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Effects of Artificial Destratification on Water Quality at East Sidney Lake, New York

by Steven L. Ashby, Robert H. Kennedy
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Prepared for Headquarters, U.S. Army Corps of Engineers
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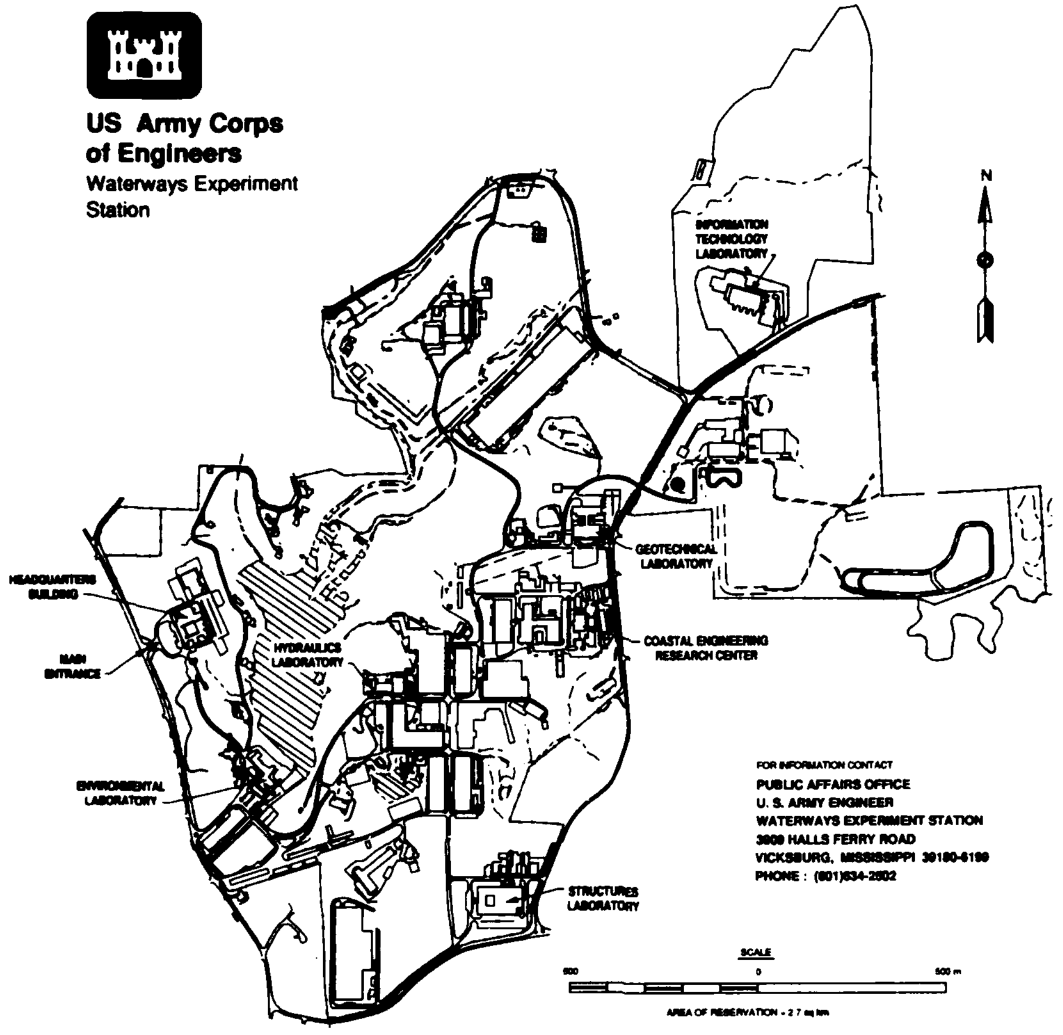

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Contents

Preface

The study described in this report was sponsored by the U.S. Army Engineer District, Baltimore (NAB), and was conducted by the U.S. Army Engineer Waterways Experiment Station (WES) under Intra-Army Order No. E86888033, dated 4 April 1988. Mr. Pete Juhle, former Chief of the Water Control Management Section, NAB, and Mr. Richard Olin, current Chief, served as the point of contact.

Additional support was provided as part of the Water Quality Management for Reservoirs and Tailwaters Demonstration of the Water Operations Technical Support (WOTS) Program. The WOTS Program is sponsored by the Headquarters, U.S. Army Corps of Engineers (HQUSACE), and is assigned to the WES under the purview of the Environmental Laboratory (EL). Funding was provided under Department of the Army Appropriation No. 96X3123, Operations and Maintenance. The WOTS Program is managed under the Environmental Resources Research and Assistance Programs (ERRAP), Mr. J. L. Decell, Manager. Ms. Carolyn B. Schneider was Assistant Manager, ERRAP, for the WOTS Program. Technical Monitors during this study were Messrs. Pete Juhle and Rixie Hardy, HQUSACE.

This report was prepared by Mr. Steven L. Ashby and Dr. Robert H. Kennedy of the EL. Participating in the conduct of the study were Messrs. Joe H. Carroll, William E. Jabour, and Dr. John J. Hains of the EL and Mr. Ed Robertson and Ms. Cynthia Huffstetler of AScI Corporation. Field sampling and chlorophyll analyses were conducted by Messrs. Jim Mitchell, Gary Harmon, Dennis Smith, and Richard Feldman, State University of New York (SUNY), Binghamton; Ms. Wendy Swears, Franklin, NY; Ms. Donna Dustin, Cornell University, Ithaca, NY; and Messrs. Bill Cook and Henry Brunelle, NAB. Phytoplankton identification and enumeration were accomplished by Dr. Barbara Speziale, Newry, SC. Analytical and field assistance was provided under the supervision of Dr. John Titus, SUNY, Binghamton; Dr. Leonard Sohacki, SUNY, Oneonta; and Drs. Ray Oglesby and Ed Mills, Cornell University. Mr. Tim Bruni, AScI Corporation, assisted in the preparation of figures and tables.

Technical reviews of this report were provided by Drs. Richard E. Price and John J. Hains.

The study was conducted under the direct supervision of Dr. Thomas L. Hart, former Chief, Ecosystem Processes and Effects Branch, and under the general supervision of Mr. Donald L. Robey, Chief, Environmental Processes and Effects Division, and Dr. John Harrison, Director, EL.

At the time of publication of this report, Director of WES was Dr. Robert W. Whalin. Commander was COL Bruce K. Howard, EN.

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1 Introduction

Background

Management of reservoirs requires an understanding of the physical, chemical, and biological processes that impact water quality at the reservoir and the area impacted by releases. Processes described for lakes (Hutchinson 1957, Ruttner 1963, Wetzel 1983) have also been characterized specifically for reservoirs (Thornton, Kimmel, and Payne 1990). In general, annual temperature cycles result in vertical gradients in water quality (due to density differences) and are common to deep (greater than 7 to 10 m, Wetzel 1983) lakes and reservoirs. This process, known as stratification, results in development of zones, referred to as the epilimnion (surface), metalimnion (middepth), and hypolimnion (bottom), which vary in depth as a function of energy inputs (thermal, wind-induced, and inflow dynamics) and outputs (cooling or discharge).

Stratification can limit vertical exchange within the water column, impacting chemical and biological processes. Of particular significance is the change in chemical processes in the hypolimnion as biological respiration exceeds dissolved oxygen production and reaeration and the available dissolved oxygen is depleted (anoxia). As oxygen depletion progresses to anoxia, oxidized metals in the surface layer of bottom sediments are reduced to soluble forms and iron-bound phosphorus is solubilized (e.g. Mortimer 1941, 1942; Stumm and Morgan 1981; Boström, Jannson, and Forsberg 1982). This source of phosphorus, referred to as internal loading, can account for 50 to 100 percent of the mass phosphorus increase in eutrophic lakes (Garber and Hartman 1985). Additionally, phosphorus has been identified as the most common nutrient that limits phytoplankton growth (Vollenweider 1968, Likens 1972, Schindler 1974). Consequently, controlling inputs of phosphorus is a major objective in managing the quality of water resources (Olem and Flock 1990).

Inflow processes such as mixing and material loading also impact water quality. The addition of inorganic nutrients, organic carbon (Likens 1972), and silt (Cooke et al. 1986) is known as eutrophication. Symptoms of eutrophication include elevated nutrient concentrations, increased oxygen depletion, increased solubilization of sedimentary nutrients and metals, excessive biological production, and reduced water clarity.

Recognition of the importance of water resources resulted in numerous studies of eutrophication processes (e.g., Likens 1972, Schindler 1974, Dillon and Rigler 1974, Chapra and Robertson 1977). Numerous techniques for restoration of eutrophic systems have been evaluated and are summarized in Cooke et al. (1986) and Ryding and Rast (1989). Techniques applied to reservoirs managed by the Corps of Engineers have focused primarily on in-reservoir or in-lake techniques (Price 1990).

Artificial destratification has been used to ameliorate adverse water quality associated with eutrophic conditions in thermally stratified lakes and reservoirs. Artificial destratification mixes the water column to develop or maintain isothermal conditions and minimizes gradients in density. The extent of mixing may be localized or lake-wide depending upon operation and system design. Methods of destratification commonly employed include mechanical mixing, hydrostatic mixing, and pneumatic mixing. Detailed reviews of artificial destratification have been provided by Lorenzen and Fast (1977), Pastorok, Ginn, and Lorenzen (1981a,b), Pastorok, Lorenzen, and Ginn (1982), Cooke et al. (1986), and others.

Effects on water quality, due to artificial destratification, are related primarily to mixing of the water column. Specifically, isolation of bottom waters is avoided, and reaeration at the air-water interface is maximized with sufficient mixing. Aerobic conditions are maintained in the water column and at the sediment-water interface. Aerobic conditions minimize the release of soluble metals from the sediments, which occurs via reduction during oxygen depletion (Mortimer 1941, 1942), thereby decreasing phosphorus solubilization associated with iron reduction (Stumm and Morgan 1981). Mixing can also result in increased concentrations of CO_2 and decreases in pH values (Shapiro 1979). Habitat for aerobic organisms is also increased as the result of expanded oxygenated regions. Phytoplankton, zooplankton, fish, benthic, and macrophyte communities may also be impacted by reduced nutrient availability (Cooke et al. 1986). Improvements in release quality, such as lower concentrations of metals and nutrients and increased dissolved oxygen concentrations, would be expected with improved in-lake water quality (Price and Meyer 1992).

Impacts of artificial destratification on the phytoplankton community have been evaluated for potential as a control technique. Generally, reduced internal nutrient loading, increased concentrations of CO_2, decreased pH, decreased time in the photic zone, and disruption of density gradients, which favor buoyant species, should result in lower concentrations of phytoplankton or a shift in dominant species (Cooke et al. 1986). Both positive (decreased biomass of undesirable algae or a shift to a more desirable species) and negative (increased biomass of undesirable species) results of artificial destratification on phytoplankton communities have been summarized by Pastorok and Grieb (1984). Failure at attempts to reduce phytoplankton biomass have been attributed to insufficient mixing (Osgood and Stiegler 1990) and operation during stratification (Gulliver and Stefan 1982). Insufficient mixing results in intermittent anoxia, increased concentrations of reduced metals and soluble nutrients, transport of sedimentary phosphorus to the photic zone, and increased phytoplankton production. Increased phosphorus transport from bottom to surface waters also occurs when mixing is initiated during stratification when chemical gradients exist. Algal control has been considered successful in applications in Australia (Burns 1981) and by others (Pastorok, Lorenzen, and Ginn 1982) when the destratification system was designed and operated to maintain isothermal conditions.

Study Objectives

Artificial destratification has been applied at East Sidney Lake to enhance water quality of the reservoir and tailwater. The objective of artificial destratification was to maintain near-isothermal temperatures in the reservoir, thereby maintaining oxygenated conditions in the bottom waters via atmospheric exchange during mixing. Maintenance of aerobic bottom waters should decrease solubilization of sedimentary metals, reduce internal loading of sedimentary phosphorus, improve discharge quality, and lessen the severity of excessive algal populations. This report compares water quality at the project prior to system operation and during several summer seasons of operation. Short-term impacts on thermal structure, dissolved oxygen regimes, metals and nutrient dynamics, and tailwater quality are evaluated, and data from phytoplankton enumeration are presented and discussed.

2 Project Location and Description

East Sidney Lake is located in the watershed of the North Branch of the Susquehanna River in south-central New York (Figure 1). The project was authorized by the Flood Control Act of 1936 and provides flood protection for towns in southern New York and eastern Pennsylvania. The lake was filled in 1950 and maintains a conservation pool near 347 m National Geodetic Vertical Datum (NGVD) from 1 December to 14 May for flood control and a summer recreation pool near 350 m NGVD. An average inflow rate of 4.9 $m^3\ sec^{-1}$ is derived primarily from surface runoff from the 264-km^2 drainage area. The volume of the lake at the summer pool elevation is 4.1 m^3 with a mean and maximum depth of 4.9 and 15.7 m, respectively, and a hydraulic residence time of 9.8 days. Lake surface area is approximately 85 ha during the summer. Recreation facilities at East Sidney Lake include one public use area with a swimming area and beach, boat ramp and docks, dressing facilities, sanitary facilities, and areas for picnicking and camping.

East Sidney Dam was constructed with controlled outlet conduits in a concrete gravity section with an uncontrolled ogee weir as a spillway. The outlet works consist of five 1.1- by 1.8-m gate-controlled conduits with inverts at 339.85 m NGVD. Consequently, at summer pool elevations, discharge waters originate from near 11 m of depth. This depth is well below the mean depth, and summer releases result in the removal of cooler bottom water with high concentrations of nutrients and metals (Ashby and Kennedy 1990).

Water quality during the summer growing season (June through August) is characterized by high phosphorus concentrations, hypolimnetic anoxia, excessive phytoplankton populations, and reduced water clarity (Kennedy et al. 1988). Anoxic conditions, which favor the mobilization of sedimentary phosphorus, develop during thermal stratification and result in the accumulation of soluble metals and nutrients in bottom waters. Internal loading of sedimentary phosphorus contributes more phosphorus to the reservoir than does external loading during thermal stratification (Ashby and Kennedy 1990). The potential for transport of sedimentary phosphorus to the photic zone during wind-induced mixing events is considered to be high, because of the low thermal stability during stratification (Kennedy et al. 1988). This source of phosphorus, along with external sources, is considered a major contributor to excessive concentrations of phytoplankton at the lake.

