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LAKE TWELVE QUALITY, NUTRIENT LOADING AND MANAGEMENT

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Water Resources Series
Technical Report No.135
April 1993

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Phase I Diagnostic Study of Lake Twelve,
for King County Surface Water Management Division;
funded in part by the Washington State Department of Ecology
Centennial Clean Water Grants with Additional Contributions from
Lake Twelve Association and Pacific Coast Coal Company

April, 1993

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Executive Summary

Lake Twelve was studied during 1991-1992 in order to determine its quality level (or trophic state) in terms of nutrient content, amount of algae, water transparency and the abundance of rooted macrophytic plants and the causes for that quality level. There was concern that runoff from a constructed noise berm and leachate from septic tank drainfields may be adversely affecting the lake's quality. Most of the lake's 161 ha watershed (including the 17.5 ha lake) is in some stage of forest growback (86%), but a portion (23 ha) of that has been recently rezoned to one dwelling per 2 ha. Runoff from developed land could pose a threat to lake quality in the future.

Total phosphorus (TP) concentrations in the lake were quite low, averaging 6.3 μg/L during the summer, indicating a high quality level or oligotrophy (= "poor nutrition"). An average summer transparency of 3.6 m also indicated a similar quality level. The amount of algae as chlorophyll a (summer mean = 7.3 μg/L), however, indicated a slightly lower quality level (mesotrophy = "moderate nutrition"), although some of that "excess" chl a (high chl a:TP) may have resulted from compensation by the algal cells for poor light penetration in the tea-colored (humics) water. That contention, however, was not supported by other algal cell indicators. For example, chl a:C was low, indicating nutrient limitation.

Summer blooms of algae in the surface water did not occur and concentrations of chl a and TP in the hypolimnion (6 m) during the summer of 1991 were not as high as in 1989. Moreover, Dinobryon, an alga found in nutrient poor waters, was the dominant genera in 1991, and potentially nuisance genera (Oscillatoria and Gonyostimun) that occurred in high concentrations in the hypolimnion in 1989 did not occur in 1991. Also, surface water TP in 1991 was lower than in past years. Thus, there is apparently considerable variation from year-to-year in hypolimnetic constituents that could influence quality of the surface water. The causes for that variation, and whether or not the surface water is affected, are probably dependent on wind conditions. P availability can be dependent on wind in shallow lakes with depth being just sufficient to stratify.

Neither groundwater nor runoff from the noise berm via the PCCC ponds were significant sources of TP, together, contributing less than an estimated 3% of the total loading. Also, there was surprisingly no internal loading of P, although 1991 was probably an unusual year (hypolimnetic TP was much higher in 1989). Nevertheless, water collected from some wells did contain high TP (maximum = 172 µg/L, mean = 75 µg/L) and Cl was several times higher than in the lake. The high TP and Cl were probably due to leachate from septic tank drainfields, but loading to the lake was low because the groundwater contribution to the water budget was estimated to be small (0.2%).

Sediment core analyses show that the sedimentation rate in the lake is low (~ 2 mm/y) compared to other area lakes (3-5 mm/y). Moreover, the sediment organic content is high (40-50%) and rather constant with depth, showing no indication of external inputs of inorganic sediment in recent years. Sediment load and organic content and radiolead dating all suggest that 30 cm of sediment have deposited since about 1880, which is about when deforestation activity and the ASARCO smelter began. The current picture from the sediments is consistent with a lake in a relatively undeveloped watershed having a relatively low water (and sediment) input.

Nearly all the P entering the lake (89%) was estimated to come from surface runoff from the watershed and the quantity was typical of undeveloped forested watersheds. Even if 16% of the watershed, rezoned for five-acre (= 1 per 2 ha) parcels, were developed, the resulting increase in TP yield with runoff (from 31 to 36 $mg/m^2 \cdot y$) would be expected to increase lake TP by only 1.3 µg/L. That small increase would not noticeably affect lake quality. However, development sufficient to double the TP yield would raise lake concentration to about 16 µg/L and likely degrade lake quality.

The quality problem of concern to lake users currently is the amount and kind of rooted aquatic plants. The exotic Eurasian water milfoil occurs in the lake albeit at a low level relative to other lakes in the Puget Sound area. The average, area-weighted mean dry-weight biomass of all species was 63 g/m^2, determined in August 1991. Maximum and mean milfoil biomass was 73 and 25 g/m^2, respectively. Macrophyte abundance in other lakes commonly exceeds 200 g/m^2 and milfoil biomass in Green Lake, Seattle has increased from undetectable levels to nearly 500 g/m^2 in ten years. According to observations and coverage estimates by Metro, milfoil has apparently existed at about the same level in Lake Twelve since the mid 1970s. Nevertheless, bioassays showed that Lake Twelve sediments are equally as conducive (nutrition, texture, etc.) to milfoil growth as are Union Bay (Lake Washington) and Green Lake sediments.

There are in-lake techniques to control P release from sediments, but because internal loading was determined not to have been significant in 1991, treatment of that source is not recommended. If it does become a problem in the future, alum or hypolimnetic aeration would be likely choices.

As for macrophytes (especially milfoil) and the filling process accelerated by their increase, the most effective technique to "turn the lake's clock back" is by dredging. Deepening the lake area between depths of 1-2 m and 4 m, the average depth of visibility and maximum depth of macrophyte colonization, to slightly greater depths would have several beneficial effects: 1) lessen the effect of increased sediment accrual if milfoil increases, 2) increase P retention because water retention time would increase and the hypolimnetic volume and its anoxic area would not increase, and 3) reduce the area where plants reach the surface and interfere with recreational use. The very great disadvantage of dredging is cost. Other techniques are bottom covers, mechanical harvesting, grass carp, water level drawdown and herbicides, none of which would provide such a long-term, ecologically rational solution as would dredging.

The approach to macrophyte control in this and other shallow lakes should probably be selective for species and area treated. Complete removal of macrophytes is not recommended because their presence tends to produce a clear-water condition (low algae). Therefore, a combination of the techniques listed could be applied in a selective manner to reduce the competitive advantage of milfoil and keep intensive-use areas relatively clear of all species.

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INTRODUCTION

Purpose of the Study

During the past decade, residents around Lake Twelve, a small, shallow lake near Black Diamond, Washington, have perceived its water quality deteriorating and an increased macrophyte abundance. Analysis of existing data indicated that runoff from a noise berm, erected between the lake and the site of an open-cut surface coal mine begun by Pacific Coast Coal Company (PCCC) in 1985, had significantly increased phosphorus loading to the lake (Smayda, 1988). Although algal blooms were reported by lake residents, lake surface samples collected by Metro's (Municipality of Metropolitan Seattle) volunteer monitoring program showed a maximum chlorophyll a (chl a) of only 12 µg/L (June, 1988)and summer means well below the eutrophic state threshold of 9 µg/L. Nevertheless, chl a concentrations over 100 µg/L were observed at 6 and 7 m in the summer of 1989, produced primarily by two algae; Gonyostomum a green flagellate, and Oscillatoria, a filamentous blue green, both potential nuisance-bloom formers. These high algal concentrations were probably the result of high phosphorus levels that accumulated in the small-volume hypolimnion. Both organisms are capable of rising through the water column, thereby producing a surface bloom. Such a process was considered to be an explanation for blooms previously reported by residents, with the source of phosphorus being the release from sediments into an anoxic hypolimnion during thermal stratification in the summer. Whether the release of sediment phosphorus was a result of enrichment by runoff from the noise berm, leachate from septic tank drainfields, or other sources around the lake or was simply a natural phenomenon resulting from the morphometric character (small hypolimnetic volume) of the lake, was not determined.

This Phase I study was proposed to resolve the issues of 1) the magnitude of the water quality problem, 2) cause(s) for the problem, and 3) the corrective measures required. The objectives were to: 1) determine the present level of lake quality or trophic state, 2) determine historical changes in lake quality, 3) construct water and phosphorus budgets and calibrate a phosphorus predictive model, and 4) evaluate potential effects of restorative measures for current conditions and future watershed development.

Lake and Watershed Development

Lake Twelve has an area of 17.5 ha (43 acres), an average depth of 3 m (10 ft), a reported maximum depth of 8.5 m (28 ft) and a volume of 598 x 10^3 m^3. The lake's watershed of 161 ha (398 acres) has 86% that is forested, 3% is used for a noise berm (PCCC), and 11% is lake surface (Ellis, personal communication). Within the forested portion, 45% has been harvested within the past 15 years and is in various stages of regrowth, while 41% has not been harvested since the 1920s-1930s. Recently, about 16% (23 ha) of the non-lake watershed was rezoned largely (95%) to the general 5-acre category, i.e. one dwelling per 5 acres (0.5/ha). An increase in development of the lake's forested watershed represents a long-term threat to existing lake quality. Land developed to single family, multiple family, or commercial land use has the potential for an increased rate of surface water runoff, because previously permeable surface areas are replaced by impervious surfaces such as pavement and buildings. Phosphorus and nitrogen content in the runoff from urban/residential land tends to be much greater (Reckow and Chapra, 1983; Omernick 1977) than from natural forested watersheds (47 ± 34 vs. 13 ± 10 µg/L) and may increase the rate of lake eutrophication. Development to the 5-acre category (without livestock) should result in only a small increase in nutrient export, however.

Historical Perspective

Coal was discovered in the Black Diamond area in 1884. The open-cut surface coal mine operated by PCCC lies on an area of 209 ha (516 a) to the west of, but largely out of the immediate Lake Twelve watershed. Forest on the mining site was clearcut in 1983 with actual mining operations beginning in 1985 (Smayda, 1988). In an attempt to minimize the noise impact from the mining operations on lake residents, PCCC constructed a noise berm (Figure 1) on the eastern edge of the mine site in 1986. The berm lies within the lake's watershed, although the mining operation lies outside. Surface water runoff due to precipitation onto the noise berm is collected in two sedimentation ponds (Pond A and Pond A') for treatment (settling) before being discharged to the lake.

Recreational Potential

Lake Twelve has a public access boat launch located on the southeast shore of the lake. The Washington State Department of Wildlife manages the lake as a put-and-take trout fishery through a program of yearly stocking. In 1992 2,352 one pound rainbow trout were stocked approximately 2 weeks before the opening day of fishing season. The lake received extensive fishing pressure on opening day; in 1992 over 100 boats were present on the lake with an average of 2.5 fishermen per boat. During a three-hour period on opening day seventy-six fishermen were polled at the public fishing ramp when leaving the lake. The average catch of those polled was 2.96 trout/person.

As part of the final EIS (1984) for the mining operation, a baseline fishery assessment was conducted by the University of Washington School of Fisheries. The study was conducted on April 5-6, 1983 using gillnets, trap nets, and a 150 foot beach seine. The gillnets collected 27 rainbow trout, 2 cutthroat trout, and 2 yellow perch. The trap nets captured 69 brown bullhead, 38 pumpkin seed, 7 yellow perch, 4 cutthroat trout, and 2 rainbow trout. The beach seine collected 65 pumpkin seed, 2 brown bullhead, 2 cutthroat trout, and 1 rainbow trout. Although the lake is obviously a natural warm-water fishery, cutthroat trout reproduction apparently occurs.

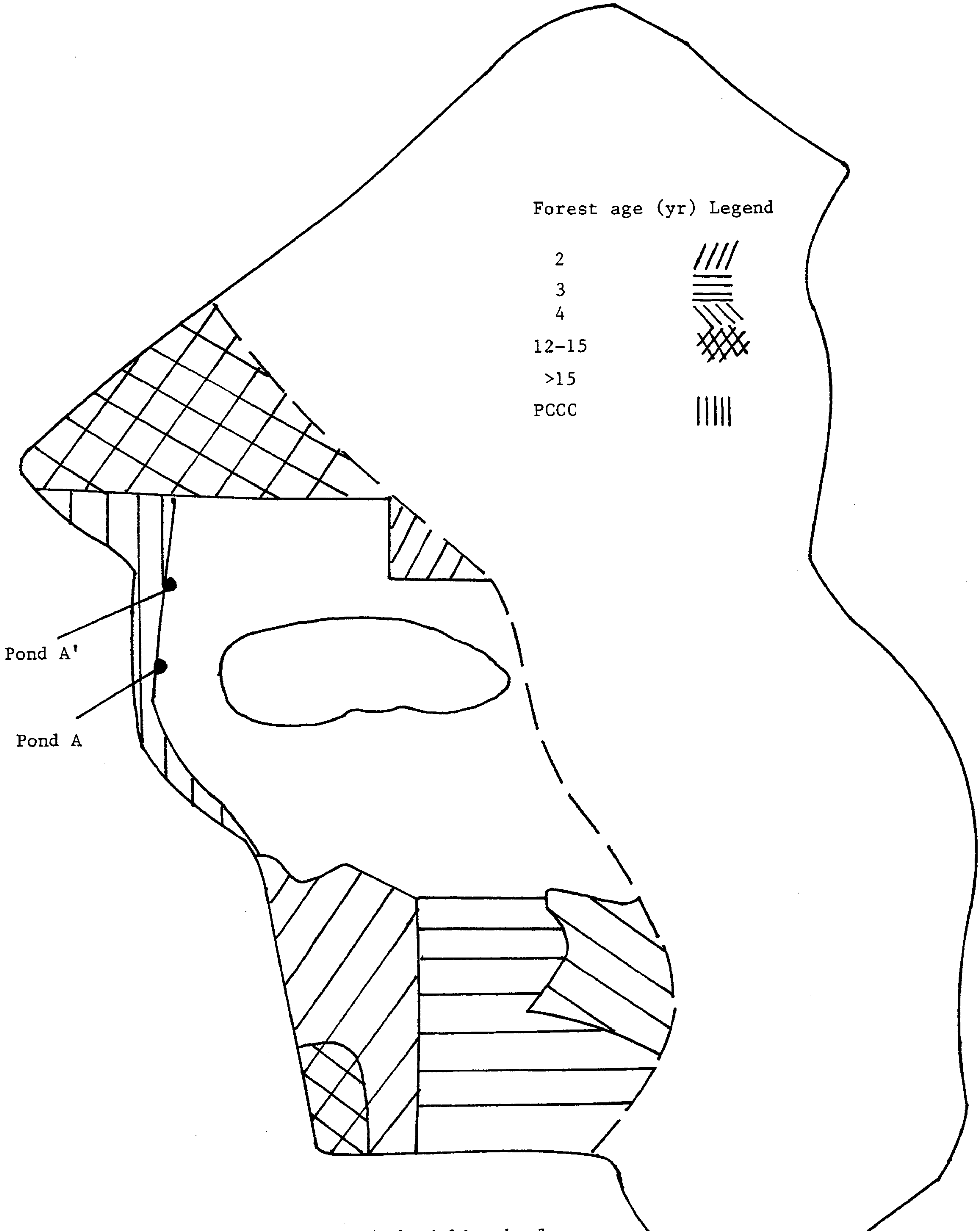


Figure 1. Lake Twelve watershed within the larger catchment that includes the wetland area down gradient from the lake. Forest age for respective areas since the last harvest are shown along with PCCC property and retention ponds (Ellis, personal communication).

Although no designated swimming beach is present at Lake Twelve, there are approximately 12 floating platforms for swimming in the lake and the public boat launch provides swimming access from the shore (Personal communication, Esko Cate, June 26, 1992).

METHODS AND MATERIALS

Field Sampling

Lake Water Quality

Water sampling to define lake quality was conducted on a twice-monthly basis from April, 1991 through October, 1991 and on a monthly basis from November, 1991 through March, 1992. Samples were collected at the deepest point in the lake and at the east end adjacent to the wetland where water exits the lake (Figure 2). The water quality and biological constituents monitored were: total phosphorus (TP), soluble reactive phosphorus (SRP), pH, specific conductance, nitrate+nitrite-nitrogen ($NO_3^- + NO_2^-$-N), ammonium nitrogen (NH_4^+-N), total nitrogen (TN), chlorophyll a (chl a), alkalinity, Secchi disk transparency, dissolved oxygen (DO) and temperature. Water samples were collected for phytoplankton biovolume determination and net hauls for zooplankton abundance (Table 1). Once each quarter, during the study period, samples were collected at two-meter intervals for calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium Na^+), potassium (K^+), sulfate (SO_4^{2-}), chloride (Cl^-), iron (Fe), and total aluminum (Al).

Temperature and dissolved oxygen profiles were determined *in situ* at one-meter intervals using a Yellow Springs Instrument Company (YSI) model 57 oxygen meter with an oxygen-sensitive membrane electrode. The DO oxygen meter was routinely calibrated *in situ* using the azide modification of the Winkler method (APHA, 1989) at random depths in the water column. Water column transparency was measured using a 20 cm diameter black/white Secchi disk.

