen are probably not conservative (i.e., may decrease significantly after deposition) because of bacterial decomposition and loss to the atmosphere (Wetzel, 1983). Therefore, sedimentation rates of carbon and nitrogen from core data may underestimate the actual deposition rate. Historical sedimentation rates of TP,

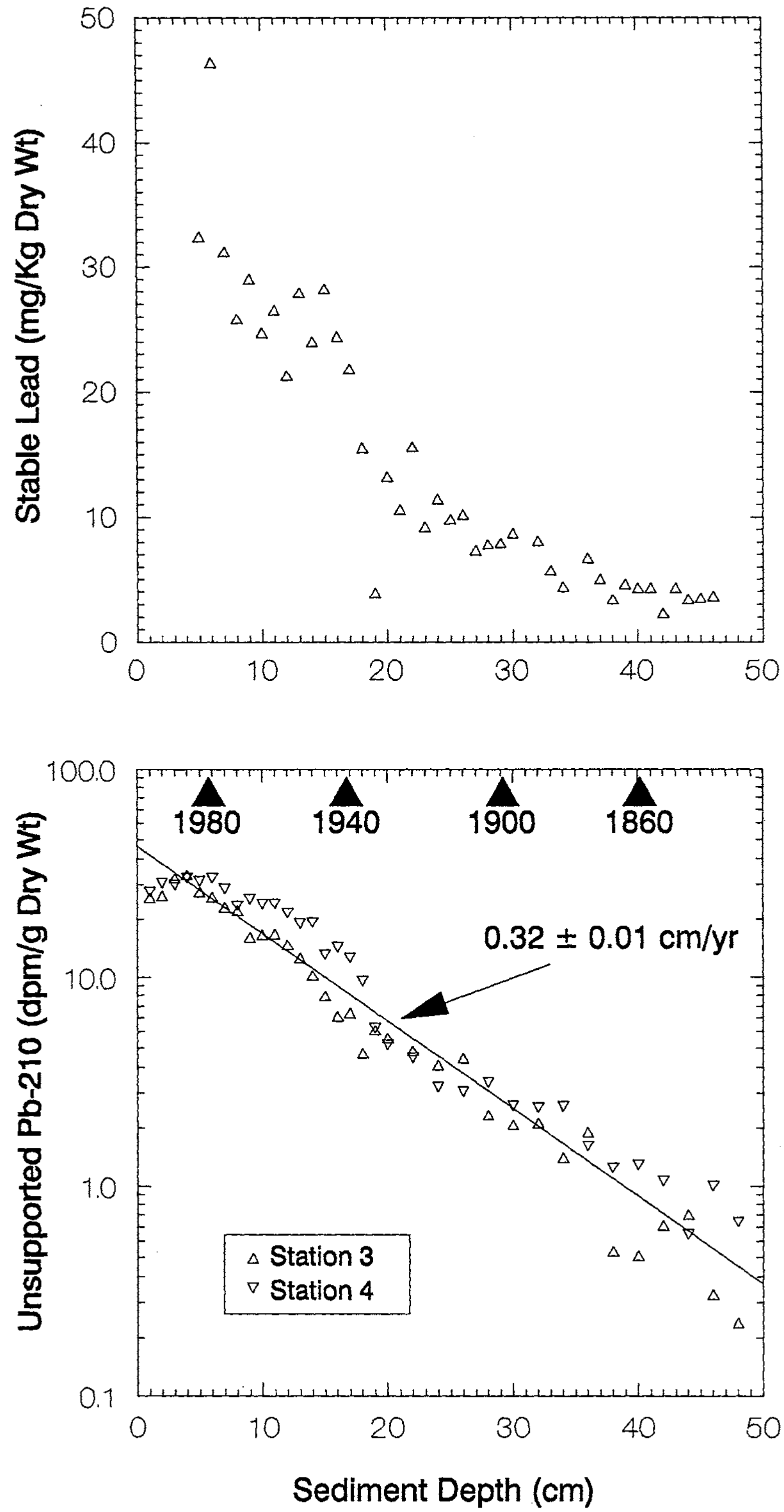


Figure 5.20. Unsupported Pb-210 and stable Pb sediment profiles. Dates shown are based on the CRS model (see text). Regression line (CIC model) shows long-term profundal sedimentation rate of 0.32 ± 0.01 cm/yr.

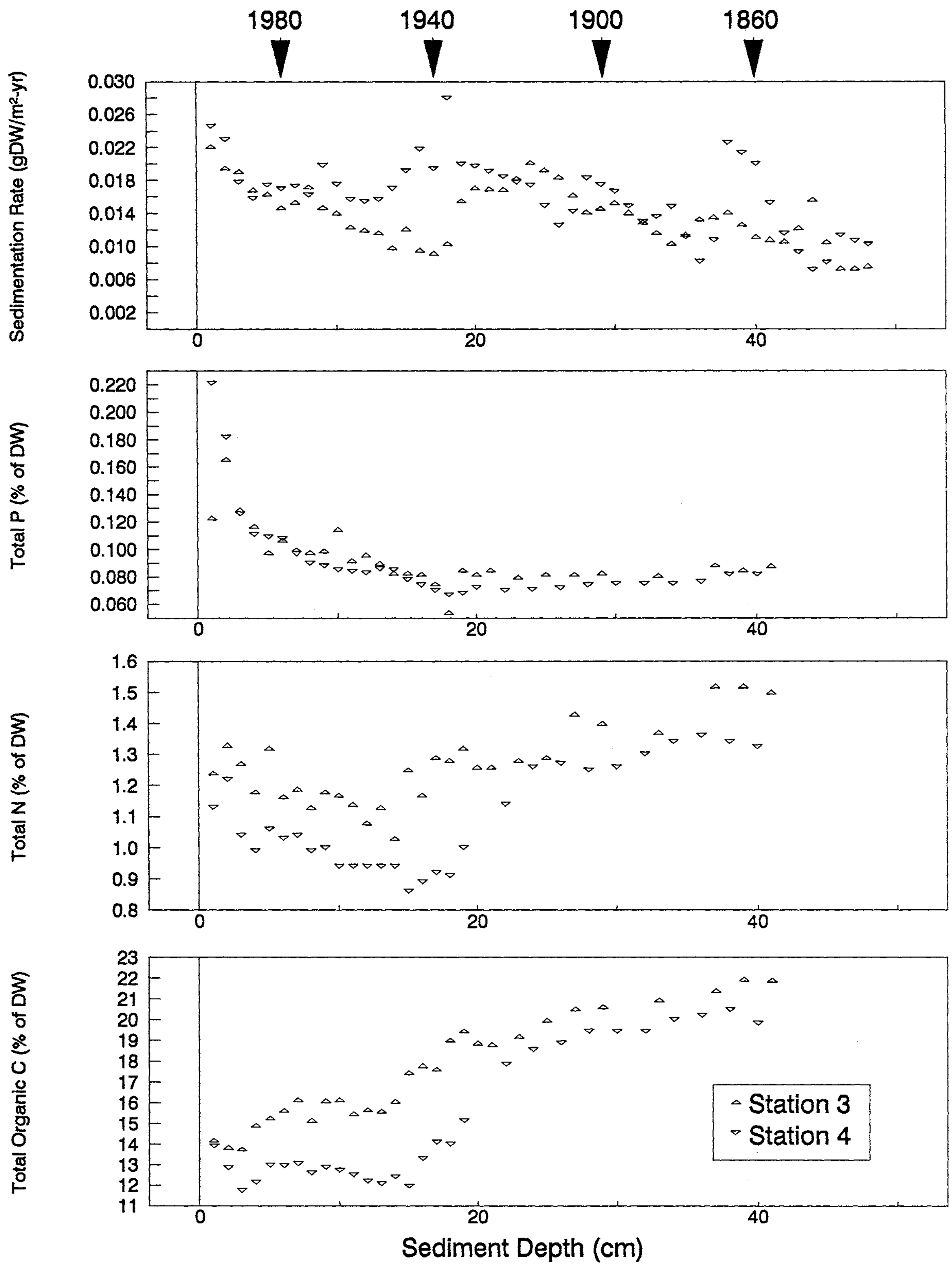


Figure 5.21. Profiles of whole-lake sediment accumulation of total solids, and concentrations of total P, total N, and total organic C.

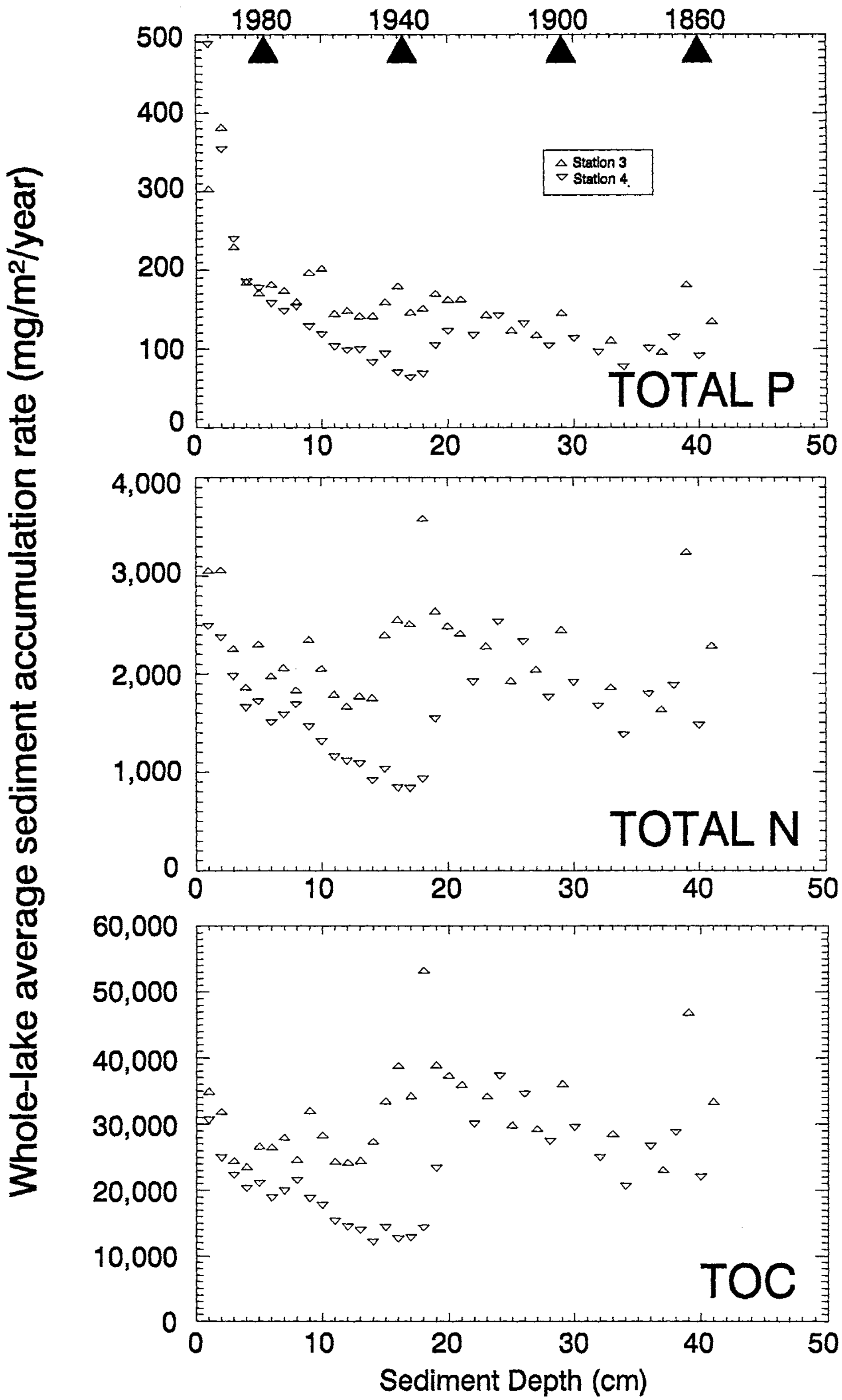


Figure 5.22. Changes in whole-lake average sediment accumulation rates for total P, total N, and TOC.

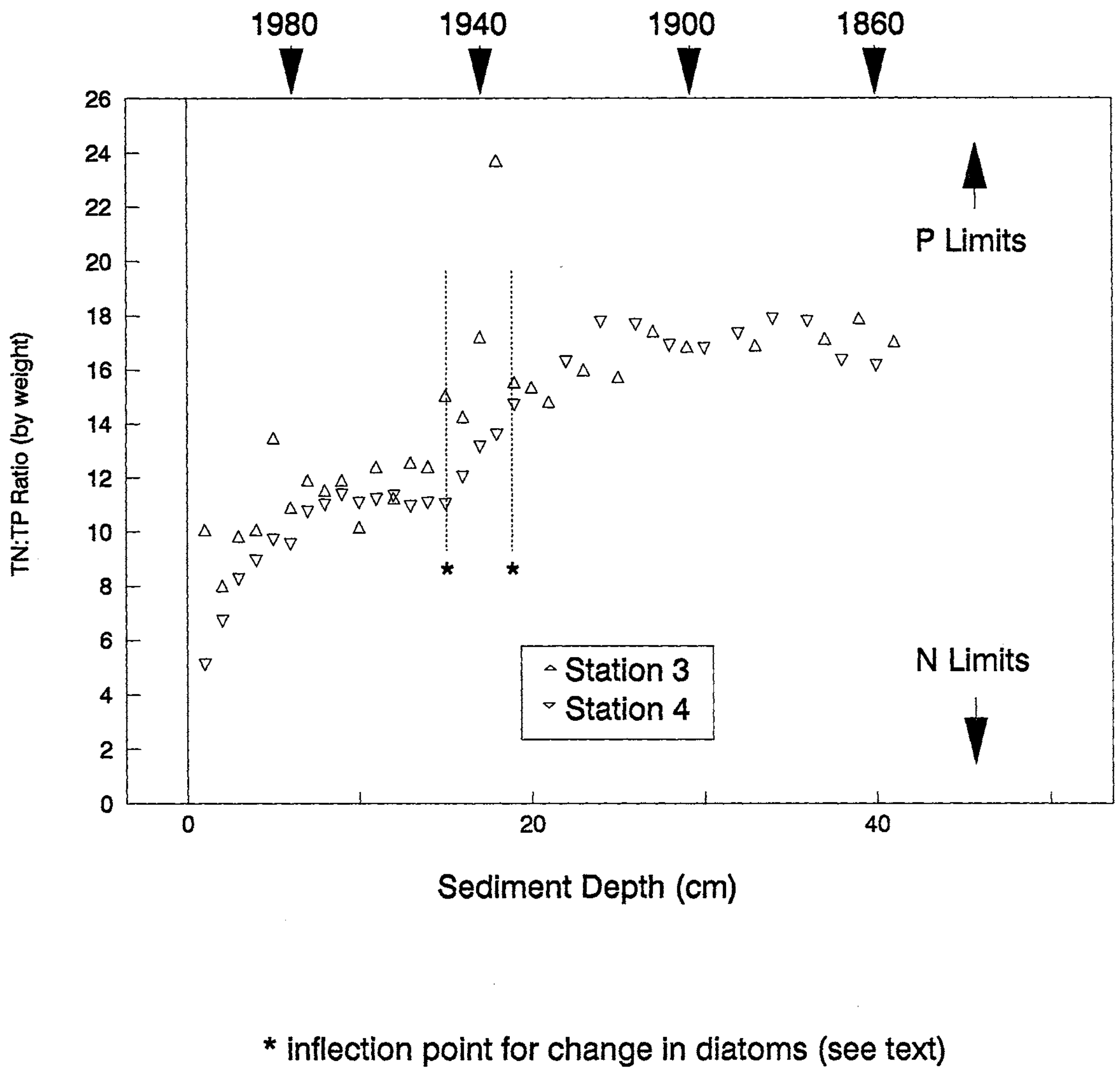


Figure 5.23. Profiles of sediment Total N:P ratios.

TN, and TOC were estimated for the whole lake by combining concentration profile data with total solids sedimentation rates based on Pb-210 after correcting for focusing (Figure 5.22). Significant increases in sedimentation rates immediately following start-up of the Black Diamond WTP (Figure 5.22) suggest nutrient loading increases. Phosphorus sedimentation after WTP start-up appears to be higher than at any other time in the past 150 years. However, TN and TOC sedimentation rates in the late 1800s and early 1900s were as high as present rates, possibly due to logging and development activities. Both TN and TOC histories show decreases in the mid 1900s with very recent increases coincident with WTP start-up.

5.3.2 Sediment Diatom Profile

Selected sediment core intervals were analyzed for diatom species and densities. The siliceous frustule remains of diatoms are useful paleoecological indicators (Wetzel, 1983). Appendix G presents the findings of the analysis, including a species list, a synopsis of dominant and common diatom species, and similarity indices comparing adjacent core intervals.

Three distinct diatom communities were evident throughout the core. The lowest portion of the core from 60 to 20 cm depth was dominated by *Cyclotella ocellata* (37% average abundance), while *Melosira italica* (34%) was predominant from the 18 to 15 cm depth and *Cyclotella comta* (36%) was dominant from 14 cm to the top of the core. Shifts from one predominant species to another were sudden, indicating possible periods of hastened environmental change. All shifts took place after European-ancestor human settlement around Lake Sawyer. Some of the lowest similarity indices resulted from the rather abrubt diatom community changes. The two most distinct community shifts are presented in Figure 5.23 correlating with shifts in sediment N:P ratios sometime around 1940. During this time period, the majority of the Ravensdale Creek watershed was apparently logged, evident from early aerial photographs.

Densities of diatoms ranged from about 49,000 to 246,000/mg dry weight of sediment throughout the core. The average density was 154,000 $\pm$ 9840/mg dry weight. Densities were generally higher in the deeper, older sediments, however the most recent density (i.e. in the top 1-2 cm of sediment) was the second highest observed (232,000/mg dry weight of sediment). Of notable interest was a solitary interval (29-30 cm interval) corresponding to the early 1900s of very low density (58,400/mg dry weight) and a period of low densities (75,100 $\pm$ 7,220/mg dry weight) from 5 cm through 14 cm which corresponded to the period when *Cyclotella comta* became dominant.