Edmondson, W. T. (1977). "Trophic equilibrium of Lake Washington,"

3 Materials and Methods

Water Quality Sampling and Analytical Methods

Data collection was conducted at inflow, in-lake, and outflow stations (Figure 2) from May through October in 1988, 1989, 1990, and 1991 on a weekly or biweekly schedule. Sampling was conducted several times a day on a daily basis in association with initial operation of the destratification system and during runoff events in August and September of 1991. Temperature, dissolved oxygen, and specific conductance were measured at middepth for inflow and outflow stations and 1-m profiles at in-lake stations. In situ measurements were conducted with model 57 and 50b meters for temperature and dissolved oxygen and a model 33 meter for specific conductance (Yellow Springs Instrument Corporation, Ohio). Instruments were calibrated prior to each sampling trip. Water samples for chemical analyses (total iron, manganese, phosphorus, nitrogen, and sodium) were collected as discrete samples on the same schedule except that in-lake profiles were conducted at 2-m intervals. Discrete samples were collected with a Van Dorn sampler at in-lake stations and a bucket at inflow and outflow stations. Samples for phytoplankton identification and enumeration were collected as integrated samples taken from the surface to twice the Secchi disk depth. A 3.8-cm (inside diameter) polyvinyl chloride (PVC) pipe fitted with a one-way valve was used to collect the integrated sample. Samples for chlorophyll pigments were collected as discrete and integrated samples.

Samples for total iron, manganese, and sodium were digested with a nitric acid reflux procedure (American Public Health Association (APHA) 1985) and analyzed with an atomic adsorption spectrophotometer employing an air/acetylene flame (model 4000, Bodenseewerk Perkin-Elmer and Company, Uberlingen, West Germany). Samples for total phosphorus and nitrogen were digested with a persulfate oxidation in an autoclave prior to analysis (APHA 1985).

Total phosphorus analysis was conducted with automated colorimetric procedures (Technicon AAII System, Technicon Industrial Systems, Tarrytown, NY) using the ascorbic acid reduction method at 880 nm (APHA 1985). Total nitrogen was determined on an aliquot of the digested sample following a 24-hr reduction to ammonium with DeVarda's alloy (50 percent copper, 45 percent aluminum, and 5 percent zinc) (Raveh and Avnimelech 1979). Automated colorimetric analysis was conducted with the phenate method at 630 nm (APHA 1985). Samples for chlorophyll analysis were filtered within 4 hr of collection and were stored on ice or frozen until analysis. Chlorophyll was analyzed spectrophotometrically using the trichromatic method (APHA 1985). Phytoplankton, preserved in Lugol's (APHA 1985), were identified and enumerated using an inverted microscope.

Data Analysis

Pool elevation and discharge data provided by the U.S. Army Engineer District, Baltimore, were converted to daily mean values (inflow data were calculated by Baltimore District as a function of discharge and pool elevation). Inflows for Ouleout Creek and Handsome Brook were calculated as 75 and 25 percent of the total calculated inflow, respectively. Percentages used were based on estimated areas of the watersheds for each inflow. External loads of phosphorus to and from the lake during the summer growing season were calculated with daily mean flows and weekly concentration data using FLUX (Walker 1987). Mean values of inflow and outflow concentrations were compared for the summer seasons of each year of the study using Duncan's Multiple Range Test. Internal loading of phosphorus was calculated for the summer growing season using the following mass balance equation:

$$(P)int = (P)chng - [(P)in - (P)out]$$

where

$(P)int$ = net internal load, kg
$(P)chng$ = change in storage, kg
$(P)in$ = external load, kg
$(P)out$ = loss due to discharge, kg

4 System Design and Operation

Design

System design, based on lake volume, temperature stratification, and thermal stability of the lake during summer 1988, followed the guidance of Davis (1980). The system consists of a 15-hp (11,185-W) compressor with a timer, a main supply line of 121.9 m of 5.0-cm galvanized pipe and 304.8 m of 3.8-cm flexible rubber hose, a PVC distribution manifold, and eight 30.5-m sections of 3.8-cm PVC pipe as the linear diffuser. The original design called for only four sections of diffuser. Four additional sections were added to test various diffuser configurations. Each section of the diffuser received air from the manifold through separate hoses. The diffuser is drilled every 0.3 m with 0.8-mm holes that serve as the diffuser ports. The system is designed to deliver 64 cu ft (1.8 m^3) of air per minute at a pressure of 50 psi (345 kPa).

Destratification of the water column is accomplished by the upward movement of air bubbles from the diffuser, which induces vertical and lateral circulation as the bubbles rise to the lake surface. The volume of water circulated is based on the depth and volume of air delivered to each diffuser port. The system was designed to maintain a temperature difference between the surface and bottom of less than 2 °C. Complete design criteria, specifications, and operating guidelines are described in Meyer, Price, and Wilhelms (in preparation).

Operation

The pneumatic mixing system was installed in summer 1989, and the diffuser was placed in the downstream region of the reservoir along a 9.1-m bottom contour, parallel to the shoreline (Figure 3). A break in the diffuser line during installation resulted in operation of only six sections of the system. Initial operation began 27 July with 23 hr of operation followed by a 1-hr period of nonoperation. Approximately 2 weeks later, it was discovered that the timer on the compressor malfunctioned and the system was shut down for repairs. Operation of the destratification system in 1990 included various operating schedules. In 1991, the system was operated for 23 hr per day. Additional breaks in the diffuser system resulted in operation of only four sections of the diffuser during 1991. System operation for each year is summarized in Table 1.

5 Results and Discussion

Hydrology and Inflow Quality

The inflow hydrograph of Ouleout Creek can be divided into a high-flow period associated with winter and spring runoff events and a low-flow period having occasional summer and fall runoff events (Figure 4). Initial water quality in the lake for the summer growing season is defined by the quality of the high flow retained for establishment of a summer recreation pool since this results in the addition of 95 percent of the total lake volume. In each of the study years, runoff events in May provided the water for the summer pool. Retention time using an average inflow of 5 m^3 sec^{-1} is about 10 days at the summer pool elevation. Runoff events (near 10 m^3 sec^{-1} or greater) reduced retention times to 5 days or less. At low-flow conditions (2 m^3 sec^{-1} or less), retention times were increased to 24 days or more. The lake was "hydraulically flushed" during the summer growing seasons five times in 1988 and 1991, which were low-flow years, 15 times in 1989, and 13 times in 1990.

Elevated nitrogen and phosphorus concentrations in the inflows were observed coincident with runoff events (Figures 4 and 5). Similar responses were observed in the two major inflows; however, nitrogen concentrations in Handsome Brook (Figure 5) were generally lower than concentrations in Ouleout Creek. Mean values for the summer growing season are reported in Table 2. Summer mean concentrations of total phosphorus and total iron were highly variable between years of the 4-year study (131 and 109 percent, respectively, for Ouleout Creek; 92 and 149 percent, respectively, for Handsome Brook) and were not statistically different ($p > 0.05$). Summer mean concentrations of total nitrogen were less variable between study years (23 percent for Ouleout Creek and 34 percent for Handsome Brook) and were significantly different ($p < 0.05$) between years. Total manganese summer mean concentrations were less than the detection limit (0.05 mg L^{-1}) in each year of the study. Impacts of external loading are discussed in context with internal loading evaluations in the section "In-lake Water Quality."

In-lake Thermal Structure

Operation of the destratification system had a pronounced impact on the thermal structure of the lake (Figure 6). In 1988, prior to installation and operation of the destratification system, stratification began in late June. Maximum temperature gradients occurred in mid-July, when temperatures ranged from 26 to 18 °C, and a thermocline was established at a depth between 5 and 6 m. Seasonal cooling of surface waters began in mid-August, and complete mixing or "autumnal turnover" occurred in late August.

In 1989, thermal gradients established in mid-June (Figure 6). Temperatures ranged from 20 to 14 °C, with a thermocline near 3 m. Thermal gradients were disrupted coincident with a notable runoff event in late June, and temperatures were approximately 20 °C throughout the water column. By mid-July, temperatures ranged from 23 to 20 °C, and weakly stratified conditions existed until autumnal turnover in mid-September. Weakly stratified conditions may be the result of mixing and flushing of the lake in late June, the occurrence of warm (19 to 20 °C) bottom waters in mid- and late-July, operation of the destratification system, passage of a cold front with wind and rain in late July, and cool (23 °C) surface water temperatures.

The response of thermal gradients to changes in operation of the destratification system was apparent in 1990. Isothermal conditions were maintained throughout June, when the destratification system was operated for 23 hr per day. Operation was decreased to 12 hr per day in mid-June to evaluate the impact of reduced mixing. As thermal gradients began to establish in mid-July, operation of the destratification system was increased from 12 to 18 hr. Within 7 to 10 days, surface temperatures decreased from 25 to 23 °C. A runoff event in early August 1990, a few days after changes in operation of the destratification system, may have also influenced the vertical distribution of temperature. Temperatures remained between 23 and 19 °C until autumnal turnover in late September.

Temperatures observed in June 1991 were warmer (between 18 and 20 °C) than temperatures observed in June of other years. Temperatures were between 22 and 23 °C for most of the summer growing season. Maintenance of near-isothermal temperatures in 1991 was the result of operation of the destratification system for 23 hr per day.

Generally, thermal gradients greater than 2 °C were not observed in late-July 1989 during operation of the destratification system and throughout 1990 and 1991 during system operation. However, surface temperatures more than 2 °C higher than bottom temperatures were observed intermittently on warm, calm days during the study period. Temperature differences between surface and bottom waters were greater in 1990 than in 1991. This indicates that operation periods greater than 12 hr per day may be required to prevent stratification.

In-lake Water Quality

Summer oxygen gradients reflected impacts of thermal structure and operation of the destratification system (Figure 7). Anoxic conditions in the hypolimnion were observed from July to late-August 1988, when thermal structure was well-defined. Prior to autumnal turnover in late August, anoxic conditions extended from the bottom to a depth of 4 m. However, oxygen concentrations remained above 2 mg L^{-1} throughout the water column during the following years when the destratification system was in operation. Effects of the destratification system on dissolved oxygen concentrations were apparent in early August 1989 when bottom concentrations increased from between 3 and 4 mg L^{-1} to between 4 and 6 mg L^{-1} following a brief period of system operation. A similar response was observed in August and September 1990, when middepth and bottom concentrations increased by 1 mg L^{-1} when operation of the system was increased.

Changes in manganese and iron concentrations in bottom waters reflected temporal changes in dissolved oxygen concentrations. Manganese concentrations, which ranged from 0.2 to 1.0 mg L^{-1} in 1988 during anoxia, were generally less than 0.5 mg L^{-1} in 1989, 1990, and 1991 when dissolved oxygen concentrations were greater than 2.0 mg L^{-1} (Figure 8). In 1988, total iron concentrations in bottom waters, which ranged from 0.6 to 2.0 mg L^{-1}, were generally less than 1.0 mg L^{-1} during years when the destratification system was in operation (Figure 9). Increases in dissolved oxygen concentrations during 1989 and 1990, associated with operation of the destratification system, resulted in a decrease of about 0.2 mg L^{-1} in total iron concentrations at mid and bottom depths. Maximum total iron concentrations remained near 0.6 mg L^{-1} in bottom waters during the summer 1991 growing season.

Distribution patterns of total phosphorus were similar to those of total iron (Figure 10). Bottom concentrations of total phosphorus exceeded 0.16 mg L^{-1} in 1988, but remained near 0.03 to 0.06 mg L^{-1} during years when the destratification system was in operation. Operation of the destratification system in 1989 and increased operation in 1990 resulted in a decrease in total phosphorus concentration of 0.01 to 0.02 mg L^{-1}. Similar positive changes (i.e., increased oxygenated areas and decreased nutrients and metals) for other systems have been summarized by Pastorok and Grieb (1984).

Calculated external and internal loads for the summer growing season varied each year of the study (Table 3). Phosphorus loads were lowest in 1988, highest in 1989 and 1990, and intermediate in 1991. In 1988, 1989, and 1990, variability of external loads at the Ouleout Creek station was less than 10 percent. In 1991, a year with few summer high-flow events, and at the Handsome Brook station, variability was higher (near 25 percent), suggesting increased variability with decreased flow. Internal loading rates between 0.7 and 8.1 mg m^{-2} day^{-1} were consistent with other reported values (3.7 and 9.9 mg m^{-2} day^{-1}, Garber and Hartman 1985). Because of the annual variability in loading, a comparison between years was made using the percentage that the internal phosphorus loads contributed to the total load of the summer growing season. These calculations indicated that operation of the destratification system reduced the internal load from 70 percent in 1988 to between 50 and 25 percent of the total when the system was in operation.

Total nitrogen values were generally between 0.6 and 1.2 mg L^{-1} (Figure 11). Concentrations between 1 and 2 mg L^{-1} in bottom waters were observed only during periods when the destratification system was not in operation (1988 and July 1989). Concentration gradients were associated with low dissolved oxygen concentrations in bottom waters and high concentrations of phytoplankton in surface waters. Concentrations between 1 and 2 mg L^{-1} in surface waters occurred in 1989 and 1991, coincident with excessive phytoplankton blooms (described later in this section).

Chlorophyll *a* concentrations exceeded 30 $\mu g\ L^{-1}$ during the summer growing season in all years of the study (Figures 12 and 13). Vertical gradients, observed in May, August, and September 1989, August 1990, and July 1991 (vertical sampling was not conducted in 1988), were generally confined to the upper 4 m of the lake. A concentration peak at a depth of 8 m in July 1991 was an exception. Maximum concentrations were observed in 1989. Peak concentrations of integrated samples were available for all years, and allow for a comparison among years (Figure 13). Peak concentrations of chlorophyll *a* from integrated samples were considerably greater in 1989 and 1991 (greater than 40 $\mu g\ L^{-1}$) than in 1988 and 1990. The concentration in the integrated sample collected in mid-August 1991 (near 90 $\mu g\ L^{-1}$) exceeded those of discrete samples (see Figure 12), suggesting the existence of a dense layer of phytoplankton at a depth different than that from which the discrete samples were collected. The presence of a dense layer of phytoplankton would suggest that these organisms were able to overcome effects of the destratification system.

Temporal and spatial distribution of chlorophyll *a* concentrations in integrated samples varied annually during the study period (Figure 14). Except in 1991, concentrations in the photic zone were highest in September. In 1991, concentrations were generally higher in late July and early August. Concentrations in 1990 were lowest and were uniformly distributed longitudinally. With the exception of a concentration maximum at station 2 (the near-dam station) in May and a minimum concentration at station 2 in October, concentrations were uniformly distributed in 1989 as well. In 1991, longitudinal gradients in concentrations were apparent for most sampling events. With the exception of two samples in late July and mid-August, concentrations at station 2 were less than at the upstream stations.