Water samples for phytoplankton were collected at 2 and 6 m depths with a 2 L Van Dorn bottle and placed in acid-washed 120 mL polyethylene sample bottles and preserved with a 1% Lugol's solution in the field. Zooplankton samples were collected with a 0.5 m diameter plankton net (76 µm mesh), using two vertical net hauls from a

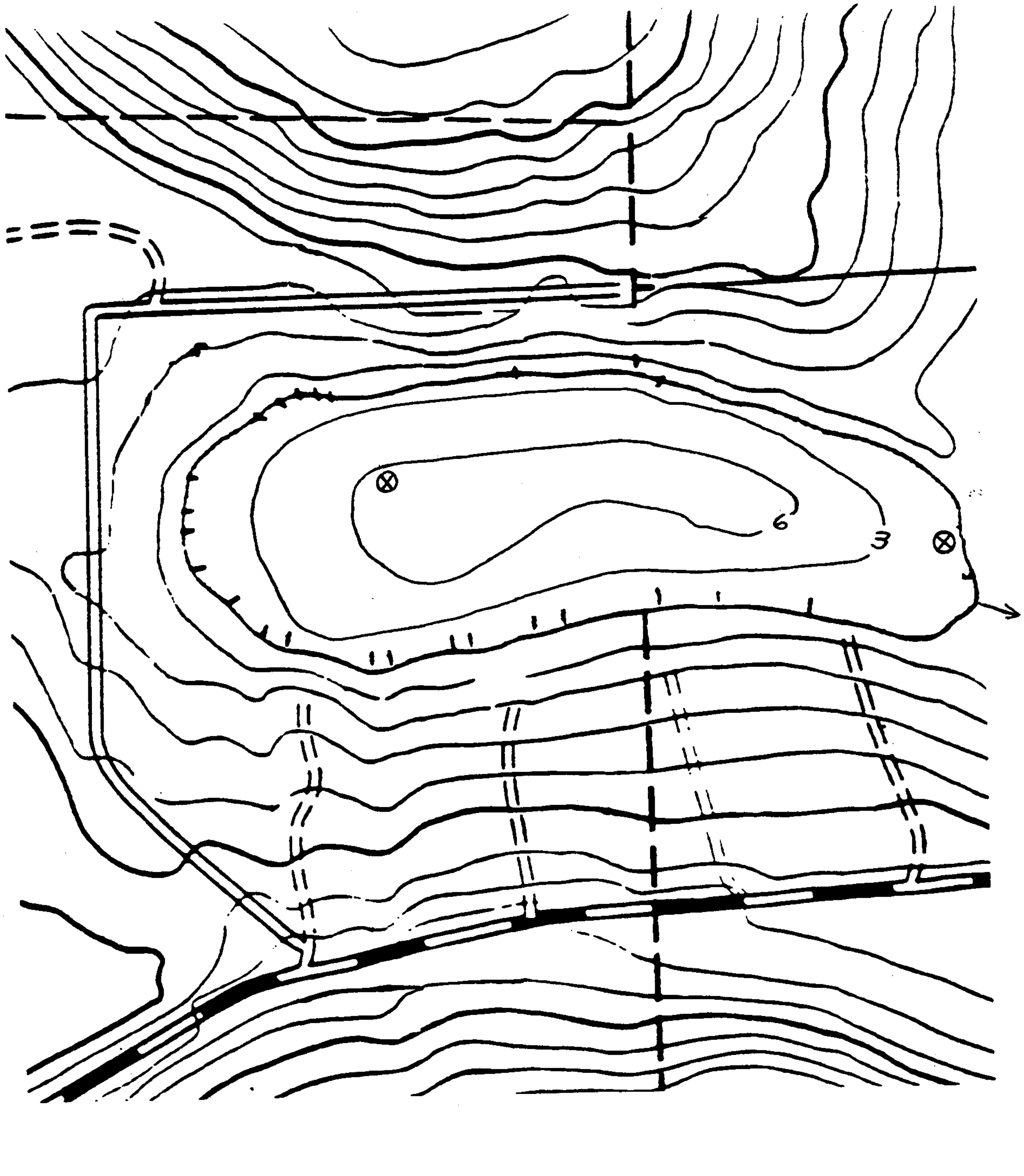


Figure 2. Lake water quality sample collection locations. Contour depths (meters) from Washington Department of Game survey in 1949 (Wolcott, 1961).

depth of 5 m. The samples were transferred to 0.5 L glass bottles and preserved with a 50% propanol solution in the field.

Three bottom grab samples for benthic invertebrate enumeration were collected on October 19, 1991 and March 21, 1992 with a model 196 F10 Wildlife Supply Company 0.25 ft^2 (m^2) Ekman grab sampler. The samples were transferred to 0.5 L glass bottles and preserved with a 50% propanol solution in the field.

Lake water samples were collected at discrete depths of 0.1, 2.0, 4.0 and 6.0 m (and 6.5 m during high lake levels) with a 2 L Van Dorn bottle and placed in acid-washed 2 L polyethylene bottles which were stored in a cooler until return to the laboratory. The pH was determined in the field at each of the depths using a Cole-Palmer pH meter. Subsamples were taken from the 2 L sample to be analyzed for chl a, alkalinity, pH, specific conductance, TP, SRP, TN, NO_3^- + NO_2^--N and NH_4^+-N. The samples for SRP, NO_3^- + NO_2^--N and NH_4^+-N were filtered through pre-soaked 0.45 μm Millipore filters, placed in 120 mL acid-washed polyethylene bottles and frozen until analyzed. The TP samples were placed in 120 mL acid-washed polyethylene bottles and preserved with two drops of 36 N H_2SO_4. Samples for TN were placed in 120 mL acid-washed polyethylene bottles and frozen until analyzed. Samples for chl a were filtered onto a 47 mm glass fiber filter with 2 drops of $MgCO_3$. The filters were then stored frozen in a darkened desiccator until analyzed.

The samples for Ca^{2+}, Mg^{2+}, Na^+, K^+, Fe, and Al were placed in 120 mL acid-washed polyethylene bottles and preserved with 36 N H_2SO_4. The samples for SO_4^{2-}, and Cl^- were filtered through pre-soaked 0.45 μm Millipore filters, placed in 120 mL acid-washed polyethylene bottles and frozen until analyzed.

Surface Water Inflow

Sampling inflow for water quality was conducted on a twice-monthly basis from April through October, 1991 and on a monthly basis from November, 1991 through

March, 1992 at the two primary surface inflows (008 and 010 are the same stream) to the lake (Figure 3). Measurements for DO and pH were made *in situ* in the culverts where the streams pass under the lake perimeter road in the same manner as in the lake. The water samples were collected by submersing acid-washed 2 L polyethylene bottles. which were then stored in a cooler until return to the laboratory. Subsamples were taken from the 2 L primary sample to be analyzed for alkalinity, TP, specific conductance, SRP, TN, $NO_3^- + NO_2^-$-N , and NH_4^+-N (Table 1). The samples for individual constituents were preserved for analysis in the same manner as the lake-water samples.

Storm Water

Two storm events were sampled by a lake-side resident during the study period; the first on January 30 and the second on March 17-18, 1992. The January 30 storm event was sampled for 10 hours at 1-hour intervals. The March 17-18 storm event was sampled for 12 hours at 2-hour intervals. Five stations were sampled during the storms and are shown in Figure 4. Grab samples were collected in 120 mL acid-washed polyethylene bottles which were frozen immediately after collection and later returned to the laboratory for analysis. A composite storm sample was constructed for each sample location by flow-volume weighting and then analyzed for TP, SRP, and TN.

Ground Water

A set of ground water monitoring wells (MW) was installed on September 23-25, 1991, along the lake shoreline and in the shallow littoral zone of the lake (Figure 5) to assess the movement and quality of ground water entering the lake (WP). Well placement criteria according to Dunn (1991) included:

1. Year around use of household septic systems
2. Location of septic systems in relation to the lake
3. Feasibility of the drilling-rig access

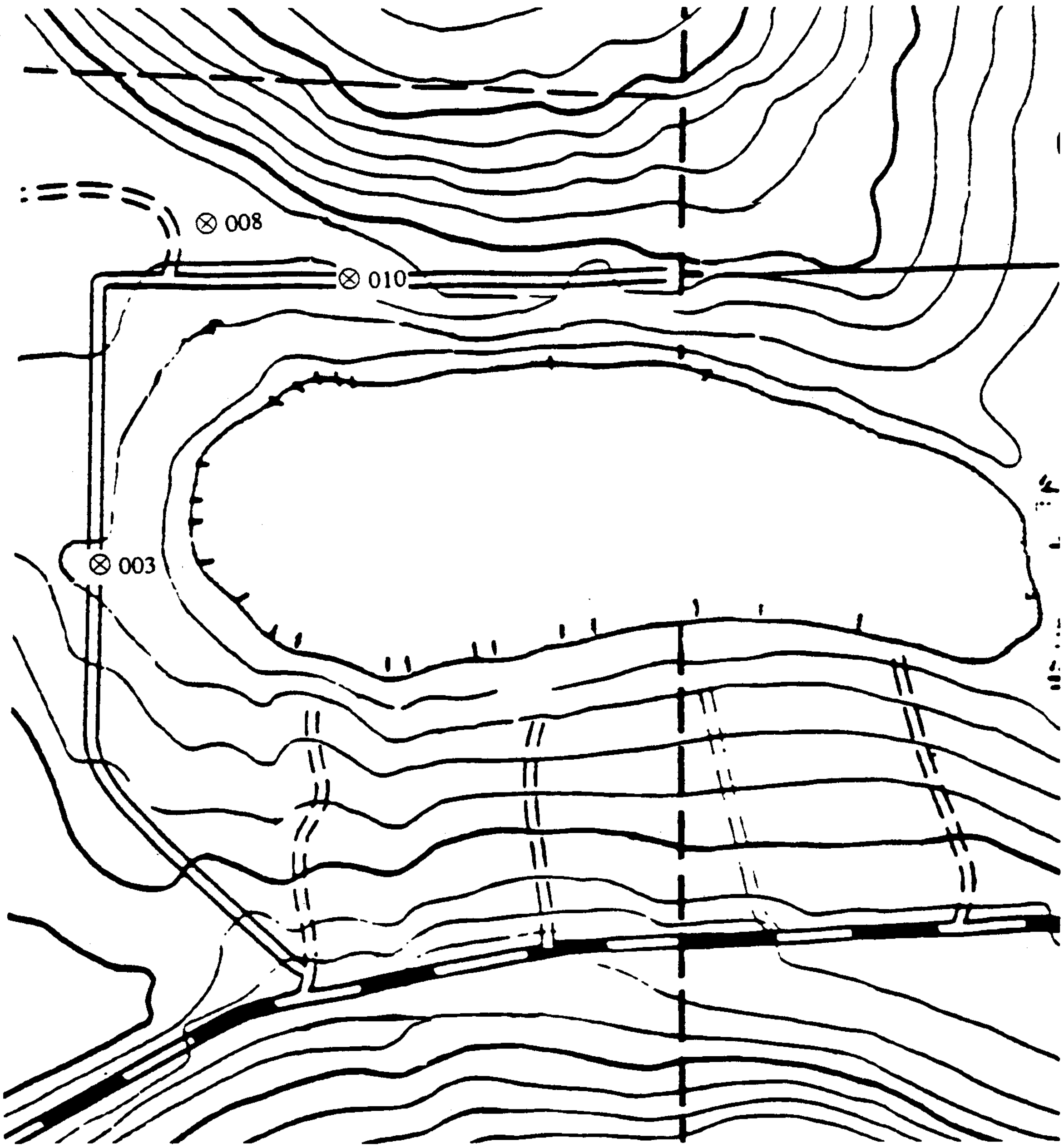


Figure 3. Surface water inflow sample locations.

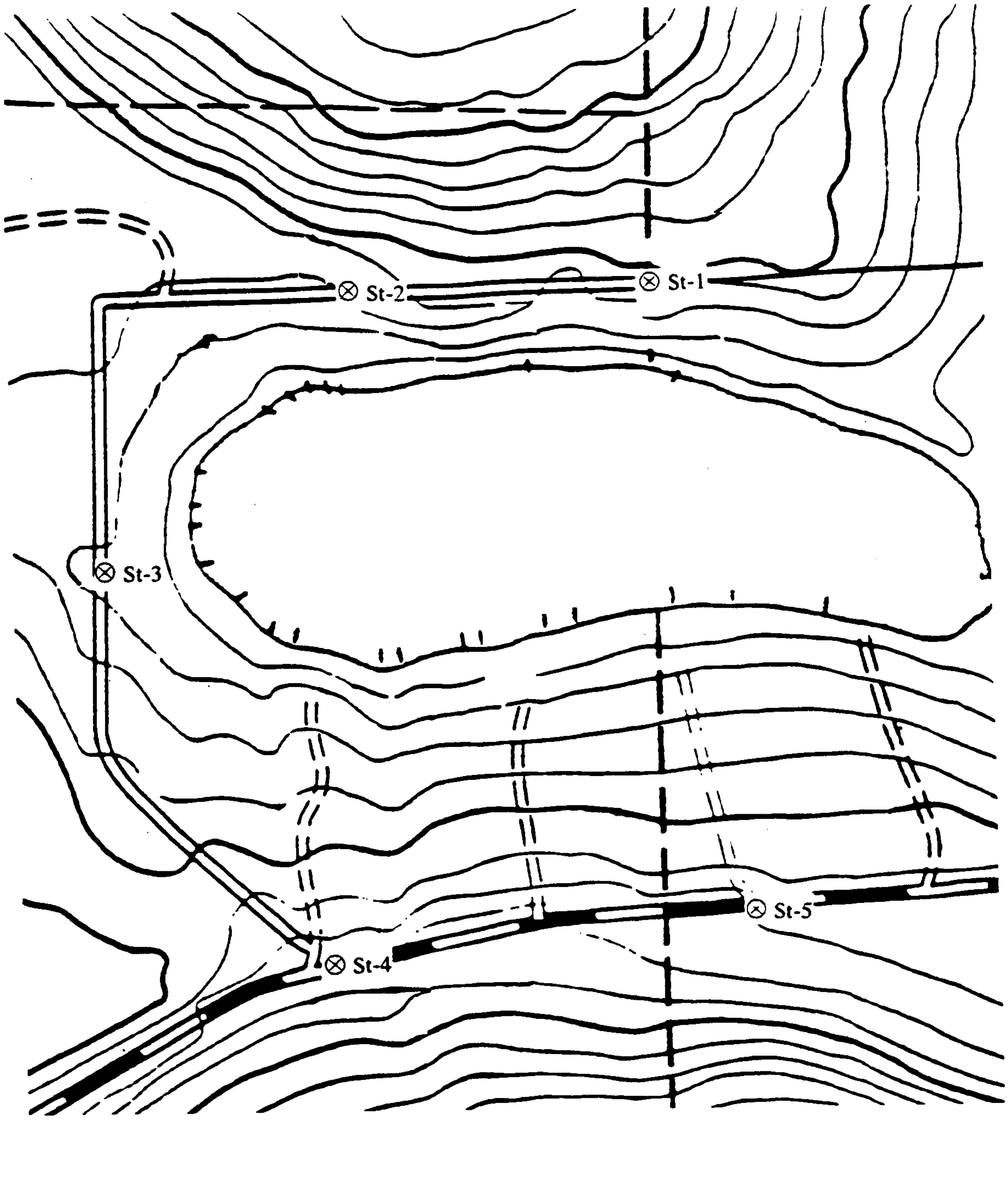


Figure 4. Storm water sample locations.

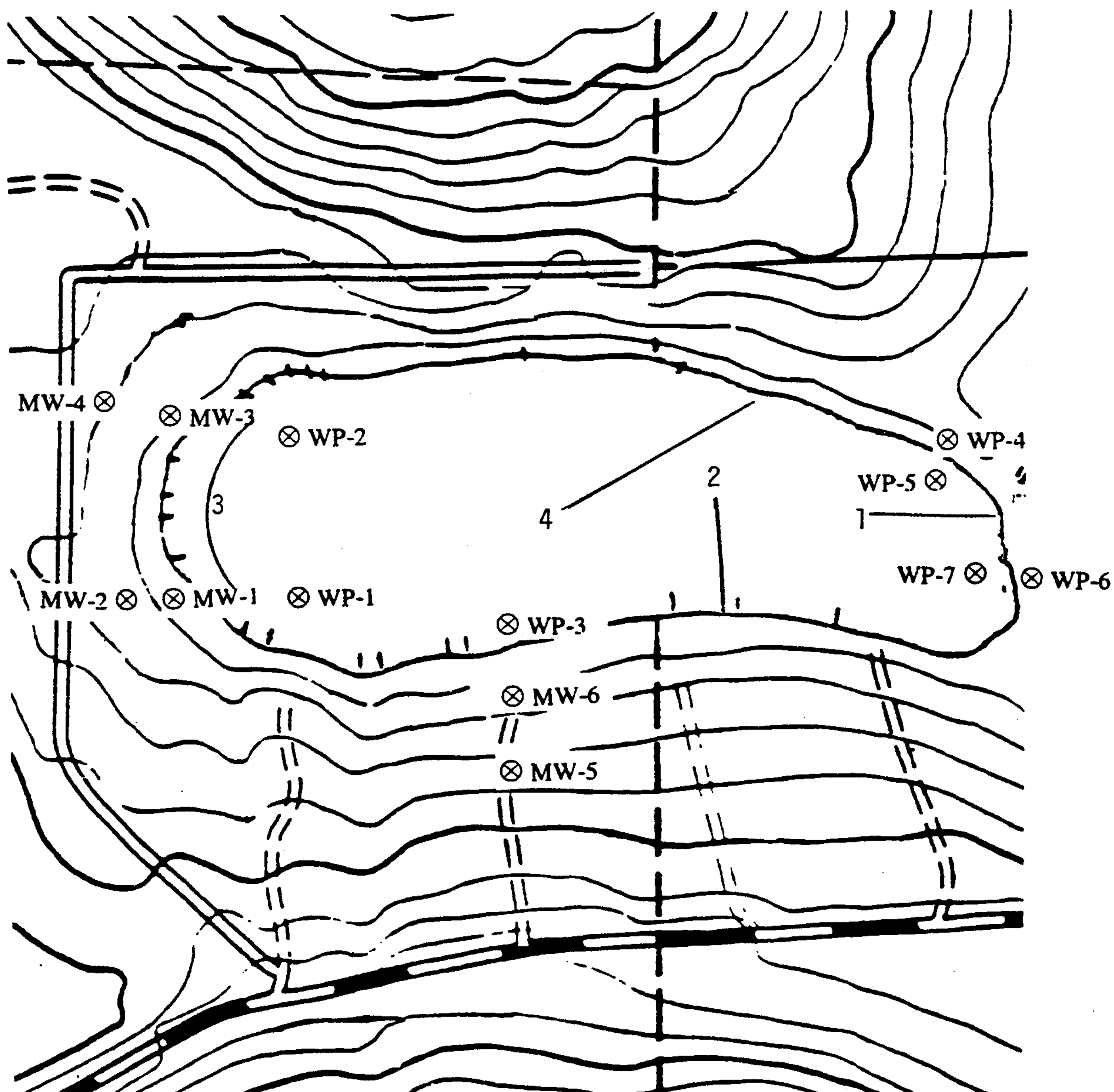


Figure 5. Ground water well locations and transects for macrophyte sampling (1-4).

4. Permission of the landowner

Water levels and water quality samples were collected from the wells on a monthly basis from October, 1991 through March, 1992. Water levels were measured using a 12-foot Stanley tape. After the water level was measured, a peristaltic pump, or dedicated PVC well bailer, was used to evacuate the wells (wells that were very slow to recover), or remove a minimum of three pore volumes prior to collecting a water quality sample. The samples were placed in acid-washed 1-L polyethylene sample bottles which were stored in a cooler until return to the laboratory. Samples for Cl, TP, TN, and specific conductance were taken from the 1-L bottle upon return to the laboratory. Samples were prepared for analysis or storage in the same manner as lake water quality samples.