Stockner (1971) proposed the use of a Araphidineae/Centrales (A/C) index as an indicator of lake productivity. In general, Centrales were considered to reflect oligotrophy while the Araphidineae were considered to reflect eutrophy. Based on samples from some temperate lakes of North America and England, a classification was developed where A/C ratios of 0 to 1 suggested oligotrophic conditions, 1 to 2 suggested mesotrophic conditions and >2 suggested eutrophic conditions. Stockner (1972) as well as others (e.g. Brugam, 1979; Wetzel, 1983) stressed that this classification was not suitable for all lakes and caution was warranted in any interpretation. In general, Lake Sawyer meets the criteria for use of the index (i.e. sufficient

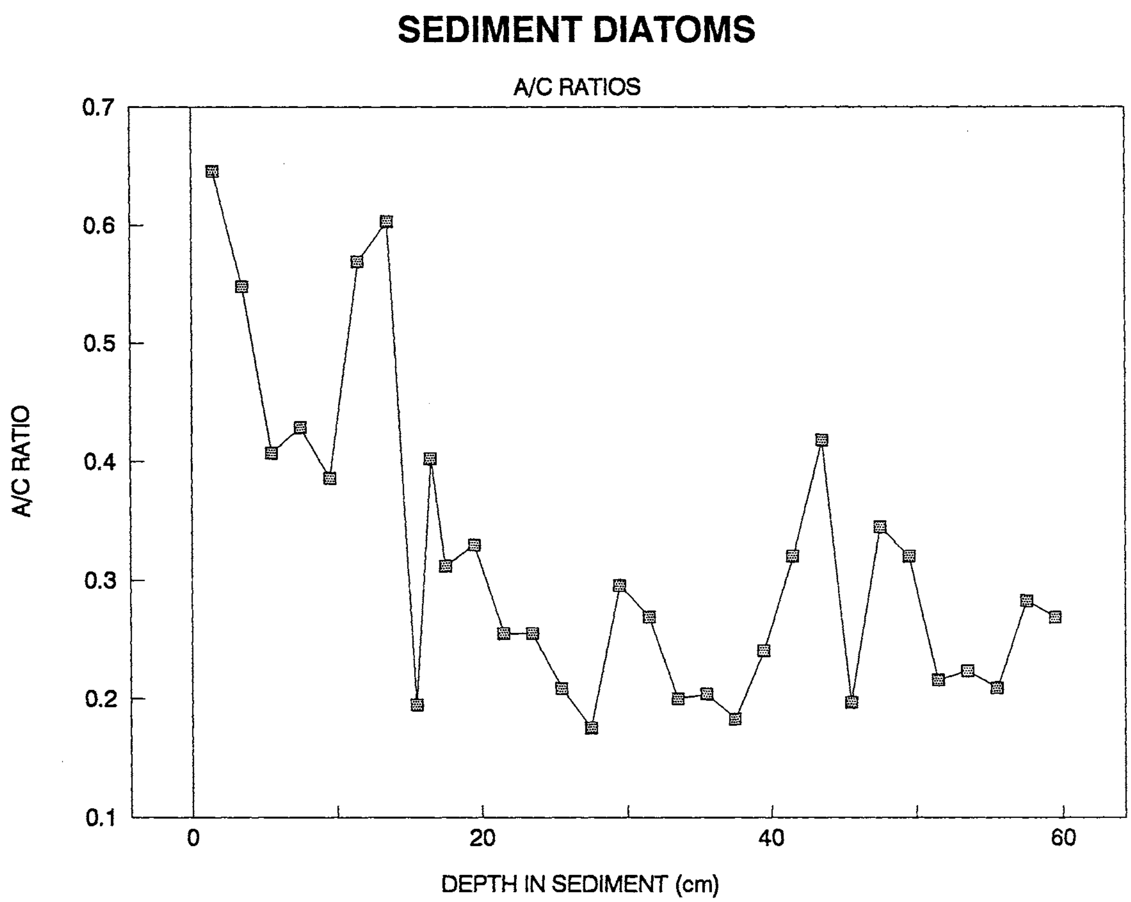


Figure 5.24. Profile of sediment diatom Araphidineae/ Centrales ratios.

depth, monomictic, moderate P concentration and low alkalinity). The A/C diatom ratios in Lake Sawyer sediment intervals were all below one (<1), suggesting oligotrophic conditions using the above index. Whether or not the index is suitable for Lake Sawyer, there was a noticeable recent trend towards more eutrophic conditions as the most recent sediments had the highest relative A/C ratios (Figure 5.24).

Indications of human disturbance to Lake Sawyer seem to be supported by diatom species remains. Increasing percentages of the araphidinate species *Fragilaria crotonensis* were noticed following human settlement of Shagawa Lake, Minnesota (Bradbury and Megard, 1972; Bradbury and Waddington, 1973). *Fragilaria crotonensis* was only found commonly in the most recent Lake Sawyer sediments from 14 cm and less. Brugam (1979) found the species *Stephanodiscus hantzschii* to be an indicator of human disturbance in lakes with TP greater than 15 μg/L and alkalinity greater than 75 mg/L as $CaCO_3$. Lake Sawyer did not exhibit high alkalinity, however, *Stephanodiscus hantzschii* did appear once at the top of the core (1-2 cm interval) in low abundance (about 1% of the total diatom biovolume).

5.3.3 Sediment Fe and Al

During the macroinvertebrate sampling on June 27, 1990, sediment dredge samples were taken for iron (Fe) and aluminum (Al) analysis. The top 4-5 cm of sediment were analyzed. Aluminum concentrations were 11,200 $\pm$ 350 mg/kg dry weight of sediment at Station 3 and 14,800 $\pm$ 1400 mg/kg dry weight of sediment at Station 4. Iron concentrations were 25,200 $\pm$ 450 mg/kg dry weight of sediment at Station 3 and 21,800 $\pm$ 550 mg/kg dry weight of sediment at Station 4. A lowest effect level (i.e. the level at which actual toxic effects become apparent) for Fe of 30,000 mg/kg dry weight was proposed by the Ontario Ministry of Environment (Persaud *et al.* 1989), therefore concentrations of iron in Lake Sawyer appear to be at a non-toxic level.

5.4 Tributary Water Quality

Regressions of TP and TN concentration versus flow were developed for Ravensdale Creek and Rock Creek, as described in Section 3.3. The shape of the concentration-discharge curve (Figure 5.25) indicates the general process of nutrient loading (Galat, 1990). For example, the TP versus discharge curve for Rock Creek, with concentration increasing dramatically as flow decreases, is indicative of dilution of a relatively constant point source (i.e. Black Diamond WTP effluent). The TP curve for Ravensdale Creek, and the TP and TN curves for both Ravensdale and Rock Creeks, are indicative of flow-driven or diffuse inputs from primarily nonpoint sources.

The background TP and TN loads to Rock Creek, from nonpoint sources other than the Black Diamond WTP, were significantly reduced (>99% probability based on ANOVA) by about 30% after installation of the WTP. Figure 5.26 presents a comparison of historical background water quality data from Rock Creek prior to 1982 (KCM, 1982) with background Rock Creek quality during the 1989-90 study year. Current background concentrations in Rock

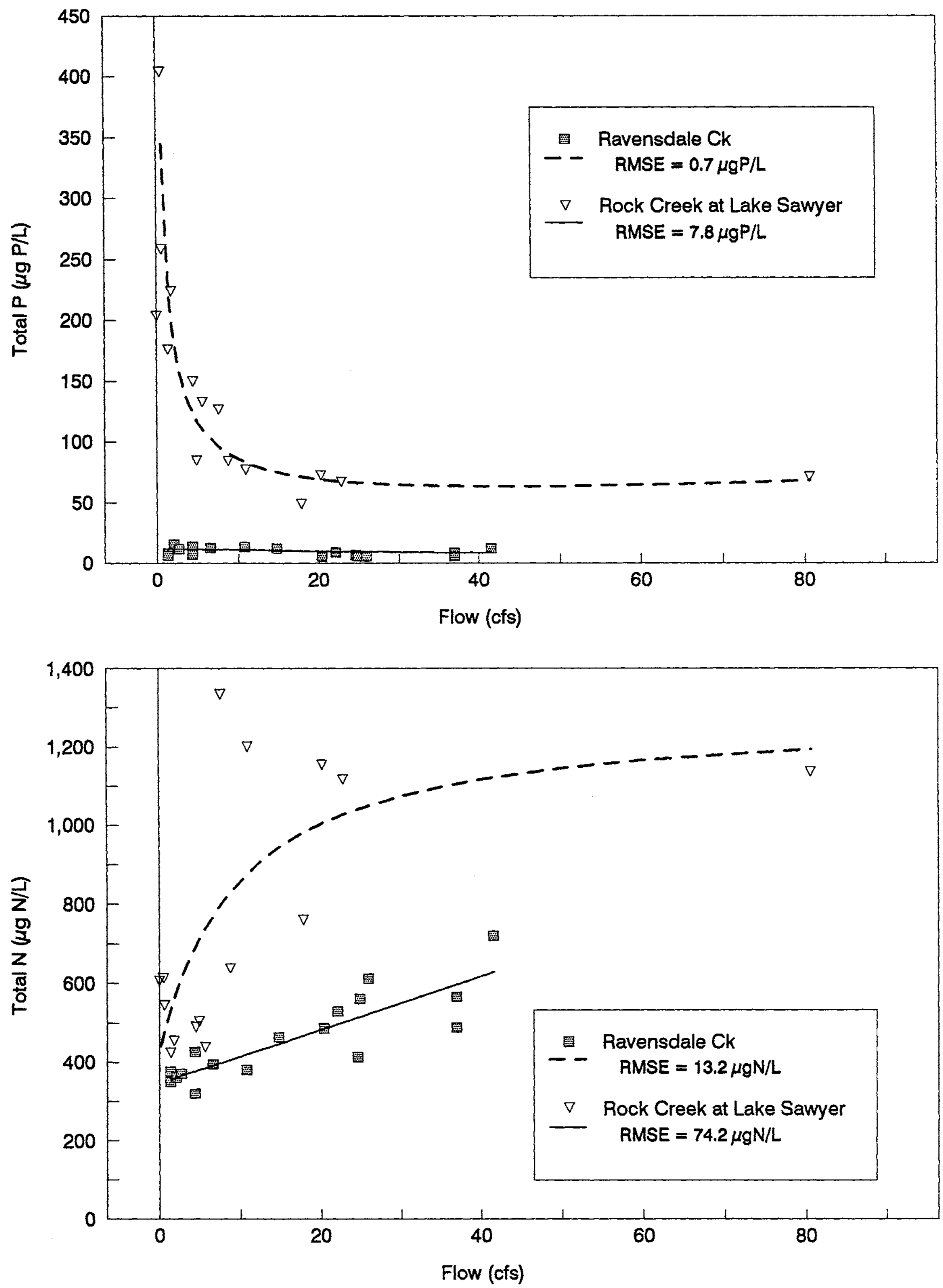


Fig 5.25. Relationships between flow, total phosphorus, and total nitrogen at Ravensdale Creek and Rock Creek at Lake Sawyer.

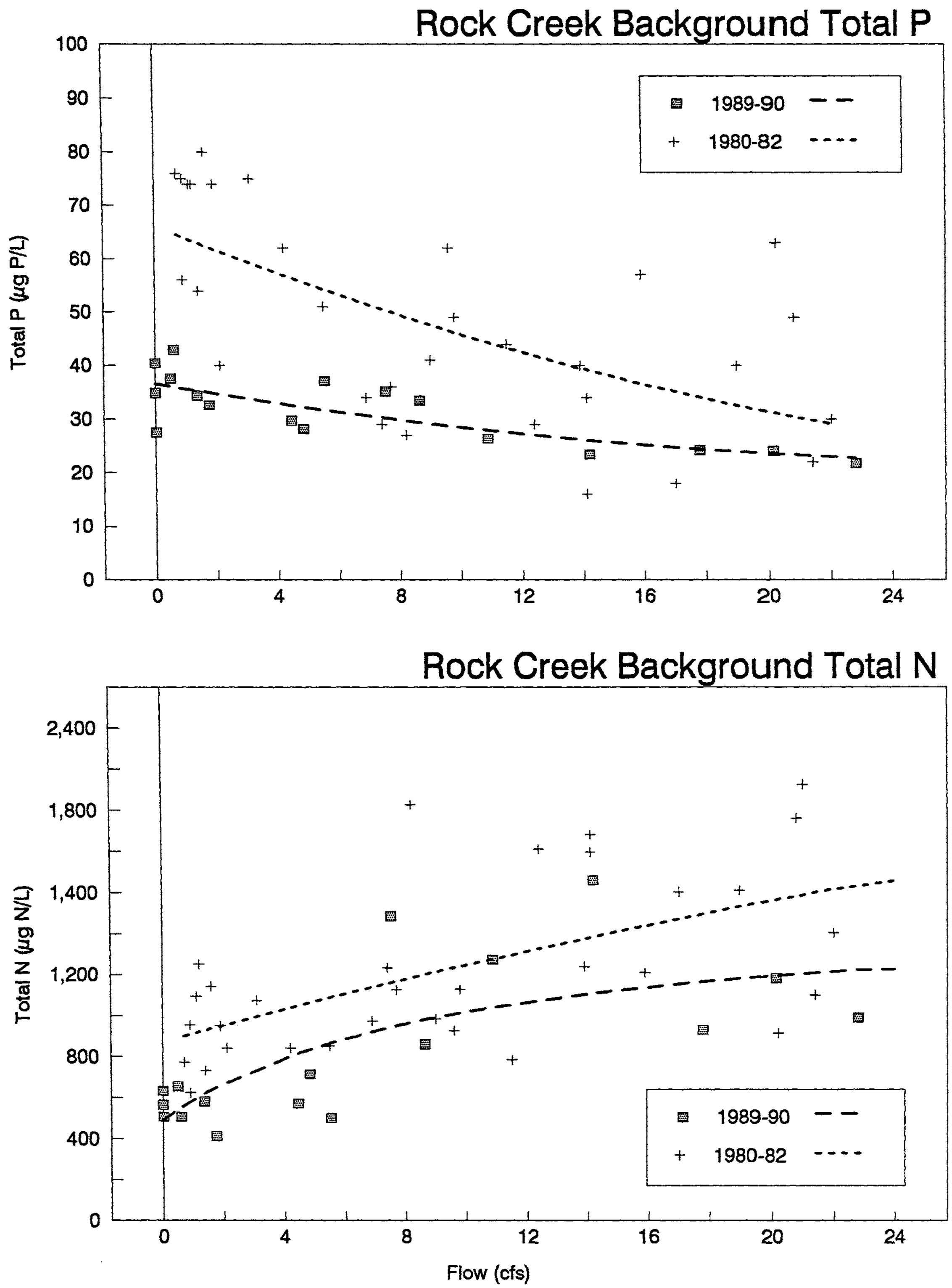


Figure 5.26. Comparison of Rock Creek background (i.e. separate from the Black Diamond WTP) nutrient concentrations before and after WTP start-up.

Table 5.3. Summary of tributary water quality data from February 1989 through March 1990.

	Number of samples	Mean *	Std Dev	Min	10%tile	Median	90%tile	Max
TEMPERATURE (degrees C)								
Ravensdale Creek	19	10.1	2.9	5.8	6.1	9.3	14.5	14.9
** Ginder Creek	9	11.7	3.0	7.7	7.7	12.0	16.4	16.4
** Rock Creek at Abrams Road	17	11.1	4.4	3.6	3.9	11.2	17.0	17.0
** Rock Creek at Morganville Bridge	19	10.2	4.3	3.0	3.2	9.4	15.0	17.4
pH (S.U.)								
Ravensdale Creek	19	7.1	0.5	6.5	6.6	7.0	7.8	8.4
Ginder Creek	9	7.2	0.4	6.7	6.7	7.4	7.6	7.6
Rock Creek at Abrams Road	17	7.0	0.5	6.4	6.4	6.9	7.8	8.5
Rock Creek at Morganville Bridge	19	6.9	0.4	6.4	6.5	6.9	7.6	7.9
DISSOLVED OXYGEN (mg/L)								
Ravensdale Creek	18	10.9	0.7	9.6	9.6	10.9	11.9	12.1
** Ginder Creek	9	10.0	1.0	8.2	8.2	10.2	11.4	11.4
** Rock Creek at Abrams Road	16	8.2	2.6	4.4	4.7	8.7	11.6	11.7
** Rock Creek at Morganville Bridge	18	8.1	2.5	4.7	4.9	7.8	11.5	12.2
DISSOLVED OXYGEN SATURATION (%saturation)								
Ravensdale Creek	18	98	3	92	93	98	103	104
** Ginder Creek	9	94	13	71	71	96	119	119
Rock Creek at Abrams Road	16	75	19	46	47	78	97	103
Rock Creek at Morganville Bridge	18	72	18	47	48	73	101	105
TURBIDITY (NTU)								
Ravensdale Creek	19	5.6	14.9	0.4	0.5	0.7	35.0	58.0
Ginder Creek	8	4.1	1.5	1.6	1.4	3.8	5.9	6.5
Rock Creek at Abrams Road	22	5.0	7.5	1.6	2.1	3.2	5.9	38.0
Rock Creek at Morganville Bridge	23	3.3	3.5	1.4	1.8	2.6	3.7	19.0
FECAL COLIFORM (CFU/100mL)								
Ravensdale Creek	19	3	11	1	1	3	24	42
** Ginder Creek	12	30	61	2	2	69	159	163
** Rock Creek at Abrams Road	22	74	382	4	11	108	441	1820
** Rock Creek at Morganville Bridge	23	52	149	7	7	58	397	590
FECAL COLIFORM:STREPTOCOCCI RATIO								
Ravensdale Creek	19	0.7	0.7	0.01	0.04	0.5	2.3	2.5
Ginder Creek	12	3.3	3.7	0.46	0.50	1.7	11.1	13.0
Rock Creek at Abrams Road	22	3.1	6.0	0.25	0.29	1.2	14.0	24.0
Rock Creek at Morganville Bridge	23	2.1	3.2	0.28	0.58	1.0	4.4	16.0

* Arithmetic mean for all parameters except fecal coliform, for which geometric mean is reported.

** Water quality limited with respect to Class AA standard.

Creek were estimated as the flow-weighted average concentrations from sources to Rock Creek other than the Black Diamond WTP (i.e. Rock Creek upstream from the WTP, Black Diamond Lake Creek, Morganville Marsh, and Palmer Spring; Figure 3.1).

The significant reduction of nonpoint nutrient inputs to Rock Creek following the WTP start-up may be explained by elimination of wastewater inputs from failing community and individual septic systems in use prior to 1982. However, even though nonpoint nutrient loads were reduced by installation of the WTP, total loading in Rock Creek increased significantly, as is apparent from comparison of Figure 5.25 and 5.26. Nutrient loads are presented in more detail in Section 6.3.

A summary of water quality data from Lake Sawyer tributaries is presented in Table 5.3. Ravensdale Creek water quality was better than Rock Creek for all parameters monitored, which is consistent with a lower level of watershed activities for Ravensdale Creek. Rock Creek and Ginder Creek were found to be water quality limited with respect to Class AA standards for temperature, dissolved oxygen, and fecal coliform. The source of fecal contamination in Rock Creek appears to be nonpoint in origin, since concentrations upstream from the WTP (Rock Creek at Abrams Road) were typically higher than downstream (at Morganville Bridge), and significant fecal contamination was also found in Ginder Creek. Similarly, the cause of depressed dissolved oxygen in Rock Creek is probably not attributable to the WTP since upstream (Abrams Road) dissolved oxygen is not significantly higher than downstream (Morganville Bridge).