Seasonal succession of phytoplankton at station 2 followed a pattern of Chrysophyta in May and June; occasional peaks of various groups (primarily Pyrrophyta and Chlorophyta [August 1990]) during June, July, and August; and Cyanophyta and Chrysophyta in August-October (Figure 15). Cyanophyta was the dominant group in each year of the study with similar biovolume maxima near 0.02 to 0.03 $mm^3\ cm^{-3}$ with notable exceptions. In August and September 1990 and 1991, biovolumes were greater than 0.04 $mm^3\ cm^{-3}$, and approached 0.10 $mm^3\ cm^{-3}$ in 1989. Maximum biovolumes occurred after the autumnal mixing. The decline in biovolume of Pyrrophyta and Cyanophyta in August 1989 occurred after operation of the destratification system and a storm event. Temporal distributions of phytoplankton at station 3 were similar to those of station 2 (Figure 16), with the exception of the peak in Chrysophyta in August 1990 coincident with the downstream peak in Chlorophyta.

Temporal distributions of Chrysophyta species varied annually (Figure 17). *Cryptomonas* dominated in May 1988 and in August and September 1990, while *Synura* was dominant in August and September 1989 and 1991. Of interest is the relatively low biovolume of all Chrysophyta species in 1989. *Anabaena planctonica* and *Aphanizomenon flos-aquae* were the major species of Cyanophyta and, except in 1990, *A. planctonica* was the dominant species (Figure 18). In late July 1989, *Aphanizomenon* was dominant prior to operation of the destratification system; however, *Anabaena* dominated following system operation.

Water clarity, as measured by Secchi disk transparency, was generally highest in June and July with minimum values (near 1 m) occurring by mid-August and continuing through mid-October (Figure 19). Minimum water clarity coincided with maximum concentrations of blue-green algal species. Reduced water clarity resulted in closure of the swimming area in years with and without operation of the destratification system. The beach was closed 30 August 1987, 7 September 1988, 28 August 1989, 16 August 1990, and 15 August 1991. Secchi disk transparency depths were 1 m or less when water clarity at the beach was considered unacceptable for swimming.

Discharge Water Quality

Changes in discharge quality were not clear for concentrations of nitrogen and phosphorus (Figure 20) nor for iron and manganese (Figure 21). Variability between years was greater than 50 percent for total phosphorus, iron, and manganese, and differences between summer means were not significant ($p > 0.05$) (Table 4). Differences between total nitrogen summer mean concentrations were significant ($p < 0.05$) between years, with higher values occurring in 1991 and 1989.

Maximum summer temperatures in the discharge, which historically were near 26 °C, were below 23 °C during the study (Figure 22). The higher discharge temperature observed in 1991 may be attributed to warmer in-lake temperatures coincident with the warm summer of that year.

6 Summary and Conclusions

Thermal gradients in the lake were minimal in 1990 and 1991 as a result of operation of the destratification system, and water quality was improved compared to previous years. Dissolved oxygen concentrations were maintained above 2 mg L^{-1} and were generally above 4 mg L^{-1}, thereby increasing the habitat for aquatic organisms. As a consequence, contributions of metals (iron and manganese) and nutrients (phosphorus and nitrogen) to bottom waters from anoxic sediments were greatly reduced. The contribution of the internal phosphorus load to the total phosphorus load during the summer growing season decreased from 70 percent in 1988 to 50 percent in 1990 and 25 percent in 1991.

Although the importance of internal phosphorus loading was reduced during the summer growing season, nuisance blue-green algal blooms continued to occur in the late summer. Water clarity during such blooms was often less than 1 m, prompting closure of the swimming area. This result may reflect the impact of increased external phosphorus loads, which were 3 to 10 times higher during summers when the destratification system was in operation. Other possible mechanisms for continued nuisance blooms exist. Internal processes, such as aerobic release of sedimentary phosphorus (Boström, Ahlgren, and Bell 1985) or biological recycling (Brabrand, Faafeng, and Nilssen 1990; Kitchell, Koonze, and Tennis 1975), may be sufficient to sustain algal populations. Other factors that may contribute to the proliferation of blue-green algal blooms include interactions among phytoplankton, zooplankton, and fish communities (e.g., Carpenter, Kitchell, and Hodgson 1985). Physiology of the algal species, such as palatability to grazing zooplankton (Fulton 1988) and buoyancy mechanisms (Walsby and Klemer 1974), may provide a competitive advantage over other algal species.

The impacts of external loading on in-lake water quality have not been fully assessed. Phosphorus concentrations were mostly above 10 μg L^{-1} in the inflow (a value suggested by Edmondson (1977) as a level adequate to sustain blue-green algal blooms), suggesting that external loading may contribute to increased algal populations. Increased phosphorus concentrations, coincident with runoff events, may also contribute to increased algal population, as suggested by the algal peak in August 1990 that followed a high-flow event. Impacts of hydraulic turnover (flushing) on in-lake nutrient concentrations and phytoplankton populations were not discernible. During high-flow summers

(1989 and 1990), flushing occurred approximately three times more than in 1988 and 1991, yet surface concentrations of phosphorus were similar for all years and nuisance blue-green blooms occurred each year.

In general, water quality of the tailwater was similar with and without operation of the destratification system. Only total nitrogen concentrations were significantly different for the summer growing season, with no obvious pattern related to operation of the destratification system. Discharge temperatures during operation of the destratification system were not higher than historical values, suggesting no impacts to the thermal regime of the tailwater.

7 Recommendations

Although successful at reducing phosphorus loading from deep sediments, operation of the destratification system was not singularly sufficient as an algal control method. If reductions in phosphorus loading from other sources can be achieved, operation of the destratification system may be more effective for algal control. Water quality studies should continue as a means for evaluating other nutrient loading processes. Other potential sources include external loading, biological inputs, and shallow-water sediment processes.

Additional meteorologic and hydrologic data should be collected during a season of nonoperation of the destratification system. Such data would allow the assessment of design criteria and impacts on thermal structure, in support of numerical methods being used to evaluate system performance.

Additional methods of algal control should be evaluated. These methods include manipulations of fish communities that impact plankton community structure and chemical addition for algal control. Manipulation of fish communities could be evaluated with a review of existing biological data, a literature review, and laboratory and field studies if the method is deemed feasible. Although less desirable to the public, chemical control of algal populations may be a method that would be acceptable during the late-summer recreation season.

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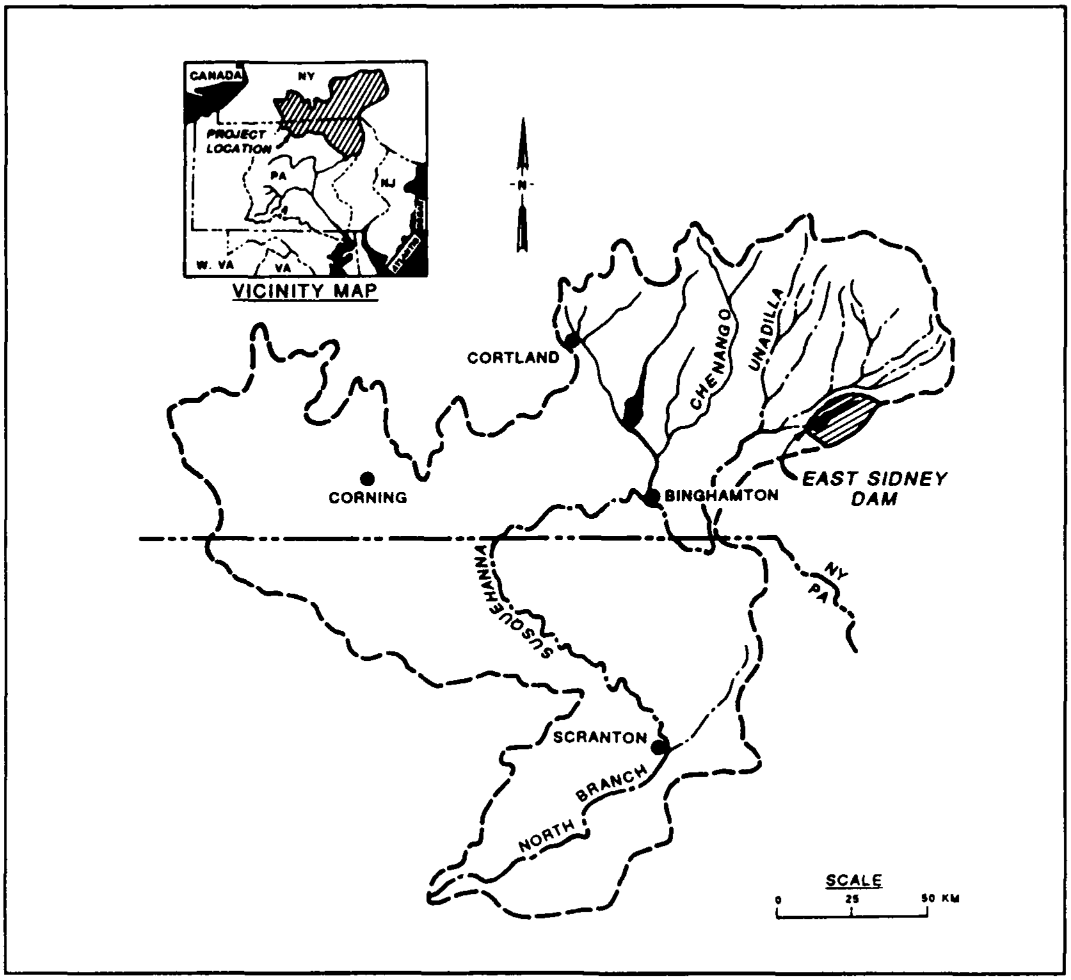