Precipitation

Precipitation was monitored on a daily basis by the PCCC as part of their NPDES waste discharge permit. In addition, precipitation was measured and samples collected for analysis by two lakeside residents from November, 1991 through March, 1992. A composite precipitation sample was analyzed at both collection sites for TP, NO_3^- + NO_2^--N, and TN.

Macrophytes

Forty random samples for macrophyte analysis were collected on August 16, 1991. Four separate transects, as shown in Figure 5, were established and sampled to characterize the population abundance of the major species present.

Samples were collected using one half of a steel drum (area 0.255 m^2), which was lowered over the side of the boat at the sample stations. A diver using SCUBA removed the macrophytes from the unit area, enclosed them in an attached netting, and brought them to the surface where they were placed in plastic sample bags and stored in a cooler.

After returning to the laboratory, the macrophyte samples were washed, separated by species and dried for approximately 24 hours at 80° C. The dried macrophytes were then weighed using a Mettler P1200 balance. The data are presented on a mass dry weight/area (g/m^2) basis in Appendix 4.

Sediment Core Collection

Three sediment cores were collected with a piston-type corer from the deepest point of the lake on September 24,. 1991. The top 50 cm of each core was sectioned at 1-cm intervals and weighed after sectioning. The 1-cm sections were then dried at 105° C for 24 hours and reweighed to determine % water content. The cores were subsequently analyzed for TP, total organic carbon, stable lead, radioactive lead-210, and total aluminum, zinc, and iron.

Benthic Invertebrates

Benthic macroinvertebrates were collected with an Ekman grab sampler at three different sites on October 19, 1991 and March 21, 1992. Samples were collected on two occasions to insure that populations were not depleted due to emergence. Sampling depth is the principal determinant of benthic populations, so a range of depths were selected: 4, 6 and 2 m on October 19 and 1, 1 and 2 m on March 21 for samples 1, 2 and 3, respectively (Appendix 10). Samples were preserved in ethanol and enumerated later in the laboratory with the aid of a dissecting scope.

Table 1. Water quality and biological constituents sampled routinely at eight depths at the mid-lake location, the lake near the outflow and two inflows.

Constituent	0 m	1 m	2 m	3 m	4 m	5 m	6 m	6.5 m	Out flow	Pond A	Pond A'
Temperature	X	X	X	X	X	X	X	X	X	X	X
Dissolved Oxygen	X	X	X	X	X	X	X	X	X	X	X
pH	X		X		X		X	X	X	X	X
Total Phosphorus	X		X		X		X	X	X	X	X
Soluble Reactive Phosphorus	X		X		X		X	X	X	X	X
Total Nitrogen	X						X				
Nitrate and Nitrite Nitrogen	X		X		X		X	X	X	X	X
Ammonium Nitrogen	X						X		X	X	X
Specific Conductance	X		X		X		X	X	X	X	X
Alkalinity	X		X		X		X	X	X	X	X
Chlorophyll *a*	X		X		X		X	X			
Phytoplankton			X				X				
Zooplankton (net haul)						X					

Sample Analysis

A summary of the methods used to analyze for water quality constituents are presented in Table 2.

SRP was determined with the ascorbic acid, molybdenum blue method according to Standard Methods (APHA, 1989). TP was measured by analyzing for SRP in unfiltered water samples after persulfate digestion. The absorbance for P analysis was determined on a Milton Roy Co. Spectronic 1001 spectrophotometer at 885 nm in a 10 cm cell.

TN was determined with a persulfate digestion in a basic solution, followed by analysis in an ALPKEM RFA-300 continuous flow autoanalyzer according to Standard Methods (APHA, 1989). NH_4^+-N was analyzed with indophenol blue color formation produced by the automated phenate method (APHA, 1989), and absorbance read at 630 nm on the ALPKEM RFA-300 continuous flow autoanalyzer. Nitrate+nitrite-N was determined as nitrite after cadmium reduction and absorbance was read at 520 nm on the ALPKEM RFA-300 continuous flow autoanalyzer (APHA, 1989).

Analytical accuracy for sample analysis of nutrients is shown in Table 3 in terms of % recovery. Except for TN, % recoveries were very near 100%. Expressed as % error, the mean values were ± 7% for TP and NH_4, and ± 3% for SRP and NO_3. The error for TN was ± 14%. Precision error, which includes both sampling and analytical variability and is based on replicate samples (n = 4-11) collected throughout the study period, averaged ± 16%, for TP, ± 10% for SRP, ± 31 for NO_3^- + NO_2^-, ± 26% for NH_4^+-N, ± 4% for TN and ± 19% for chl a.

Alkalinity expressed as $CaCO_3$ was determined by titrating 100 mL of sample to pH 4.6 with a standardized 0.01 N H_2SO_4 solution (APHA, 1989). Potentiometric titrations were performed with a Corning glass electrode calibrated at a pH of 4 and 7. Specific conductance was determined in 100 mL samples with a model PM-70CB Sybron/Barnstead conductivity bridge with a cell constant of 0.1.

Ca^{2+}, Mg^{2+} Na^+, K^+, Fe, and TAl samples were analyzed on a Jovin-Yvon Model JY-50 inductively-coupled argon plasma (ICP) emission spectrophotometer (APHA, 1989). The SO_4^{2-} and Cl^- samples were analyzed on a Dionex QIC ion exchange chromatograph (APHA, 1989). Precision error for these constituents ranged from zero for TAl to ± 10.5% for Fe, while % recovery ranged from 90% for Ca to 99% for K.

Zooplankton samples were stained with Iosin Y, and concentrated by filtering (100 μm mesh) prior to counting. The samples were then brought up to 500 or 100 mL with tap water depending on the amount of zooplankton present in the sample. Two 5-mL subsamples were then placed in a counting chamber and counted using a Nikon binocular dissecting microscope. The zooplankton were identified to genera and enumerated as No./L.

A 30 mL subsample from the phytoplankton samples was centrifuged for 20 minutes and aspirated to concentrate the samples ten fold prior to counting. The sample was then mixed and aliquots placed in a 0.1 mL Palmer Maloney counting cell, with 50 grids counted per sample. The phytoplankton were identified to genera, enumerated, and biovolume determined as mm^3/L.

Chlorophyll a was determined by grinding the samples in a $MgCO_3$-saturated, 90% acetone solution. Following a 24-hour, 90%-acetone extraction, absorbance was read at 665 and 750 nm on a Perkin-Elmer Model 3 scanning spectrophotometer (APHA, 1989).

Table 2. Analytical methods for physical and chemical water quality constituents.

Constituent	Method
Soluble reactive phosphorus (SRP)	Molybdate blue ascorbic acid
Total phosphorus (TP)	Persulfate digestion/molybdate blue ascorbic acid
Total nitrogen (TN)	Persulfate digestion
Ammonium nitrogen (NH_4^+)	Indophenol blue color formation
Nitrate+nitrite-nitrogen (NO_3^-+NO_2^-)	Cadmium reduction
Alkalinity	Potentiometric titration
Specific Conductance	Conductivity bridge
pH	Corning probe
Dissolved oxygen	Winkler azide modification
Transparency	Secchi disk
Chlorophyll *a*	Acetone extraction
Phytoplankton	Direct count
Zooplankton	Direct count

Table 3. Quality assurance results using EPA standards.

Total P

Date	True Value	Measured	% Recovery
9/23/91	3.75	4.2	112.0
	18.75	21.3	113.6
3/23/92	18.75	18.7	99.7
	75	74.9	99.9
		mean	106.3

SRP

Date	True Value	Measured	% Recovery
9/23/91	5	5.4	108.0
	25	25.5	102.0
3/23/92	6.25	3.1	97.6
	25	24.5	98.0
		mean	101.4

NO3

Date	True Value	Measured	% Recovery
11/8/91	19.8	20.8	105.1
	198	201.7	101.9
2/12/92	20	18.9	94.5
	200	191.7	95.9
3/19/92	25	25.5	102.0
	100	98.8	98.8
		mean	99.7

NH4

Date	True Value	Measured	% Recovery
7/20/91	20	17.9	89.5
	100	108.9	108.9
2/12/92	20	18.9	94.5
	200	191.7	95.9
		mean	97.2

TN

Date	True Value	Measured	% Recovery
2/12/92			
Inorganic & Organic	250	195.7	78.3
Organic	100	106.9	106.9
Organic	250	221.1	88.4
Total Soluble	400	341.6	85.4
		mean	89.8

Hydrologic Budget

The hydrologic budget was constructed as a mass balance of inflows and outflows to/from the lake. The budget equation that defines the continuity of inputs and outputs is as follows:

$$\mathbf{DP + SI + GW + \Delta ST = EV + OW}$$

where DP = Direct precipitation on the lake

SI = Surface inflows to the lake

GW = Ground water inflow to the lake

ΔST = Change in lake storage

EV = Evaporation from the lake

OW = Surface outflow from the lake to the wetland

Direct Precipitation

Precipitation was measured on a daily basis locally by PCCC personnel as part of their NPDES waste discharge permit. The direct precipitation component of the hydrologic budget was calculated by multiplying the daily precipitation values by the surface area (17.5 ha) of the lake.

Surface Inflows

The surface water inflows to the lake were divided into three separate components. The first component represents the surface inflows from PCCC's sedimentation ponds. The surface inflows from sedimentation Ponds A (003) and A' (008) were measured on a daily basis throughout the study period by PCCC personnel as part of their NPDES waste discharge permit. The numbers 003 and 008 are NPDES designations.

The second component of the surface inflows is a composite flow made up of discharge from pond A' and a portion of the natural watershed to the north of the lake. This inflow is referred to as 010 on PCCC's NPDES waste discharge permit. A linear regression equation was developed from daily measurements reported by PCCC personnel for the discharge from sedimentation Pond A' to predict the flow at location 010. The regression equation was developed from daily flow data (January 14-February 12, 1992) collected by PCCC personnel at locations 008 and 010 for flows at 008 greater than or equal to 0.01 cfs and took the form:

$$\text{FLOW composite 010 (cfs)} = 1.33 + 23.2 * \text{FLOW 008 (cfs)}, (r^2 = 0.86).$$

The inflows used for location 010 (north watershed) in the hydrologic budget were those predicted by the above equation minus the portion of the flow originating at sedimentation pond A' as reported by PCCC.

Ungauged inflow/outflow represents the third component of the surface inflows and was calculated as the residual of the known hydrologic components. A positive value of ungauged inflow/outflow was taken as ungauged inflow and a negative value as ungauged outflow. This portion of the hydrologic budget consisted largely of storm generated flow that originated as overland flow in the forested portion of the watershed outside of the lake perimeter roads. When this overland flow reached the lake perimeter road, it was channeled into storm culverts that pass under the roads and into the lake. The ungauged inflow was estimated with the following equation:

$$\mathbf{UI = EV + OW - (DP + GSI + GW + \Delta ST)}$$

where UI = Ungauged inflow to the lake

GSI = Gauged surface inflows to the lake

Other symbol definitions as before

Ground Water

The ground water component of the hydrologic budget for the period October, 1991 through March, 1992 was calculated using the water level data from the six monitoring wells and the driven well point (WP-6) located in the wetland. Because the ground water wells were not installed until September, 1991, the ground water component of the hydrologic budget for the period from April, 1991 through September, 1991 was estimated based on the actual calculated values from the later period. These ground water estimates were based on the observed rainfall data. The months of July, August, September, and October all received less than 2.5 inches of rainfall, and were assigned the calculated ground water flow value for October. The months of April, May, and June received more than 2.5 inches of rain and were assigned ground water inflow values equal to the average of the calculated winter values.

The hydraulic conductivity, K, expressed in m/sec, was estimated *in situ* using a variable head slug test (Lambe, 1979). The specific discharge v (m/sec) was calculated at each set of monitoring wells and at WP-6 in the wetland using Darcy's law (Freeze, 1979), which may be expressed as:

$$v = -K\left(\frac{dH}{dL}\right)$$

where H is the hydraulic head (m), L is the distance between the wells (m), and (dH/dL) is the hydraulic gradient. The ground water discharge was calculated on a monthly basis for the western, southern, and eastern shorelines. Because a well network was not installed on the north side of the lake, ground water flows were estimated to be equal to those calculated for the south shore.

The ground water flow was calculated using another form of Darcy's law (Freeze, 1979)

$$Q = v*A$$

where Q is flow in m^3/sec and A is the area through which flow occurs. That area was estimated as the depth of the aquifer at the monitoring wells times the length of the lake shore adjacent to the wells. The west shore was estimated to be 288 m, which was split evenly for the flow calculations between the MW-3/MW-4 and MW-1/MW-2 well sets (Figure 5). The south shore is approximately 768 m and was used for the MW-5/MW-6 well set. The depth of ground water flow from the wetland was estimated by Dunn (personal communication) to be 4.6 m at the center, sloping to 1.5 m at the outer edges. The length of the east shore, adjoining the wetland, was estimated to be 124 m.

Change in Storage

The change in lake storage was calculated on a monthly basis. Lake level measurements were taken from a staff gauge located at Esko Cate's residence (south shore) at approximately two-week intervals, beginning in June, 1991 and continuing through the end of the study period. The lake level for the April/May 1991 period was estimated based on the actual measurements during the subsequent period. The change in lake storage was calculated by multiplying the change in lake level by the lake surface area (17.5 ha), which was assumed to remain constant independent of the lake level.

Evaporation

Class A pan evaporation data from the Seattle Maple Leaf Reservoir was used to calculate evaporative losses from the lake during the study period (Phillips, 1968). A pan-to-lake evaporation correction factor of 0.75 was used to adjust the Class A pan evaporative losses to that of a shallow lake (Dunne and Leopold, 1978).

Outflow to Wetland

The primary loss of water is discharge into the wetland that is adjacent to the east end of the lake. Outflow measurements were hampered by a poorly delineated channel and low velocities. Nevertheless, the flow into the wetland was estimated using a stage-discharge relationship that predicted the surface outflow based on the measured lake level and only three flow measurements. Surface flow discharge measurements to the wetland were made using a wading rod and either a Pygmy Price or a Marsh-McBirney velocity meter. Each of the flow measurements taken corresponded to a different lake-level stage, including a no-flow condition at a lake level of 20 in, which is considered to be the threshold no-flow lake level. The regression equation for the stage-discharge relationship for three data points is:

$$\textbf{Discharge (cfs)} = -8.3 + 0.4^{*}\ \textbf{Lake level (in)} \qquad (r^2 = 0.88)$$

Phosphorus Budget

The P budget was calculated by multiplying the TP concentrations determined in each of the sampling intervals by their respective flows from the hydrologic budget. A mass balance approach was then used to calculate the only unknown, net sedimentation of TP, on a monthly basis. The mass balance equation is:

$$S = I - O - \Delta TP$$

where	S	=	Net sedimentation of TP, kg
	I	=	Inflow of TP, kg
	O	=	Outflow of TP, kg
	ΔP	=	Change in mass storage of TP in the lake, kg

A positive value of S indicates a net sedimentation of TP, whereas a negative value represents internal loading to the lake from the sediments.

TP loading from precipitation was determined by multiplying the monthly precipitation values reported by PCCC by the average TP concentration in precipitation (19 µg/L). TP loading from PCCC's sedimentation ponds was calculated using the TP concentrations from routine (monthly/twice-monthly) sampling at the perimeter road culverts and means of the respective period flows based on daily flows reported by PCCC.

TP loading from the culvert at location 010 was calculated by multiplying the TP concentrations measured at the culverts by the period flow estimates derived in the hydrologic budget. Loading from ungauged inflow was calculated by multiplying the positive flow estimates determined in the hydrologic budget by the average flow weighted TP concentration (24 µg/L) measured in the two storm events. The TP concentrations from the storm events were used because most ungauged inflows produce runoff only during heavy rains.

TP loading from ground water was calculated using a combination of the measured and estimated flow data from the hydrologic budget and the average TP concentration (75 µg/L) measured in the ground water wells during the study period.

Outflow TP was calculated by multiplying estimated lake discharge values, predicted from the stage-discharge relationship developed in the hydrologic budget, by the TP concentrations measured in the lake near the wetland discharge channel.

The change in lake TP mass was calculated by multiplying the lake volume weighted TP concentrations by the change in lake volume calculated in the hydrologic budget.

Phosphorus Model

The primary source controlling the lake TP concentration was surface water runoff, which occurred primarily during the November through April period. In order to determine the importance of the winter surface water runoff, versus summer internal loading, on the overall trophic state of the lake, a non-steady state mass balance model was calibrated for the 1991-1992 data.

The mass-balance model was initially developed by Vollenweider (1969) and later modified by Larsen et al. (1979) to include internal loading. The model calculates the change in TP concentration with time, which is equal to the external loading plus internal loading minus sedimentation and washout of TP. The model is represented by:

$$\frac{dTP}{dt} = \frac{L_{ext}}{\bar{z}} - \rho TP - \sigma TP + \frac{L_{\text{int}}}{\bar{z}}$$

where $\frac{dTP}{dt}$ = the change in TP concentration with time

L_{ext} = external loading of TP in mg/m^2-week

$\bar{z}$ = the average lake depth in m

ρ = the lake flushing rate in week^{-1}

σ = the sedimentation rate coefficient in week^{-1}

L_{int} = the internal loading in mg/m^2-week

The constraints of the model require that: 1) the lake is completely mixed, 2) the lake level is constant, and 3) water inflow equals water outflow. Although these constraints were not precisely satisfied, the associated errors from assuming compliance are not considered large so the model was nevertheless calibrated to the lake TP content for the study period.

The model was constructed by dividing the hydrologic and phosphorus concentration data into one week time steps in an effort to minimize the time interval variability due to calculation. The model was then calibrated to the whole lake mean TP data by selecting the proper sedimentation rate coefficient, σ, which was formulated as a function of the lakes flushing rate, ρ.

This approach has been successful in the following western Washington lakes: Green Lake (Mesner, 1985), Lake Sammamish (Welch et al., 1986), Long Lake (Kelly, 1987), and Silver Lake (Welch et al., 1988). Verification is not possible with only one year of data, although the approach of calibrating the sedimentation rate coefficient as a function of flushing rate ($\sigma = \rho^x$) was verified for Lake Sammamish.