The ratio of fecal coliform to fecal streptococci (FC/FS) is used to discriminate between human versus animal fecal contamination (EPA, 1985; Geldreich and Kenner, 1969). A FC/FS ratio in excess of approximately 4 suggests human fecal material, a ratio of less than 1 suggests animal feces, and a ratio between 1 and 4 does not conclusively indicate origin. Table 5.3 shows that tributary FC/FS ratios were typically less than 1, which suggests non-human fecal contamination. However, greater than 10 percent of the observations in Ginder and Rock Creeks had ratios in excess of 4, which suggests nonpoint human fecal contamination at times.

This page is purposely left blank for duplex printing.

6.0 NUTRIENT BUDGETS AND TROPHIC RESPONSE

6.1 Nutrient Limitation

Nutrients are required for algal development and growth. The nutrients of primary importance are phosphorus and nitrogen as they have generally been shown to limit growth (i.e they are in least available supply compared to their need) in freshwater lakes (Wetzel, 1983). Algal productivity directly and indirectly influences a range of other water quality characteristics in a lake (OECD, 1982). From a lake management perspective, it is most practical to control the most limiting nutrient to manage the water quality of a lake.

The ratio of nitrogen to phosphorus in lake waters is generally accepted as an indicator of which nutrient may be the more limiting to algal growth (Forsberg, 1980; Healey and Hendzel, 1980). While the concept of using N:P ratios in order to determine N or P nutrient limitation is generally accepted, the published ratio cutoff points which define either N or P limitation vary slightly. Most commonly, lake water is determined to be N limited when the N:P ratio is less than 10:1 (all discussion of nutrient ratio values in this report are by weight) and P limited when the N:P ratio is greater than 17:1. If N:P ratios are between 10:1 and 17:1 either one or both of the nutrients may be limiting (Sakamoto, 1966; Forsberg, 1980; Vallentyne, 1974). While this is the most commonly reported range, the upper cutoff point defining P limitation has also been reported as 20:1 (Healey and Hendzel, 1980), 21:1 (Smith, 1979) and 10:1 (Chiandani and Vighi, 1974) while the lower cutoff point defining N limitation has also been reported as 13:1 (Smith, 1979) and 5:1 (Chiandani and Vighi, 1974).

Figure 6.1 presents the N:P ratios at Lake Sawyer during the study period. Both total and dissolved fraction ratios are plotted, though the standing crop of N and P (e.g. total nitrogen and total phosphorus) is typically the most useful ratio comparison for water quality management (Forsberg, 1980; Cole, 1979). The TN:TP ratios indicated the lake water to be P limited for algal growth from the beginning of the study period until the end of May 1989 when the TN:TP ratios fell and remained in the uncertain zone of either P and/or N limitation until January 1990. The most commonly reported cutoff points of 17:1 and 10:1 are used here.

Following a similar temporal trend, the DIN/SRP (soluble fractions of N and P) ratios showed a P limitation at first and then plunged well below 10:1 as the water column became practically devoid of nitrate and nitrite (discussed in section 6.1.6). Ratios of DIN/SRP may not be useful indicators of limitation since the concentrations of dissolved nutrient forms are affected by remineralization of particulate nutrients as well as rapid algal uptake, whereas TN:TP ratios are more indicative of cellular nutrient content.

In most cases, phosphorus is found to be the most limiting nutrient for algal growth in lake water (OECD, 1982). This is probably due to the limited sources of phosphorus compared to nitrogen and rapid sedimentation of phosphorus in aquatic ecosystems. In such cases, N:P ratios are generally high and distinctly fall into a P limiting category. The N:P ratios at Lake Sawyer do not clearly indicate that phosphorus is the limiting nutrient year-round. Additionally, a

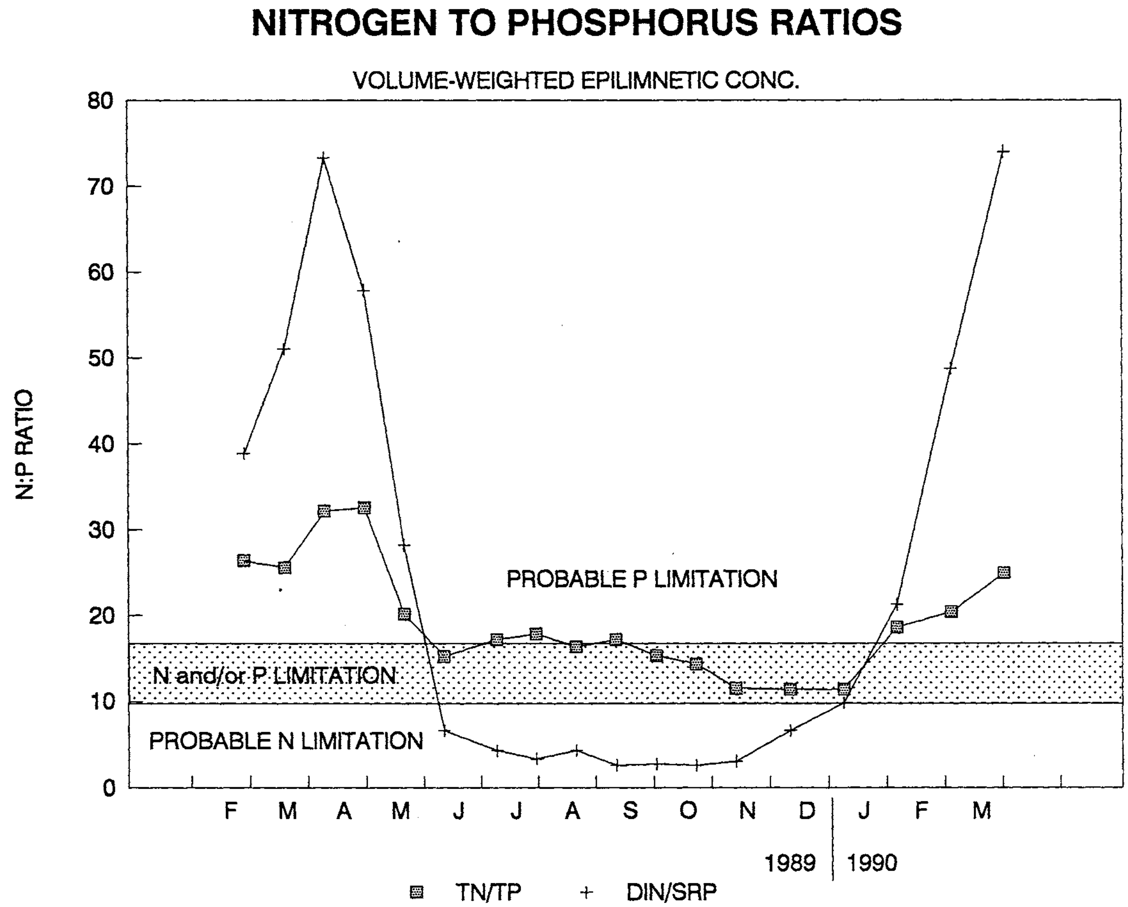


Figure 6.1. Volume-weighted epilimnetic total and soluble N:P ratios for Lake Sawyer, February 1989 through March 1990.

review of the sediment TN:TP ratios (Figure 5.23) suggests a trend since the 1940s from P limitation towards co-limitation of N and P in Lake Sawyer.

Schindler (1978) found that nitrogen fixation within lake waters would allow proportional phosphorus uptake should the N:P ratio for the incoming load be reduced to 5:1. The annual N:P ratio for the incoming load (discussed in more detail below) to Lake Sawyer during the study year (April 1989 through March 1990) was approximately 19:1. Therefore, Lake Sawyer does not appear to be nitrogen limited based on the N:P ratio of the incoming load to the lake. Also, in view of the imminent removal of the Black Diamond WTP load to Lake Sawyer, the N:P ratio of the incoming load to the lake will increase, decreasing the likelihood of nitrogen limitation.

The most probable explanation of nutrient limitation presently in Lake Sawyer is a combination of both P and N limitation, with the limiting factor occasionally being only P or N in the short term. In cases of cultural eutrophication, such as at Lake Sawyer, it is common to find a historical long-term P limited condition in the lake which develops, with the onset of additional P loading, into a nitrogen limited condition (either occasional or short-term). This might be attributed to the rather low TN:TP ratio found in sewage. In such cases, the control of P as the limiting nutrient (possibly with additional control of N) may still impart the greatest regulation of algal productivity (OECD, 1982; Welch, 1980; Cole, 1979). Since N and P both appear to potentially limit productivity, predictive models for estimating algal biomass and transparency for various loading scenarios will be developed with both N and P as controlling variables.

6.2 Trophic State Indicator Predictive Models

The general approach to predicting trophic indicators of interest (e.g. chlorophyll *a* and Secchi depth) is to employ lake-specific regression equations (Mancini *et al.*, 1983). Predictive models for estimating trophic state indicator values were developed from Lake Sawyer data. Chlorophyll *a* and Secchi disk transparency were selected as indicators of trophic status. Models to predict these variables from water column TP and TN are described below.

6.2.1 Chlorophyll *a*

A regression analysis of growing season (March-October) concentrations of chlorophyll *a*, TN, and TP showed a strong dependence of chlorophyll *a* on both nutrients. A similar finding was reported by Smith (1982) for 228 north latitude lakes. Multiple regression, including both N and P as independent variables, allows the prediction of chlorophyll *a* to reflect co-limitation of both nutrients. A log-transformed multiple linear regression model provided the best fit to the Lake Sawyer data:

$$\text{Log (Chla)} = 1.63 * \text{Log (TP)} + 0.753 * \text{Log (TN)} - 3.12 \qquad \text{(eqn 6.1)}$$

$$r^2 = 0.88$$

where "Log" denotes the base 10 logarithm, "Chla" is chlorophyll *a* in μg/L, and TP and TN are in μg P/L and μg N/L, respectively.

6.2.2 Secchi Disk Transparency

The most useful basis for relating Secchi depth to algal biomass is the Beer-Lambert law for light extinction in water, which says that light intensity attenuates exponentially with depth as in (Reckhow and Chapra, 1983):

$$I_z = I_o \exp(-nz) \qquad \text{(eqn 6.2)}$$

where I_z is light intensity at depth z, I_o is light intensity at the lake surface, n is the total light extinction coefficient (m^{-1}; base e), and z is water depth (m). The value of n is commonly assumed to be a function of algal and non-algal properties (Lorenzen, 1980):

$$n = a + b\,(\text{Chla}) \qquad \text{(eqn 6.3)}$$

where "a" represents non-algal extinction (e.g. from color) and "b" is the incremental extinction from algae. The values of "a" and "b" were determined for Lake Sawyer from linear regression (Figure 5.3), such that during the growing season (March-October):

$$n = 0.366 + 0.0433\,(\text{Chla}) \qquad \text{(eqn 6.4)}$$

$$r^2 = 0.89$$

Equation 6.2 can be rearranged to solve for depth (z):

$$z = \frac{-\ln(I_z/I_o)}{n} \qquad \text{(eqn 6.5)}$$

The fraction of incident light that corresponds to the Secchi depth (I_z/I_o), which was observed to be 10.3% $\pm$ 0.9% of incident light for Lake Sawyer during the growing season (Figure 5.2). The value of I_z/I_o at the Secchi depth is typically about 10% for most lakes (Reckhow and Chapra, 1983). Equation 6.4 and 6.5 can be used to predict Secchi depth in Lake Sawyer for any given chlorophyll *a* concentration during the growing season. The potential for water clarity in Lake Sawyer is also evident from equations 6.4 and 6.5, assuming chlorophyll *a* is zero, which yields a maximum possible Secchi depth of 6.2 $\pm$ 1.0 meters. The relatively high non-algal light extinction (and relatively low maximum possible transparency) in Lake Sawyer is probably due to the water color from naturally occurring dissolved humic substances derived from the tributary wetlands. For comparison, Secchi transparency in pristine lakes with less humic color than Lake Sawyer is generally much higher than the maximum possible value in Lake Sawyer (e.g., typical transparency of 29 meters in Crater Lake, 24 meters in Lake Tahoe, 13 meters in Lake Chelan, Patmont *et al.*, 1989).

6.3 External Nutrient Loading

6.3.1 Wetland Treatment Efficiency

The ability of natural wetlands to assimilate nutrient loads from wastewater is a function of the loading rate and wetland treatment area. At low loading rates, natural wetlands have been reported to remove substantial quantities of N and P from domestic wastewater (Nichols, 1983). Numerous wastewater land application studies have shown that wastewater P does not move far in the soil but is retained primarily in the unsaturated surface zone. It is well known that soluble inorganic P is readily immobilized in soils by adsorption and precipitation reactions with aluminum, iron, calcium, and clay minerals. Bacterial nitrification and denitrification is one mechanism for removing wastewater N. Uptake of nutrients by wetland vegetation is generally greatest during periods of active growth. Release of nutrients from decaying vegetation may result in a pattern of high removal in the growing season and lower removal (or net release) during the non-growing season.

In order to make predictions of Lake Sawyer's response to the Black Diamond WTP diversion, an examination was made of nutrient attenuation performance in the Rock Creek wetlands. Specifically, an assessment was made to determine what fraction of the WTP's effluent load to the wetland entered Lake Sawyer. To answer the question of wetland performance, two modeling approaches were considered: (1) a simple mass balance model; and (2) use of a conservative effluent tracer (chloride) to estimate effluent volume fractions in downstream samples and calculate treatment efficiency at the downstream location from measured background, effluent and downstream nutrient concentrations. The attenuation of both total phosphorus (TP) and total nitrogen (TN) were estimated for each survey using both model approaches.

The steady state mass balance model assumes that all loading inputs to the wetland have equal attenuation. The treatment efficiency for this model applies to the wetland's ultimate treatment performance for all incoming loads. The steady state mass balance equation is written as:

$$M_{wl} = M_p + M_{rca} + M_{mm} + M_{ps} + M_{bdlc} + M_{le} + M_{gwi} + M_{rcmb} \qquad \text{(eqn 6.6)}$$

where M denotes mass loading of TN or TP, and subscripts are as follows:

wl = wetland storage accumulation/release;
p = precipitation;
rca = Rock Creek at Abrams Road;
mm = Morganville Marsh;
ps = Palmer Spring;
bdlc = Black Diamond Lake Creek;
le = Black Diamond WTP lagoon effluent;
gwi = net groundwater inflow; and
rcmb = Rock Creek at Morganville Bridge.

The retention (R) or "treatment efficiency" may be calculated as:

$$R = 1 - \frac{\text{output}}{\text{input}} \qquad \text{(eqn 6.7)}$$

where output is the output flux (Rock Creek at the Morganville Bridge) and input is the sum of all inputs in equation 6.6.

An alternative to the simple mass balance model is the use of chloride as a conservative tracer of effluent at the downstream sampling point. Measurement of upstream background (i.e. input sources in equation 6.6 other than M_{le}), lagoon effluent, and downstream concentrations of chloride, TP and TN allows the effluent volume fraction and nutrient attenuation in the wetland to be calculated. The chloride tracer technique is more versatile than the simple mass balance equation (equation 6.6) because it allows various assumptions about background and effluent retention to be made. Also, nutrient retention calculated using the chloride tracer technique is expected to be less variable than from equation 6.6 because instantaneous equilibrium of input and output discharges is not essential since the downstream effluent volume fraction is directly calculated from chloride.

The chloride tracer technique was used to estimate wetland treatment efficiency two ways: 1) assuming that background and effluent nutrient loads are equally retained in the wetland; and 2) estimating WTP effluent load attenuation by assuming that background nutrient load is not retained in the wetland. The first step in the chloride tracer technique is to solve for the volume fraction of effluent (V_e) present at the downstream sampling point:

$$V_e = (C_m - C_b) / (C_e - C_b) \qquad \text{(eqn 6.8)}$$

where "C" denotes chloride concentration and subscripts "m", "b", and "e" represent downstream, background, and effluent, respectively. After the volume fraction of effluent is determined, the nutrient retention (R) can be calculated one of two ways. The first way, assuming equal retention of effluent and background loads, is represented by:

$$R = P_m / [(1 - V_e) * P_b + V_e * P_e] \qquad \text{(eqn 6.9)}$$

where "R" is the fraction of effluent and background nutrient loads that is retained in the wetland, "P" denotes TP or TN concentration, and subscripts are as above. Results based on equation 6.9, assuming equal retention of effluent and background loads, are expected to be equivalent to equation 6.6 with the exception of decreased variability because of deviations from equilibrium of input and output flows. As expected, the retention calculated by equation 6.9 using Rock Creek wetland data was not significantly different from equation 6.6 and 6.7, and variability was lower using equation 6.9.

Table 6.1. Calculated Total P removal efficiency for the Rock Creek wetland between the WTP discharge and Rock Ck at Lake Sawyer (RCLS).

	Bkg Total P (ug P/L)	Bkg Chloride (mg/L)	Effluent Total P (ug P/L)	Effluent Chloride (mg/L)	RCLS Total P (ug P/L)	RCLS Chloride (mg/L)	RCLS Effluent Volume Fraction	Fraction Total P Removed
27-Feb-89	23.4		4774		81.4			
21-Mar-89	21.8	2.27	4006	15.5	67.4	2.90	0.0476	0.755
11-Apr-89	24.2	2.01	3662	15.1	49.0	2.40	0.0302	0.768
02-May-89	29.7	2.18	5462	20.2	150.0	3.18	0.0555	0.598
23-May-89	37.1	1.78	5893	23.1	133.0	2.90	0.0524	0.684
13-Jun-89	34.4	1.71	6307	23.2	176.1	3.10	0.0646	0.647
12-Dec-89	26.3	2.56	3571	15.9	77.1	3.35	0.0591	0.752
09-Jan-90	64.9	1.77	4113	15.0	71.1	2.15	0.0291	0.932
06-Feb-90	24.1	2.06	3686	15.3	72.6	2.40	0.0256	0.480
06-Mar-90	33.4	2.03	4219	17.5	84.5	3.05	0.0659	0.808
03-Apr-90	28.2	1.71	5054	18.5	85.0	2.65	0.0559	0.793
WET SEASON (Dec-Jun) SUMMARY STATISTICS:				N:				10
				Avg:				0.722
				SE:				0.040
11-Jul-89	32.6	1.83	6902	27.4	223.8	3.65	0.0713	0.607
01-Aug-89	42.9	1.74	7567	29.9	258.7	4.10	0.0838	0.654
22-Aug-89	37.6	2.14	7774	29.0	404.1			
12-Sep-89	40.4	1.77	7265	29.0	137.0			
03-Oct-89	34.8	2.04	8513	31.7	300.8	9.90	0.2650	0.878
24-Oct-89	27.5	2.26	7870	29.6	203.8	11.35	0.3330	0.929
14-Nov-89	35.2	2.95	7101	26.4	126.8	5.48	0.1082	0.876
DRY SEASON (Jul-Nov) SUMMARY STATISTICS:				N:				5
				Avg:				0.789
				SE:				0.066
ANNUAL SUMMARY STATISTICS:				N:				15
				Avg:				0.744
				SE:				0.034

Table 6.2. Calculated Total N removal efficiency for the Rock Creek wetland between the WTP discharge and Rock Ck at Lake Sawyer (RCLS).