Figure 1. Location of East Sidney Lake

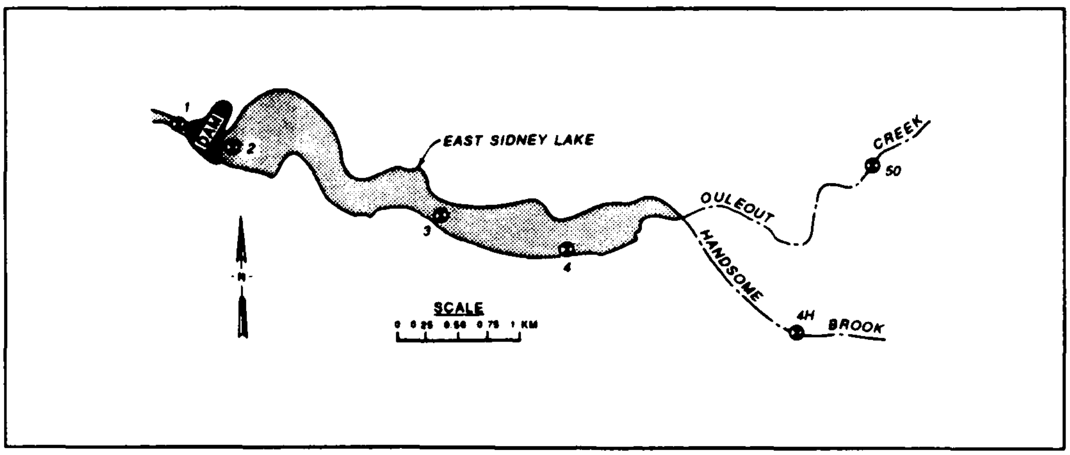

Figure 2. Station locations, East Sidney Lake

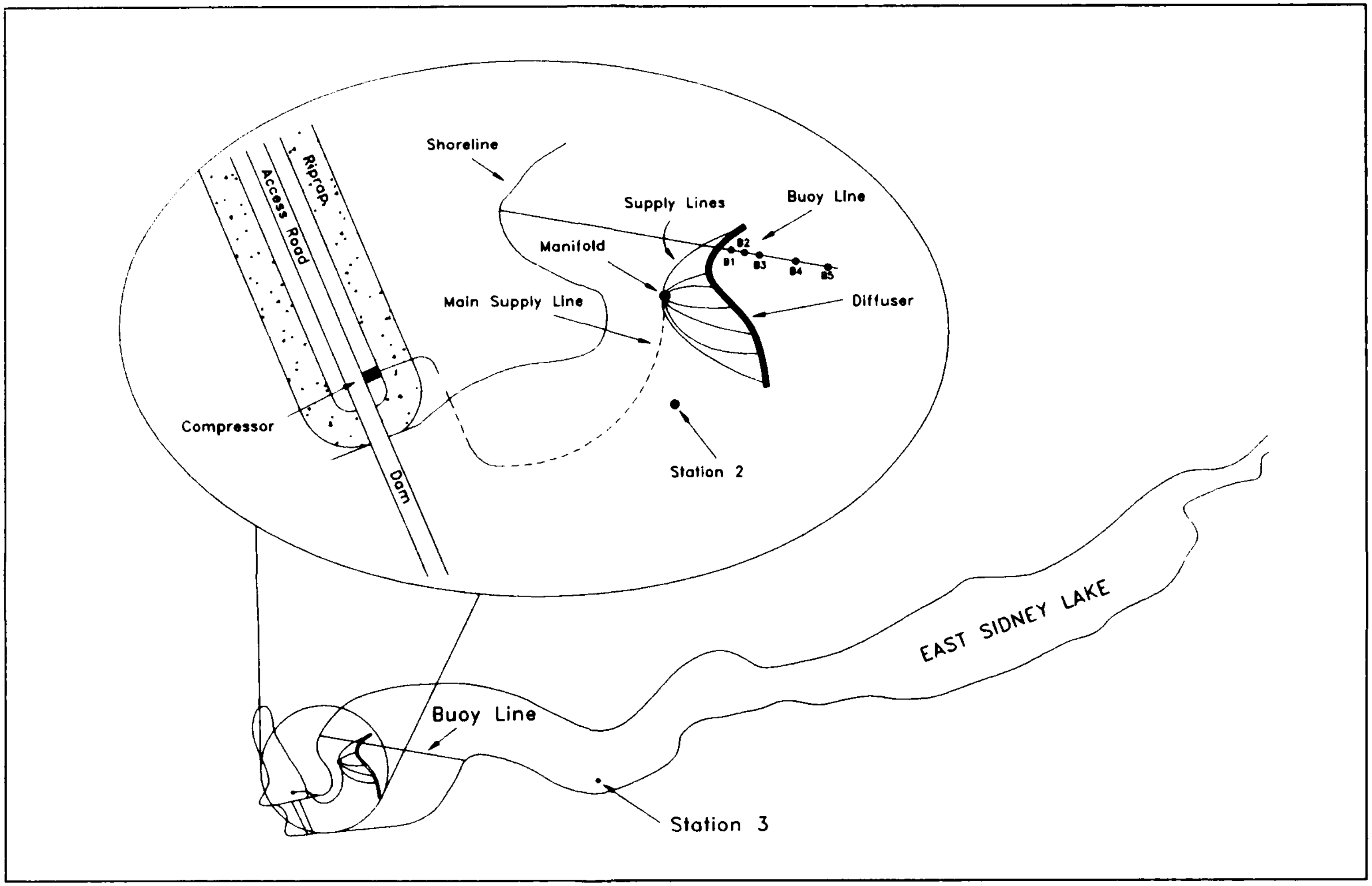

Figure 3. Schematic of diffuser system and location

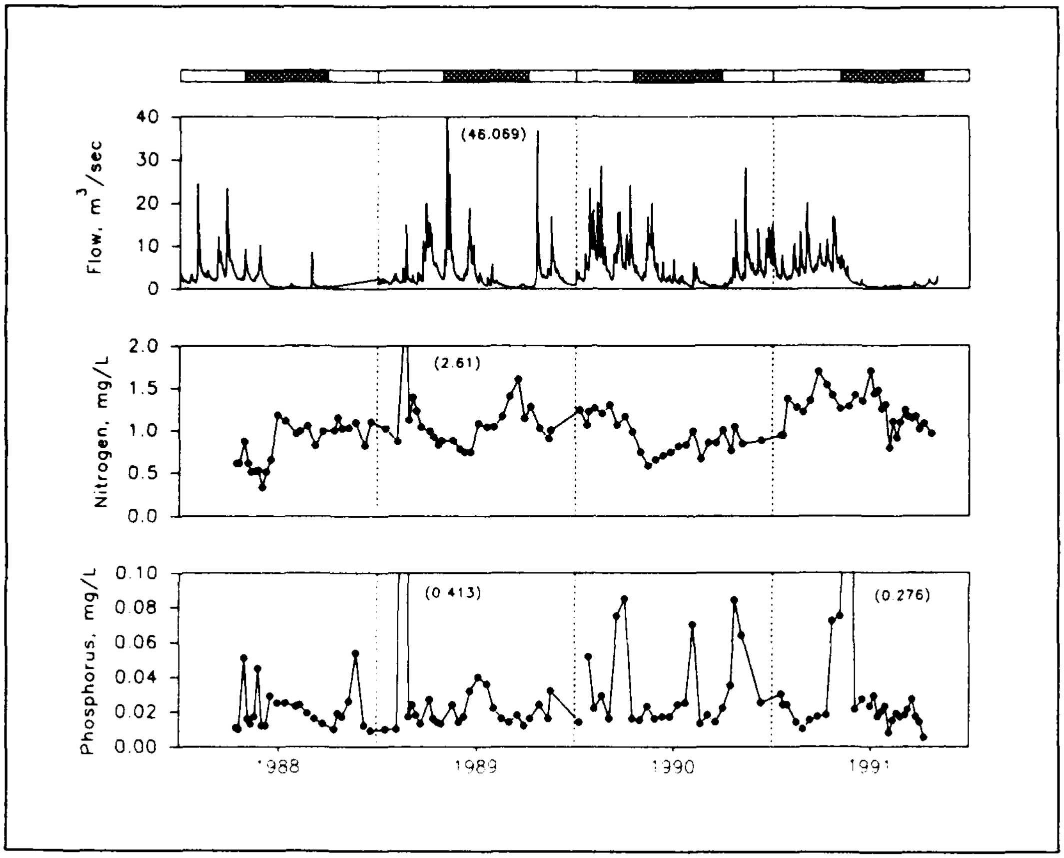

Figure 4. Inflow hydrograph, total nitrogen, and total phosphorus concentrations, Ouleout Creek, 1988-1991 (hatched area denotes May-October)

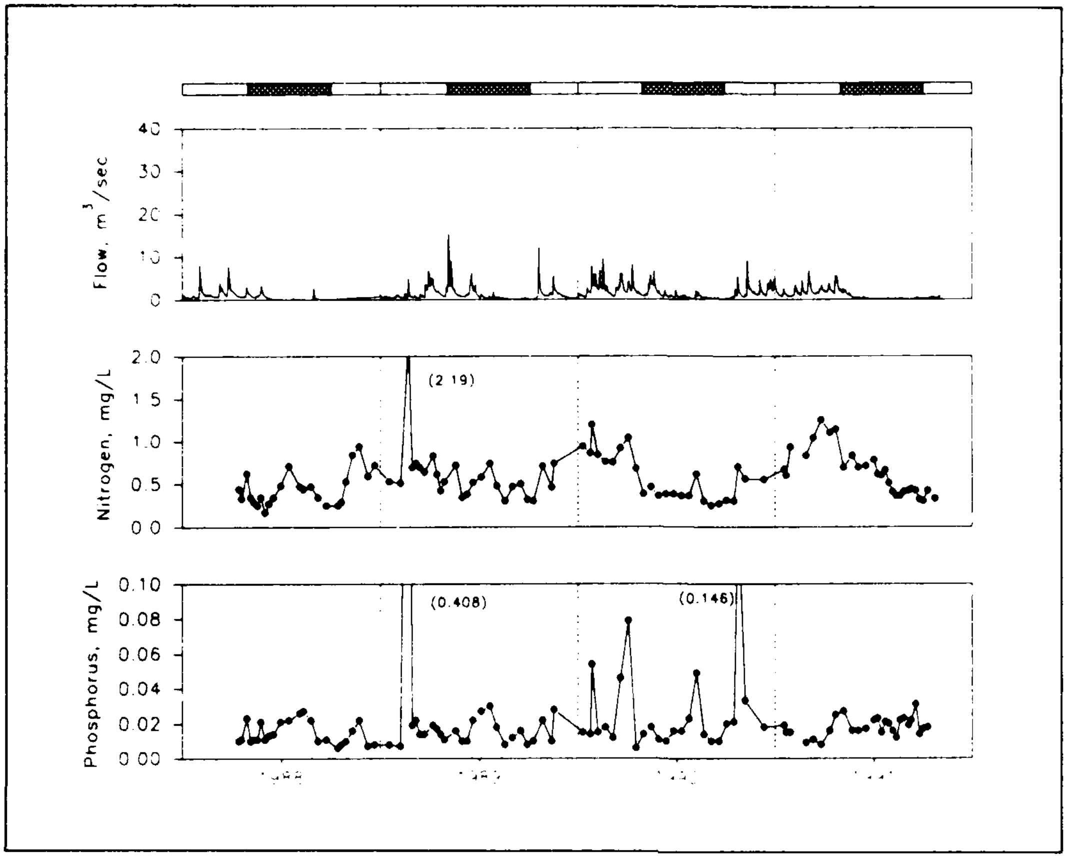

Figure 5. Inflow hydrograph, total nitrogen, and total phosphorus concentrations, Handsome Brook, 1988-1991 (hatched area denotes May-October)