Sediment Analysis

TP, stable lead, aluminum, zinc and iron were determined by nitric acid digestion of 100 mg of dry subsample from each 1-cm depth interval in a four-step process similar to the procedure used by Borlelson and Lee (1972). After soaking in 5 mL of nitric acid overnight, the subsamples were digested in a block digester at 150°C for about 5-6 hours. The clear, yellowish subsamples were filtered through pre-rinsed #4 Whatman paper and the volume restored to 25 mL with a 1% nitric acid solution. Analysis was performed by ICP (APHA, 1989).

A subsample was also combusted at 550°C and weight loss determined as measure of organic matter.

Analytic precision was less than ± 3% for all constituents and % recovery (accuracy) on EPA standards ranged from 81% for lead to 93% for aluminum.

RESULTS

CHEMICAL AND PHYSICAL CHARACTERISTICS

Temperature/Dissolved Oxygen

Surface temperature in Lake Twelve reached a high of 26.2° C on August 22, 1991. Thermal stratification began in the middle of April and continued until the end of September (see Appendix 1).

Hypolimnium DO declined to effectively zero concentration during summer stratification. Hypolimnetic DO levels remained below 2 mg/L from the middle of July until the beginning of October and were inversely related to the increase in RTRM (relative thermal resistance to mixing; the difference in density between surface and bottom relative to the 4°C to 5°C difference) as shown in Figure 6. However, anoxia did not persist for more than a month in mid-summer, indicating that stratification was not very stable; i.e., DO was apparently being entrained into the hypolimnium by wind acting on the lake's relatively shallow water column (6-7 m). Lakes in the Puget Sound lowland, that are on the order of 10-15 m maximum depth, normally do not begin to destratify until October following significant cooling that reduces RTRM (e.g., Pine Lake, Anderson and Welch, 1991). Although DO declined with stratification, oxygen deficit rate was not calculated because the rate would be greatly underestimated with such instability.

Nutrients

TP concentrations remained at relatively low values throughout the study period. Figures 7 and 8 depict whole-lake, volume-weighted and hypolimnetic (6-7 m)/epilimnetic (0-4 m) average TP concentrations. Mean whole-lake TP concentration ranged from 4.9 to 12.3 µg/L with the highest values occurring in the winter months during periods of higher external loading. The whole-lake TP concentration had yearly, summer, and winter averages of 8.2, 6.3, and 9.6 μg/L

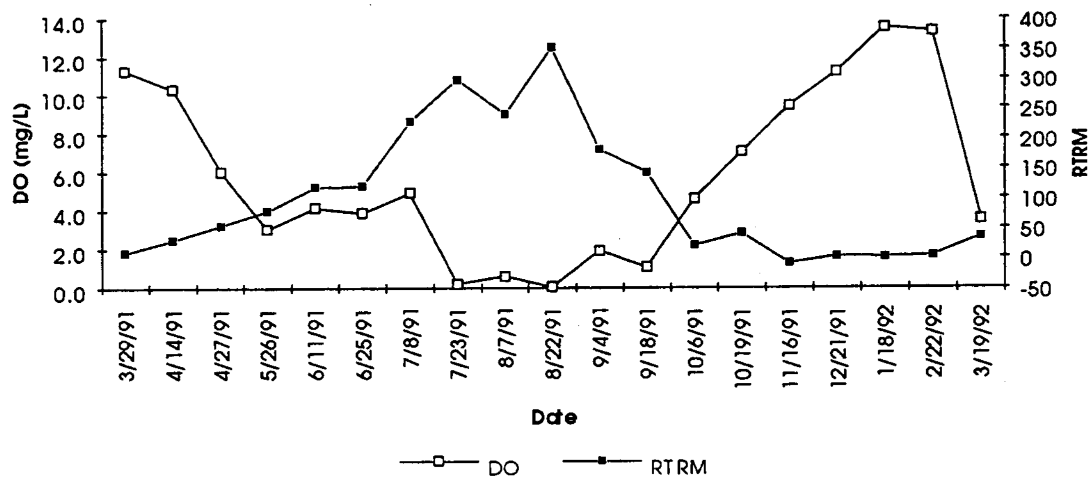


Figure 6. Hypolimnetic dissolved oxygen and relative thermal resistance to mixing. DO at 6 m.

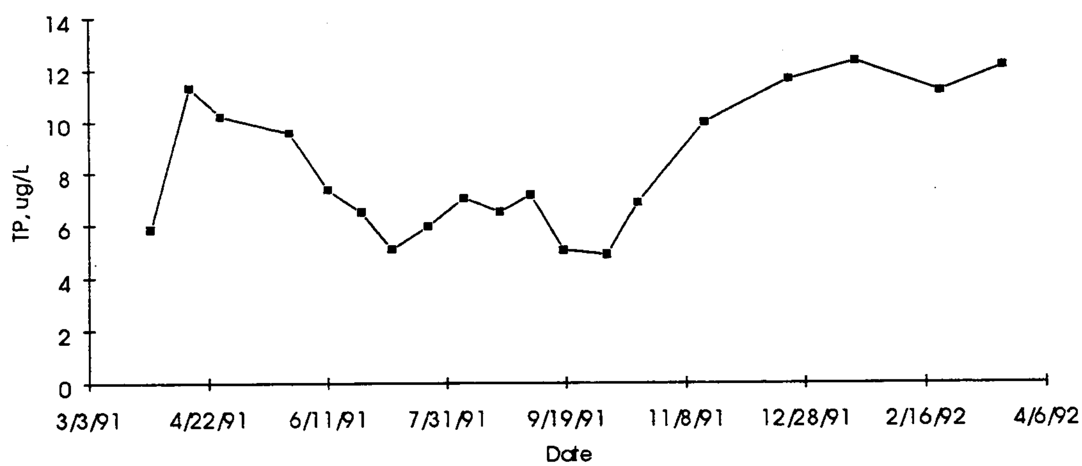


Figure 7. Whole-lake, volume-weighted TP concentrations.

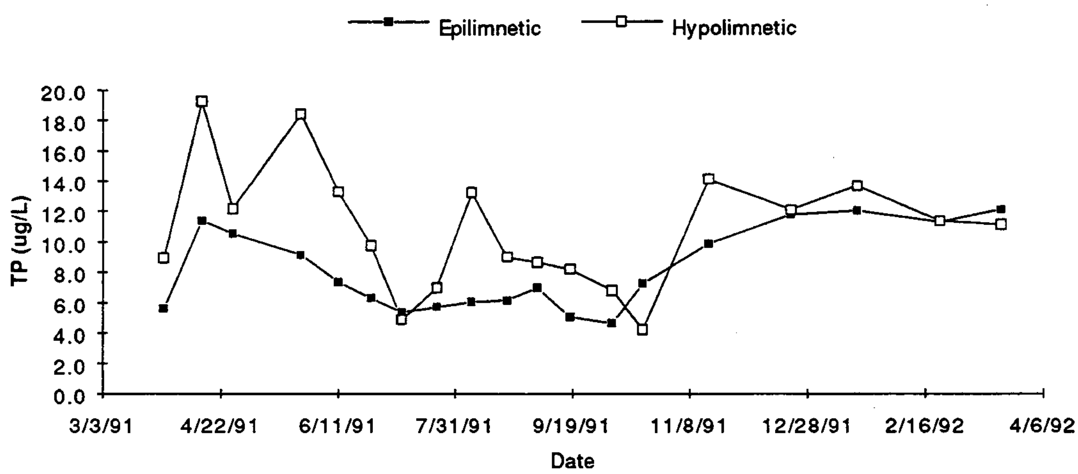


Figure 8. Epilimnetic and hypolimnetic total phosphorus concentrations.

respectively. The lowest concentration occurred in summer, when the potential for algal growth is greatest. These values are all well below both the oligotrophic and eutrophic TP thresholds of 14 and 25 µg/L, respectively (Porcella et al., 1980).

Hypolimnetic TP was usually higher than epilimnetic TP, but not markedly higher. Surprisingly, the highest hypolimnetic TP levels occurred in spring, shortly after statification and prior to anoxia. Hypolimnetic TP increased briefly in late July, probably in response to anoxic conditions, but subsequently declined as oxic conditions were restored in September (Figure 6, 8).

SRP also remained low throughout the study period. Figures 9 and 10 depict whole-lake, volume-weighted and hypolimnetic/epilimnetic average SRP concentrations. Mean whole-lake SRP concentration ranged from 1.6 to 4.2 µg/L without the definite seasonal trend shown by TP. The whole-lake mean SRP concentrations for the year, summer, and winter were 2.8, 3.0, and 2.6 µg/L, respectively. The hypolimnetic SRP was lower than or approximately equal to the epilimnetic SRP values throughout most of the study period (Figure 10). This indicates that there was no significant internal P loading via the iron redox process by which the reduction of iron in anoxic surficial sediments results in the mobilization and release of SRP.

Figure 11 depicts the ratio of SRP/TP in the hypolimnium and the epilimnium. That ratio is higher in the epilimnium than in the hypolimnium throughout most of the summer. The ratio would normally be higher in the hypolimnion where algal uptake of SRP is restricted due to low light. This again may indicate that P release from the sediments is minimal. On the other hand, the lack of a SRP increase, associated with TP during anoxic conditions, may simply reflect adequate light for algal uptake in this shallow hypolimnion (note the decline in hypolimnetic SRP/TP: in July).

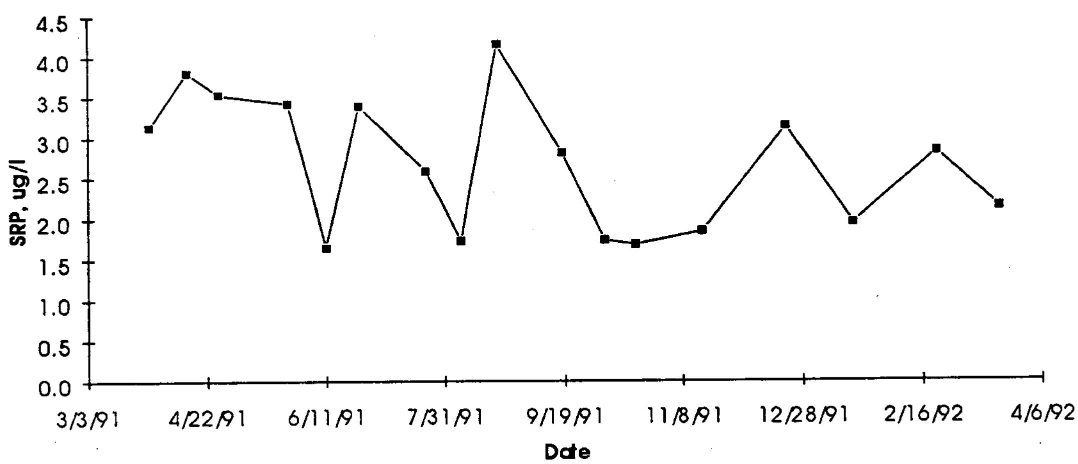


Figure 9. Whole-lake, volume-weighted SRP concentration

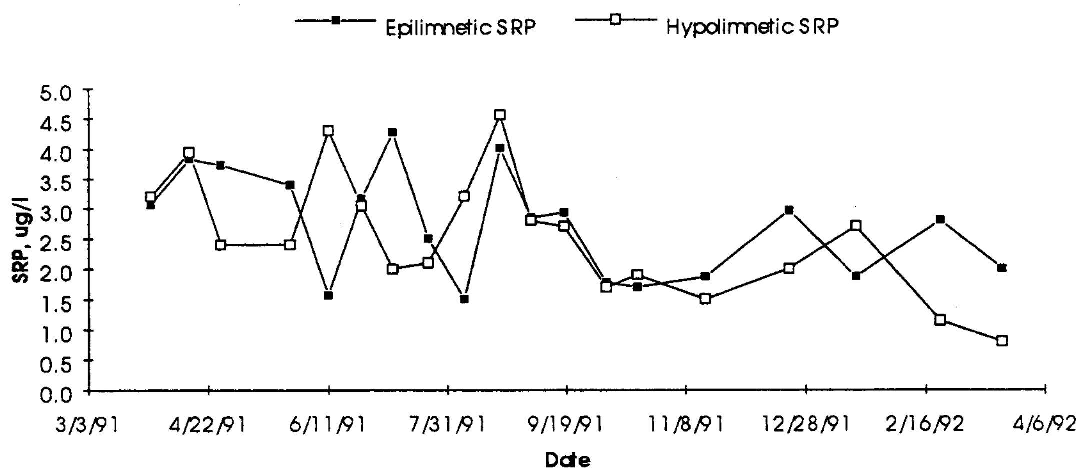


Figure 10. Mean epilimnetic and hypolimnetic SRP concentrations.

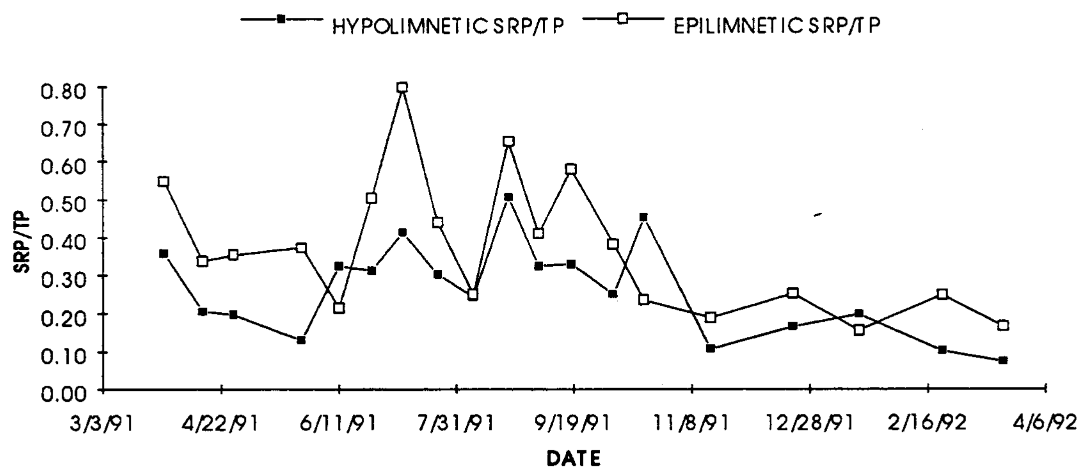


Figure 11. Epilimnetic and hypolimnetic SRP/TP ratios.

TN was rather high throughout the study period with average water column values ranging from a low of 306 µg/L in July, to a high of 1,185 µg/L in April (Figure 12). The yearly, summer, and winter average TNs were 497, 380, and 601 µg/L, respectively. The seasonal differences were similar to those for TP, being higher in winter months when inflow and external loading are greatest. These average values all greatly exceed the eutrophic threshold for total nitrogen of 180 µg/L (Porcella et al, 1980). However, that is insignificant, because the yearly and

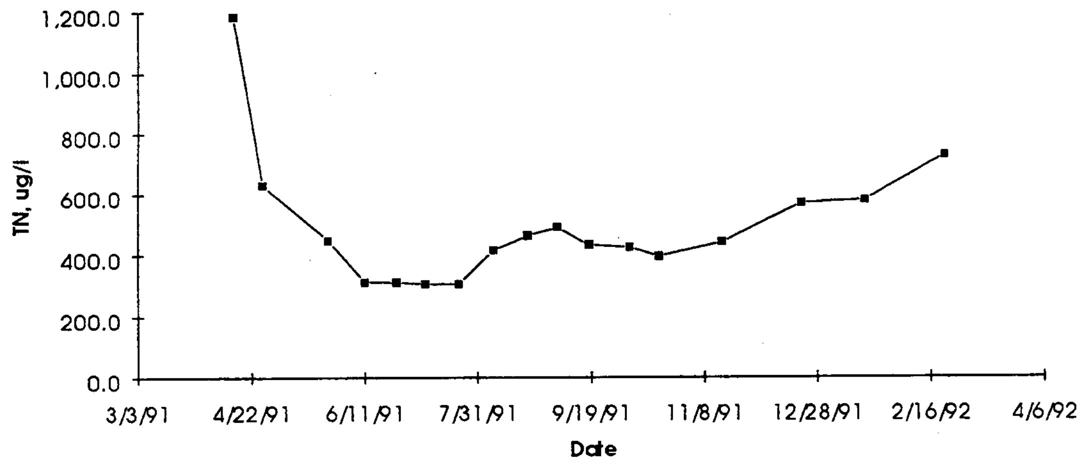


Figure 12. Mean water column TN concentrations

summer TN:TP ratios are about 60:1, making P clearly the limiting nutrient in the lake.

Nitrate+nitrite-N concentration ranged from 1.9 µg/L in August to 871 µg/L in March (Figure 13). Nitrate+nitrite-nitrogen followed the seasonal inflow trends with yearly, summer, and winter averages of 197, 14.3, and 314 µg/L, respectively. Depleted nitrate content during summer usually reflects increased uptake by algae in the epilimnion and/or loss through denitrification and lack of nitrification in the anoxic hypolimnion. Because low nitrate extended over a longer period than anoxic conditions (see Figure 6), these low levels most probably reflect algal uptake.

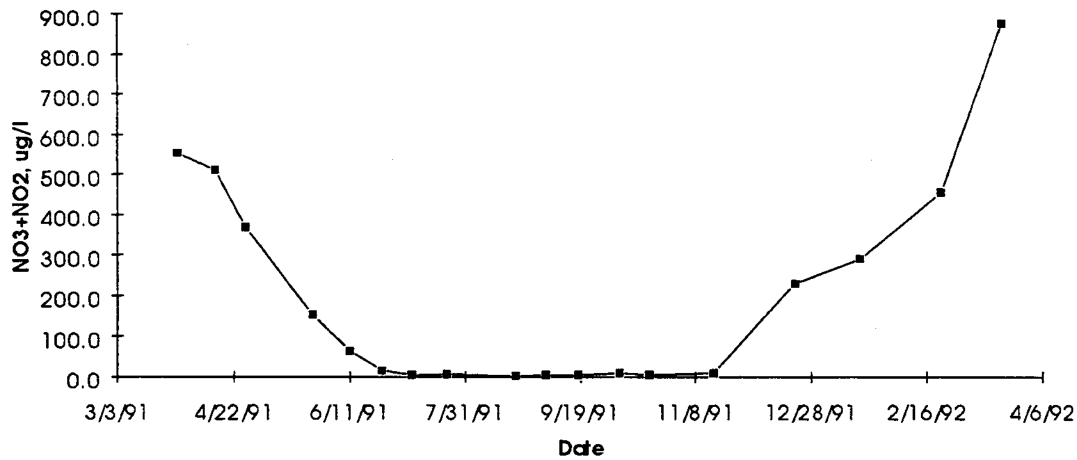


Figure 13. Whole-lake mean nitrate+nitrite-nitrogen concentrations.