	Bkg Total N (ug N/L)	Bkg Chloride (mg/L)	Effluent Total N (ug N/L)	Effluent Chloride (mg/L)	RCLS Total N (ug N/L)	RCLS Chloride (mg/L)	RCLS Effluent Volume Fraction	Fraction Total N Removed
27-Feb-89	1661		20125	17.9	1020			
21-Mar-89	991	2.27	13870	15.5	1117	2.90	0.0476	0.304
11-Apr-89	932	2.01	13067	15.1	761	2.40	0.0302	0.414
02-May-89	574	2.18	16535	20.2	490	3.18	0.0555	0.665
23-May-89	503	1.78	19840	23.1	439	2.90	0.0524	0.711
13-Jun-89	582	1.71	21461	23.2	425	3.10	0.0646	0.780
12-Dec-89	1275	2.56	14465	15.9	1201	3.35	0.0591	0.416
09-Jan-90	1406	1.77	17980	15.0	1136	2.15	0.0291	0.398
06-Feb-90	1183	2.06	14805	15.3	1155	2.40	0.0256	0.247
06-Mar-90	862	2.03	16649	17.5	639	3.05	0.0659	0.664
03-Apr-90	717	1.71	16463	18.5	505	2.65	0.0559	0.684
WET SEASON (Dec-Jun) SUMMARY STATISTICS:				N:				10
				Avg:				0.528
				SE:				0.061
11-Jul-89	415	1.83	22250	27.4	454	3.65	0.0713	0.770
01-Aug-89	507	1.74	15043	29.9	546	4.10	0.0838	0.683
22-Aug-89	656	2.14	16707	29.0	613			
12-Sep-89	634	1.77	16563	29.0	542			
03-Oct-89	567	2.04	16413	31.7	626	9.90	0.2650	0.869
24-Oct-89	507	2.26	16607	29.6	607	11.35	0.3330	0.897
14-Nov-89	1487	2.95	17709	26.4	1335	5.48	0.1082	0.588
DRY SEASON (Jul-Nov) SUMMARY STATISTICS:				N:				5
				Avg:				0.761
				SE:				0.057
ANNUAL SUMMARY STATISTICS:				N:				15
				Avg:				0.606
				SE:				0.052

The alternative assumption, estimated effluent load retention (R) with conservative background nutrient load, is represented by:

$$R = [\, P_m - P_b * (1 - V_e)\,] \,/\, (V_e * P_e) \qquad \text{(eqn 6.10)}$$

Historical data, sixteen observations at approximately monthly intervals, from the Rock Creek wetland prior to installation of the Black Diamond WTP (KCM, 1982) were used to evaluate whether equation 6.9 or equation 6.10 should be used. For TP, comparison of upstream (Rock Creek at Abrams Road) with downstream (Rock Creek at Morganville Bridge) concentrations revealed no significant retention within the wetland (paired sample t-test; probability < 80%). This finding suggests that background TP is not retained in the Rock Creek wetland and equation 6.10 is applicable for estimating retention of TP loading from the WTP. For TN, a highly significant background retention (paired sample t-test; probability > 99%) was observed, which suggests that equation 6.9 is appropriate for TN retention calculations. The finding of significant N removal as opposed to insignificant P removal for background loads may be a result of possible N limitation of wetland vegetation (Mitsch and Gosselink, 1986). This finding is also consistent with the primary mechanisms for removal of N and P. Retention of P is probably greatest in the unsaturated surface soils and may be insignificant in the saturated zone, especially if anaerobic conditions develop. Retention of N is favored by conditions in the saturated soils because of the requirements for nitrification and denitrification, which may result in substantial losses of nitrogen to the atmosphere.

Table 6.1 and 6.2 present calculated nutrient removal efficiency in the Rock Creek system between the WTP lagoon effluent discharge point and the shore of Lake Sawyer (RCLS) during the study. Average annual TP retention was 74% ± 3%. Wet season (December-June) TP retention was only slightly lower than dry season (July-November). Average annual TN retention was 61% ± 5%, and dry season TN retention was considerably higher than wet season. The large discrepancy between dry and wet season TN retention suggests that summer plant uptake may be a dominant removal mechanism for TN. The less seasonal retention of TP suggests that physical/chemical processes may be more significant for TP retention than for TN.

The WTP lagoons provide an additional TP removal of 16.5 ± 4.5% and TN removal of 30.2 ± 7.6% based on six samplings of the lagoon influent and effluent between March 1989 and April 1990. Overall reduction of raw sewage TP and TN from the lagoons and natural wetlands is therefore 77.0 ± 5.5% and 72.5 ± 9.2%, respectively, surpassing original design removal expectations.

6.3.2 External Loads

Monthly and annual external loads of TN and TP from all sources were estimated for the study year. Loading from Rock Creek was estimated as the sum of attenuated WTP lagoon effluent

Table 6.3. Summary of study-year Black Diamond WTP and background nutrient loads to Rock Ck and Lake Sawyer.

<----- TOTAL P (Kg P) ----

	Black Diamond WTP Effluent Load to Rock Ck Wetland	Rock Ck Background Load to Rock Ck Wetland	Wetland Retention of WTP Load (Conserv. bkg)	Black Diamond WTP Load to Lake Sawyer after Wetland Retention	Rock Ck Background Load to Lake Sawyer after Wetland Retention	Total Rock Ck Load to Lake Sawyer During Study Year
March 1989	142 ± 27	35.8 ± 6.6	78% ± 3.5%	31.8 ± 7.8	35.8 ± 6.6	67.6 ± 10.3
April	130 ± 26	26.9 ± 4.7	72% ± 3.3%	36.6 ± 8.5	26.9 ± 4.7	63.5 ± 9.7
May	137 ± 26	10.7 ± 1.4	65% ± 3.0%	48.2 ± 9.9	10.7 ± 1.4	58.9 ± 10.0
June	126 ± 24	6.4 ± 0.9	64% ± 2.9%	44.8 ± 9.2	6.4 ± 0.9	51.2 ± 9.2
July	105 ± 20	3.5 ± 0.4	62% ± 2.9%	39.5 ± 8.1	3.5 ± 0.4	43.0 ± 8.1
August	104 ± 19	1.5 ± 0.2	67% ± 3.1%	34.5 ± 7.2	1.5 ± 0.2	36.0 ± 7.2
September	113 ± 21	0.9 ± 0.1	85% ± 3.9%	16.6 ± 5.4	0.9 ± 0.1	17.5 ± 5.4
October	121 ± 23	3.0 ± 0.4	90% ± 4.1%	11.6 ± 5.5	3.0 ± 0.4	14.5 ± 5.5
November	130 ± 26	23.2 ± 3.9	87% ± 4.0%	17.5 ± 6.2	23.2 ± 3.9	40.6 ± 7.3
December	142 ± 27	45.1 ± 6.0	73% ± 3.4%	37.7 ± 8.6	45.1 ± 6.0	82.9 ± 10.5
January 1990	151 ± 28	92.5 ± 9.7	58% ± 2.7%	62.5 ± 12.5	92.5 ± 9.7	155.0 ± 15.8
February	125 ± 25	31.2 ± 6.0	58% ± 2.7%	51.8 ± 11.0	31.2 ± 6.0	82.9 ± 12.6
ANNUAL TOTAL: (Mar89-Feb90)	1524 ± 85	281 ± 16		433 ± 30	281 ± 16	714 ± 34

<----- TOTAL N (Kg N) ----

	Black Diamond WTP Effluent Load to Wetland	Rock Ck Background Load to Rock Ck Wetland	Wetland Retention of Total Load (Noncons. bkg)	Black Diamond WTP Load to Lake Sawyer after Wetland Retention	Rock Ck Background Load to Lake Sawyer after Wetland Retention	Total Rock Ck Load to Lake Sawyer During Study Year
March 1989	486 ± 88	1848 ± 408	49% ± 4.2%	249 ± 50	947 ± 223	1196 ± 229
April	449 ± 83	1230 ± 282	48% ± 4.1%	235 ± 47	645 ± 157	881 ± 164
May	399 ± 73	282 ± 85	69% ± 6.0%	122 ± 33	86 ± 31	208 ± 45
June	349 ± 63	144 ± 50	77% ± 6.7%	81 ± 27	33 ± 15	114 ± 31
July	267 ± 49	63 ± 25	74% ± 6.4%	69 ± 21	16 ± 8	85 ± 23
August	259 ± 47	23 ± 10	69% ± 6.0%	79 ± 21	7 ± 3	86 ± 21
September	293 ± 53	14 ± 7	85% ± 7.3%	45 ± 23	2 ± 1	47 ± 23
October	321 ± 58	55 ± 22	87% ± 7.5%	41 ± 25	7 ± 5	48 ± 26
November	455 ± 84	987 ± 236	61% ± 5.3%	177 ± 41	385 ± 106	562 ± 113
December	538 ± 98	1704 ± 360	43% ± 3.7%	309 ± 60	977 ± 216	1286 ± 224
January 1990	586 ± 105	2977 ± 583	36% ± 3.1%	377 ± 70	1915 ± 386	2292 ± 393
February	473 ± 92	1541 ± 370	38% ± 3.3%	294 ± 59	957 ± 235	1250 ± 242
ANNUAL TOTAL: (Mar89-Feb90)	4876 ± 267	10867 ± 959		2077 ± 149	5978 ± 581	8055 ± 600

Table 6.4. Summary of external loads of total P and total N (means ± standard error).

TOTAL P (Kg P)

	Study Year (1989-1990)						Total External Load	
	Ravensdale Ck	Rock Ck	Precipitation	Calculated Groundwater Inflow (1)	Septic Systems (1)	Water Budget Residual Load (2)	STUDY YEAR	TYPICAL YEAR FOLLOWING WTP DIVERSION (3)
March 1989	19.8 ± 2.5	67.6 ± 10.3	10.1 ± 4.0	0.10 ± 0.2	1.4 ± 0.6	0.8 ± 1.8	99.8 ± 11.5	74.0 ± 10.9
April	18.4 ± 2.3	63.5 ± 9.7	5.4 ± 1.2	0.01 ± 0.0	0.7 ± 0.3	2.4 ± 1.7	90.3 ± 10.2	59.5 ± 6.1
May	11.3 ± 1.3	58.9 ± 10.0	5.8 ± 0.7	0.03 ± 0.1	0.7 ± 0.3		76.7 ± 10.2	36.6 ± 4.4
June	8.5 ± 1.1	51.2 ± 9.2	5.0 ± 2.4	0.07 ± 0.1	0.7 ± 0.3		65.5 ± 9.6	20.4 ± 3.5
July	5.0 ± 0.6	43.0 ± 8.1	5.0 ± 3.2	0.03 ± 0.1	0.4 ± 0.2		53.3 ± 8.7	12.0 ± 3.7
August	2.7 ± 0.3	36.0 ± 7.2	5.0 ± 2.1	0.05 ± 0.1	0.4 ± 0.2		44.0 ± 7.5	6.4 ± 2.2
September	1.6 ± 0.2	17.5 ± 5.4	5.0 ± 2.1	0.04 ± 0.1	0.4 ± 0.2		24.5 ± 5.8	7.4 ± 2.5
October	0.8 ± 0.1	14.5 ± 5.5	6.0 ± 3.7	0.03 ± 0.1	0.4 ± 0.2		21.7 ± 6.6	9.0 ± 4.2
November	4.1 ± 0.4	40.6 ± 7.3	10.1 ± 6.9	0.02 ± 0.0	0.4 ± 0.2		55.3 ± 10.1	37.6 ± 12.3
December	14.3 ± 1.6	82.9 ± 10.5	5.3 ± 4.1	0.05 ± 0.1	0.7 ± 0.3		103.3 ± 11.4	87.3 ± 17.8
January 1990	19.3 ± 2.4	155.0 ± 15.8	0.7 ± 0.3	0.17 ± 0.4	1.4 ± 0.6		176.7 ± 16.0	90.0 ± 14.4
February	19.0 ± 2.5	82.9 ± 12.6	2.8 ± 1.1	0.10 ± 0.2	1.4 ± 0.6	1.0 ± 1.8	107.3 ± 13.0	75.8 ± 7.4
ANNUAL TOTAL: (Mar89-Feb90)	124.6 ± 5.5	713.7 ± 33.6	66.1 ± 11.1	0.7 ± 1.4	9.0 ± 4.0	4.1 ± 3.0	918 ± 36	516 ± 30

TOTAL N (Kg N)

	Study Year (1989-1990)						Total External Load	
	Ravensdale Ck	Rock Ck	Precipitation	Calculated Groundwater Inflow (1)	Septic Systems (1)	Water Budget Residual Load (2)	STUDY YEAR	TYPICAL YEAR FOLLOWING WTP DIVERSION (3)
March 1989	1430 ± 129	1196 ± 229	95 ± 43	7 ± 15	30 ± 14	49 ± 118	2808 ± 292	2216 ± 347
April	1295 ± 114	881 ± 164	85 ± 59	3 ± 6	15 ± 7	153 ± 107	2431 ± 234	1905 ± 225
May	537 ± 48	208 ± 45	66 ± 23	3 ± 7	15 ± 7		829 ± 70	922 ± 134
June	365 ± 41	114 ± 31	43 ± 17	3 ± 6	15 ± 7		539 ± 55	438 ± 77
July	187 ± 20	85 ± 23	31 ± 12	1 ± 3	8 ± 4		313 ± 33	216 ± 56
August	91 ± 10	86 ± 21	52 ± 28	1 ± 2	8 ± 4		238 ± 37	90 ± 33
September	52 ± 6	47 ± 23	52 ± 28	1 ± 2	8 ± 4		159 ± 37	108 ± 43
October	26 ± 3	48 ± 26	53 ± 32	1 ± 2	8 ± 4		135 ± 41	121 ± 57
November	158 ± 14	562 ± 113	62 ± 44	1 ± 3	8 ± 4		792 ± 122	906 ± 408
December	811 ± 66	1286 ± 224	33 ± 26	4 ± 9	15 ± 7		2150 ± 235	2896 ± 580
January 1990	1619 ± 120	2292 ± 393	5 ± 2	10 ± 21	30 ± 14		3957 ± 411	3280 ± 539
February	1400 ± 127	1250 ± 242	22 ± 10	8 ± 17	30 ± 14	66 ± 114	2777 ± 297	2656 ± 291
ANNUAL TOTAL: (Mar89-Feb90)	7972 ± 263	8055 ± 600	598 ± 108	46 ± 92	190 ± 90	268 ± 196	17129 ± 704	15755 ± 928

1 Groundwater inflow and septic system loading from Hart Crowser, 1990.

2 Water budget residual load was estimated using the net inflow discharge from the water budget residual and the concentrations in Ravensdale Creek (total soluble P = 7 ± 0.8 ugP/L; total soluble N = 453 ± 31 ugN/L).

3 "Typical Year" loads following WTP diversion were estimated based on scaled study year loads after subtracting attenuated WTP input to Lake Sawyer. Scaling of Ravensdale and Rock Ck background annual study year loads was based on the ratio of 30-year average to study-year precipitation at Landsburg. Monthly loads were estimated from scaled annual loads based on the distribution of long-term (1954-59) monthly average flows at Covington Ck .

loads and background Rock Creek loads (Table 6.3), where background loading was based on regression of flow-weighted background input concentrations (i.e. Rock Creek upstream from Abrams Road, Palmer Spring, Black Diamond Lake Creek, Morganville Marsh, and precipitation; Figure 5.26) with flows in Rock Creek at Lake Sawyer. As described above, Rock Creek background TP loads were assumed to be conservatively transported to Lake Sawyer, while background TN loads were retained as calculated by equation 6.9. The Rock Creek nutrient loads to Lake Sawyer calculated in this way were not significantly different (t-test, probability ≥ 95%) from an independent regression estimate based on concentrations and flows measured directly in Rock Creek at the shore of Lake Sawyer (Figure 5.25), which supports the validity of the approach used. Rock Creek loads calculated as the sum of attenuated WTP and background inputs were used so that WTP loads could be subtracted for evaluation of loading following WTP diversion.

Table 6.4 presents a summary of monthly and annual external TP and TN loads from all sources to Lake Sawyer during the study year. Typical-year loads following WTP diversion were also estimated based on scaling of study year loads after subtracting attenuated WTP loads from the Rock Creek loading estimate. The ratio of long-term (30-year average) to study-year precipitation at Landsburg was used to adjust the annual study-year loads to estimated typical-year loads. Monthly loads for a "typical-year" were then apportioned from the annual load based on the long-term monthly fractions of annual discharge at Covington Creek (1954-59 USGS data) for nutrient loads from Ravensdale Creek and Rock Creek, and from long-term monthly fractions of annual precipitation at Landsburg for loading from precipitation. Study year loads from groundwater inflow and septic systems (Hart Crowser, 1990), which were relatively minor, were assumed to be representative of typical loads from these sources.

6.4 Nutrient Budgets and Mass Balance Models

The budget of nutrients (TP and TN) is an accounting of all inputs and outputs. In addition to the external loads discussed above, internal cycling from lake sediments and net sedimentation also influence water column concentrations of nutrients. In order to estimate the total nutrient budget, a mass balance model relating all sources and losses of nutrients must be developed (Cooke *et al.*, 1986; Mancini *et al.*, 1983). Two types of mass balance models were used in the evaluation. A time-variable model was used to evaluate seasonal relationships between external and internal processes and examine seasonal trophic state indicators. A steady-state annual model was used to more clearly illustrate the relative magnitude of various loading sources and to predict long-term average conditions.

6.4.1 Time Variable Model

Time variable analysis of the Lake Sawyer nutrient concentrations included whole-lake and two-layer (epilimnion and hypolimnion) modeling. A whole-lake finite difference model was constructed to model the seasonal nutrient budget dynamics for the whole year. Results of the whole-lake time-variable model were used to estimate total internal sediment feedback of TP and

TN. A whole-lake finite difference mass balance model presented by Chapra and Reckhow (1983) was applied to Lake Sawyer. The whole-lake mass balance is described as:

$$V_{wl} \frac{dC_{wl}}{dt} = W - Q\, C_{wl} - v_{swl}\, A\, C_{wl} + I_h \qquad \text{(eqn 6.11)}$$

where:

W = external load of TP or TN (mg mo^{-1}).
V_{wl} = whole lake volume (8.62 X 10^6 m^3)
C_{wl} = whole lake concentration of TP or TN (mg m^3)
t = time (months)
Q = outflow discharge rate (m^3 mo^{-1})
v_{swl} = apparent TP or TN settling velocity (m mo^{-1})
A = lake surface area (1.11 X 10^6 m^3)
I_h = internal sediment feedback load of TP or TN (mg mo^{-1}).