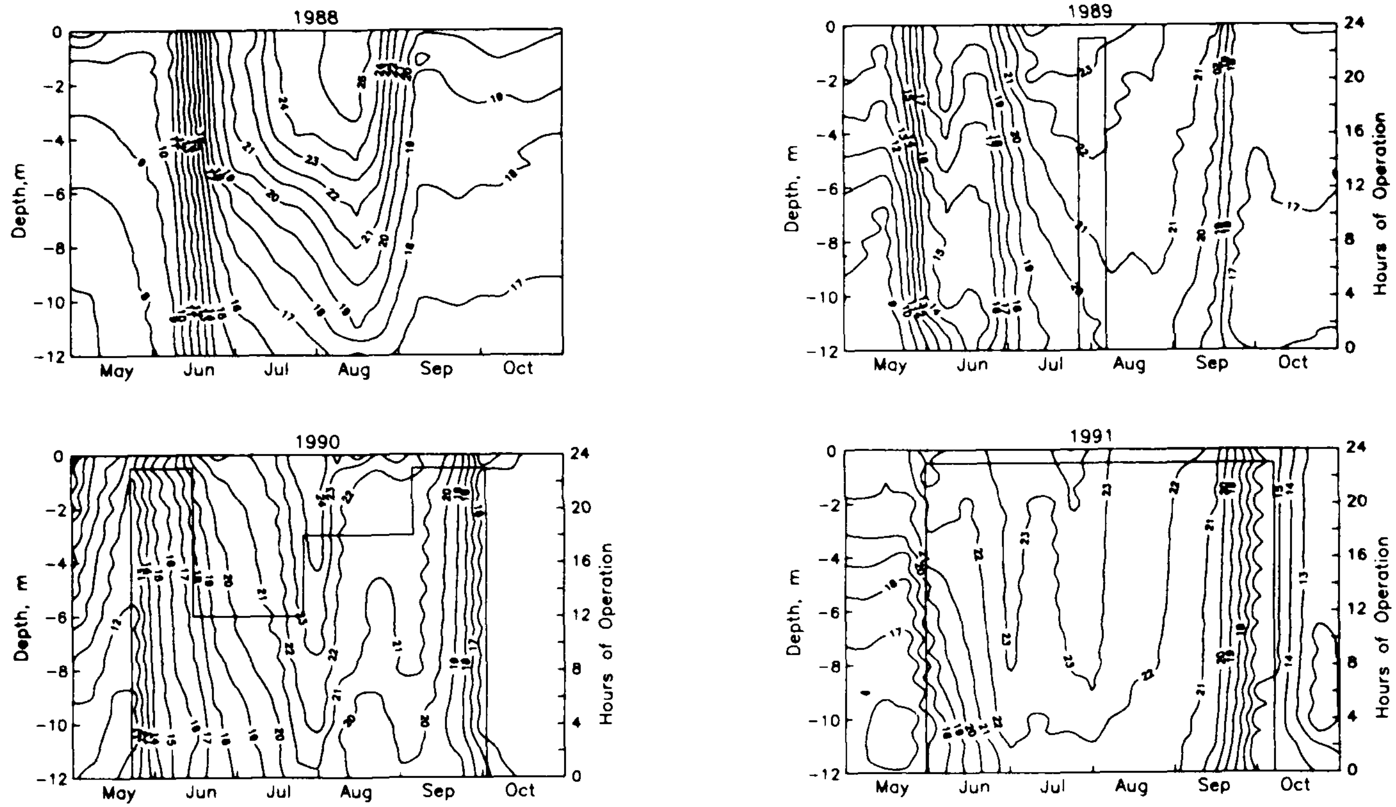

Figure 6. Temperature isopleths, station 2, May-October, 1988-1991

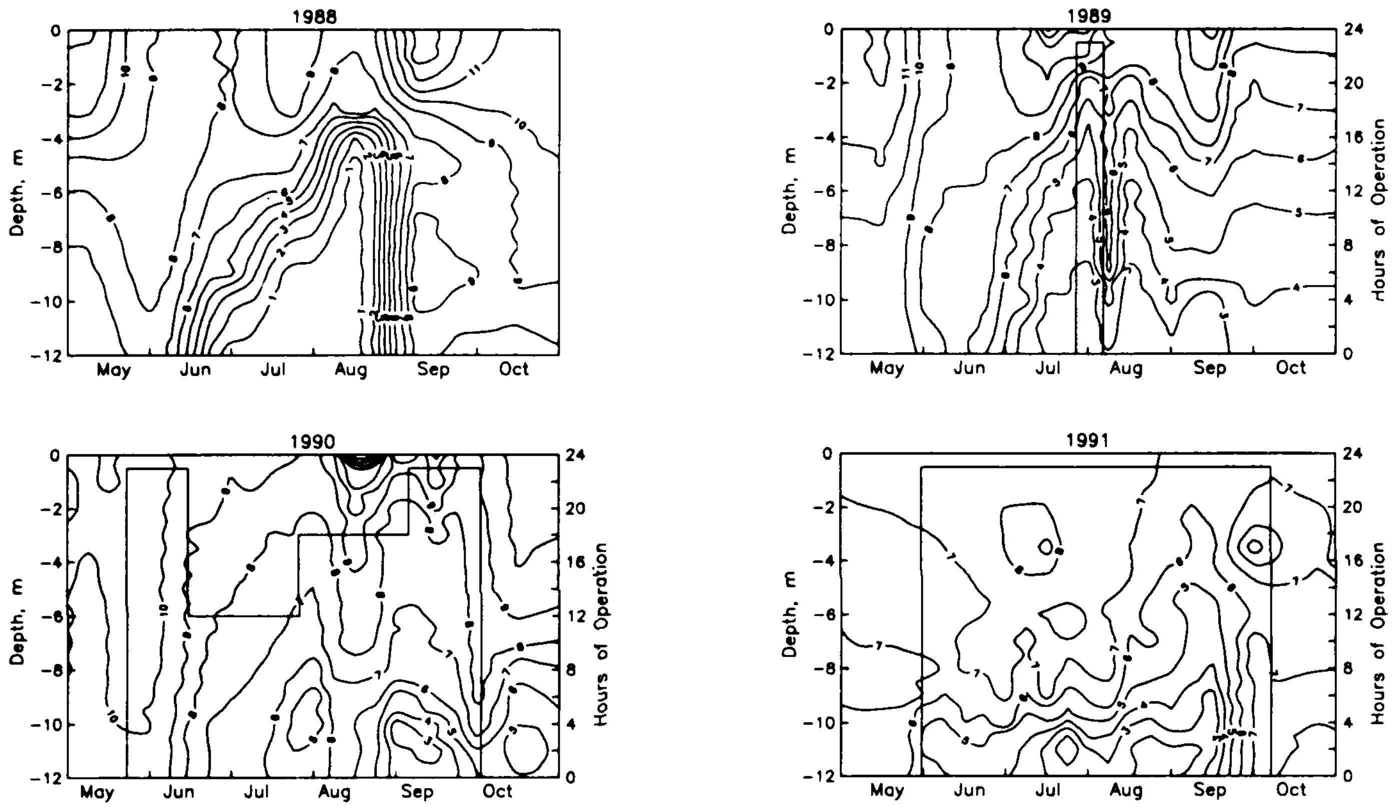

Figure 7. Dissolved oxygen isopleths, station 2, May-October, 1988-1991

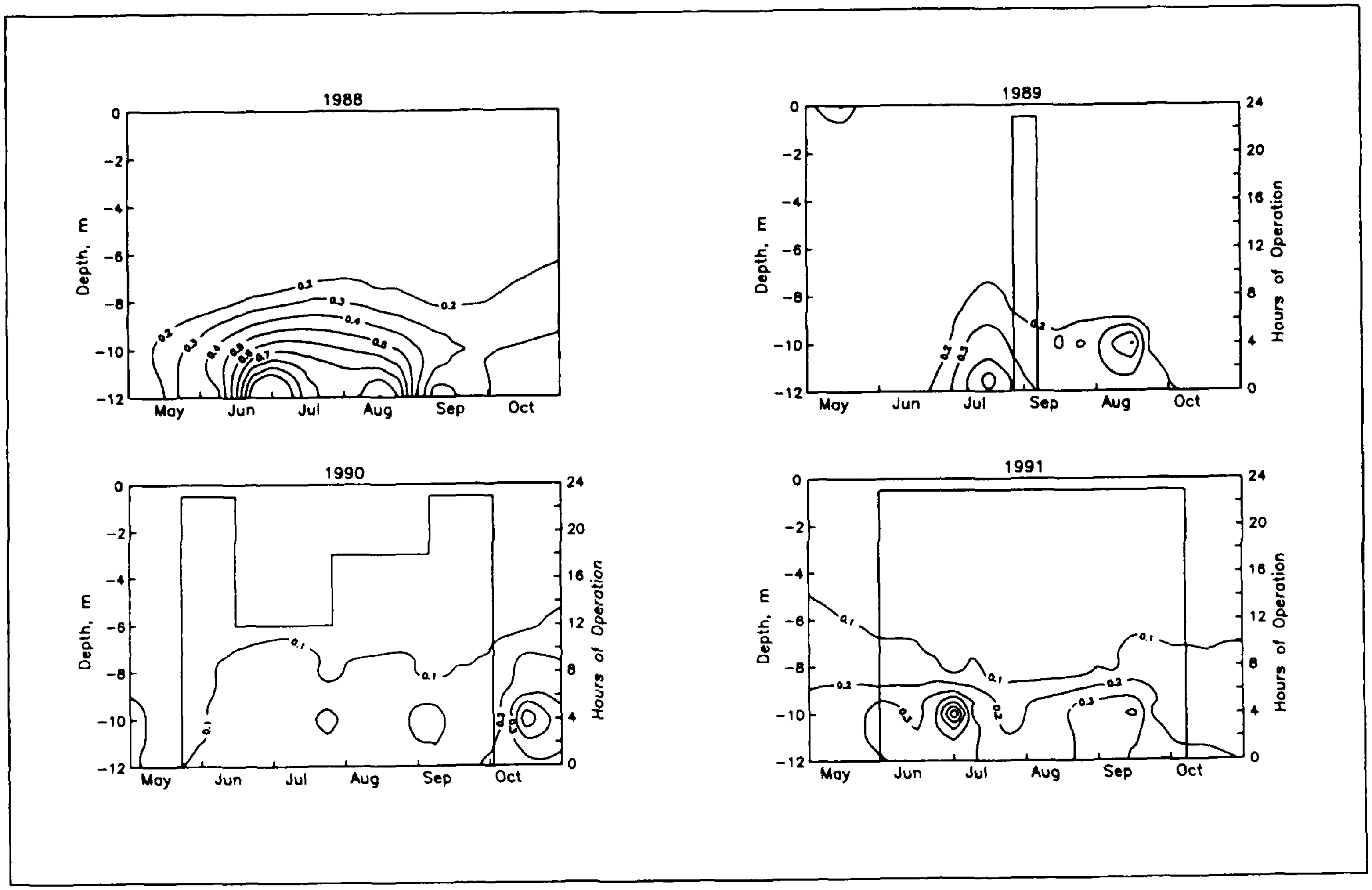

Figure 8. Total manganese isopleths, station 2, May-October, 1988-1991

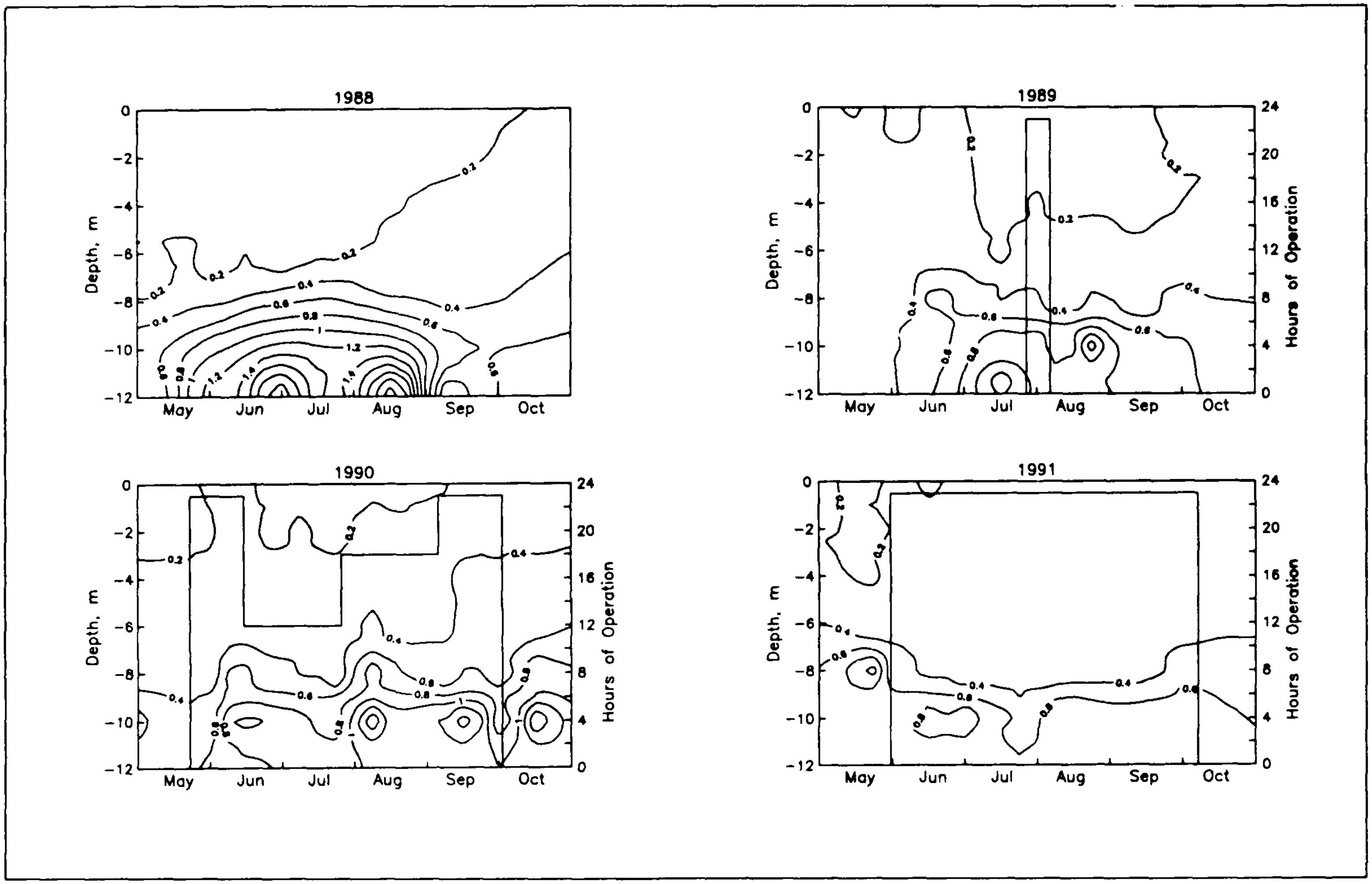

Figure 9. Total iron isopleths, station 2, May-October, 1988-1991

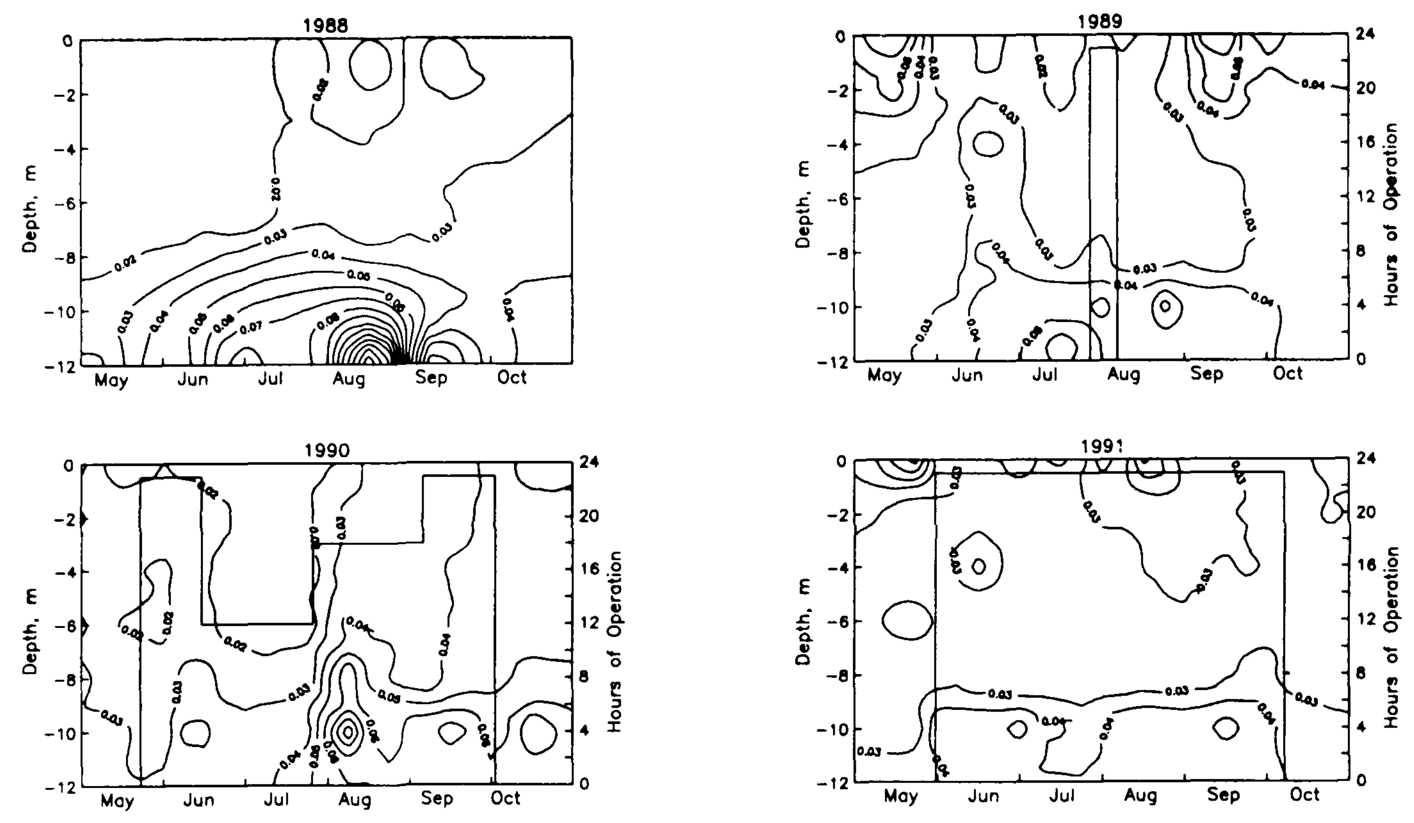

Figure 10. Total phosphorus isopleths, station 2, May-October, 1988-1991