Ammonium-nitrogen concentrations varied from 5.5 to 45 μg/L, although there was no seasonal trend similar to that of TN or nitrate (Figure 14). The mean ammonium concentration during the year, summer, and winter were 23.4, 24.7, and 22.3 μg/L, respectively.

Both nitrate and ammonium are available for uptake by algae. But do these low summer concentrations indicate growth limitation? Although inorganic N (NO_3^- $+NH_4^+$) was rather low in the summer (mean 29 μg/L), the mean IN:SRP ratio was about 10:1, indicating that P and N were limiting algal growth rates simultanously. Moreover, the very low concentrations present mean that if one nutrient were increased, growth would quickly be limited by the other nutrient. In spite of immediate growth rate effects, the very high TN:TP ratio represents ultimaate availability and at 60:1 is well above any possibility of long-term N limitation in this lake.

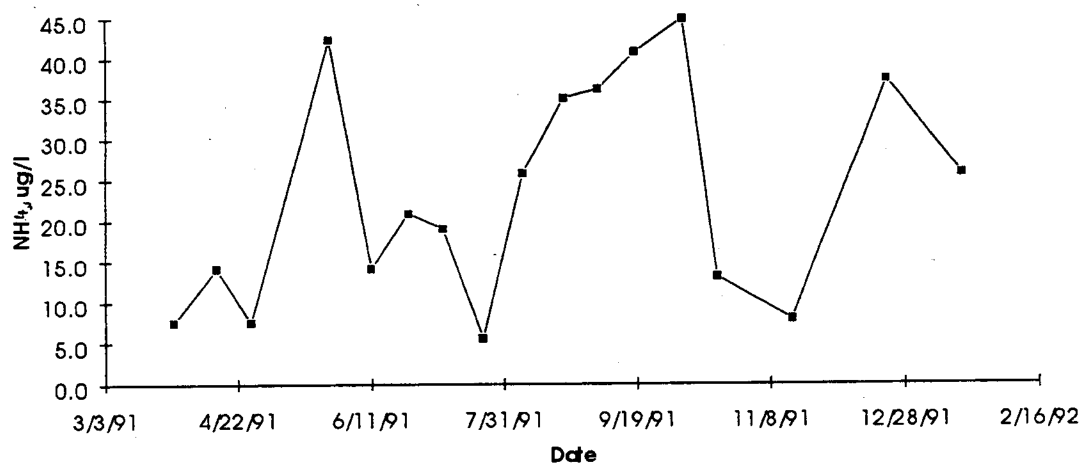


Figure 14. Whole-lake, volume-weighted mean ammonium nitrogen concentration

Secchi Disk Transparency

SD ranged from 6.0 m on June 25 to 1.4 m on September 4 (Figure 15). There was little difference among seasonal means, with average yearly and summer SDs of 3.7 m and 3.6 m. respectively. Nevertheless, there was a strong inverse relationship between chl a, which represents phytoplankton biomass, and transparency during most of the study period and especially in the summer (Figure 16). That is, the lowest transparency occurred during July and August when chl a was greatest. However, the lake was still quite clear; the summer average value of 3.6 meters is on the border line between an oligotrophic and mesotrophic state (Carlson, 1977).

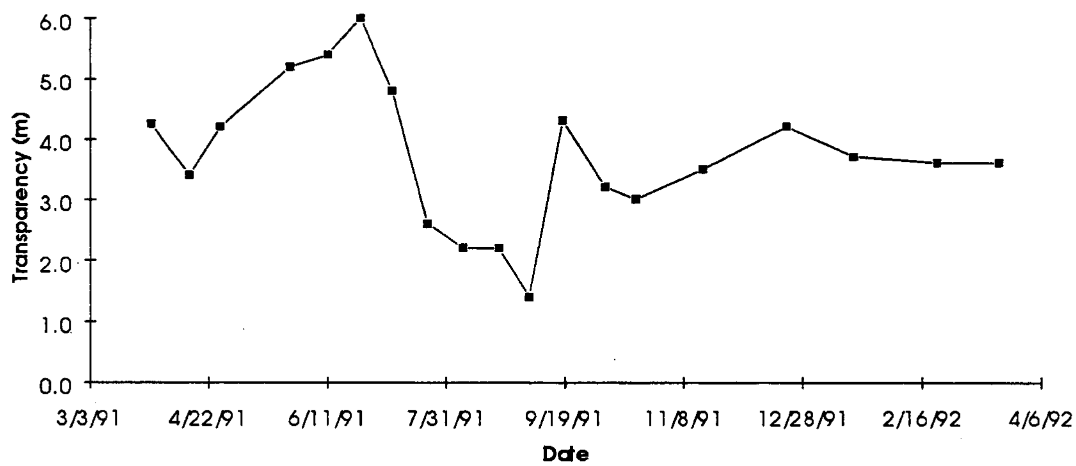


Figure 15. Secchi disk transparency.

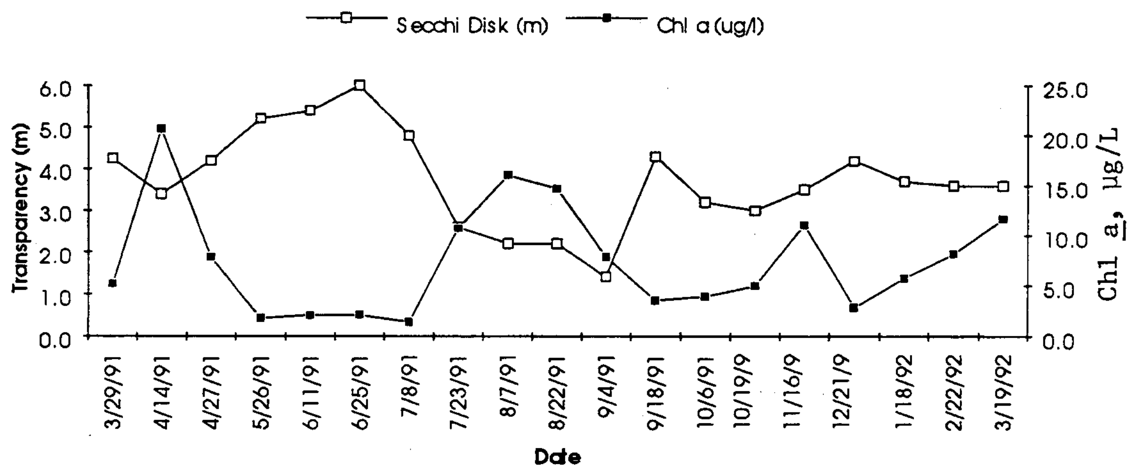


Figure 16. Secchi disk transparency and whole lake, volume weighted chl *a* concentration

Chlorophyll *a*

Whole-lake, volume-weighted mean chl a ranged from 1.4 to 20.7 µg /L. Figures 17 and 18 show whole-lake and hypolimnetic/epilmnetic average values for chl a. Chl a averaged 7.5, 7.3, and 7.6 µg/L during the year, summer, and winter, and

7.6 µg/L, respectively. These seasonal averages are all slightly below the eutrophic threshold of 8.7 µg/L chl a suggested by Chapra and Tarapchak (1976).

During most of the study period there was little difference in chl a between the hypolimnium and epilimnium (Figure 18). However, there was a very high peak observed in the hypolimnium on August 7 (42 µg/L). That peak was due to a large population of *Dinobryon*, a yellow-green alga, typical of oligotrophic waters and not known to cause nuisance blooms. The dense, hypolimnetic population of the green flagellate, *Gonyostomum,* observed in 1989, was not observed in 1991.

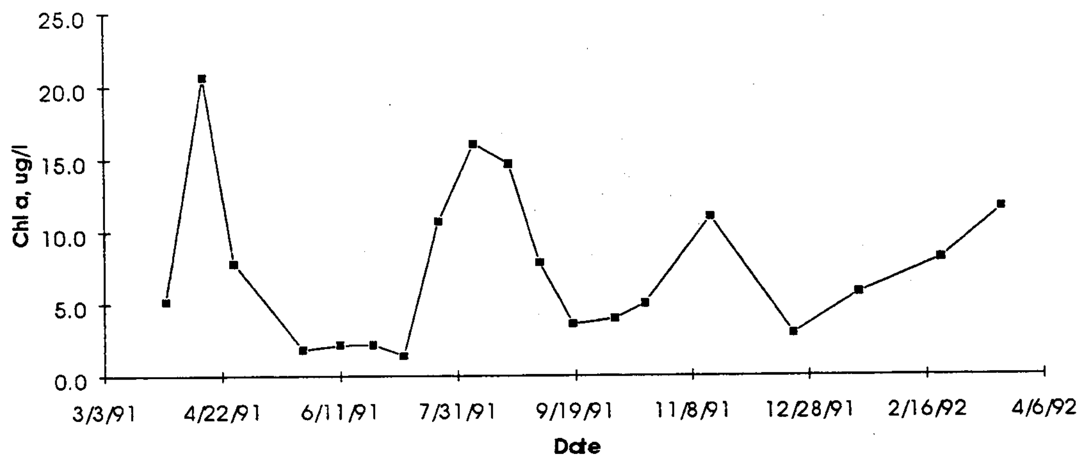


Figure 17. Whole lake, volume weighted mean chlorophyll *a* concentrations.

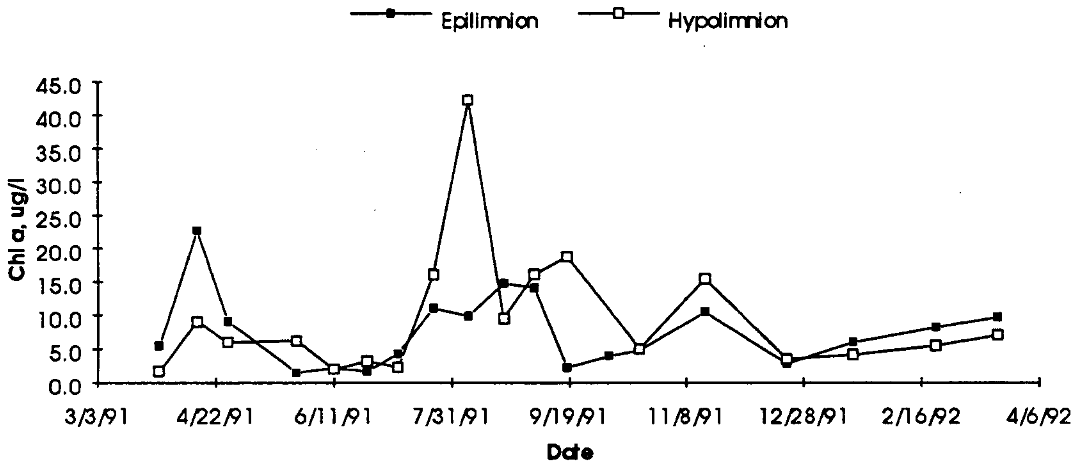


Figure 18. Mean epilimnetic and hypolimnetic chlorophyll a concentrations.

BIOLOGICAL CHARACTERISTICS

Phytoplankton

During the spring months (March-April) the green flagellate *Chlamydomonas*, and yellow-green *Dinobryon* were the dominant phytoplankton present in the lake. *Dinobryon* was also the dominant taxa during the summer months (June-September) reaching concentrations of 111 mm^3/L at 6 meters and 29 mm^3/L at 2 meters. *Chlamydomonas* and *Gloeocystis* were also abundant during the summer. The summer mean at 2 m for all genera was 12 mm^3/L. The concentrations of each taxa are presented in Appendix 2.

Diatoms, such as *Asterionella*, *Fragilaria* and *Melosira* typically form the spring bloom in temperate lakes, with *Dinobryon* being less represented (Wetzel, 1975). *Fragilaria* and *Melosira* were present but not in bloom proportions. *Stephanodiscus*, another common diatom, formed a large bloom at depth in October. Overall, the dominance of the phytoplankton by yellow-green and green algae and desmids (*Chlosterium*, and *Cosmarium*) indicate a relatively unproductive condition (Wetzel, 1975).

Anabaena, Aphanizomenon and *Coelosphaerium* are blue-green algae that were present in the lake but at rather low and non-bloom concentrations. Except for June 25, when *Anabaena* and *Coelosphaerium* accounted for 75% of the biovolume, blue greens were always less than 25% of the biovolume during the summer growth period (May-September).

Zooplankton

Cyclopoid copepods were the dominant zooplankton in the spring while nauplia larvae were very abundant from July through September. *Bosmina* were most abundant during the winter months. Figure 19 shows the total zooplankton abundance and that for *Daphnia*, the important algal grazer, during the study period. For most of the year, zooplankton were rather abundant. However, except for late May and early June, *Daphnia* were relatively low in abundance. The complete zooplankton data are presented in Appendix 3.

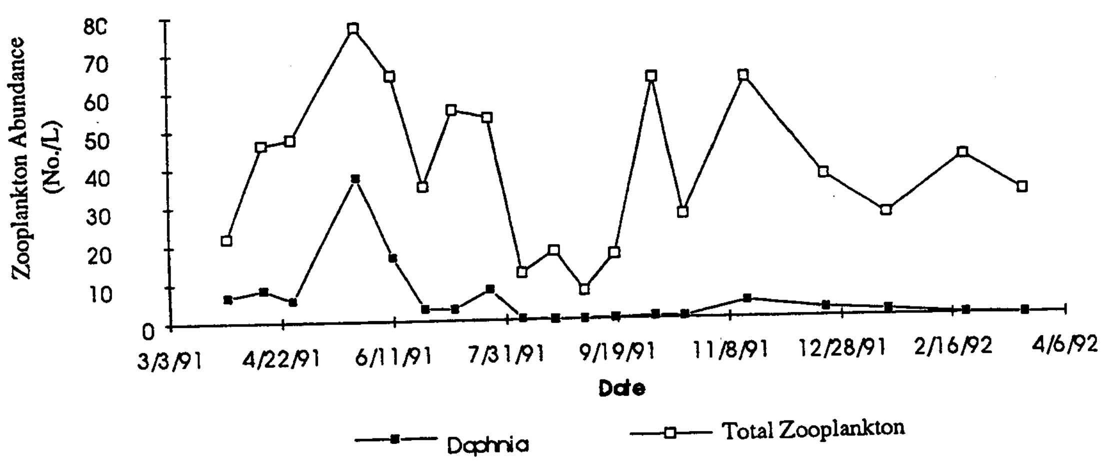


Figure 19. Total zooplankton and *Daphnia* abundance

Macrophytes

Submersed, rooted macrophytes have colonized most areas of the lake where the depth is less than 4 m. Additionally, an unidentified, partially decaying, grass-like macrophyte was collected from the lake bottom at depths in excess of 5 m and at densities as high as 44 g/m^2 (mean 16 g/m^2). The white water lily, (*Nymphaea odorata*), Eurasian water milfoil, (*Myriophyllum spicatum*), and the watershield (*Brasenia*) were the most common macrophyte species collected in the lake, at biomass levels as high as 197, 73, and 148 g/m^2, respectively. Water lilies were concentrated in the shallow areas, while milfoil was well distributed throughout the lake. Mean biomass levels for water lilies and milfoil, in samples where they occurred, were 88.7 and 25.6 g/m^2, respectively. The area-weighted mean for all species from 0 to 4m was 63 g/m^2. The complete macrophyte biomass and species composition data are presented in Appendix 4.

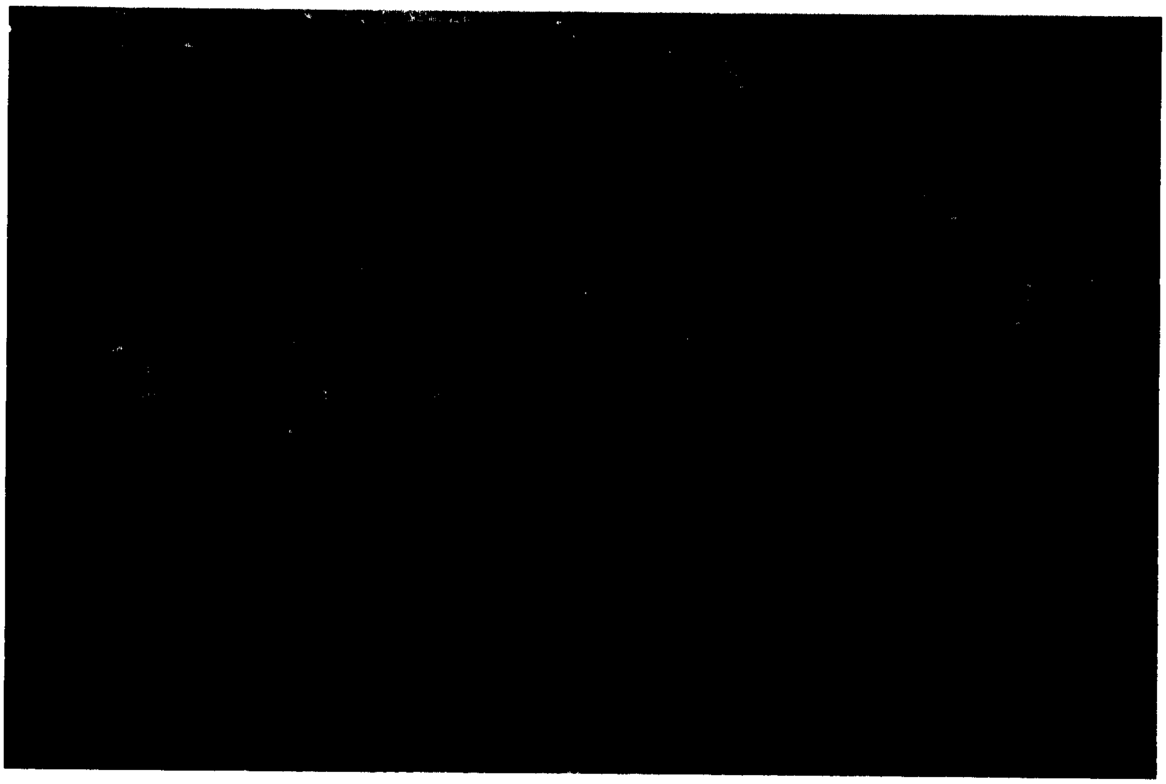
Because macrophytes obtain most of their nutrition from the sediments, light is usually the important factor that limits growth and distribution. While biomass is not predictable, a regression model developed by Canfield et al. (1985), based on 108 lakes, may be used to predict the maximum depth of colonization (MDC) of rooted submersed macrophytes:

$$\log MDC = 0.62 \log SD + 0.26$$

where MDC = Maximum depth of colonization in m and SD = Secchi depth in m

Using the average annual SD of 3.7 m, macrophytes would be expected to colonize the portions of the lake with depths of less than 4.1 m. As noted above, that is precisely what was observed. Floating leaf macrophytes were most concentrated near the south and east side, bordering the wetland, because that area was shallowest (see plate 1). However milfoil was more evenly distributed throughout the lake between 2 and 4 m (see Appendix 4).