Internal sediment feedback occurs primarily in the hypolimnion during anaerobic conditions or in the littoral zone. Phosphorus retention in sediments is dependant primarily on iron content, redox potential, and pH. Anoxic conditions typical of monomictic Pacific Northwest lowland lakes, as well as midwest dimictic lakes, can result in relatively large rates of phosphorus release from sediments (Welch, 1977; Cooke *et al.*, 1977). Significant sediment feedback of TP can occur in oxygenated littoral waters at high pH (Jacoby *et al.*, 1982), or with reduced sediment/water interface conditions caused by macrophyte decay (Carpenter, 1983). Sediment feedback of TN also can be significant, especially under anaerobic conditions as bacterial nitrification of ammonia to nitrite and nitrate ceases and adsorptive capacity of sediments is greatly reduced (Wetzel, 1983).

Table 6.5 presents a summary of calibration data used in the finite difference modeling. Equation 6.11 was calibrated to the study year data by rearranging and solving for v_{swl} (Table 6.6). The value of v_{swl} calculated during the months of September through December was negative, and therefore was not a valid estimate of v_{swl} during the anaerobic period because of the internal sediment feedback. The amount of internal sediment feedback during September through December was estimated by assuming that v_{swl} during that period was equal to the average of August and January calculated values, which represents the period immediately before and after anaerobic conditions. A similar technique for mass balance model calibration was used by Shuster *et al.*, (1986) for Lake Sammamish. Results of Monte-Carlo simulation of whole-lake TP and TN for the calibration data set are shown in Figure 6.2.

Internal loading of TP and TN probably is derived mainly from hypolimnetic processes. Macrophyte decomposition may also contribute, but probably represents a minor fraction of the total internal load if macrophytes are assumed to release about 20% of the standing stock (Table 5.2) of nutrients during senescence (Welch *et al.*, 1979).

Table 6.5. Summary of calibration input data for total P and N mass balance models.

	Time-wtd monthly Whole-Lake Concentration	Time-wtd monthly Epilimn Concentration	Time-wtd monthly Hypolimn Concentration	External Load	Outflow Discharge Rate	Vertical Diffusion Coefficient
Model Symbol:	Cwl	Ce	Ch	We	Qe	E
Units:	mg/m3	mg/m3	mg/m3	mg/mo	m3/1000	cm2/sec
<--------- TOTAL P --->						
Mar-89	22.5 ± 1.5	23.0 ± 1.4	21.3 ± 2.0	9.98E+07 ± 1.15E+07	4272 ± 124	1.7E-01 ± 7.5E-01
Apr-89	16.7 ± 1.1	17.3 ± 1.3	15.4 ± 0.6	9.03E+07 ± 1.02E+07	3865 ± 112	3.4E-02 ± 7.5E-01
May-89	15.4 ± 1.3	16.9 ± 1.5	11.9 ± 1.1	7.67E+07 ± 1.02E+07	1571 ± 139	8.1E-03 ± 2.1E-03
Jun-89	18.2 ± 1.9	19.6 ± 1.2	14.9 ± 4.0	6.55E+07 ± 9.60E+06	1098 ± 103	3.9E-03 ± 1.5E-03
Jul-89	19.0 ± 2.5	17.8 ± 1.1	22.0 ± 5.7	5.33E+07 ± 8.70E+06	617 ± 56	3.3E-03 ± 3.5E-04
Aug-89	20.1 ± 2.7	17.0 ± 1.1	27.5 ± 6.5	4.40E+07 ± 7.50E+06	504 ± 46	3.4E-03 ± 2.8E-04
Sep-89	22.1 ± 2.7	14.7 ± 1.0	39.6 ± 6.5	2.45E+07 ± 5.80E+06	337 ± 35	2.8E-02 ± 7.2E-01
Oct-89	29.3 ± 5.4	15.3 ± 0.8	62.1 ± 16.4	2.17E+07 ± 6.60E+06	380 ± 24	-- -- --
Nov-89	32.7 ± 5.3	19.9 ± 0.8	63.0 ± 16.0	5.53E+07 ± 1.01E+07	786 ± 93	-- -- --
Dec-89	39.9 ± 4.8	32.4 ± 1.3	57.6 ± 13.9	1.03E+08 ± 1.14E+07	2952 ± 173	-- -- --
Jan-90	38.8 ± 1.7	38.5 ± 1.4	39.4 ± 2.7	1.77E+08 ± 1.60E+07	4918 ± 296	-- -- --
Feb-90	33.8 ± 1.6	33.9 ± 1.5	33.4 ± 1.6	1.07E+08 ± 1.30E+07	4292 ± 124	-- -- --
Mar-90	26.4 ± 2.0	26.3 ± 2.0	26.8 ± 2.0	-- -- --	-- --	-- -- --
<--------- TOTAL N --->						
Mar-89	618 ± 16	614 ± 15	626 ± 21	2.81E+09 ± 2.92E+08	4272 ± 124	1.7E-01 ± 7.5E-01
Apr-89	563 ± 19	557 ± 16	578 ± 25	2.43E+09 ± 2.34E+08	3865 ± 112	3.4E-02 ± 7.5E-01
May-89	463 ± 24	424 ± 20	556 ± 33	8.29E+08 ± 7.00E+07	1571 ± 139	8.1E-03 ± 2.1E-03
Jun-89	401 ± 26	327 ± 22	574 ± 37	5.39E+08 ± 5.50E+07	1098 ± 103	3.9E-03 ± 1.5E-03
Jul-89	359 ± 14	290 ± 11	521 ± 23	3.13E+08 ± 3.30E+07	617 ± 56	3.3E-03 ± 3.5E-04
Aug-89	322 ± 8	268 ± 8	450 ± 10	2.38E+08 ± 3.70E+07	504 ± 46	3.4E-03 ± 2.8E-04
Sep-89	251 ± 9	229 ± 6	305 ± 17	1.59E+08 ± 3.70E+07	337 ± 35	2.8E-02 ± 7.2E-01
Oct-89	232 ± 14	219 ± 6	264 ± 33	1.35E+08 ± 4.10E+07	380 ± 24	-- --
Nov-89	256 ± 25	234 ± 20	309 ± 36	7.92E+08 ± 1.22E+08	786 ± 93	-- --
Dec-89	378 ± 29	370 ± 26	397 ± 36	2.15E+09 ± 2.35E+08	2952 ± 173	-- --
Jan-90	511 ± 20	518 ± 18	493 ± 24	3.96E+09 ± 4.11E+08	4918 ± 296	-- --
Feb-90	634 ± 22	644 ± 18	609 ± 32	2.78E+09 ± 2.97E+08	4292 ± 124	-- --
Mar-90	575 ± 20	577 ± 17	572 ± 28	-- -- --	-- --	-- -- --

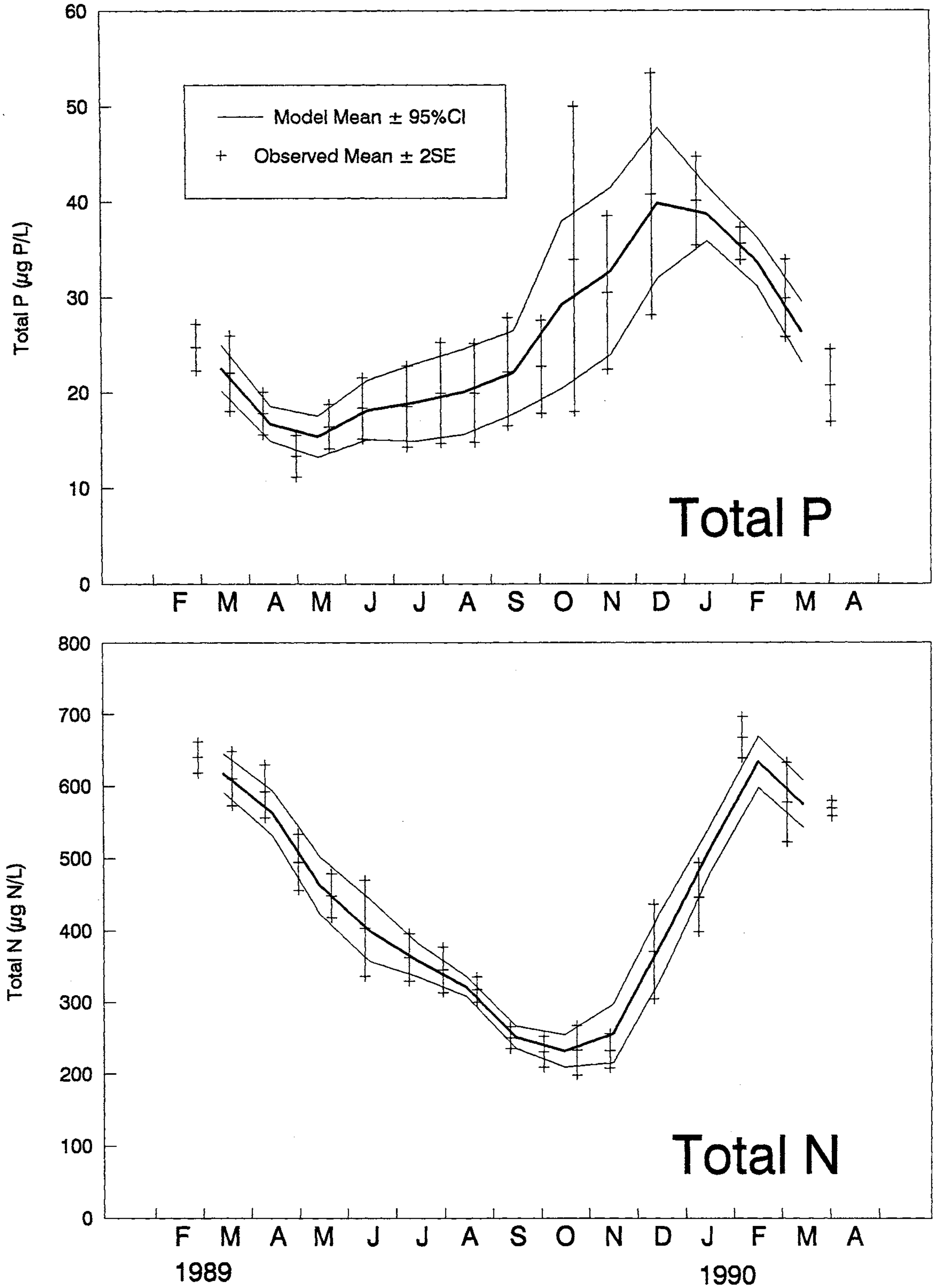


Figure 6.2. Results of Monte-Carlo simulation of study-year calibration data for whole-lake total P and total N.

Table 6.6. Calibration results for time-variable whole-lake total P and N models.

	External Load	Flushing	Measured Whole-Lake Change	Calibrated Settling Velocity	Assumed Settling Velocity (1)	Internal Load
Model Symbol:	We/Vwl	(Qe/Vwl)Cwl	dCwl/dt	vswl	vswl	I
Units:	mg/m3/mo	mg/m3/mo	mg/m3/mo	m/mo	m/mo	Kg/mo
<--------- TOTAL P MODEL --------->						
Mar-89	11.6	11.1	-5.8	2.13	2.13	0.0
Apr-89	10.5	7.5	-1.3	1.99	1.99	0.0
May-89	8.9	2.8	2.8	1.67	1.67	0.0
Jun-89	7.6	2.3	0.8	1.90	1.90	0.0
Jul-89	6.2	1.4	1.1	1.51	1.51	0.0
Aug-89	5.1	1.2	2.0	0.74	0.74	0.0
Sep-89	2.8	0.9	7.1	-1.81	0.71	61.9
Oct-89	2.5	1.3	3.5	-0.59	0.71	42.3
Nov-89	6.4	3.0	7.2	-0.90	0.71	58.4
Dec-89	12.0	13.7	-1.2	-0.11	0.71	36.2
Jan-90	20.5	22.1	-5.0	0.68	0.68	0.0
Feb-90	12.4	16.8	-7.4	0.69	0.69	0.0
Annual Total:						199 ± 132 Kg/y
<--------- TOTAL N MODEL --------->						
Mar-89	326	306	-55	0.94	0.94	0
Apr-89	282	252	-100	1.79	1.79	0
May-89	96	84	-62	1.24	1.24	0
Jun-89	63	51	-42	1.03	1.03	0
Jul-89	36	26	-37	1.03	1.03	0
Aug-89	28	19	-71	1.92	1.92	0
Sep-89	18	10	-19	0.86	1.30	123
Oct-89	16	10	24	-0.62	1.30	495
Nov-89	92	23	122	-1.63	1.30	832
Dec-89	249	130	133	-0.26	1.30	655
Jan-90	459	291	123	0.68	0.68	0
Feb-90	322	316	-58	0.80	0.80	0
Annual Total:						2105 ± 721 Kg/y

1) September-December settling velocity was assumed equal to the average of August and January.

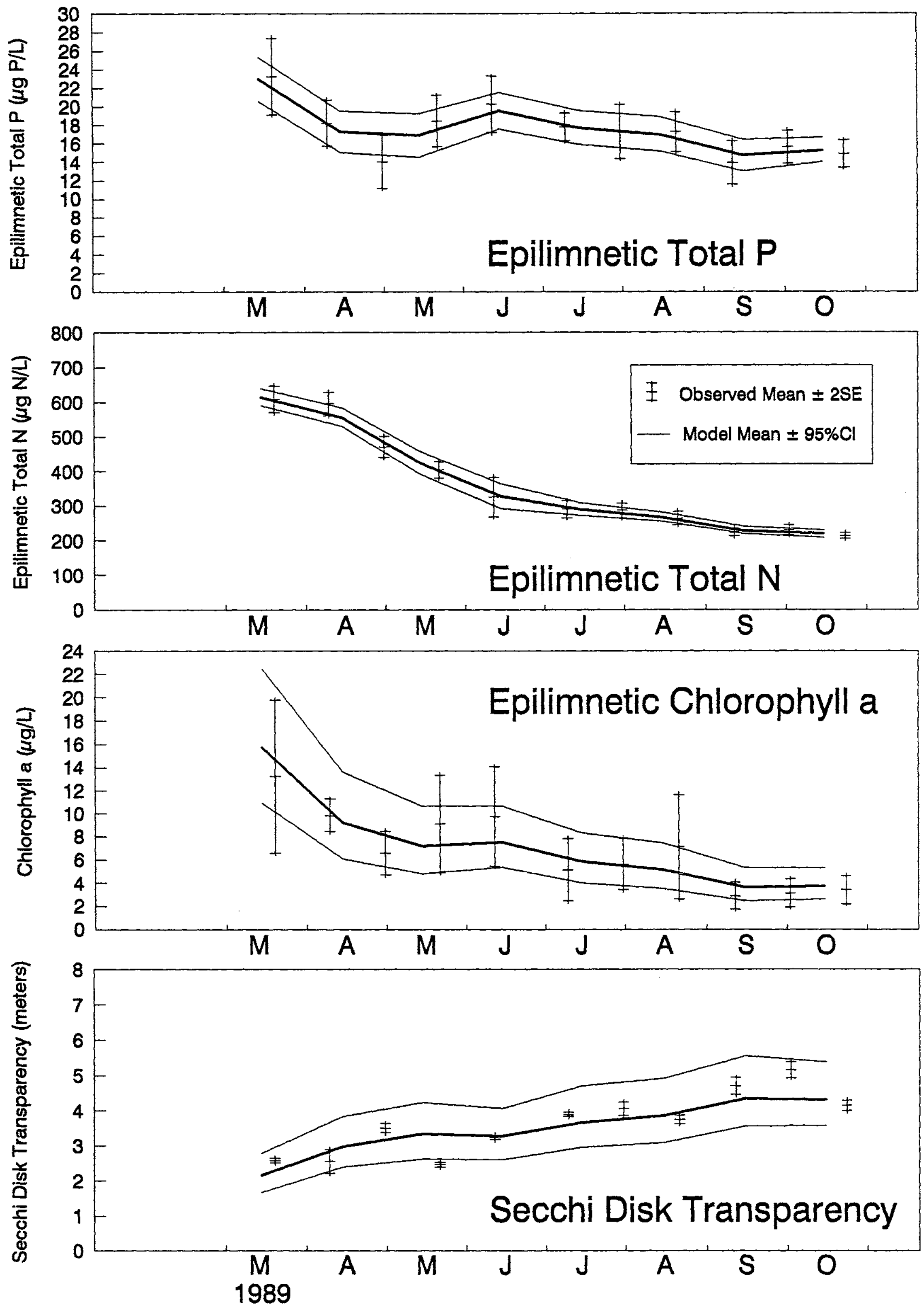


Figure 6.3. Results of Monte-Carlo simulation of study-year epilimnetic water quality for model calibration.

A two-layer (epilimnion and hypolimnion) finite difference numerical solution to the nutrient mass balance described by Chapra and Reckhow (1983) was also applied to Lake Sawyer to allow predictions of growing season (March-October) dynamics of the epilimnion. For the epilimnion, the mass balance equation is:

$$V_e \frac{dC_e}{dt} = W - Q\,C_e + \frac{E_t A_t}{z_{bar}} (C_h - C_e) - v_{se} A_t C_e \qquad \text{(eqn 6.12)}$$

where:

V_e = volume of the epilimnion (6.05×10^6 m^3)
C_e = epilimnetic TP or TN (mg m^{-3})
C_h = hypolimnetic TP or TN (mg m^{-3})
W = external TP or TN load (mg mo^{-1})
Q = outflow discharge rate (m^3 mo^{-1})
E_t = vertical diffusive exchange coefficient across the thermocline (m^2mo^{-1})
A_t = surface area of the thermocline (0.564×10^6 m^2)
v_{se} = epilimnetic TP or TN settling velocity (m mo^{-1}).

The mass balance for the hypolimnion is described as:

$$V_h \frac{dC_h}{dt} = \frac{E_t A_t}{z_{bar}} (C_e - C_h) - v_{sh} A_t C_h + I_h \qquad \text{(eqn 6.13)}$$

where:

v_{sh} = hypolimnetic TP or TN settling velocity (m mo^{-1})
I_h = internal TP or TN sediment feedback (mg mo^{-1}).

Calibration of the two-layer model was accomplished by first rearranging equation 6.12 to solve for v_{se}. Next, equation 6.13 was rearranged to solve for v_{sh}. Monte Carlo simulation of growing season epilimnetic TP and TN calibration data are shown in Figure 6.3. Also show in Figure 6.3 are comparisons of predicted and observed epilimnetic chlorophyll *a* and Secchi depth for the calibration data set using equations 6.1, 6.4 and 6.5 with predicted TP and TN from equation 6.12. In general, Figures 6.2 and 6.3 illustrate calibration of the time-variable model in fitting the study year data set.