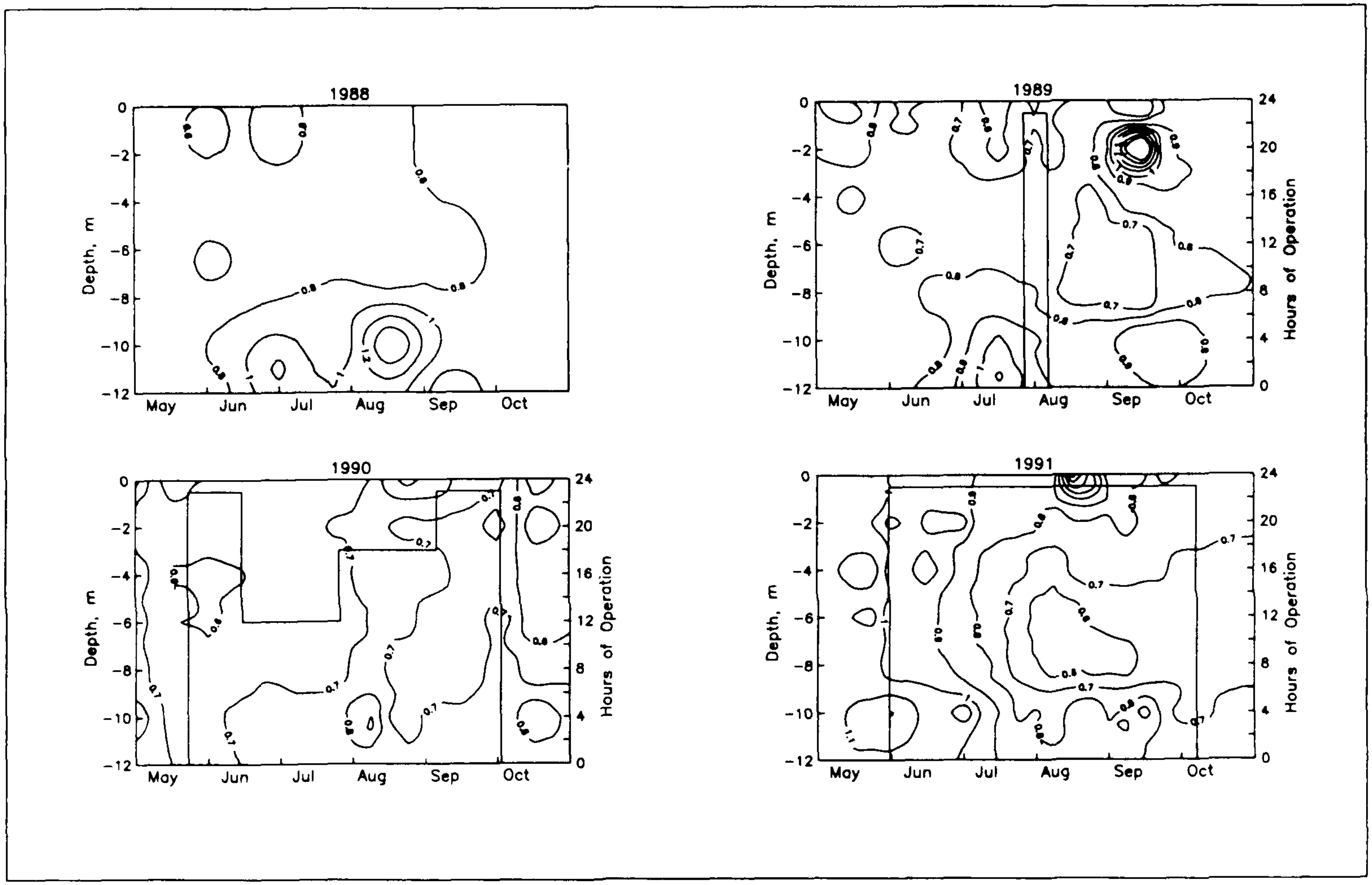

Figure 11. Total nitrogen isopleths, station 2, May-October, 1988-1991

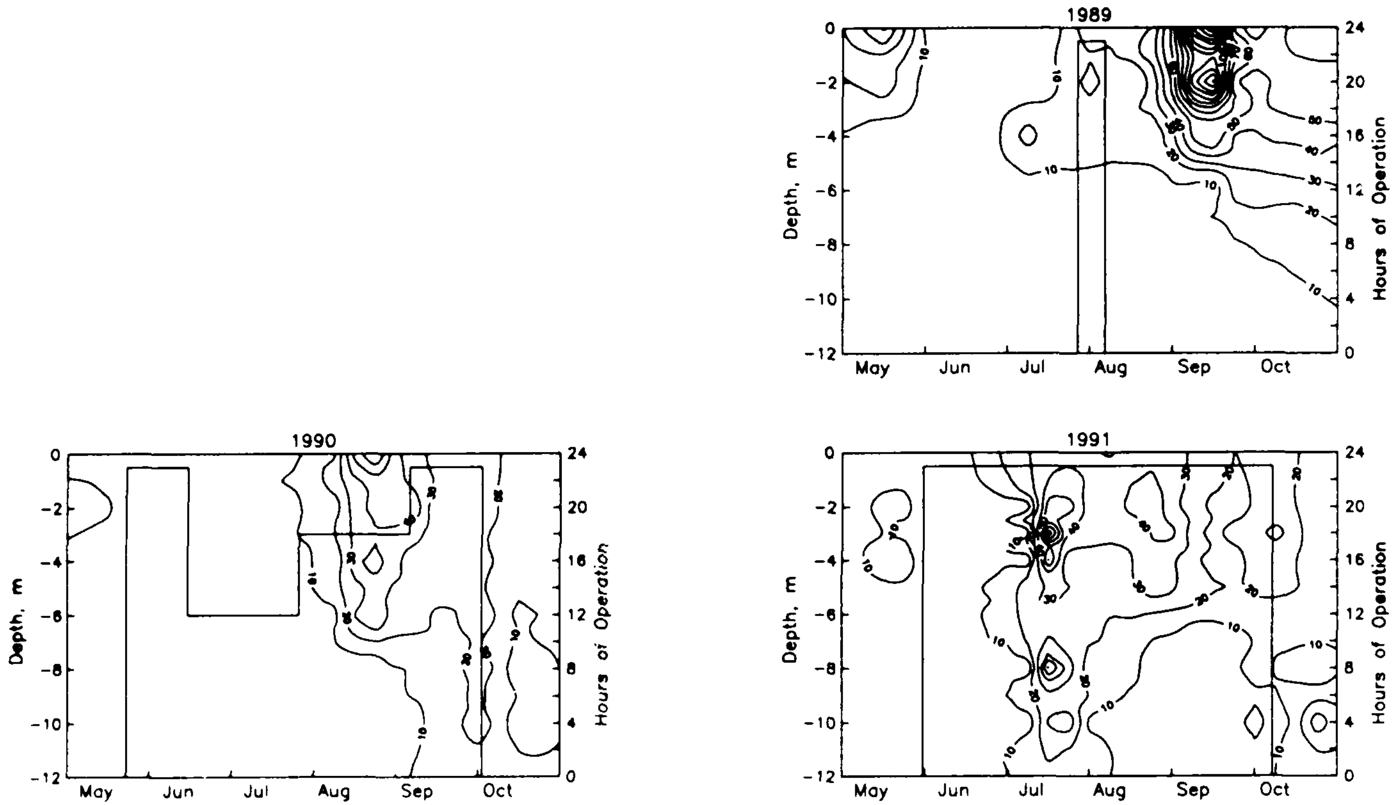

Figure 12 Chlorophyll *a* isopleths, station 2, May-October, 1988-1991

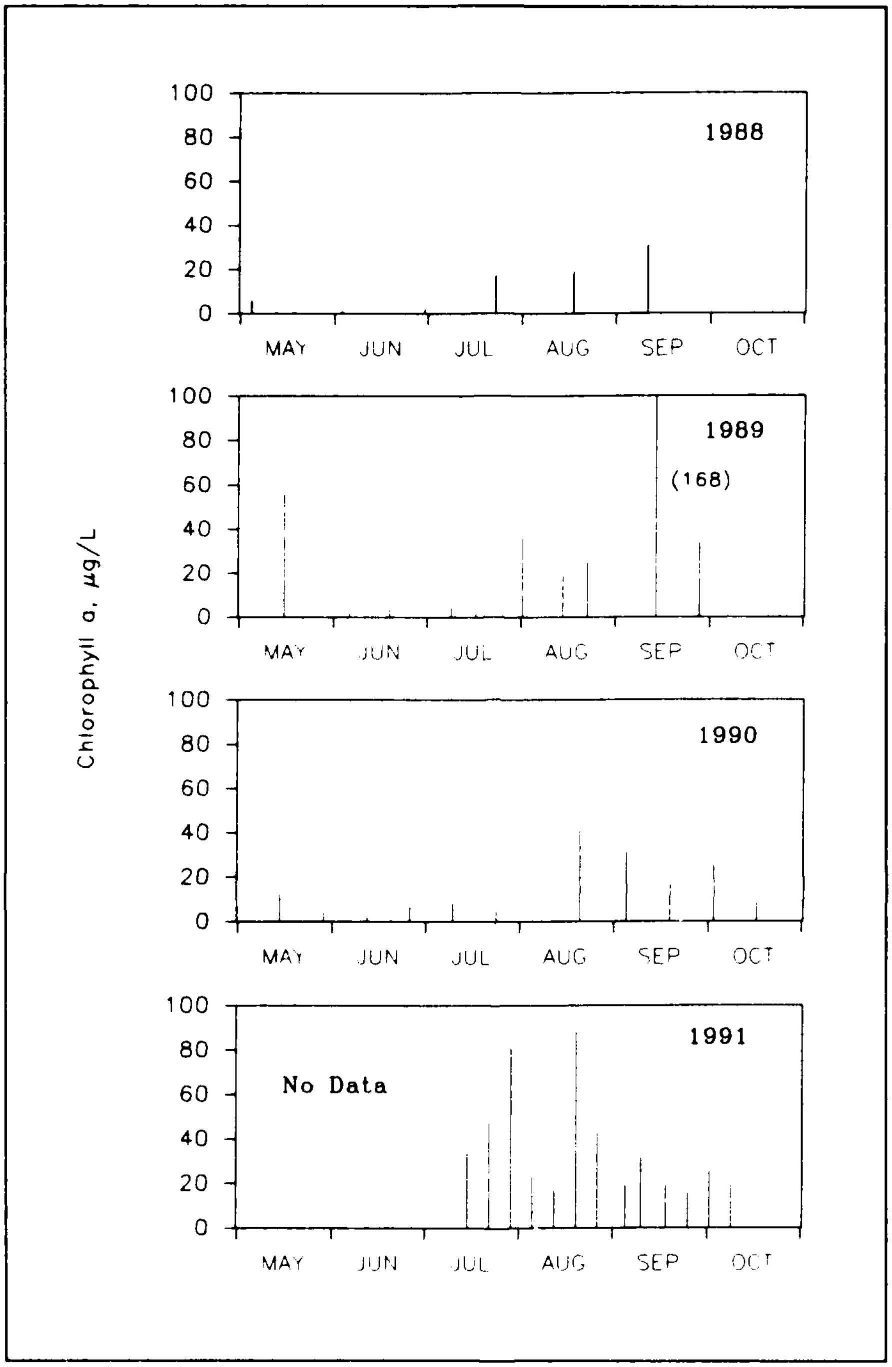

Figure 13. Chlorophyll *a* concentrations in integrated surface samples, station 2, May-October, 1988-1991

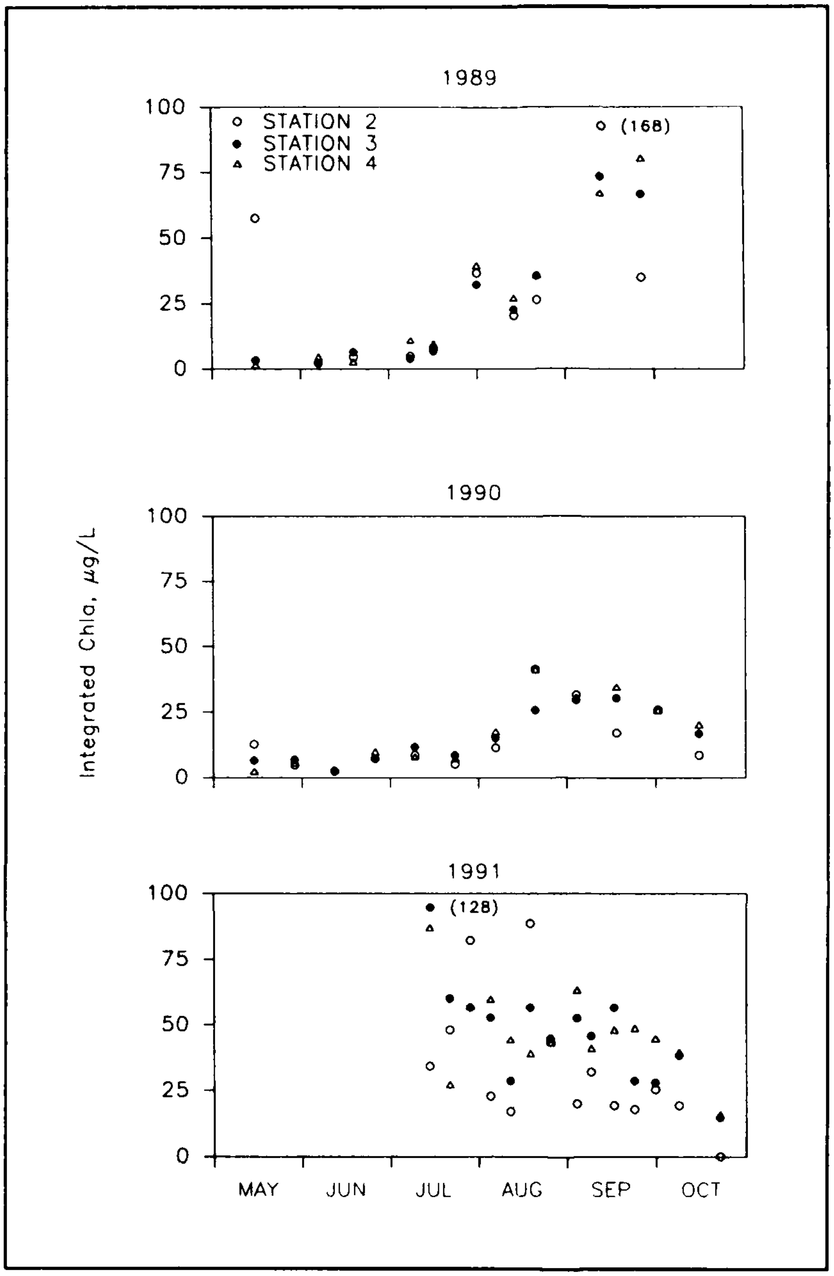

Figure 14. Chlorophyll *a* concentrations in integrated surface samples, stations 2-4, May-October, 1989-1991