Plate 1. View of Lake Twelve floating vegetation from east to west.



Hydrologic Budget

The monthly quantities and annual summary of the hydrologic budget are provided in Tables 4 (m^3) and 5 (acre-feet). The complex hydrologic characteristics of the lake (e.g., relatively undefined outflow to a large wetland) created some unusual uncertainties in the budget. Some quantities are known rather accurately, however. For example, precipitation and discharge from the PCCC sedimentation ponds were measured on a daily basis, but they represented only 14% of the total estimated input to the lake. Because these sources were measured on a daily basis, they are considered accurate to within 5% of the true value.

The composite north-shore flow component represented 34% of the total inflows to the lake during the study period. The flows were predicted with a linear equation regressing them against Pond A' flows of at least 0.1 cfs (see Appendix 5). The actual calculated values for the north-shore composite flows are considered to be fairly accurate because the correlation coefficient for the regression was high (r^2=0.86). However, there was a relatively large potential error when Pond A' was not producing sufficient flow to predict the flow at the downstream location. So the north shore flows are considered accurate to within 20% of the true values.

The ground water component represented an insignificant 0.2% of the total flows to the lake. Although the actual values of the ground water component may deviate considerably from the calculated values, due to large variations in the hydrogeologic characteristics of the watershed and few well sites, the relatively small contribution from ground water makes any error introduced insignificant in the overall hydrologic budget. Ground water well and flow data are given in Appendix 6.

Table 4. Lake Twelve hydrologic budget in cubic meters.

	Inflow						Outflow			
Date	Precipitation	Pond A (003)	Pond A' (008)	North Shore, 010-008	Ungauged Inflow*	Ground Water	Ungauged Outflow*	Evaporation	Discharge to Wetland	Change in Storage
Apr-91	38,855	8,264	3,577	160,105	0	305	38,942	10,608	156,671	-4,885
May-91	14,309	740	0	0	133,007	305	0	15,789	127,687	-4,885
Jun-91	11,718	294	0	0	103,361	305	0	17,146	94,979	-3,552
Jul-91	2,220	0	0	0	62,622	116	0	22,573	25,067	-17,318
Aug-91	6,538	0	0	0	20,557	116	0	17,886	0	-9,325
Sep-91	2,344	0	0	0	20,484	116	0	11,842	0	-11,102
Oct-91	10,978	0	0	0	0	116	12,204	5,551	0	6,661
Nov-91	39,842	3,084	1,110	61,321	0	160	103,613	2,344	52,848	53,287
Dec-91	16,776	2,667	1,174	104,166	110,175	184	0	1,727	230,307	-3,108
Jan-92	35,648	6,044	2,220	81,966	164,082	305	0	1,974	315,823	27,532
Feb-92	17,269	4,317	2,097	130,770	168,021	434	0	2,960	291,084	-28,864
Mar-92	10,978	1,234	493	53,164	119,713	441	0	5,921	168,113	-11,990
Total	207,475	26,644	10,671	591,492	902,022	2,902	154,759	116,321	1,462,578	-7,549

* Residual of Inflow - Outflow

Table 5. Lake Twelve summary hydrologic budget in acre-feet.

Inflow	Quantity	% of Total	Outflow	Quantity	% of Total
Precipitation	168	11.9%	Evaporation	94	6.7%
Pond A	22	1.5%	Discharge to Wetland	1,186	84.0%
Pond A'	9	0.6%	Ungauged Outflow	126	8.9%
North Shore	480	34.0%	Change in Storage	6	0.4%
Ungauged Inflow	731	51.8%			
Ground Water	2	0.2%			
Totals	1,412		Totals	1,412	

The loss of water to evaporation represented 6.7% of the total outflow from the lake during the study period. The calculations for evaporation were based on an average of 16 years of record from the Seattle Maple Leaf Reservoir (Phillips, 1968). The evaporative losses are estimated to be accurate to within 10% of the true values.

The discharge to the wetland represented 84% of the total outflow from the lake. The discharge was calculated with a stage-discharge linear regression equation (r^2=0.88) that predicted outflow based on lake level (see Appendix 7). The flow measurements in the wetland are estimated to be accurate to within 10% of the actual values. However, the flow measurements were taken at relatively low lake levels and the uncertainty in the stage-discharge relationship may be greater at higher lake levels and associated outflows throughout the winter months. Overall, the discharge to the wetland component is considered to be accurate to within 25% of the actual values.

The change in lake storage represented an insignificant 0.4% of the total discharge from the lake, and consequently is not considered to have an effect on the total accuracy of the hydrologic budget.

The ungauged inflow/outflow component of the hydrologic budget represented 51.8% and 8.9% of the respective inflow and outflow annual totals. The ungauged component of flow was calculated as the residual of the known/estimated values described above and, therefore, are only considered to be as accurate as the combined error in each of the known values. This would yield an ungauged inflow error of approximately 25% and ungauged outflow error of approximately 35%.

The overall error in the hydrologic budget may be estimated as the sum of all errors in the individual components, which yields a potential total error of 120%.

The ratio of total annual outflow to lake volume $\left(\frac{1.74 \times 10^6 m^3}{6.98 \times 10^6 m^3}\right)$ provides an estimate of average flushing rate of 2.5/year, assuming complete mixing. The reciprocal, 0.4 years, is the average detention time.

Phosphorus Budget

The TP budget for the lake was developed according to the procedures outlined in Methods and Materials and is presented in Tables 6 and 7. The lake received a total of 44.8 kg of TP from external sources during the year which is equivalent to an areal loading of 256 mg/m^2-y. The external TP loading observed during the study period may be compared to the critical TP loading for a lake with a eutrophic/mesotrophic threshold of 20 µg/L (mg/m^3) as predicted by Vollenwieder's (1976) model, which is expressed as:

$$L_c = 20\, mg/m^3\, \bar{z}\left(\rho + \sqrt{\rho}\right)$$

where: L_c = critical TP loading, mg/m^2· y

$\bar{z}$ = average depth, m

ρ = flushing rate, ^{1}/year

For Lake Twelve, with a mean depth of 3 m and a flushing rate of 2.5/year, the critical TP loading is 245 mg/m^2-y, which is essentially equal to the observed loading of 256 mg/m^2-y. This result would indicate that the lake is highly sensitive to an increase in external loading, i.e., any increase would move the lake toward an eutrophic state (TP = 20 µg/L). However, the lake TP content was only 8 µg/L. Thus, that sedimentation coefficient is not appropriate for the lake and a model more specific to the lake is called for.

A TP mass balance based on the hydrologic budget and the respective TP concentrations yields a net retention of 7.8 kg of TP/year (Table 7). That is, the lake retained 17% of the external load of TP during the year which is relatively low. Lakes commonly retain over 50% of externally loaded TP (Welch, 1992). TP concentrations observed in ground water and stormwater are given in Appendix 8.

Net internal loading of TP occurs if the residual in the mass balance, or net flux across the sediment surface, is negative. Thus, there was no net internal loading during the summer when high temperature, microbial activity, anoxic conditions and algal sediment-to-water migrations are most apt to occur (Table 6). Although anoxic conditions did occur, and TP increased briefly (Figures 6 and 8), the anoxic period was apparently not long enough to produce a significant internal load. The high negative flux in November resulted from the large ungauged outflow following the start of fall rains and not a sediment-to-water transport of P.

Phosphorus Model

The non-steady state TP model described in the Materials and Methods was calibrated to the whole-lake, mean TP concentration on a weekly basis during the year. The calibration was accomplished by selecting the proper sedimentation rate coefficient, σ, which was made a function of the lakes flushing rate, ρ. Welch et al.

(1986) found that the sedimentation rate coefficient for Lake Sammamish was best approximated by $\rho^{0.78}$, while $\rho^{0.71}$ was the best estimate for the rate in Green Lake (Mesner, 1985). The Lake Twelve TP model was constructed by dividing the hydrologic and annual TP budget data into one week time steps in an effort to minimize unnatural time-step fluctuations and consequent variability in calculated TP. The summer period was included to estimate σ, because there was no significant internal loading. Thus, gross internal loading in the model was not a rate to be determined through calibration and was considered as zero. Data for the calibration are given in Appendix 9.

Table 6. Lake Twelve Monthly Phosphorus Budget (grams).

	Input						Output			
Date	Precipitation	Pond A (003)	Pond A' (008)	North Shore (010-008)	Ungauged Inflow	Ground Water	Ungauged Outflow	Discharge to Wetland	Change in Storage	Net Flux Across Sediment Surface
Apr-91	738	202	55	3,794	0	23	857	2,444	-53	1,459
May-91	272	20	0	0	3,139	23	0	894	-47	2,513
Jun-91	223	9	0	0	2,439	23	0	741	-25	1,928
Jul-91	42	0	0	0	1,478	9	0	118	-95	1,316
Aug-91	124	0	0	0	485	9	0	0	-63	555
Sep-91	45	0	0	0	483	9	0	0	-68	469
Oct-91	209	0	0	0	0	9	155	0	39	102
Nov-91	757	99	27	5,090	0	12	11,397	5,813	533	-10,694
Dec-91	319	67	28	4,219	2,644	14	0	2,856	-36	4,398
Jan-92	677	100	27	1,107	3,938	23	0	3,474	339	2,736
Feb-92	328	63	121	1,948	4,032	33	0	6,142	-323	61
Mar-92	209	112	14	1,994	2,873	33	0	2,017	-146	3,070
Totals	3,942	671	271	18,152	21,513	219	12,409	24,499	54	7,913

Table 7. Lake Twelve Summary Phosphorus Budget (kg).

Inflow	Quantity	% of Total	Outflow	Quantity	% of Total
Precipitation	3.9	8.8%	Discharge to Wetland	24.5	66.3%
Pond A	0.7	1.5%	Ungauged Outflow	12.4	33.6%
Pond A'	0.3	0.6%	Change in Storage	0.1	0.1%
North Shore	18.2	40.5%			
Ungauged Inflow	21.5	48.1%			
Ground Water	0.2	0.5%			
Totals	44.8		Totals	37.0	
Net Retention					7.8

The sedimentation rate coefficient was initially set equal to $\rho^{0.5}$, which Larsen and Mercier (1976) found to closely approximate σ on an annual basis for a large number of lakes. The sum of the squares of the differences between the predicted TP concentrations and the measured TP concentrations was then calculated. The sedimentation coefficient was incrementally increased until the sum of the squares of the differences was minimized. The optimal sedimentation coefficient, showing the best fit between observed and predicted TP, was determined to be $\rho^{0.74}$.

Figure 20 shows the actual TP concentrations and those predicted by the model. During the summer months when external loading was at a minimum, the calibrated model tends to predict slightly greater TP concentrations than were actually observed. Conversely, the model underestimates TP concentration during the winter months when external loading was greatest. This seasonal inconsistency is due to the usually higher flushing rate in the winter months, when the actual sedimentation rate coefficient (σ) would be less than the yearly average used to calibrate the model. One could use more than one σ to account for such differences, but such modification is considered unwarranted considering the small error involved.

Although the model is capable of predicting internal loading, analysis of the actual TP budget and the lack of hypolimnetic SRP increase during the summer indicate that internal loading was insignificant during the study period. The model was calibrated for zero net internal loading to reflect the observed TP concentrations.

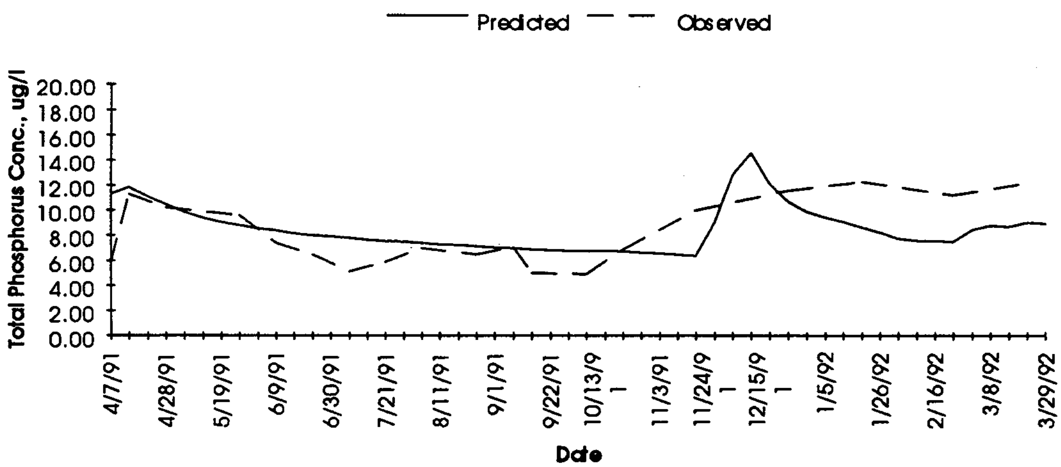


Figure 20. TP observed and predicted by a mass-balance model

Sediment Rates and Chronology

Results from the physical and chemical characterization of three deep-water sediment cores (cores A, C and L) collected in September, 1991 are included in Appendix 11. Detailed description of the sediment collection and analysis is presented elsewhere (Siwadune, 1991).

Core L was used for ^{210}Pb (radiolead) dating performed by Dr. Ahmed E. Nevissi (Nevissi et al., 1989). Cores A and C were used for chemical characterization. Nitric acid (HNO_3) digestion of the sediments was very effective in extracting nearly all the Pb and P from sediment (over 90%), but only partially effective in extracting Al and Fe (60 to 87%). The greater efficiency with HNO_3 rather than perchloric acid ($HClO_4$) for sediment digestion more than compensated for the lack of complete digestion by HNO_3. The distinct sediment elemental profiles obtained from the HNO_3 digestion would not be significantly different from those derived from $HClO_4$ digestion (Siwadune, 1992 and Hathaway, 1992).

The analytical precision for sediment chemical characterization was very good; precision, expressed as percent relative standard deviation, ranged from 0.1 to less than 22. The poorest precision was associated with trace metals at levels near the detection limit (Appendix 11, Table 1). The accuracy of sediment analysis (Appendix 11, Table 2), based on percent recoveries of elements from a dry municipal sludge (EPA quality control sample), ranged from 79% to 102%.

All three cores were similar in appearance with the top 8 cm being relatively dark (almost blackish) and unconsolidated. The top sediments gradually graded into a peat like material (dy) below 10-15 cm. The wet/dry ratios for core L decreased from 56 in the top cm to about 20 at 16 cm, and then gradually to 16 at about 29-30 cm and 15 to 14 from 30 cm to 55 cm with a similar trend occurring in core A (Appendix 11, Tables 3-5 and Figure 1). Although wet/dry ratios in Core C follow the same general trend, as in cores L and A, the ratios are relatively lower because weighing was not done immediately after core sectioning. The high water content of recent sediments (down to 20 cm) is indicative of a low rate of fluvial input to the lake. Fluvial inputs often contribute a large inorganic sediment load that results in compacted sediments.

The high, relatively constant high organic C content (as indicated by weight lost on ignition) in the top 30 cm or so of both cores A and C indicate that the nature and source of organic material has not changed very much during recent years. With the exception of the top 2 cm in core A (where organic matter content is about 25%), the top 30 cm of cores A and C show that organic matter content fluctuates between 39 and 45% (Appendix 11, Figure 1). The bottom 30-50 cm sections of both cores appear to be even richer in organic matter content varying from 45 to 50% while organic content in core C at those depths ranged from 40-45%. The decrease in organic matter at about 30 cm may be related to deforestation in the 1880s. The development of coal fields near Black Diamond in the early 1880s may have contributed to deforestation in the lake's watershed (Smayda, 1988). The content of Pb, Cu, Zn and Al also changes at 30 cm.

Sedimentation and deposition rates were determined by plotting the logarithm of the ^{210}Pb activity in Pci/g dry weight (after correcting for the supported ^{210}Pb) against the weight of dry sediment accumulated in the sediment core at the depth corresponding to each ^{210}Pb measurement. These results indicate that the changes in elemental profiles and organic matter content that occur at about 30 cm correspond to the period 1885-1895 (Appendix 11, Table 6, Figure 2). A regression of ^{210}Pb activity vs dry weight accumulated over a 9.35 cm^2 cross section area (the sediment core) for the 15-35 cm core section provided the basis for calculating sedimentation and deposition rates. More specifically, using the 24-25 cm sections, in which Pci/g dry weight decreases by one-half as the depth in the core increases from 25 to 29 cm, results in a deposition rate of 2.598 g/(22 y)(9.35 cm^2) or 0.0126 $g/cm^2 \cdot y$ (126 $g/m^2 \cdot y$), where 2.598 g represents the dry weight of a sediment core section 4 cm deep and 9.35 cm^2 in cross sectional areas, deposited over a 22-year period (the half-life of ^{210}Pb). Similarly, the sedimentation rate for the 25-29 cm core section corresponds to (29-25) cm/22 y or 1.82 mm/yr for the compacted sediments. After correcting for sediment compaction, the rate increases only slightly to 1.83 mm/y:

Sedimentationi rate =

$$\frac{\text{Deposition rate, } g/cm^2 \bullet y}{\text{Dry sediment density, } g/cm^3} \text{x} \frac{1}{1-\phi} = \frac{0.0126\ g/cm^2 \bullet y}{1.4\ g/cm^3} \text{x} \frac{1}{1-0.9507} = 1.83\ cm/y$$

where the porocity (ϕ) of 0.9507 was calculated from:

$$\phi = \frac{(\%\ H_2O)\left(\text{dry sediment density, g/cm}^3\right)}{(\%\ H_2O)(\text{dry sediment density}) + \left(1\text{-}\%\ H_2O\right)\left(H_2O\ \text{density}\right)}$$

$$= \frac{(93.23\%)\left(1.4\ \text{g/cm}^3\right)}{(93.23\%)\left(1.4\ \text{g/cm}^3\right) + (1-93.23\%)(1.00)} = 0.9507$$

Both sediment water and dry sediment densities were assumed rather than calculated.