6.4.2 Steady-state Model and Annual Nutrient Budget Summary

The most commonly used form of the nutrient mass balance model is the "steady-state" model, which assumes that the average nutrient concentration in a lake is in equilibrium with the annual rates of input (external and internal loads) and loss (sedimentation and outflow). The steady-state solution of the whole lake mass balance model (equation 6.11) is described by Reckhow and Chapra (1983) as:

$$C_{wl} = \frac{W + I_h}{v_{swl} A + Q} = \frac{L}{v_{swl} + q_s} \qquad \text{(eqn 6.14)}$$

where:

$L = (W + I_h) / A$ = areal TP or TN load (mg m^{-2} y^{-1})
$q_s = Q / A$ = areal water loading (m y^{-1})

The apparent settling velocity (v_{swl}) describes the net settling velocity over the lake surface area, and is often used as a variable to calibrate the model to a specific lake since all other terms in equation 6.14 are more readily measured. Values of v_{swl} for TP ranging from 10 to 16 m y^{-1} have been reported (Mancini *et al.*, 1983). Literature reporting settling rates for TN are more scarce than for TP, but Mancini *et al.*,(1983) suggest that TN and TP settling rates are of similar magnitude.

Equation 6.14 was calibrated to the Lake Sawyer study year data by rearranging to solve for v_{swl}. Table 6.7 presents a summary of the study year TP and TN budgets along with the calibration of equation 6.14 to Lake Sawyer. The Lake Sawyer TP settling velocity (16.1 ± 6.5 m y^{-1}) was found to closely agree with the range of values in the literature summarized by Mancini *et al.*, (1983). The Lake Sawyer TN settling velocity (18.6 ± 2.9 m y^{-1}) also appears to be reasonably similar to the TP settling velocity.

Table 6.7 shows that the Black Diamond WTP is the single largest source of TP to Lake Sawyer, and it provides about half of the external TP load and 12% of the external TN load. Since the Black Diamond WTP makes up a greater portion of the TP than TN budgets, TP loading is expected to decrease proportionally more than TN following diversion. Internal loads of TP and TN currently account for less than 20% of the total. Table 6.8 shows the expected "typical-year" nutrient loads to Lake Sawyer following WTP diversion. Also shown in Table 6.8 are the predicted steady-state concentrations of TP and TN based on equation 6.14.

Historical data from Lake Sawyer and Rock Creek prior to installation of the Black Diamond WTP were used to test the validity of the steady-state model calibration shown in Table 6.7. TP loading from Rock Creek was estimated to be 429 ± 32 Kg P/year for a typical year prior to WTP installation based on regression analysis (Figure 5.26) of concentration data from KCM (1982) applied to typical year flows. Substituting this TP load from Rock Creek for the typical-year load from Rock Creek expected after WTP diversion (Table 6.8) provides an estimate of total typical loading prior to 1982.

The steady-state TP concentration in Lake Sawyer for typical loading prior to 1982, predicted using equation 6.14 calibrated to 1989-90 study year data, is 19.2 ± 4.5 μg P/L. This predicted pre-1982 concentration is in excellent agreement with semi-monthly multiple-depth sampling data

Table 6.7. Summary of study-year annual total P and total N budgets for Lake Sawyer and annual steady-state model calibration.

BUDGET SOURCE	STUDY YEAR — TOTAL P: Kg P/year	STUDY YEAR — TOTAL P: % of total load	STUDY YEAR — TOTAL N: Kg N/year	STUDY YEAR — TOTAL N: % of total load
Rock Creek				
- Black Diamond WTP	433 ± 30	(39% ± 5%)	2,077 ± 149	(11% ± 1%)
- Background and Nonpoint	281 ± 16	(25% ± 3%)	5,978 ± 581	(31% ± 3%)
- Rock Creek Total	714 ± 34		8,055 ± 600	
Ravensdale Creek	125 ± 6	(11% ± 1%)	7,972 ± 263	(41% ± 3%)
Atmospheric Deposition	66 ± 11	(6% ± 1%)	598 ± 108	(3% ± 1%)
Background Groundwater	0.7 ± 1.4	(0% ± 0%)	46 ± 92	(0% ± 0%)
Septic Systems	9.0 ± 4.0	(1% ± 0%)	190 ± 90	(1% ± 0%)
Residual External Load	4.1 ± 3.0	(0% ± 0%)	268 ± 196	(1% ± 1%)
TOTAL EXTERNAL LOAD	918 ± 36	(82% ± 10%)	17,129 ± 704	(89% ± 6%)
INTERNAL LOAD	199 ± 124	(18% ± 11%)	2,105 ± 678	(11% ± 4%)
TOTAL LOAD (Kg/year):	1,117 ± 129		19,234 ± 977	
STEADY-STATE MODEL CALIBRATION:				
TOTAL AREAL NUTRIENT LOAD (L; mg/m2/year):	1006 ± 116		17328 ± 881	
WHOLE-LAKE CONCENTRATION (C; mg/m3):	25.7 ± 3.1		416 ± 20	
AREAL HYDRAULIC LOAD (qs; m/y):	23.1 ± 0.4		23.1 ± 0.4	
CALIBRATION APPARENT SETTLING VELOCITY (vs = L/C - qs; m/y):	16.1 ± 6.5		18.6 ± 2.9	

Table 6.8. Summary of "typical-year" post-WTP-diversion annual total P and total N budgets for Lake Sawyer and annual steady-state model predicted concentrations.

BUDGET SOURCE	TYPICAL YEAR AFTER WTP DIVERSION: TOTAL P Kg P/year	TOTAL P % of total load	TOTAL N Kg N/year	TOTAL N % of total load
Rock Creek				
- Black Diamond WTP	- ± -		- ± -	
- Background and Nonpoint	304 ± 26		6,241 ± 737	
- Rock Creek Total	304 ± 26	(43% ± 8%)	6,241 ± 737	(35% ± 5%)
Ravensdale Creek	135 ± 8	(19% ± 4%)	8,643 ± 437	(48% ± 4%)
Atmospheric Deposition	66 ± 11	(9% ± 2%)	598 ± 108	(3% ± 1%)
Background Groundwater	0.7 ± 1.4	(0% ± 0%)	46 ± 92	(0% ± 1%)
Septic Systems	9.0 ± 4.0	(1% ± 1%)	190 ± 90	(1% ± 1%)
Residual External Load	0.6 ± 4.9	(0% ± 1%)	37 ± 314	(0% ± 2%)
TOTAL EXTERNAL LOAD	516 ± 30	(72% ± 14%)	15,755 ± 928	(88% ± 8%)
INTERNAL LOAD	199 ± 124	(28% ± 18%)	2,105 ± 678	(12% ± 4%)
TOTAL LOAD (Kg/year):	715 ± 128		17,860 ± 1,149	
STEADY-STATE MODEL PREDICTIONS:				
TOTAL AREAL NUTRIENT LOAD (L; mg/m2/year):	644 ± 115		16090 ± 1035	
AREAL HYDRAULIC LOAD (qs; m/y):	23.2 ± 2.3		23.2 ± 2.3	
CALIBRATION APPARENT SETTLING VELOCITY (vs; m/y):	16.1 ± 6.5		18.6 ± 2.9	
PREDICTED WHOLE-LAKE CONCENTRATION (C = L / [vs + qs]; mg/m3):	16.4 ± 4.1		384 ± 42	

collected by Metro (Brenner, 1989) from May 1979 to March 1980, which showed whole-lake average TP of 18.3 ± 4.0 μg P/L. The USGS also collected multiple-depth sampling prior to WTP installation (March through September, 1973; McConnell *et al.*, 1976), although at fewer depths and time intervals than Metro. If the USGS data are combined with Metro data, then the pooled average whole-lake TP concentration prior to 1982 was approximately 20 ± 3 μg P/L. The close agreement of predictions from equation 6.14 with the Metro, USGS, and KCM pre-1982 data validates the model calibration. Typical TP concentrations in the lake following WTP diversion are predicted to be 16.4 ± 4.1 μg P/L (Table 6.8), which is somewhat lower than the concentration present prior to 1982.

The TP load to Lake Sawyer from Rock Creek prior to WTP installation was probably significantly lower than the current load, but higher than the load expected following WTP diversion. The difference between loads before WTP installation and following WTP diversion may be explained by TP contributions from wastewater entering the creek from failing community septic systems in use prior to 1982. Therefore, diversion of the WTP from Rock Creek probably will result in TP loading to Lake Sawyer that is significantly lower than the load immediately prior to WTP installation.

Aside from the pre-1982 studies by Metro and USGS, the present Ecology study of 1989-90 is the only other multiple-depth sampling study which provides characterization of seasonally averaged whole-lake concentrations. Metro has been collecting multiple-depth samples once or twice each year (generally in spring) since 1982, in addition to monthly surface sampling during summer, but the Ecology 1989-90 data provide a more representative average of the annual whole-lake concentrations. Therefore, the annual average whole-lake TP and TN concentrations from the 1989-90 study year are considered to be the best estimate of steady state conditions following WTP start-up.

The steady state model may be used to estimate the time required for a lake to respond to loading changes. The response time for a lake may be calculated as (Chapra and Reckhow, 1983):

$$t_\phi = \frac{\ln 100/(100 - \phi)}{\alpha} \qquad \text{(eqn 6.15)}$$

where:

ϕ = percent of new steady state attained
t_ϕ = response time to reach ϕ% of new steady state
$\alpha = (Q + v_{swl}\,A) / V_{wl}$.

Because of the relatively rapid flushing rate in Lake Sawyer (approximately 3 lake volumes per year) and the sedimentation of nutrients, response to changed loading is predicted to be rapid using equation 6.15. The lake is expected to attain about 99% of the new steady state condition within the first year after a change in loading rate. Therefore, decreases in lake concentrations should be observed soon after nutrient loading reductions.

6.5 Trophic Response To Nutrient Loading Changes

The assumed management goal for Lake Sawyer is to minimize the eutrophication process. Although trophic descriptions (e.g. eutrophic, mesotrophic) have no absolute meaning, they are generally used by many lake investigators and managers to denote the nutrient status of a waterbody, or describe the effects of nutrients on water quality (Cooke *et al.*, 1986). Consequently, several attempts have been made to relate descriptive trophic terms to specific boundary values for key water quality parameters. The most rigorous attempt at such a classification scheme was presented by OECD (1982).

The OECD scheme is based on a probabilistic evaluation of an extensive limnological database collected from lakes and reservoirs throughout the northern temperate zone, and reflects the judgement of a large number of limnologists about the relationship between commonly measured parameters (e.g. TP, TN, chlorophyll *a*, etc.) and trophic status categories. An example of the resultant probability distributions of trophic status based on the most highly correlated parameter, in-lake TP, is presented in Figure 6.4. The overlap between trophic categories is substantial, and attests to the uncertain nature of trophic classifications.

Numerous fixed boundaries have been proposed to delineate the trophic classifications (OECD, 1982; Mancini *et al.*, 1983; EPA, 1974; Brezonik, 1976; Dobson *et al.*, 1974; Wetzel, 1983). Table 6.9 presents a summary of the more commonly used relationships between lake total phosphorus concentrations, trophic status, and lake use for north temperate lakes. As a practical management goal, OECD (1982) recommends that for water uses which do not require high purity conditions (e.g. drinking water), a mesotrophic condition should generally provide adequate protection against impacts to important water uses such as recreation and fisheries production. EPA (1986) reached a similar conclusion in recommending that in-lake TP concentrations less than 25 μg P/L should generally protect against undesirable water quality conditions associated with eutrophy.

Table 6.10, Figure 6.5, and Figure 6.6 present a summary of trophic status indicators observed during the study year (1989-90) and predicted for typical conditions following WTP diversion. A significant improvement is predicted based on likely changes in TP, chlorophyll *a*, and Secchi transparency. The ratio of TP:TN is also likely to change significantly toward a more clearly phosphorus limited condition, which supports the use of TP as a trophic indicator.

The Lake Sawyer WLA (Pelletier and Joy, 1989) recommended an in-lake upper-limit TP concentration criterion of 25 μg P/L to protect the water quality of Lake Sawyer. Based on the assessment of this diagnostic study, there is presently an in-lake TP concentration of 25.7 $\pm$ 3.1 μg P/L (i.e. representing the upper limit of the suggested criterion). If the present lake

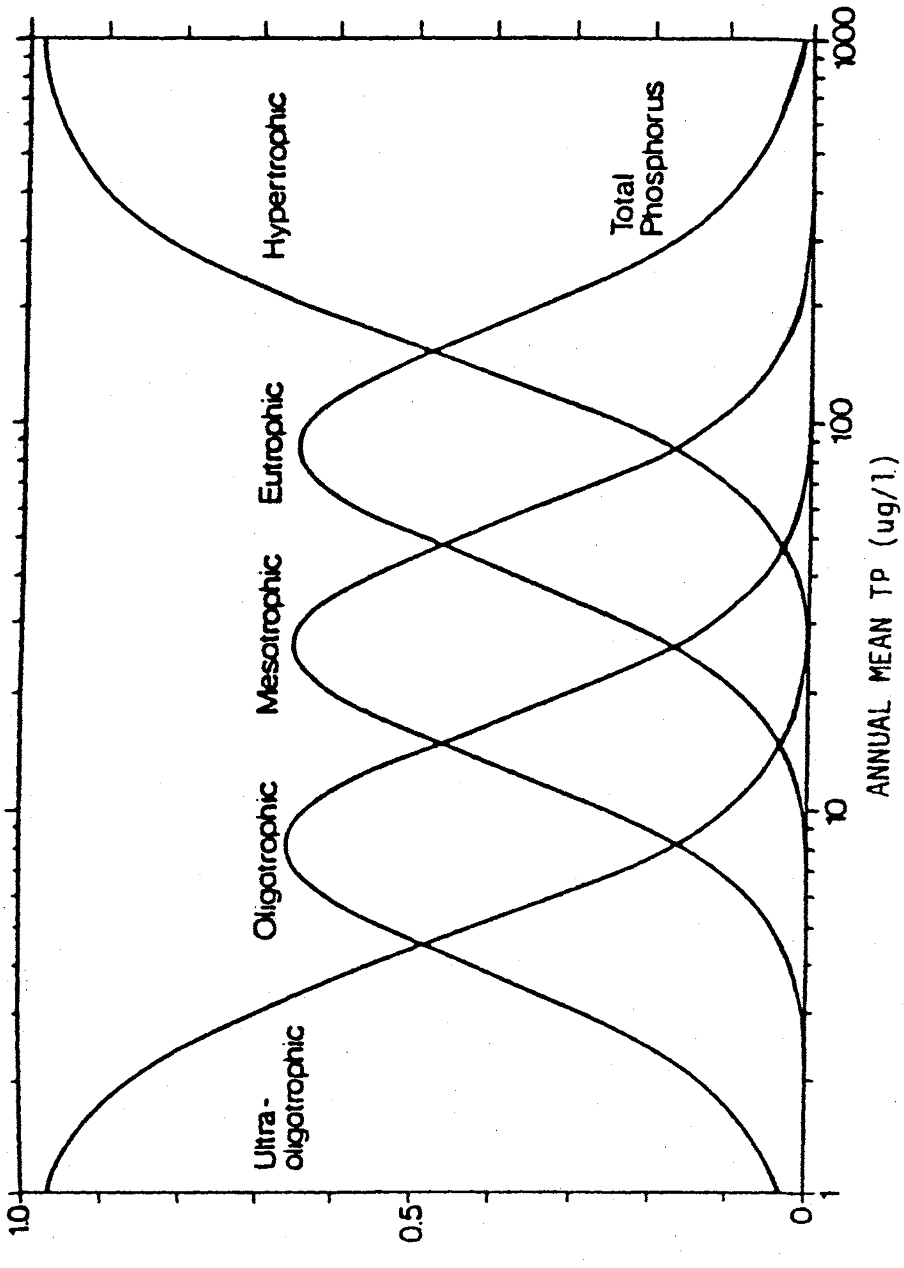


Figure 6.4. Probability distributions of trophic categories relative to average annual total phosphorus concentrations (OECD, 1982).

Table 6.9. Summary of proposed relationships among phosphorus concentration, trophic state and lake use for north temperate lakes.

In-Lake Total P (ug P/L)			
Mancini *et al.* (1983)[1]	OECD (1982)	Trophic State	Lake Use
<10	<10	Oligotrophic	Suitable for water-based recreation and propagation of cold water fisheries, such as trout. Very high clarity and aesthetically pleasing.
10-20	10-35	Mesotrophic	Suitable for water-based recreation but often not for cold water fisheries. Clarity less than oligotrophic lake.
20-50	35-100	Eutrophic	Reduction in aesthetic properties diminishes enjoyment from body contact recreation. Generally very productive for warm water fisheries.
>50	>100	Hypereutrophic	A typical "old-aged" lake in advanced succession. Some fisheries, but high levels of sedimentation and algae or macrophyte growth may be diminishing open water surface area.

1) Hypereutrophic category suggested by Reckhow and Chapra (1983).

Table 6.10. Summary of observed and predicted trophic state indicators for Lake Sawyer before and after WTP diversion.

INDICES	Study Year 1989-90	Typical-year Post-WTP-Diversion	Statistically Significant Change? (1)	Trophic State Criteria (2)		
			Prob >=90%	Oligotrophic	Mesotrophic	Eutrophic
ANNUAL WHOLE-LAKE						
Total P (ug P/L)	25.7 ± 3.1	16.4 ± 4.1	YES	(=< 10)	(10 - 35)	(>= 35)
Total N (ug N/L)	416 ± 20	384 ± 42	no	661	753	1875
Total N:P Ratio (wt:wt)	16.2 ± 2.1	23.4 ± 6.4	no	--	--	--
MARCH - OCTOBER EPILIMNION						
Total P (ug P/L)	17.7 ± 1.2	9.8 ± 1.8	YES *	--	--	--
Total N (ug N/L)	366 ± 14	333 ± 35	no	--	--	--
Total N:P Ratio (wt:wt)	20.7 ± 1.6	34.0 ± 7.2	YES	--	--	--
Average Chlorophyll a (ug/L)	7.3 ± 1.0	3.2 ± 1.7	YES	(=< 2.5)	(2.5 - 8)	(>= 8)
Peak Monthly Chlorophyll a (ug/L)	15.8 ± 3.6	8.3 ± 3.1	no	(=< 8)	(8 - 25)	(>= 25)
Average Secchi Disk Transparency (meters)	3.5 ± 0.3	4.8 ± 0.3	YES *	(>= 6)	(6 - 3)	(=< 3)
Minimum Monthly Secchi Disk (meters)	2.2 ± 0.4	3.1 ± 0.6	no	(>= 3)	(3 - 1.5)	(=< 1.5)

1) Statistical significance based on t-test. Asterisk (*) denotes significance at 95% probability level, otherwise 90% probability was used.

2) Criteria from OECD (1982). The single value for each trohic category represents the maximum likelihood from an open-boundary classification. The ranges given in parentheses are fixed boundary values for trophic category ranges.