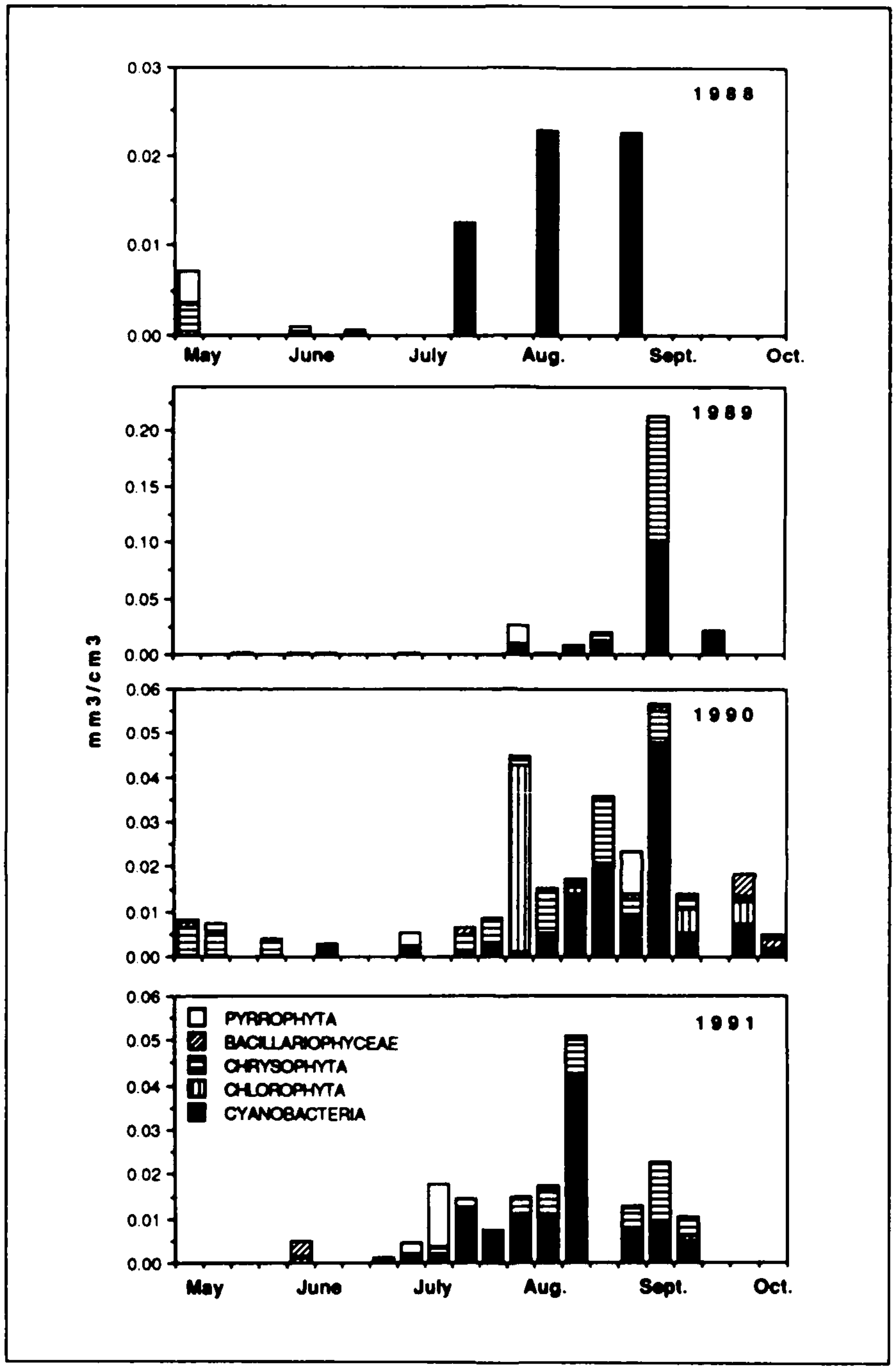

Figure 15. Temporal distribution of phytoplankton groups, station 2

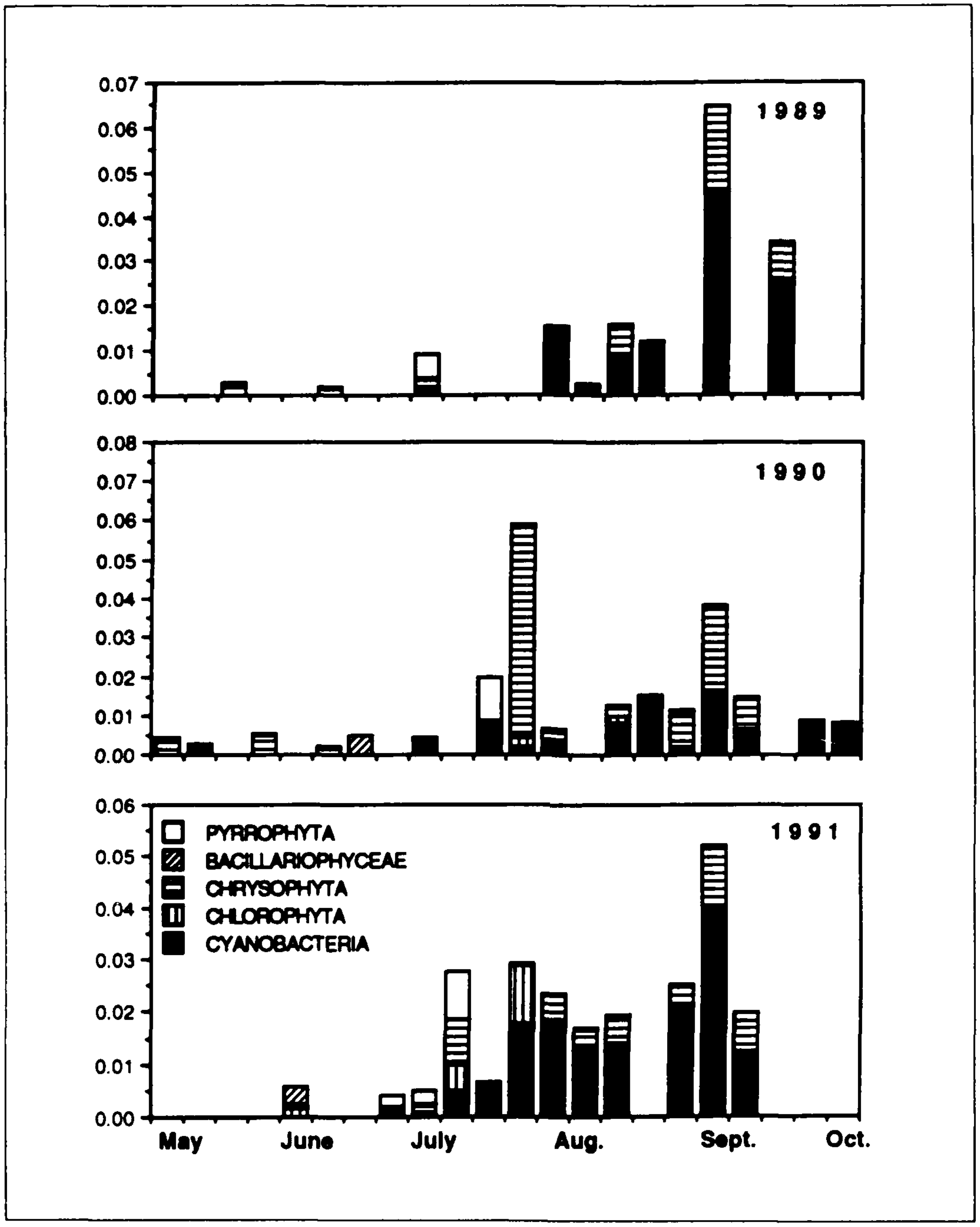

Figure 16. Temporal distribution of phytoplankton groups, station 3

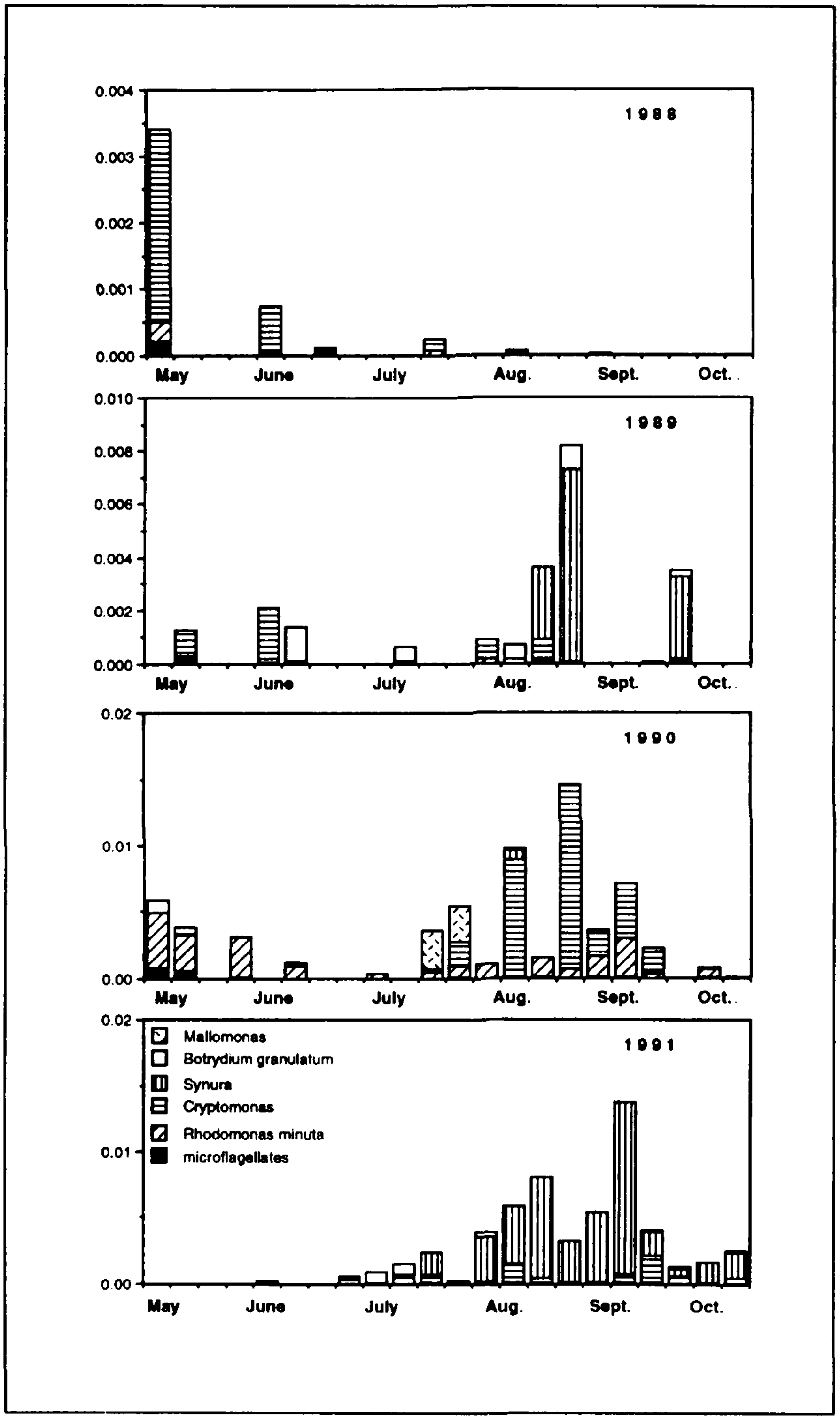

Figure 17. Temporal distribution of Chrysophyta species, station 2

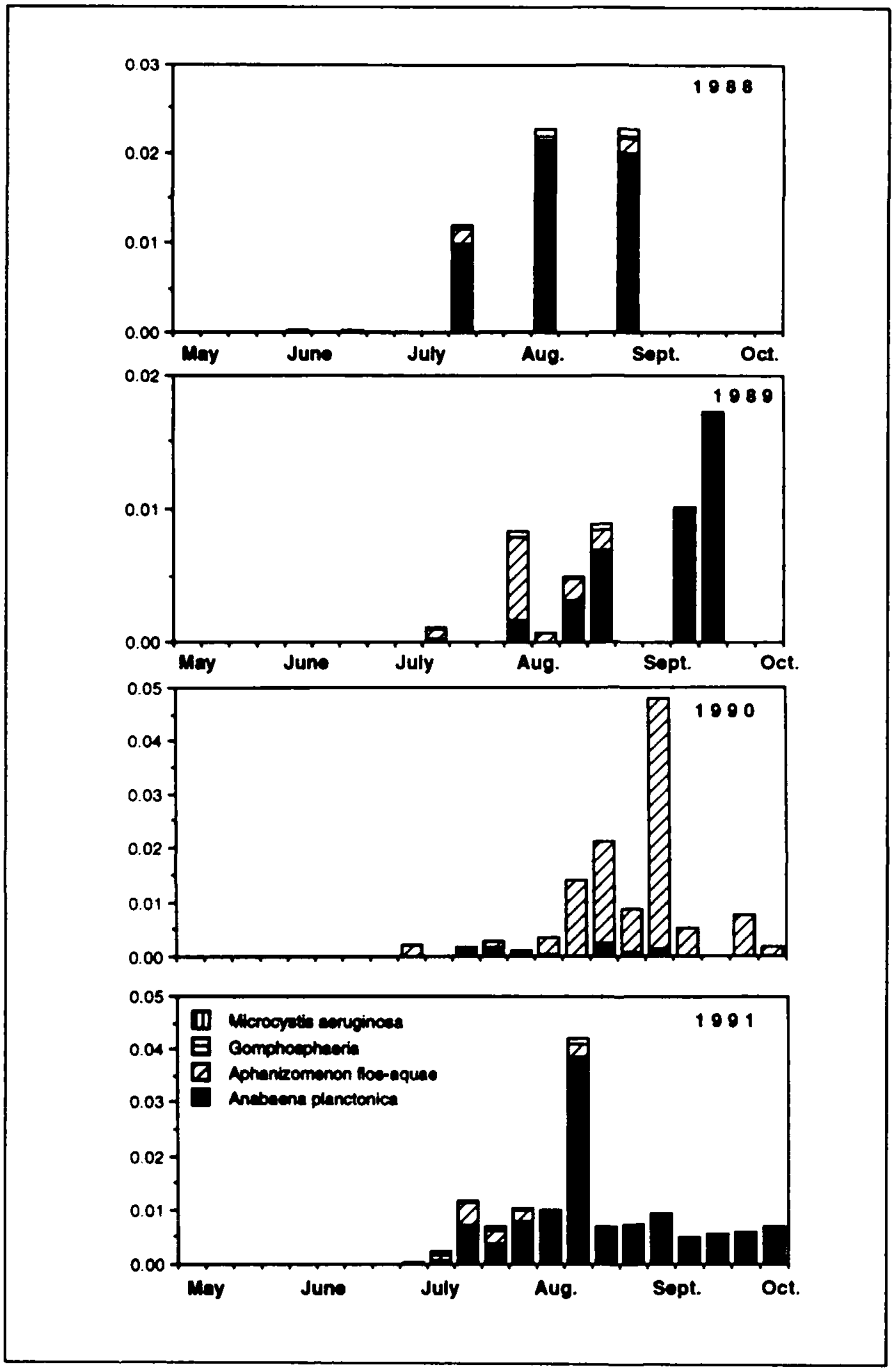

Figure 18. Temporal distribution of Cyanophyta species, station 2

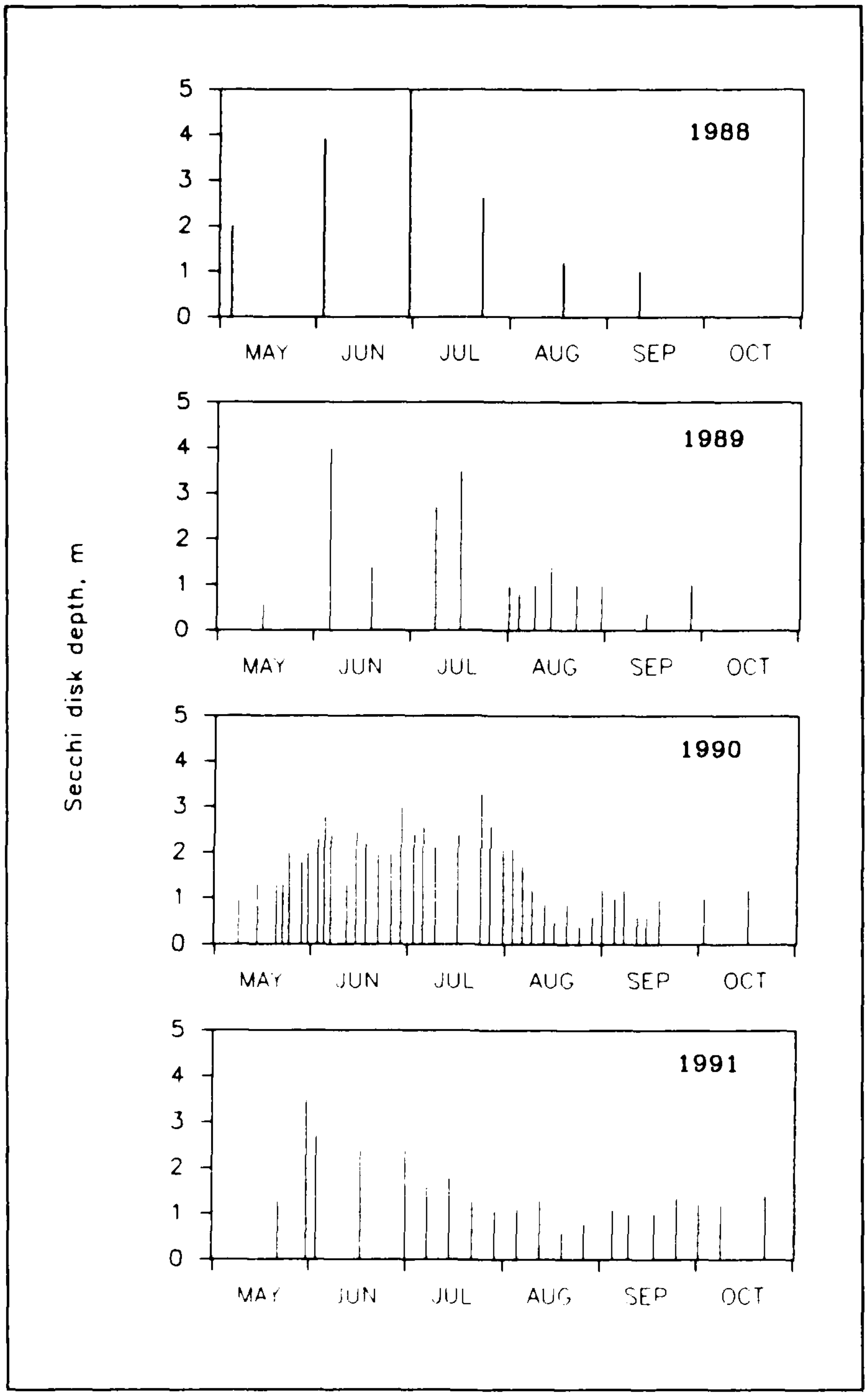

Figure 19. Secchi disk depths, station 2, May-October, 1988-1989

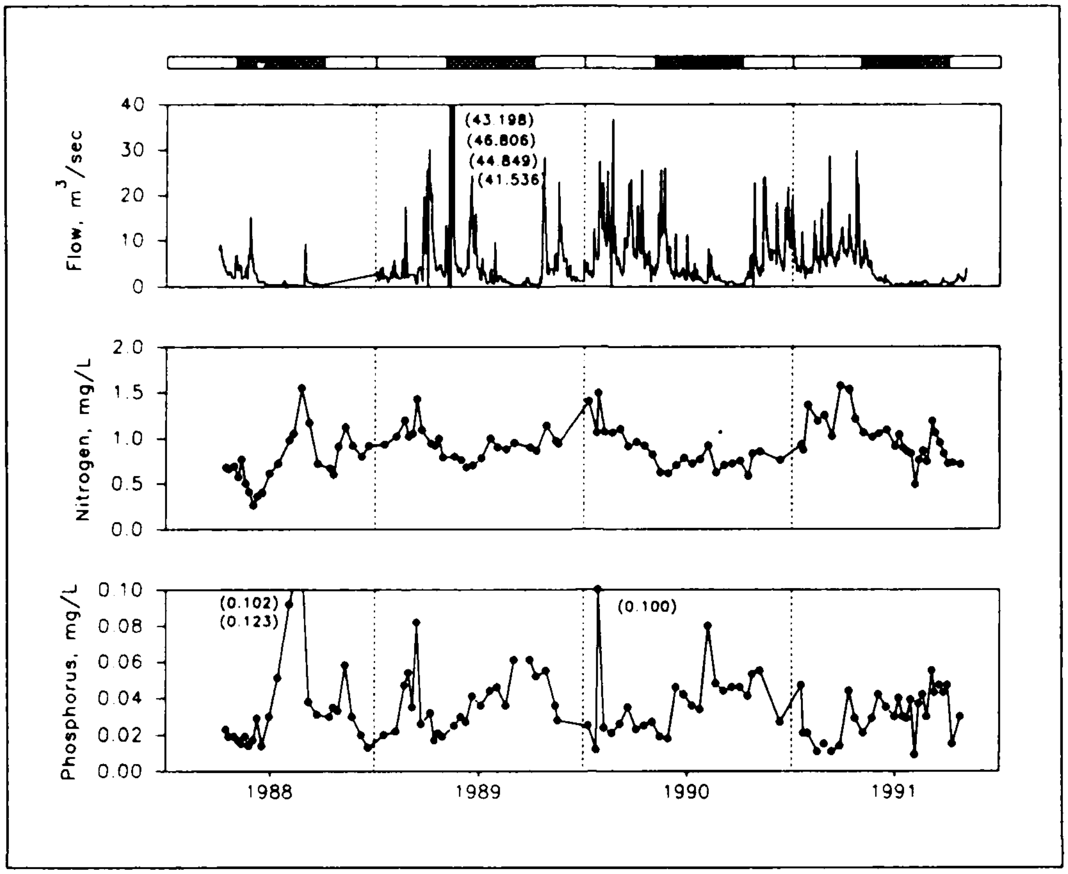

Figure 20. Discharge hydrograph, total nitrogen, and total phosphorus concentrations, East Sidney lake, 1988-1991 (hatched area denotes May-October)

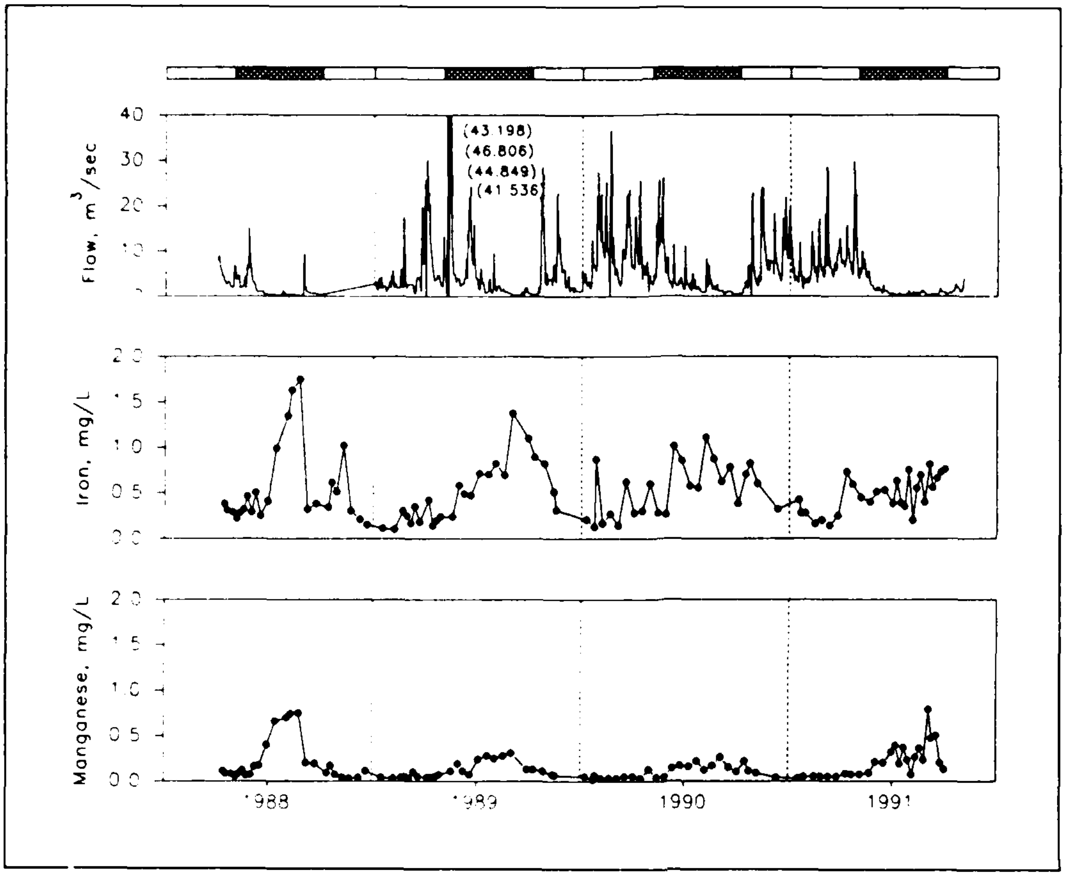

Figure 21. Discharge hydrograph, total iron, and total manganese concentrations, East Sidney lake, 1988-1991 (hatched area denotes May-October)

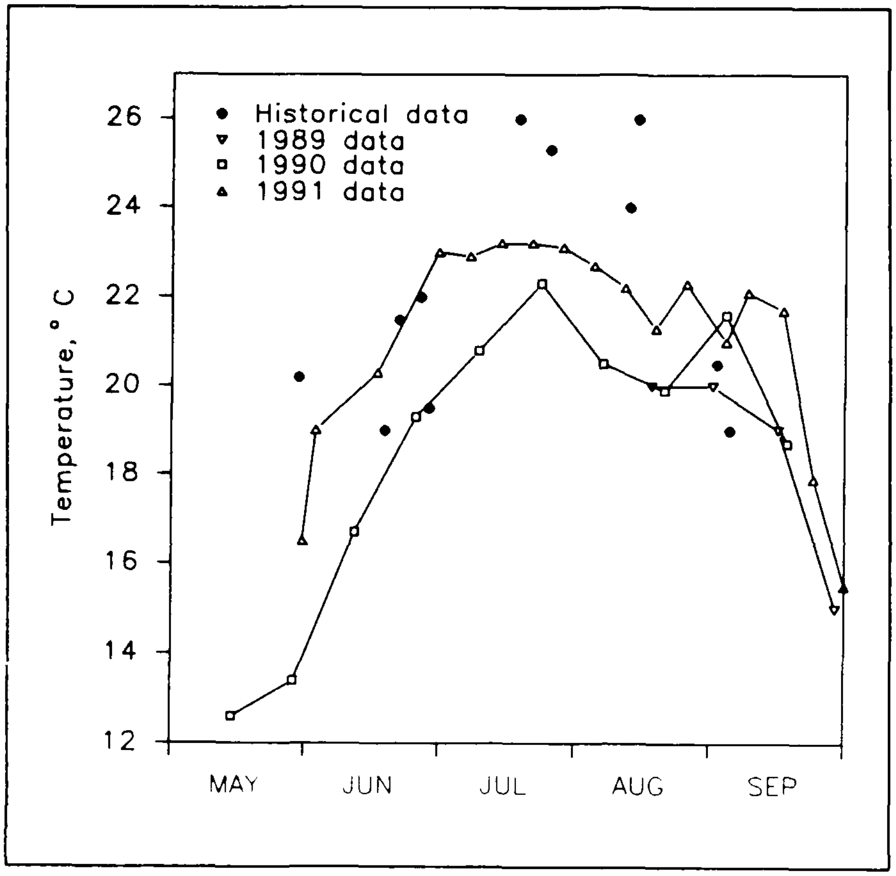

Figure 22. Discharge temperatures, 1988-1991

Table 1
Mixing System Operation at East Sidney Lake

Julian Date	Calendar Date	Action		
		Compressor Operation	On	Off
208	27 Jul 89	23 hr/day	2400 hr	2300 hr
215	3 Aug 89	Off		
122	2 May 90	Thermistor chain installed		
124	4 May 90	Weather station installed		
141	21 May 90	23 hr/day	2400 hr	2300 hr
166	15 Jun 90	12 hr/day	0800 hr	2000 hr
207	26 Jul 90	18 hr/day	0600 hr	2400 hr
248	5 Sep 90	23 hr/day	0100 hr	2400 hr
275	2 Oct 90	Off		
148	28 May 91	23 hr/day	1200 hr	1100 hr
344	10 Oct 91	Off		

Table 2
Summer Mean Concentrations (mg L^{-1}) and Variability for Inflows at East Sidney Lake

Handsome Brook				Ouleout Creek			
Duncan Group	Mean	n	Year	Duncan Group	Mean	n	Year
Dependent Variable: Total Phosphorus							
Coefficient of Variation (CV) = 92%				CV = 131%			
A	0.027	14	90	A	0.035	20	91
A	0.019	20	91	A	0.028	14	90
A	0.016	13	89	A	0.022	13	89
A	0.015	17	88	A	0.020	17	88
Dependent Variable: Total Nitrogen							
CV = 34%				CV = 23%			
A	0.51	20	91	A	1.20	20	91
AB	0.49	13	89	A	1.07	13	89
BC	0.39	14	90	B	0.82	17	88
C	0.37	17	88	B	0.80	14	90
Dependent Variable: Total Iron							
CV = 149%				CV = 109%			
A	0.26	14	90	A	0.28	18	91
A	0.20	11	89	A	0.26	14	90
A	0.17	18	91	A	0.17	12	89