With an average deposition rate of 0.0126 g/cm$^2 \cdot$ y, it would have taken approximately 128 y for 15.1 g of dry sediment to be accumulated over a cross sectional area of 9.35 cm^2 at 30 cm depth (Core L, Appendix 11, Table 5). At this deposition rate, the age of the sediments at 30 cm depth corresponds to about 1863 ($\pm$ 10-20 y). Normally, one can assume that the accuracy of the ^{210}Pb dating will be about $\pm$ 15 y on events occurring about 60 yr before present.

A constant rate of supply (CRS) model for ^{210}Pb deposition (which compensates for increased and decreased sediment deposition rates) provided absolute sediment dates that placed the 30 cm sediment depth at about 1880 (Appendix 11, Figure 3). However, both the 1863 and 1880 dates for the 30 cm depth could be in error as a result of bioturbation, core dragging during coring and/or loss of the top few mm of the unconsolidated sediments. Although loss of the top few mm of sediments during core retrieval apparently did not occur or was minimal, both ^{210}Pb and stable Pb profiles indicate that the top 20 cm is well mixed (Appendix 11, Figure 2), which makes accurate estimates of present day sedimentation rates difficult.

Background stable Pb levels in sediments below 30 cm are extremely variable in core A, ranging from about 25 mg/kg at around 35 cm to as low as 2 mg/kg at depths below 45 cm (Appendix 11, Figure 4). In contrast, background Pb levels at depths greater than 30 cm in core C are relatively constant, ranging from 14 to 20 mg/kg. From about 30 cm upward, Pb concentrations increase rather sharply, from about 20-30 mg/kg to about 80-110 mg/kg through the top 25 cm of core C. Pb values for the top 30 cm of core A are higher, ranging from 90 to 112 mg/kg. The higher Pb concentrations in the top 25 cm of core A are extremely variable. The higher concentrations at depths greater than 30 cm in A may be due to a failure to correct continuous drifting in the instrument during Pb analyses. Another factor that adversely influences background Pb values is due to the inability of the ICP instrument to accurately measure Pb concentrations at levels less than 0.1 mg/kg in the digestate.

In core C the initial increase in Pb occurs at the 30 cm horizon. The ASARCO Pb smelter was established in Tacoma in 1890 and is known to have influenced Pb concentrations in lake sediments over 40 km away (Crecilius and Piper, 1973). Lake Twelve lies within 30 km of the smelter and is partially downwind. Thus, the gradual increase in Pb content from 30 to 25 cm, in core C, may be coincident with the installation of the ASARCO smelter, which produced Pb bullion from its inception in 1890 until 1913 when it became a copper smelter.

Fly ash from coal burned locally, beginning when the coal fields were developed in the late 1880s, may have been another important source of Pb. However, since the advent of leaded fuel additives in 1923 increases in Pb concentrations in lake sediments closely parallel increases in local gasoline consumption. Total Pb emissions in the Puget Sound region reached a maximum in the 1973-1975 period and are currently very low (lower than the early 1940 period) because of the use of the unleaded gasoline. Assuming a uniform sediment deposition over the last 40-50 y, a maximum Pb content, corresponding to the peak of the leaded gasoline consumption in the early 1970s, would provide a reasonably good indicator for present day sedimentation. However, since the top 20 cm of Lake Twelve sediments are rather extensively mixed, the selection of any recent Pb marker is nearly impossible.

Comparison of Pb profiles in cores A and C from 1991 to a Pb profile from a deep core sediment collected in 1989 (digested with $HClO_4$) shows that Pb concentrations in the top 25-30 cm of all three cores are nearly identical (Welch and

Spyridakis, 1989). That further supports the contention that HNO_3 sediment digestion (used with 1991 cores) provides nearly complete extraction of sediment Pb. This conclusion is further substantiated by comparing the P content of the 1989 sediment core with those in the 1991 cores A and C (Appendix 11, Figure 4).

The sediment TP profiles, shown in Appendix 11, Figure 4, indicate, that both P and Al (Appendix 11, Figure 5) follow the same increasing trends from 30 cm upwards (except for the top few cm), reflecting perhaps the close association of P with Al in eroded material entering the lake. The increased TP content increases toward the surface from 0.16% at 20-10 cm to nearly 0.2% in the surficial sediments. That upward increase may reflect increased anthropogenic inputs in recent years and/or chemical/biological mobilization of P under highly reduced conditions. Increased P enrichment at the sediment surface parallels that of Fe and Mn (Appendix 11, Figures 5 and 6), indicating the close redox-imposed association with Fe and P.

Assuming that the P content (0.175%) of the 8 to 4 cm interval resulted from P loading to the lake and based on a sediment deposition rate of 126 $g/m^2 \cdot y$, a P deposition rate of $(126\ g/m^2 \cdot y)(0.175\%\ P) = 0.221\ g/m^2 \cdot y$ can be estimated for the deep portion of the lake. Further, assuming that the area of the lake at mean depth ($\bar{z} = 3$ m) represents the average of permanent sediment accumulation, lake surface area P loading that accumulated can also be estimated as follows:

$$\left(0.221\ \mathrm{gP/m^2 \bullet y}\right)\frac{3\ \mathrm{m\ (mean\ depth,\ \bar{z})}}{7.5\ \mathrm{m\ (maximum\ depth,\ z)}} = 0.0884\ \mathrm{gP/m^2 \bullet y}$$

or 88.4 mgP/m^2 y.

This accumulated surface P loading when augmented by the P leaving the lake proper (221.4 $md/m^2 \cdot y$) results in an external P loading of $(88.4 + 221.4) = 309.8$ ($\simeq$ 310) $mgP/m^2 \cdot y$, which is reasonably similar to the external surface P loading calculated from the water and P budgets (256 $mg/m^2 \cdot y$).

The discontinuity in the nature of the sediment that was observed for stable Pb, P, organic matter, Cu and Zn at about 30 cm depth (1870-1890) becomes readily apparent in Appendix 11, Figure 5, which depicts the Al profiles in cores A and C. Al is a tracer of inorganic material and is a major component of the silt and clay fractions of aluminosilicates and enters the lake primarily by fluvial means. The Al profiles indicate that Al enrichment began at a depth of 20-30 cm, increasing from background values of about 1% Al (at depths greater than 30 cm) to between 1.5% to 2% in the upper 20 cm. Both background and contemporary sediment Al content in this lake are lower than those found in the sediments of nearby Lake Meridian. However, given that the HNO_3 digestion only partially solubilizes aluminosilicates, absolute comparisons of sediment Al content between the two lakes may be risky. Nevertheless, Al content can be used as an indictor of allochthonous terrigenous inputs by comparing the ratios of % Al in the recent to background sediments in this lake (about 1.7 to 2.0) to that in Lake Meridian (about 2.7 to 3.0). This comparison shows that erosional inputs to Lake Twelve since about 1870-1890 to present were only about two-thirds as great as that measured in Lake Meridian (Barnes et al., 1978), a lake whose watershed is larger (2.2 versus 1.5 km^2) and nearly completely suburbanized. The relative uniformity of Al concentration since the early 1900s suggests that watershed and regional development following activities spurred by the discovery (1863) and later mining (1884) of coal reserves near Black Diamond, did not dramatically alter either the inorganic composition of the sediment or, in all probability, the amount of material entering the lake.

Results from sediment cores collected in 1989 (Welch and Spyridakis, 1989) suggested that lower levels of Pb and P in the surficial sediments at the west side of the lake, compared to those in the mid-lake and east, may represent dilution of sediment P and Pb in the west side of the lake from suspended sediments originating from the noise berm. However, evidence of recent increased sedimentation from erosional material could not be substantiated with these latest deep area core results.

Based on a comparison of recent Al enrichment over background in the two lakes, the background sediment deposition in Lake Twelve was about one-half (or less) of the recent sediment deposition or about 60 g/m^2 y. In Lake Meridian the precultural deposition rate was 77 g/m^2 y, and it increased, proportionately to the increase in Al, to about 250-280 g/m^2 y in recent sediment.

The profiles of Fe concentrations in cores A and C are shown in Appendix 11, Figure 5. The Fe and Al profiles show the same trends in increasing concentration over background in the top 30 cm of each core. That indicates their common source of erosional material. The Fe profiles are more uniform than those of Al, but with surficial concentrations that are more than double those for Al. This uniform nature of the Fe profile, along with the marked increase in the surficial sediments over background (from less than 1% in background to over 4% in surface sediments) is due to the mobilization of Fe in the sediments due to the reducing conditions.

Mn profiles (Appendix 11, Figure 6) reflect its extreme mobility (greater than Fe) under reducing conditions, its concentration increasing from less than .01% in the 30 cm deep sediments to nearly 0.14% at the surface.

The Cu and Zn profiles are shown in Appendix 11, Figure 7. The almost simultaneous Cu and Pb (Appendix 11, Figure 4) peaks indicate the source is probably the same (primarily atmospheric deposition). A comparison of the Zn and Pb profiles suggests that the marked initial increase in both elements occurred at about the same depth (35 cm) but while Pb concentration is relatively constant (110 $\pm$ 10 ppm) from 20 cm and upward, the Zn concentration shows a dramatic increase over Pb in the recent top 3 cm of the cores. That is demonstrated by Zn/Pb ratios increasing from about 1 to 1.5 in the 25-20 cm section to nearly 2 or more at the surface. This trend suggests that much of the Zn pollution is local and has increased in recent time. When compared to decreasing Pb inputs from the atmosphere (via Pb removal from gasoline and closure of ASARCO) the high Zn/Pb ratios in the surficial sediments are readily explained.

Zn also parallels sediment P, suggesting perhaps that some of the P enrichment in the top 2-3 cm sediments may be due to more than P mobilization (diffusion) to the sediment-water interface in a reducing environment.

DISCUSSION

Trophic State

The trophic state of Lake Twelve was determined using TP, chl *a*, SD and TN data collected during the study period. The measured values were compared to trophic state threshold values from the literature and presented in Table 8. The summer average TP concentration of 6.3 μg/L is well below the oligotrophic threshold of 14 μg/L (Porcella et al., 1980). The summer average Secchi disk transparency of 3.6 m is on the border line between an oligotrophic and mesotrophic state (Carlson, 1977). The summer average chl *a* concentration of 7.3 μg/L is slightly below the eutrophic threshold of 8.7 μg/L suggested by Chapra and Tarapchak (1976). The summer average TN concentration of 380 μg/L greatly exceeds the eutrophic threshold of 180 μg/L (Porcella, 1980), but is not considered indicitive of trophic state in this case due to the high TN:TP ratio, which demonstrates that P is the ultimate limiter to algal biomass.

The unusually high chl a :TP ratio (>1) may be indirectly due to the high dissolved color content of the lake water. Chl a:TP ratios are normally between 0.5 and 1.0 (Ahlgren et al., 1988). The high color restricts light penetration, which would cause the algae to compensate by raising their cell chl a content. Other colored water lakes in the Puget Sound lowland also have high chl a:TP ratios (Anderson and Welch, 1991). The average summer chl a:C ratio was 1.4%, which is low for healthy algae, and does not support high cell chl a as an explanation for the high chl a:TP ratio.

TP and SD indicate the lake to be oligotrophic, while chl a content suggests a mesotrophic state. Based on the most significant criterion, TP content, the lake is best described as oligotrophic. Thus, the lake's water is of very high quality. The relatively high concentrations of *Dinobryon* in the hypolimnion (>40 μg/L chl a) during 1992 was largely restricted to that water depth. Although *Dinobryon* and *Anabaena* did occur in the epilimnion during summer, they did not cause a visible nuisance near the surface. The fact that the high biomass was restricted to the hypolimnion is indicative of the strong limitation by P; the alga scavenged the P apparently released from bottom sediments during the short period of anoxia at the end of July.

Table 8. Average summer values for trophic state indicators.

Water Quality Parameter	Trophic State: Oligotrophic	Mesotrophic	Eutrophic	Lake Twelve	Reference
TP (μg/L)	<14	14-25	>25	6.3	Porcella et al., 1980
Chl *a* (μg/L)	<2.8	2.8-8.7	>8.7	7.3	Chapra and Tarapchak, 1977
TN (μg/L)	<140	140-180	>180	380	Porcella et al., 1980
Secchi depth (m)	>3.6	3.6-1.9	<1.9	3.6	Carlson, 1977

The magnitude of phytoplankton biomass and species dominance in the hypolimnion apparently varies from year-to-year. Chl a content exceeded 100 μg/L in 1989 and was composed of *Oscillatoria*, a blue-green alga, and *Gonyostomum*, a green flagellate. The reason(s) for such differences in dominance is not clear, although *Dinobryon* is known to occur in waters of low P content (Rodhe, 1948). During years with greater stability and longer periods of anoxia, larger biomass of potentially surface bloom species may occur as a result of greater P release from sediment and higher hypolimnetic P concentrations.

There is reason to believe that 1992 was a low-P year. Figure 21 shows that TP concentrations in the past were higher on the average than in 1992. That year-to-year variation may be due to wind conditions affecting water-column stability; less wind

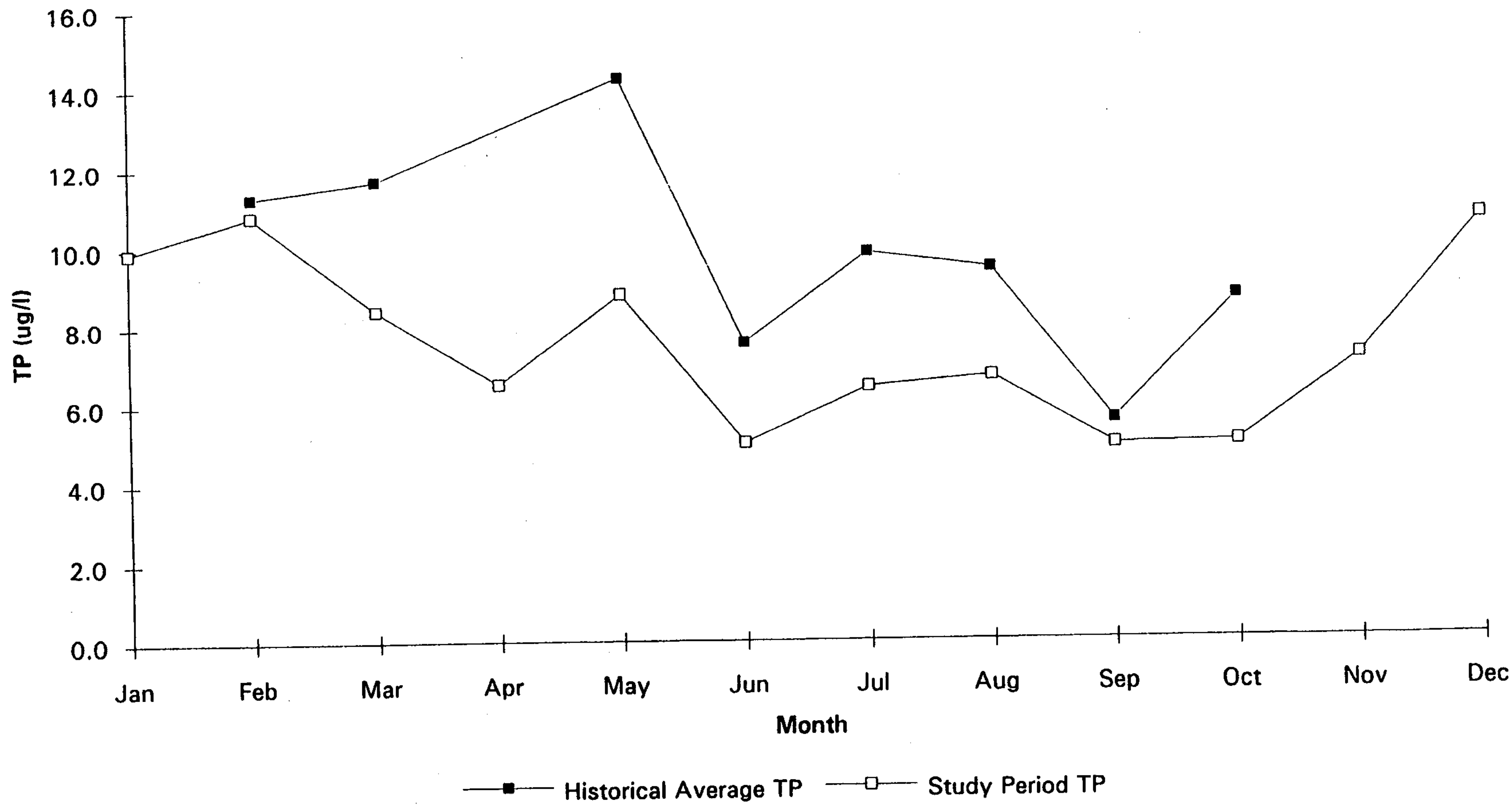


Figure 21. Historical (1976-1988) monthly averages (or single observations) and study period TP (March 1991-1992)

would produce greater stability (exert less force to mix the water column against the density gradient that resists mixing). With greater stability, the period of anoxia would be longer and sediment P release and hypolimnetic P build-up greater. TP at 6 m exceeded 100 µg/L in June and July of 1989. The shallowness of Lake Twelve, yet with sufficient depth to stratify, apparently results in this year-to-year variability in conditions that control P recycling from sediments.

The tendency of lakes to destratify and entrain hypolimnetic P into the water column and cause algal blooms has been related to the magnitude of the Osgood number (mean depth/square root of surface area; $\bar{z}/\sqrt{Km^2}$); Osgood, 1988). The Osgood number for Lake Twelve is 7.2, about the point (7.0) below which Osgood observed the magnitude of recycling (entrainment) from the hypolimnion to increase. Thus, the sensitivity of Lake Twelve to accumulate hypolimnetic P and then, on the otherhand, mix and distribute it through the water column is relatively great. With the appropriate wind conditions during summer, this lake could become eutrophic.