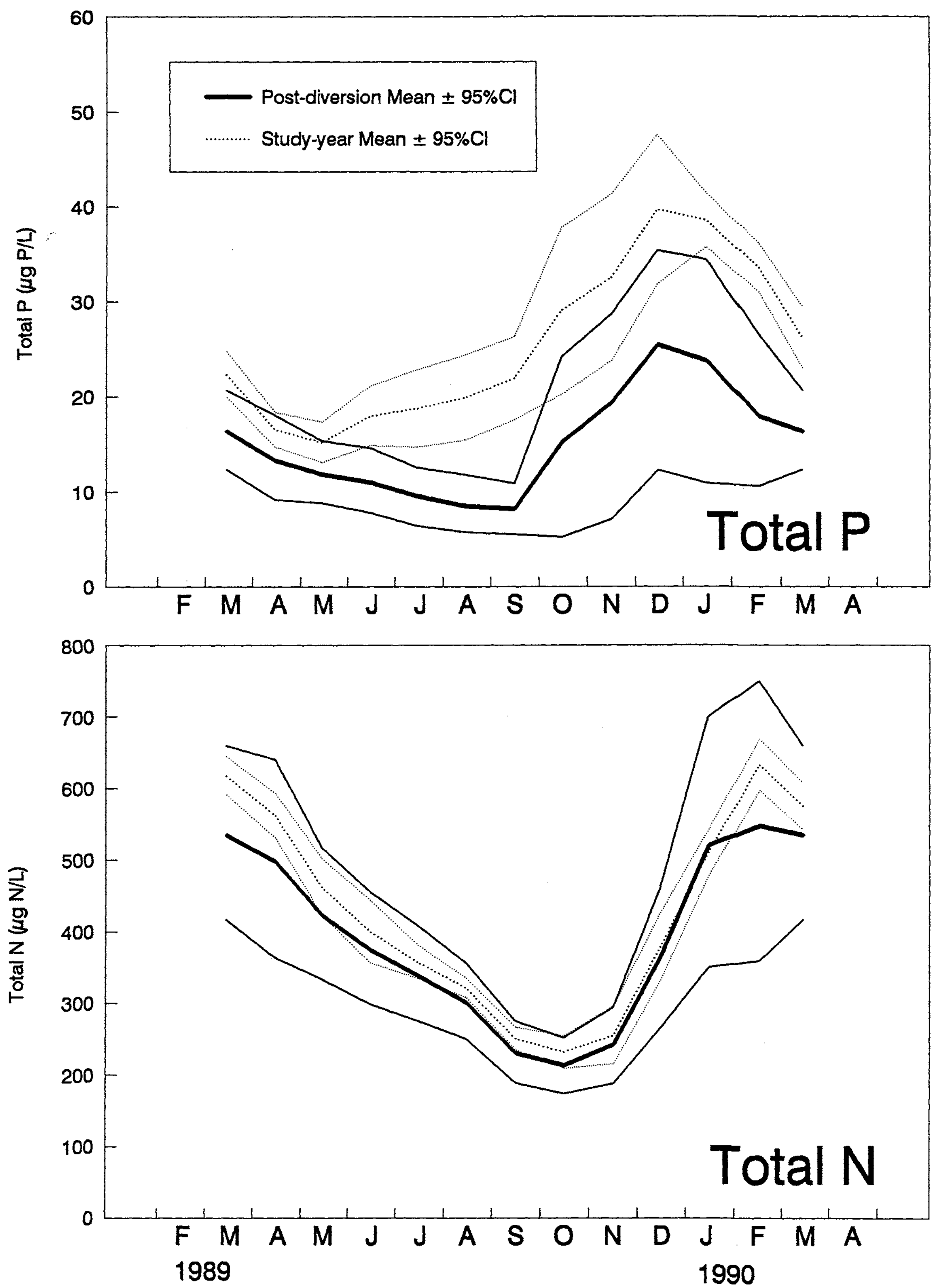


Figure 6.5. Results of Monte-Carlo simulation of "typical-year" post-diversion and study-year whole-lake total P and total N.

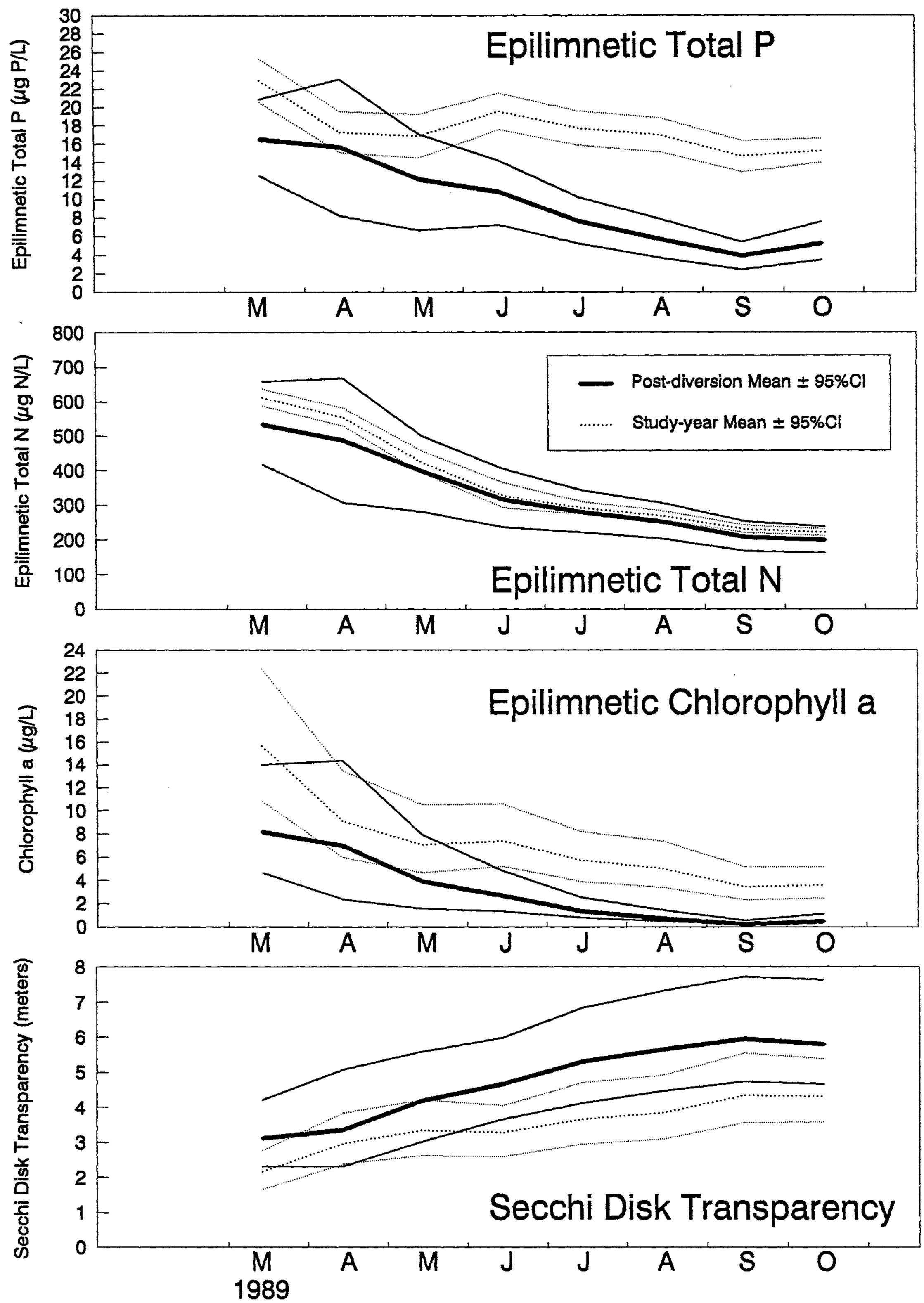


Figure 6.6. Results of Monte-Carlo simulation of typical-year post-diversion epilimnetic water quality compared with study year (1989).

condition represents an unacceptable water quality condition for Lake Sawyer, perhaps a more stringent in-lake TP criterion representing a lower risk of eutrophic conditions warrants adoption.

Ecology interpreted protection of trophic status for Lake Chelan in the context of uncertainty by stating that "acceptable" enrichment should not result in greater than a five percent chance of degradation to a higher trophic state (Patmont *et al.*, 1989). The selection of five percent as a cutoff is somewhat arbitrary, but is intended to represent a very small chance of water quality deterioration. If this guideline is applied to mesotrophic Lake Sawyer, then the goal for protection of a mesotrophic condition would be stated as achieving less than a five percent chance of a eutrophic condition. However, if the management goal is interpreted this way for Lake Sawyer, there should be a recognition that the present condition of the lake as well as the condition of the lake prior to installation of the WTP probably did not meet the goal.

Table 6.11 summarizes various scenarios of TP loading, annual whole-lake TP, and the probability of classification of either mesotrophic (or better) or eutrophic (or worse). In addition to the TP loading scenarios discussed above, Table 6.11 also includes projections for hypothetical future conditions if the planned WTP diversion does not occur. These hypothetical scenarios were based on typical-year loads presented in Table 6.8 plus WTP loads proportionally scaled from the 1989-90 study year using population projections for the service area (Brown and Caldwell, 1988). Concentrations of TP were calculated from the steady-state mass balance model (equation 6.14). Trophic classification probabilities were based on the OECD (1982) classification scheme (Figure 6.4).

Table 6.11 shows that prior to WTP installation there was a 9% risk of eutrophic conditions based the OECD scheme. The existing condition of the lake with an in-lake TP concentration of approximately 25 μg/L shows a degradation from pre-WTP status with approximately an 18% chance of eutrophic conditions. Following WTP diversion, the lake is expected to have approximately a 5% chance of eutrophic conditions, which is an improvement over pre-1982 conditions, but is near the threshold of the management goal discussed above. A 5% chance of eutrophic conditions represents an upper-limit in-lake TP criterion of about 16 μg/L.

For comparison, the hypothetical future scenarios if the planned diversion of the WTP did not occur show a substantially increased probability of eutrophic conditions, with up to a nearly seven-fold increased risk compared with pre-1982 and a predominance of undesirable conditions. This finding supports the determination of Pelletier and Joy (1989) which demonstrated that the originally permitted Black Diamond WTP load to Lake Sawyer was likely to cause a shift from mesotrophic to eutrophic designation.

Lake Sawyer is expected to recover to an acceptable mesotrophic condition if wastewater diversion is implemented as planned. However, as stated above, the equilibrium condition of the lake following diversion is likely to include about a 5% chance of eutrophic conditions, which may represent the upper limit of acceptable risk. Therefore, increases in watershed development may return Lake Sawyer to a state of increased likelihood of eutrophic conditions. consequently, any future development within the basin should incorporate management practices

Table 6.11. Summary of TP loading, annual whole-lake TP, and trophic status for various future loading scenarios.

Loading Scenario	External and Internal Total P Load (Kg P/year)	Annual Whole-lake Total P (ug P/L)	Trophic Status Probability: Mesotrophic or better	Trophic Status Probability: Eutrophic or worse
Pre-1982 Condition (before WTP start-up)	840 ± 136	19.2 ± 4.6	91%	9%
Ecology Study-Year (1989-1990)	1,117 ± 129	25.7 ± 3.1	82%	18%
Typical-year Following WTP diversion	715 ± 128	16.4 ± 4.1	95%	5%
Year 2010 if WTP is not diverted	1,576 ± 310	36.1 ± 9.5	65%	35%
Service Area Population Saturation if WTP is not diverted	2,533 ± 131	58.0 ± 10.6	37%	63%

that minimize nonpoint phosphorus inputs to the lake. Additionally, a more stringent in-lake TP criterion representing a lowered acceptable risk of eutrophic conditions should be adopted if the present conditon of Lake Sawyer represents an unacceptable condition.

7.0 CONCLUSIONS AND RECOMMENDATIONS

- The flushing rate of Lake Sawyer during the study year was relatively rapid at approximately 2.8 times per year or once every 4.2 months on the average.

- Volume-weighted whole-lake TP concentration for the study year (March through February) was 25.7 ± 3.1 μg P/L. Volume-weighted epilimnetic chlorophyll *a* concentration during the 1989 growing season (March through October) was 6.2 ± 1.5 μg/L.

- Aquatic macrophytes are and will continue to be, a nuisance in the shallower areas of Lake Sawyer and potentially could worsen in response to decreased algae and increased light penetration within the lake.

- The color of lake water due to naturally occurring dissolved humic substances results in a maximum possible Secchi disk transparency of about 6.2 meters.

- Phosphorus sedimentation appears to be higher since the start-up of the WTP in 1982 than at any time within the last 150 years. Nitrogen and organic carbon sedimentation appears to have increased since the WTP start-up; However, past sedimentation around the late 1800s and early 1900s may have been as high.

- Sediment diatom remains seem to indicate periods of probable ecological change in Lake Sawyer. One such period occurred in the late 1930s or early 1940s, corresponding to a time when the majority of Ravensdale Creek and part of Rock Creek watersheds were clearcut of timber. Recent diatom remains suggest a trend towards more eutrophic conditions within Lake Sawyer.

- Hypolimnetic dissolved oxygen depletion occurs during stratification due to the oxidation of organic and inorganic matter in the water column and sediment oxygen demand. The present rate of hypolimnetic oxygen depletion is about the same as that observed prior to WTP installation in 1982. Therefore, hypolimnetic anoxia, and internal cycling of P and N, is not expected to change because of WTP diversion.

- Algal biomass is strongly correlated to both nitrogen and phosphorus. However, phosphorus is generally more limiting than nitrogen at the present time. Phosphorus is expected to be even more important as a controlling nutrient after WTP diversion.

- Internal loads of total P and total N account for less than 20% of the total nutrient budgets. Internal recycling of nutrients probably is dominated by sediment feedback in the hypolimnion during anoxia.

- The removal of total P and total N in raw wastewater from the combined effects of the WTP lagoons and natural wetland system averaged 77 ± 6% and 73 ± 9%, respectively.

- The Black Diamond WTP presently contributes about half of the external total P load and 12% of the external total N load to Lake Sawyer.

- Septic systems treating wastewater from nearshore homes are estimated to contribute only about 1% of the total phosphorus and nitrogen loads to the lake.

- During the 1989-90 study year, the sources of total phosphorus loading to Lake Sawyer consisted of: Blank Diamond WTP (39%); Rock Creek Nonpoint (25%); Ravensdale Creek (11%); Internal Load (18%); Precipitation (6%); and Nearshore Septic Systems (1%).

- Rock Creek has experienced development within its watershed, responding with increased stormwater flow and deteriorating background water quality. Future development in the basin will continue to impair the water quality in this tributary unless significant efforts are made to control the quality of runoff from developed areas.

- Ravensdale Creek supplies the majority (57%) of hydrologic input to Lake Sawyer and is relatively pristine. The water quality impact on Lake Sawyer, should Ravensdale Creek deteriorate (e.g. to the point that Rock Creek's background water quality has), would be detrimental, counteracting present efforts and expenditures to preserve Lake Sawyer (i.e., WTP diversion).

- The equilibrium condition of the lake after WTP diversion is expected to be somewhat better than before the WTP installation because the total P load to Lake Sawyer from Rock Creek prior to WTP installation was probably lower than the present load, but higher than the load expected following WTP diversion. The difference between loads before WTP installation and following WTP diversion may be explained by total P contributions from failing treatment systems in use before 1982.

- The present trophic state of Lake Sawyer is predominantly mesotrophic with approximately an 18% expectation of eutrophic conditions. The diversion of Black Diamond WTP effluent from Rock Creek and Lake Sawyer will result in decreased likelihood of eutrophic conditions within the lake. Without diversion, the lake would have experienced a significant increase in the likelihood of eutrophic characteristics and a predominance of undesirable conditions.

- The lake is expected to respond fairly rapidly to changes in nutrient loading following diversion because of relatively rapid flushing and sedimentation rates. The lake is predicted to attain about 99% of the new steady state concentration within the first year after a change in the loading rate.

- The equilibrium condition of the lake after WTP diversion is expected to include about a 5% chance of eutrophic conditions because of loading sources other than the WTP. This may represent the upper limit of acceptable risk. Any increases in nonpoint loading which may occur in the future with increased watershed development may result in an unacceptable likelihood of eutrophic characteristics.

- Future development within the basin should incorporate management practices that minimize nonpoint phosphorus inputs to the lake. If possible, total P loading should be maintained at a level no higher than the total external and internal P load following WTP diversion (715 $\pm$ 128 Kg P/year) or a steady state in-lake total P concentration of about 16 μg P/L.

This page is purposely left blank for duplex printing.

8.0 REFERENCES

APHA, AWWA, WPCF. Standard Methods for the Examination of Water and Wastewater, 16th Edition, 1985.

Appleby, P.G. and F. Oldfield. The Calculation of Pb-210 Dates Assuming a Constant Rate of Supply of Unsupported Pb-210 to the Sediment. CATENA, Vol. 5., 1978, pp. 1-8.

Aroner, E. 1990. WQHYDRO - Environmental statistical data analysis graphics package. Washington State Department of Ecology, Sediment Management Unit, Olympia, WA.

Austin, E.R. and G.F. Lee. Nitrogen Release from Lake Sediments. Journal WPCF. Vol.45, No.5, May 1973, pp. 870-879.

Barnes, R.S., S. Lazoff, and D.E. Spyridakis. "Phosphorus Loading to Lake Meridian." In: A study of the trophic status and recommendations for the management of Lake Meridian. Municipality of Metropolitan Seattle, 1978. Seattle.

Beale, E.M.L. Some Uses of Computers in Operational Research. Industrielle Organisation, 1962. 31:51-52.

Birch, P.B. 1976. "The Relationship of Sedimentation and Nutrient Cycling to the Trophic Status at Four Lakes in the Lake Washington Drainage Basin." Ph.D thesis, University of Washington, 1976. Seattle.

Birch, P.B., R.S. Barnes, and D.E. Spyridakis. "Recent Sedimentation and its Relationship with Primary Productivity in Four Western Washington Lakes." Limnology and Oceanography, Vol. 25(2), 1980. pp. 240-247.

Bradbury, J.P., and R.O. Megard. "Stratigraphic Record of Pollution in Shagawa Lake, Northeastern Minnesota." Geo. Soc. Amer., Special Paper 171, 1972.

------ and J.C.B. Waddington. "The Impact of European Settlement in Shagawa Lake, Northeastern Minnesota, USA." In: Quaternary Plant Ecology (Ed. by H.J.B. Birks and R.G. West), 1973, pp. 238-307. Blackwells, Oxford.

Brenner, B. Personal Communication. Unpublished Data of Municipality of Metropolitan Seattle, 1989.

Brenner, R.N. and J.I. Davis. Status of Water Quality in Small Lakes, Seattle-King County Region: 1985 Survey. Municipality of Metropolitan Seattle, 1986.

------ and J.I. Davis. Status of Water Quality in Small Lakes, Seattle-King County Region: 1987 Survey. Municipality of Metropolitan Seattle, 1988.

Brezonik, P.L. Trophic Classifications and Trophic State Indices: Rationale, Progress, Prospects. Report No. ENV-07-76-01. Department of Engineering Sciences. University of Florida, 1976.