A	0.09	17	88	A	0.16	17	88

Table 3
Total Phosphorus Loading During Summer Growing Seasons at East Sidney Lake

	1988	1989	1990	1991
Load, kg				
Ouleout Creek	60.6 (0.07)[1]	658.3 (0.08)	502.1 (0.28)	162.7 (0.09)
Handsome Brook	19.9 (0.03)	165.2 (0.11)	121.9 (0.28)	58.9 (0.25)
(P)in	80.5	823.5	624.0	221.6
(P)out	210.4 (0.07)	1,573.2 (0.22)	1,099.8 (0.29)	325.1 (0.06)
(P)chng	59.5	79.5	107.6	125.0
(P)int	189.4	829.2	583.4	73.3
(P)int % of total	70	50	48	25
Loading Rate, mg m^{-2} day^{-1}				
Internal	2.98	8.06	5.45	0.69
External	1.26	7.94	5.83	2.10

[1] Numbers in parentheses denote coefficient of variation.

Table 4
Mean Concentration (mg L⁻¹) and Variability for Summer Discharge at East Sidney Lake

Duncan Grouping	Mean	n	Year
CV = 52% Dependent Variable. Total Phosphorus			
A	0.043	13	89
A	0.041	14	90
A	0.041	17	88
A	0.035	20	91
CV = 26% Dependent Variable Total Nitrogen			
A	0.89	13	91
AB	0.86	20	89
B	0.72	14	90
B	0.72	17	88
CV = 52% Dependent Variable Total Iron			
A	0.74	13	89
A	0.67	14	90
A	0.62	17	88
A	0.54	18	91
CV = 77% Dependent Variable Total Manganese			
A	0.28	13	91
A	0.28	18	88
A	0.18	17	89
A	0.15	14	90

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13. ABSTRACT *(Maximum 200 words)*

Artificial destratification has been applied at East Sidney Lake to enhance water quality of the reservoir and tailwater. Operation of the destratification system during the summer growing season maintained near-isothermal temperatures in the reservoir, thereby maintaining oxygenated conditions in the bottom waters via atmospheric exchange during mixing. Maintenance of aerobic bottom waters decreased solubilization of sedimentary metals, decreased internal loading of sedimentary phosphorus, increased available habitat for aquatic organisms, and improved discharge quality. Populations of algal species attained excessive levels in late summer, even with decreased internal phosphorus loading and disruption of thermal gradients. Consequently, improvements in water clarity were not observed.

14. SUBJECT TERMS

Algal control	Internal loading
Artificial destratification	Reservoirs
Eutrophication	Water quality enhancement

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