Brown and Caldwell. Draft Facility Plan for Wastewater Treatment System. City of Black Diamond, Washington, 1988.

Brugam, R.B. "A Re-evaluation of the Araphidineae/ Centrales Index as an Indicator of Lake Trophic Status." Freshwater Biology, Vol. 9, 1979, pp. 451-460.

Carpenter, S.R. "Submersed Macrophyte Community Structure and Internal Loading: Relationship to Lake Ecosystem Productivity and Succession." In: Lake Restoration, Protection, and Management, EPA 440/5-83-011, 1983.

Carpenter, R., J.T. Bennett, and M.L. Peterson. "Pb-210 Activities in and Fluxes to Sediments of the Washington Continental Slope and Shelf." Geochim. Cosmochim. Acta. Vol. 45, 1981, pp. 1155-1172.

------ and M.L. Peterson, and J.T. Bennett. "Pb-210 Derived Sediment Accumulation and Mixing Rates for the Washington Continental Slope." Mar. Geol., 48: 135-164, 1982.

Chapra, S.C. and K.H. Reckhow. Engineering Approaches for Lake Management. Vol.2: Mechanistic Modelling. Butterworth Publishers; Boston, MA, 1983, 492 pp.

Chiandani, G., and M. Vighi. The N:P Ratio and Tests with Selenastrum to Predict Eutrophication in Lakes. Water Res. Vol. 8, 1974, pp. 1063-1069.

Cole, G.A. Textbook of Limnology. The C.V. Mosby Company, St. Louis, MO, 1979, 426 pp.

Cooke, G.D., McComas, Waller, and Kennedy. The occurrence of internal phosphorus loading in two small, eutrophic, glacial lakes in northeastern Ohio. Hydrobiologia. 56(2), 129-135, 1977.

Cooke, G.D., E.B. Welch, S.A. Peterson and P.R. Newroth. Lake and Resevoir Restoration. Butterworth Publishers, Stoneham, MA, 392 pp., 1986.

Cornell, C.L. First order analysis of model and parameter uncertainty. In: International Symposium on Uncertainties in Hydrologic and Water Resources Systems. pp. 1245-1274, 1973.

Davis, M.B. Redeposition of pollen grains in lake sediment. Limnol. Oceanogr. 18:44-52, 1973.

D'Elia, C.F., P.A. Steudler, and N. Corwin. Determination of total nitrogen in aqueous samples using persulfate digestion. Limnology and Oceanography, 22:760-764, 1977.

Devitt, R. Upper Rock Creek, Black Diamond. Washington Department of Ecology Memorandum. Technical Services Program, 1972.

Dexter et al. A summary of knowledge of Puget Sound related to chemical contaminants. NOAA Technical Memorandum OMPA-13, Boulder, CO, December 1981, 435 pp.

Dobson, H.F., M. Gilbertson, and P.G. Sly. A summary and comparison of nutrients and related water quality in lakes Erie, Ontario, Huron, and Superior. Journal of Fishery Research Board of Canada. 31:731-738, 1974.

Dolan, D.M., A.K. Yui, and R.D. Geist. Evaluation of river load estimation methods for total phosphorus. J. Great Lakes Res., 7(3): pp. 207-214, 1981.

Dunne, T. and L.B. Leopold. Water in Environmental Planning. W.H. Freeman and Co., San Fransisco, CA., 818 pp., 1978.

Edmondson, W.T. and A.H. Litt. *Daphnia* in Lake Washington. Limnol. Oceanog., 27(2), 1982, pp. 272-293.

EPA. The relationships of phosphorus and nitrogen to the trophic state of northeast and north-central lakes and reservoirs. National Eutrophication Survey Working Paper No. 23, 1974.

EPA. Methods for chemical analysis of water and wastes. EPA-600/4-79-020, 1983.

EPA. Rates, constants, and kinetics formulations in surface water quality modeling (Second Edition). EPA/600/3-85/040, 1985.

EPA. Quality Criteria for Water. EPA 440/5-86-001, 1986.

EPA. Test methods for evaluating solid wastes. Volume 1C. Laboratory manual for physical/chemical methods. U.S. EPA, Office of Solid Waste and Emergency Response, 1986.

EPA. Advanced waste treatment review for City of Black Diamond. U.S. EPA, Region 10, July 1989.

Environmental Resources Management. Black Diamond, Washington, Wastewater Treatment System. EPA Contract No. 68-01-7108., 1986.

Forsberg, C. Present knowledge on limiting nutrients. In: Restoration of Lakes and Inland Waters. EPA 440/5-81-010. U.S. Environmental Protection Agency; Washington, D.C. p.37, 1980.

Galat, D.L. Estimating fluvial mass transport to lakes and resevoirs: avoiding spurious self-correlations. Lake and Resevoir Management, 1990 6(2):153-163. North American Lake Management Society, 1990.

Geldrich, E.E. and B.A. Kenner. Concepts in fecal streptococci in stream pollution. Journal Water Pollution Control Federation, 41(8):R336-352, 1969.

Gibbons, M.V. Pend Oreille River Eurasian Watermilfoil Control Program, 1986. Prepared for Pend Oreille County, 1986.

Goldberg, E.D. Geochronology with Pb-210. In: Radioactive Dating, IAEA-STI/PUB/86, Vienna, Austria. pp. 121-131, 1963.

Harbeck, G.E. A Practical Field Technique for Measuring Resevoir Evaporation Utilizing Mass-Transfer Theory, U.S. Geol. Sur. Prof. Pap. 272-E, pp. 101-105, 1962.

Harper-Owes. Pine Lake Restoration Analysis, Final Report. Prepared for Municipality of Metropolitan Seattle, Washington, 1981.

Hart Crowser. Lake Sawyer Hydrogeologic Study - Black Diamond, Washington. Prepared for Washington State Department of Ecology. October 5, 1990.

Healey, F.P. and L.L. Hendzel. Physiological indicators of nutrient deficiency in lake phytoplankton. Can. J. Fish. Aquat. Sci. Vol. 37. pp 442-453, 1980.

Hedges, J.I. and J.H. Stern. Carbon and nitrogen determinations of carbonate-containing solids. Limnol. Oceanogr., 29(3), 1984, 657-663, 1984.

Hilton, J., J.P. Lishman, and P.V. Allen. The dominant processes of sediment distribution and focusing in a small, eutrophic, monomictic lake. Limnol. Oceanaogr. Vol. 31(1). pp. 125-133, 1986.

Hitchcock, C.L. and A. Cronquist. Flora of the Pacific Northwest. University of Washington Press, Seattle, WA, 730 pp., 1973.

Huntamer, D. Department of Ecology Laboratory Users Manual. Washington State Department of Ecology, 1986.

Hutchinson, G.E. A Treatise on Limnology, Chemistry of Lakes. Vol 1, Part 2. Wiley and Sons Publishers, 1975.

Infante, A. and W.T. Edmondson. Edible phytoplankton and herbivorous zooplankton in Lake Washington. Arch. Hydrobiol. Beih. Vol.21, pp.161-171, 1985.

Jacoby, J.M., D.D. Lynch, E.B. Welch, and M.A. Perkins. Internal phosphorus loading in a shallow eutrophic lake. Water Research, 16:911-919, 1982.

Johnson, A.H. Estimating solute transport in streams from grab samples. Water Resources Research. Vol.15, no.5, pp. 1224-1228, 1979.

Kamiyama, K., S. Okuda and A. Kawai. Studies on the release of ammonium nitrogen from the bottom sediments in freshwater regions. II. Ammonium nitrogen in dissolved and absorbed form in the sediments. Japan. J. Limnol. 38:100-106, 1977.

KCM. Black Diamond Wastewater Facility Planning Study and Environmental Assessment. Report prepared for King County Department of Public Works, 1979.

KCM. Unpublished data from Lake Sawyer tributaries. Kramer, Chin, and Mayo. Washington Department of Ecology Municipal Grants Files, 1982.

Krishnaswamy, S. et al. Geochronolgy of Lake Sediments. Earth and Planetary Science Letters. Vol. 11. pp. 407, 1971.

Lehman, J.T. Reconstructing the rate of accumulation of lake sediment: the effect of sediment focusing. Quaternary Research. Vol 5. pp. 541-550, 1975.

Lettenmaier, D.P. and J.E. Richey. Use of first order analysis in estimating mass balance errors and planning sampling activities. In: E. Halfon ed., Theoretical Systems Ecology, pp. 79-104. Academic Press, 1979.

Linsley, R.K. et al. Hydrology for Engineers. McGraw-Hill, New York, N.Y., 482 pp., 1975.

Lorenzen, M.W. Use of chlorophyll-Secchi disk relationships. Limnol. Oeceanogr. 25(2), 371-372, 1980.

Mancini, J.L, G.G. Kaufman, P.A. Mangarella, and E.D. Driscoll. Technical Guidance Manual for Performing Waste Load Allocations. Book IV Lakes and Impoundments. Chapter 2 Eutrophication. Final report for U.S. EPA Office of Water Regulations and Standards, Contract No. 68-01-5918, 1983.

Mason, W.T. An Introduction to the Identification of Chironomid Larvae. Division of Pollution Surveillance, Federal Water Pollution Control Administration, U.S. Department of the Interior, Cincinnati, Ohio, 89 pp., 1968.

McConnell, J.B., G.C. Bortleson, and J.K. Innes. Data on selected lakes in Washington. Part 4. Washington State Department of Ecology Water Supply Bulletin 42, Part 4, 1976.

Merritt, R.W. and K.W. Cummins. An Introduction to the Aquatic Insects of North America. Kendall/Hunt Publishing Company, Dubuque, Iowa, 722 pp., 1984.

Mitsch, W.J., and J.G. Gosselink. Wetlands. Van Nostrand Reinhold Company, 1986.

Municipality of Metropolitan Seattle. A Baseline Survey of Aquatic Plants in Selected Lakes of King County, 1976.

Municipality of Metropolitan Seattle. A study of composition, growth and distribution of aquatic macrophytes in fourteen lakes and bays of King County, 1977.

Municipality of Metropolitan Seattle. Distribution and Community composition of aquatic macrophytes in selected waters of King County, 1978.

Municipality of Metropolitan Seattle. Aquatic Plants in selected waters of King County, 1979.

Municipality of Metropolitan Seattle. Aquatic Plants in selected waters of King County, 1980.

Nevissi, A.E. Measurement of Pb-210 atmospheric flux in the pacific northwest. Health Physics. Vol. 48 No. 2 (February) pp. 169-174, 1985.

National Climatic Center. Monthly normals of temperature, precipitation, and heating and cooling degree days 1951-80. Washington. NOAA/EDIS/National Climatic Center, Asheville, N.C., September 1982.

Nichols, D.S. Capacity of natural wetlands to remove nutrients from wastewater. Journal of the Water Pollution Control Federation, 55(5):495-505, 1983.

OECD (Organization for Economic Cooperation and Development). Eutrophication of waters: Monitoring, assessment and control. Final report. OECD Cooperative Programme on Monitoring of Inland Waters (Eutrophication Control), Environment Directorate, OECD, Paris, 154 pp., 1982.

Patmont, C.R., G.J. Pelletier, E.B. Welch, D. Banton, and C.C. Ebbesmeyer. Lake Chelan Water Quality Assessment. Final Report to Washington State Department of Ecology, Contract No. C0087072, 1989.

Pechlaner, R. The phytoplankton spring outburst and its conditions in Lake Erken (Sweden). Limnol. Oceanogr. 15:113-130, 1970.

Pederson, G.L. Plankton secondary production and biomass; seasonality and relation to trophic state in three lakes. PhD thesis, Civil Engineering, University of Washington, Seattle, 1974.

Pelletier, G.J. and Joy, J.W. Lake Sawyer - Black Diamond Waste Load Allocation Evaluation, Washington State Department of Ecology, 1989.

Pennak, R.W. Freshwater Invertebrates of the United States. John Wiley & Sons, Inc., New York, NY, 803 pp., 1978.

Pennington, W., R.S. Cambray, J.D. Eakins, and D.D. Harkeness. Radionuclide dating of the recent sediments of Blelham Tarn. Freshwater Biology. Vol. 6. pp. 317-331, 1976.

Persaud, D., R. Jaagumagi and A. Hayton. Provincial Sediment Quality Guidelines (A Discussion Paper on Their Development and Application), Water Resources Branch, Ontario Ministry of the Environment, Toronto, Ontario, 1990.

Prescott, G.W. How to know the aquatic plants. The pictured key nature series. Wm. C. Brown Company Publishers, Dubuque, Iowa. 158 pp., 1980.

Preston, S.D., V.J. Bierman Jr., and S.E. Silliman. Evaluation of Methods for the Estimation of Tributary Mass Loading Rates. Technical Report Number 187. Water Resources Research Center, Purdue University, IN, 50 pp., 1989.

Reckhow, K.H., and S.C. Chapra. Engineering approaches for lake management. Volume 1: Data analysis and empirical modeling. Butterworth Publishers, 1983.

Robbins, J.A., and Edgington, D.E. Determination of recent sedimentation rates in Lake Michigan using Pb-210 and Cs-137. Geochim. Cosmochim. Acta. Vol. 39. pp. 285., 1975.

R. W. Beck and Associates. Marshland Wastewater Treatment Evaluation for the City of Black Diamond. Report prepared for Black Diamond, Washington, 1985.

Sakamoto, M. Primary production by the phytoplankton community in some Japanese lakes and its dependence on lake depth. Arch. Hydrobiol. Vol. 62, pp. 1-28, 1966.

Schindler, D.W. Factors regulating phytoplankton production and standing crop in the world's freshwaters. Limnol. Oceanogr. Vol. 23(3). pp. 478-486, 1978.

Shapiro, J., W.T. Edmondson, and D.E. Allison. Changes in the chemical composition of sediments of Lake Washington, 1958-1970. Limnol. Oceanogr. Vol. 16(2). pp. 437-452, 1971.

Shuster, J.I., E.B. Welch, R.R. Horner, D.E. Spyridakis. Response of Lake Sammamish to urban runoff control. In: Lake and reservoir management. Volume II. Proceedings of the Fifth Annual Conference and International Symposium on Applied Lake & Watershed Management. North American Lake Management Society, 1986.

Smith, V.H. Nutrient dependence of primary productivity in lakes. Limnol. Oceanogr. Vol. 24(6). pp. 1051-1064, 1979.

Smith, V.H. The nitrogen and phosphorus dependence of algal biomass in lakes: An empirical and theoretical approach. Limnol. Oceanogr. 27(6), 1982, 1101-1112, 1982.

Solarzano, L. The determination of ammonia in natural water by the phenolhypochlorite method. Limnol. Oceanogr. 14(5): pp. 799-801, 1969.

Spyridakis, D.E. and R.S. Barnes. Contemporary and historical trace metal loadings to the sediments of four lakes in the Lake Washington drainage. OWRT project report. Proj. A-083-Wash, 1977.

Stockner, J.G. Preliminary characterization of lakes in the Experimental Lakes Area, northwestern Ontario, using diatom occurrence in sediments. J. Fish. Res. Bd. Can. Vol. 28, pp. 265-275, 1971.

Stockner, J.G. Paleolimnology as a means of assessing eutrophication. Vern. Internat. Verein. Limnol. Vol. 18, pp. 1018-1030, 1972.

Sweet, J.W. A survey and ecological analysis of Oregon and Idaho phytoplankton. Final Report submitted to Environmental Protection Agency, Region 10, Seattle, WA, 1986.

Thielen, J. Effects of Black Diamond on Ginder Creek. Washington Department of Ecology Memorandum. Technical Services Program, 1978.

Thomann, R.V. and J.A. Mueller. Principles of Surface Water Quality Modeling and Control. Harper and Roe, Publishers, Inc., New York, NY, 644 pp., 1987.

U. S. Environmental Protection Agency. Methods for chemical analysis of water and wastes. Environmental Monitoring Support Laboratory, Cincinnati, Ohio, EPA-600/4-79-020, 1983.

U. S. Geological Survey. Water Resources Data for Washington, Water Year 1975. USGS Report WA-75-1, 1975.

Vallentyne, J.R. The algal bowl. Misc. Spec. Publ. 22. 186 p. Dep. Environ., Fish Mar. Serv., Ottawa, 1974.

Von Damm, K.L., L.K. Benninger, and K.K. Turekian. The Pb-210 chronology of a core from Mirror Lake, New Hampshire. Limnol. Oceanogr. Vol. 24(3). pp. 434-439, 1979.

Walker, M. G. and F.M. Veach, 1964. Miscellaneous stream-flow measurements in the State of Washington from 1890 to January 1961. U.S. Geological Survey, Water-Supply Bulletin No. 23.

Washington Dept. of Ecology. National Pollutant Discharge Elimination System, Waste Discharge Permit, Town of Black Diamond. Permit No. WA-002996-3, 1980.

Welch, E.B. Nutrient diversion: resulting lake trophic state and phosphorus dynamics. Ecological Research Series, EPA-600/3-88-003. 91 pp., 1977.

Welch, E.B. and M.A. Perkins. Oxygen deficit-phosphorus loading relation in lakes. J. Water Poll. Cont. Fed. 51:2823-2828, 1979.

Welch, E.B., M.A. Perkins, D. Lynch, and P Hufschmidt. Internal phosphorus related to rooted macrophytes in a shallow lake. Proc. of a Workshop/Conference on Efficacy and Impact of Intensive Plant Harvesting in Lake Management. Madison, WI, February 14-16, 1979.

Welch, E.B. and T. Lindell. Ecological Effects of Wastewater. Cambridge University Press, Cambridge, U.K., 337 pp., 1980.

Welch, E.B. and T. Smayda. Nutrient loading and trophic state of Scriber Lake. Water Resources Series Technical Report No. 100. University of Washington. Department of Civil Engineering. Environmental Engineering and Science. Seattle, Washington, 1986.

Welch, E.B., et al, 1981. The trophic state and phosphorus budget of Pine Lake, Washington. Final Report to Municipality of Metropolitan Seattle.

Welch, P.S. Limnological Methods. Blakiston Publishers, 1948.

Wetzel, R.G. Limnology. W.B. Saunders Co.; Philidelphia, PA. 753 pp., 1983.

Wetzel, R.G., and B.A. Manny. Secretion of dissolved organic carbon and nitrogen by aquatic macrophytes. Verh. Int. Ver. Limnol. 18:162-170, 1972.

Williams and Pearson. Streamflow statistics and drainage basin characteristics for the Puget Sound Region, Washington. U.S. Geological Survey, Open File Report 84-144-A, 1985.

Wolcott, E. E. Lakes of Washington. Washington Department of Conservation, Water Supply Bulletin No. 14, 1965.

Wydoski, R.S. and R.R. Whitney. Inland Fishes of Washington. University of Washington Press, Seattle, WA. 220 pp., 1979.

Zar, J.H. Biostatistical Analysis. Prentice-Hall Publishers, 1974.