The quality of Lake Twelve during 1992, however, was quite good. Although hypolimnetic TP and algal content have been much higher than observed in 1992, there is no evidence that those hypolimnetic conditions have been transferred to the lake surface to cause a nuisance. Transparency during 1992 was close to an oligotrophic condition and was not different overall than in previous years (Figure 22). Therefore, in-lake treatment to reduce phosphorus and algae is not recommended. Continued monitoring, including the 6-m depth is recommended, however, in order to document the frequency of poor hypolimnetic quality and its possible transport to the lake surface. If such a process does occur, then an appropriate in-lake treatment should be considered.

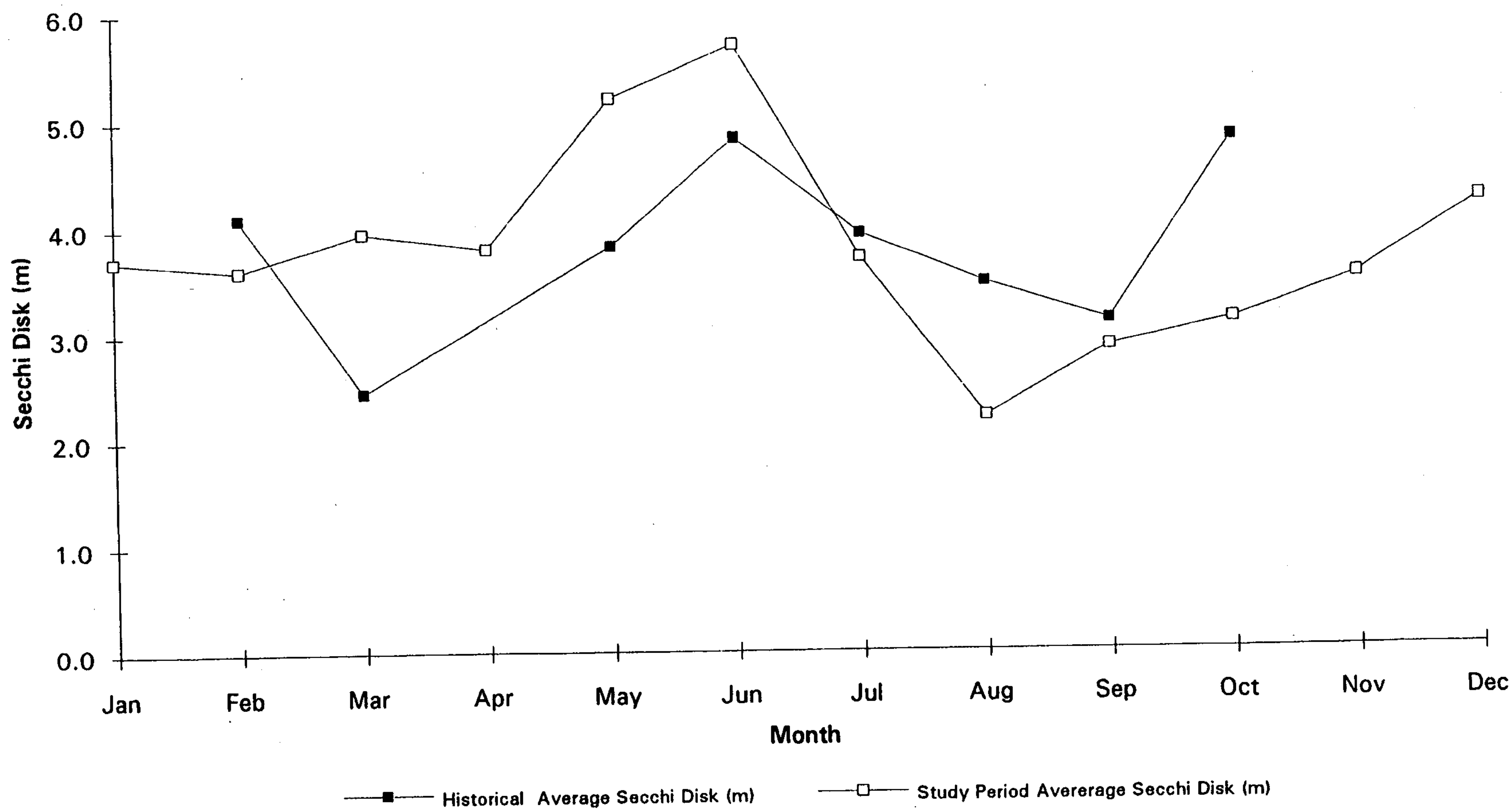


Figure 22. Historical Average & Study Period Transparency

Nutrient Sources

Because there was no observable existing problem with lake quality related to enrichment, leachate from septic tank drainfields was apparently not contributing significantly to lake enrichment. That is also indicated by the small contribution to the P budget from ground water (0.5%), which was due largely to the small ground water flows. Nevertheless, ground water TP concentrations were relatively high in some wells, ranging from 11 to 172 μg/L (Appendix 8) with a mean of 75 ± 44 μg/L and some wells definitely contained high chloride (Cl) concentrations, which probably indicates drainfield leachate contamination. Cl content in the six wells on land, over six months, averaged, in mg/L, 2.9, 3.7, 4.7, 9.0, 6.0 and 10.8 for wells 2 through 6, respectively (see Appendix 10). Wells 3 through 6 had maximum Cl concentrations of 10.2, 41.1, 18.5 and 13.7 mg/L. Cl in the lake ranged from only 2 to 3 mg/L. Thus, there is probably contamination from septic tank drainfields, but the relative magnitude is apparently not great.

Similarly, the TP contributions from PCCC's retention ponds, in terms of both loading and inflow concentration, were relatively minor (2.1% of external load). The contributions of particulate P were greater following construction of the noise berm, but apparently have subsided greatly as the berm slopes became vegetated. Sedimentation rates determined in cores from the deep area of the lake show no evidence of inorganic sediment in recent years. Although sedimentation rate is relatively low in Lake Twelve, any slight changes in the rate recently could not be discerned because the upper 20 cm was mixed presumably through bioturbation.

As stated in the Introduction, zoning has been changed from forest to 5-acre parcels on 16% of the non-lake watershed. According to the estimated TP loading (44.8 kg), the watershed yield (load/area) is 31 $mg/m^2 \cdot y$, right in the middle of values for other forested watersheds (Reckhow and Chapra, 1983; Issaquah Creek, Shuster, 1985). If a ten-fold increase in P yield, which normally occurs with land use changes from forest to single family residential (Reckhow and Chapra, 1983), were assumed for that 16%, the average yield from the whole watershed would about double, from the current 31 mg/m^2·y to 66 mg/m^2·y. The effect of such an increase should be detectable, even considering the sizable errors in the water and P budgets. Using the calibrated P model, a 3.3-fold increase in external loading (yield increase of 31 to 102 mg/m^2·y) would be required to raise summer lake TP from the current 6 to 25 μg/L, the eutrophic threshold (Table 8). Proportionately, an increase of 35 mg/m^2·y (31 to 66 mg/m^2·y) would raise lake TP by 9.4 μg/L to 15.7 μg/L (9.4 + 6.3) and would surely be detectable; the case if P yield from the 16% developed portion increased tenfold. However, P yield from one unit per 5 acres would probably not increase ten-fold to that of single-family density development. A doubling in yield at most would be more reasonable, to about 60 mg/m^2·y, which is near the lowest rate reported from urban areas (50 mg/m^2·y, Reckhow and Chapra, 1983). That would increase yield only 5 mg/m^2·year (31-36 mg/m^2·y) in the whole watershed, which would raise lake TP only 1.3 μg/L and would probably be undetectable.

The current yield estimate (31 mg/m^2·y) includes input from septic-tank leachate through ground water and ungauged inflow. The relative amount coming from septic-tank leachate is uncertain, but based on the current best estimate from six monitoring wells, it is probably not great. Nonetheless, it is a source that can and should be controlled. Currently operating drainfields can be inspected for percolation failure, and adequate effectiveness of P removal in newly constructed drainfields should be required.

Macrophytes

Macrophyte biomass levels in the lake in 1991 were moderate to low. The average, area-weighted mean of 63 g/m^2 contrasts with levels in other Puget Sound lowland lakes that exceed 200 g/m^2 (e.g., *Egeria* in Long Lake, Kitsap, Welch et al., 1988; milfoil in Union Bay, Lake Washington, Perkins et al, 1980, and milfoil in Green Lake). Milfoil is the largest contributor to whole-lake biomass in Lake Twelve and is considered to be a recent invader. Although, milfoil biomass levels are not great at present, they could increase because there is no reason to expect that their growth is limited by light availability or nutrient content or physical characteristics of the sediment, which is where they obtain most of their nutrients (Welch, 1992). Records from Metro's (1980) surveys in the lake show that milfoil was present at 38-61% of the total lake area during 1976-1980. Both areal coverage and density were reported to have decreased during that time interval. Biomass was not determined in those surveys, so comparison with the 1991 biomass level is not possible. However, milfoil has apparently existed at moderate abundance in Lake Twelve for at least 15 years.

The potential for milfoil biomass increase under good-growth conditions is suggested by its progression in Green Lake in Seattle. Milfoil was present at a few sites in Green Lake in 1981 but was not taken in any samples during an extensive survey. Thus it did not contribute to the very low whole lake biomass (all plants) of about 2.5 g/m^2 determined at that time (Perkins, 1983). By 1986, milfoil averaged 4.5 root crowns/m^2 (~30-50 g/m^2) in samples from 100 m transects located perpendicular to the shore all around the lake (Hart Crowser, 1986). Milfoil biomass in Green Lake in 1991 averaged 483 g/m^2 (KCM, 1991).

Sediment nutrient content and other characteristics in Green Lake (90% water, 20% organic matter and 1.5-2.0 mg/g TP) are similar to those in Lake Twelve (85% water, 40% organic matter and 1.5-2 mg/g TP), so milfoil growth should not be P limited. However, recent research suggests that plants may be N limited in sediments with high organic matter (Barko, personal communication). If the higher organic content of Lake Twelve sediments causes N limitation, then milfoil growth would be more restricted in Lake Twelve than Green Lake, which is suggested by the apparent stable and low milfoil abundance in Lake Twelve over the past 15 years. However, that is apparently not the case.

Bioassay results show that milfoil growth was the same in sediments from Lake Twelve, Green Lake and Union Bay of Lake Washington. Replicate, composite sediment samples from each lake were incubated at 3500 lux and 20°C for 30 days with rooted shoots of milfoil from Union Bay. Mean growth as stem elongation was slightly over 30 cm ($\pm$ 11) for each of the three sediments. There were 20-21 stems measured in each five-gallon pickle jar with sediment (see Appendix 12). Previous work with milfoil in sediment from Lake Union resulted in 41 cm stem elongation in 33 days, but at 4900 lux (Fussell-Ruthford, 1979). Normalizing for light produces good agreement between the two experiments using sediment from the same location.

Milfoil is known to senesce during summer and through decomposition release its P (which orginated from sediments) into the water contributing significantly to summer algal blooms (Smith and Adams, 1986). This process has been proposed to accelerate with the invasion of milfoil, because the increased plant production increases the rate of sediment accumulation, which in turn increases the area shallow enough for milfoil to establish. That results in greater milfoil biomass due to more available light and hence more P recycling from sediment (Carpenter, 1981). Although internal loading of P in Lake Twelve is currently not significant, it could become so in the future with a further increased biomass of milfoil.

Sediments

Lake Twelve sediments seem to provide an accurate history of activities in the immediate watershed and surrounding region. The history of the area suggests that the sediment desposition rate has remained relatively constant in recent years. The

126 g/m^2 y deposition rate may be representative as far back in time (and depth) as the current Al concentrations are maintained and consistent with erosion from watershed activities. The magnitude of the deposition rate observed seems to be in good agreement with rates meaured in other western Washington lakes when lake catchment areas, geographic setting and watershed activities are considered. Thus, the recent deposition rate (and catchment area) in Lake Twelve of 126 g/m^2 y (1.5 km^2) is consistent with the following: 260 (2.2 km^2) for Lake Meridian, 600-700 (1560 km^2) for Lake Washington, 690 (253 km^2) for Lake Sammamish, 360 (217 km^2) for Chester Morse Reservoir and 50 (2.6 km^2) for Findley Lake. The order of lakes is based on the relative trophic state of the lakes (Birch et al., 1980), varying from slightly mesotrophic (Lake Meridian) to oligotrophic (Chester Morse Reservoir and Findley Lake).

SUMMARY AND CONCLUSIONS

1. The trophic state of Lake Twelve is most accurately characterized as oligotrophic, primarily due to its low TP and relatively high transparency (summer means, respectively, 6.3 μg/L and 3.6 m).
2. P is clearly the nutrient that limits algal growth in the long term, due to very high TN:TP ratios (60:1), although soluble N and P were both quite low during summer and probably limited simultaneously in the short term. Therefore, management should emphasize P control.
3. Chl a concentrations were higher than expected from TP. Normally, chl a:TP ratios range from 0.5 to 1.0, but they exceeded 1.0 in Lake Twelve. That may be due to compensation for low light due to high water color content, although the low average chl a:C ratio (1.4%) indicates low cell chl a and moderate nutrient deficiency of the algae.
4. The principal alga present during the occurrence of high chl a levels was *Dinobyron*, a yellow-green that is more prevalent in oligotrophic than eutrophic lakes and does not cause nuisance blooms. Nuisance types that were observed in high hypolimnetic concentrations (>100 μ g/L) in 1989 were not observed in 1992.
5. TP content, and probably algal biomass as well, has varied considerably from year to year and was lower during 1992 than in previous years. That may have been due to a relatively short anoxic period and correspondingly low internal P loading. The lake's water quality in 1992 was not the worst nor probably even the average during the recent past. Morphometrically, the lake is probably very sensitive to wind that controls the period of anoxia as well as the vertical entrainment of hypolimnetic P and algae and some years may have relatively poor

water quality. Nevertheless, the evidence is not convincing enough to suggest that in-lake treatment to control P and algae is needed; but continued monitoring is recommended.

6. Although Cl and TP concentrations indicate contamination of ground water by leachate from septic-tank drainfields, the lake's oligotrophic state and relatively low contribution of groundwater to the P budget indicate that the magnitude of effect is not great. However, P removal effectiveness by drainfields should nonetheless be maintained at a high level at existing and future sites.

7. The contributions from PCCC's detention ponds to the lake's external loading was relatively small (2.1%) and apparently has declined since construction and development of vegetation on the berm. Sediment core results show no evidence of external sources of inorganic sediment in recent years.

8. Planned development in the watershed does not represent a serious threat to lake quality in the near future. Only 16% of the lake's watershed has been rezoned from forest to 5-acre parcels. Allowing for increased TP yield from the rezoned area at the lowest reported rate for single family residence density, the expected increase in lake TP (~ 1.3 μg/L) would probably be undetectable and insignificant to water quality.

9. Macrophyte biomass levels are relatively low, compared to other Puget Sound area lakes, or lakes with nuisance levels of milfoil. Bioassays showed that there is no nutritional reason for the low biomass in Lake Twelve. However, milfoil and water lily biomass levels are considered great enough to interfere with recreation and optimum use of the lake at this time. Biomass levels may increase in the future, judging from the dramatic increase in milfoil in Green Lake over 10 years to levels 20 times those in Lake Twelve currently. If that occurs, the associated organic matter sedimentation and P recycling (internal loading), which

probably occurs now through summer senescence of milfoil, could increase further and degrade water quality.

MANAGEMENT RECOMMENDATIONS

Algae and Lake Water Quality

The following techniques should be considered only if continued monitoring shows longer periods of anoxia and worse hypolimnetic and surface water quality (high TP and algae) in the future. Because there was no internal loading detected, a term for that source of P was not included in the calibrated P model. Therefore, predictions of lake TP resulting from applications of the following techniques is not possible.

a. **Artificial Circulation.** This technique is usually employed to increase mixing depth to limit plankton algal production by reducing available light. For that to work, depth should exceed 10 m or light attenuation should be high. Because algal nuisance biomass levels at the surface do not occur in Lake Twelve, and lake depth is insufficient for complete mixing to limit light, the technique is not recommended. Light and not nutrients should limit algal growth to insure success. Therefore, Lake Twelve is not a logical candidate.

   On the other hand, the technique prevents stratification if the rate of air injection is sufficient. That would maintain an oxic sediment-water interface and could be appropriate for Lake Twelve. However, there is a risk of worsening surface quality through destratification, even if oxic conditions are created; P released from sediments by processes other than iron redox would be made more available in the lighted zone and result in increased algal production. The evidence indicates that oxic sediment-P release is minimal in Lake Twelve, so the risk of increased internal loading and algal production with complete circulation may be minimal.

b. **Hypolimnetic Aeration.** Lake Twelve may be too shallow for the hypolimnion to be aerated while maintaining thermally stratified conditions, especially using

Limno full air lift or layer aeration devices (see Cooke et al., 1986; in press). However, hypolimnia of stratified lakes as shallow as Lake Twelve have been effectively aerated by removal, aeration and return of hypolimnetic water while maintaining stratification (e.g. Scriber Lake, Snohomish County). This technique would probably be less risky than complete circulation because, theoretically, entrainment would not occur and present a problem. This technique would provide a darkened, oxic refuge for large, efficient grazers, such as *Daphnia*. The lowest abundance of *Daphnia*, and zooplankton generally, occurred during the anoxic period (see Figure 6 and 19).

c. **Alum.** The addition of aluminum sulfate to water creates an aluminum hydroxide floc that sorbes and removes phosphorus and particulate matter from the water column. The settled floc forms a blanket over the sediment, preventing the release of P (internal loading) even under anoxia. The technique has proven to be highly cost-effective and safe in most stratified and unstratified lakes alike (see Cooke et al., 1986 and in press).

d. **Riplox.** The injection of calcium nitrate into anoxic reducing sediments reduces organic matter through denitrification, thereby restoring oxic conditions and the retention of P by oxidized iron. Iron is added if sediment content is too low or if sulfur, which ties up iron, is too high. The technique may be more long-lasting than alum (restores top 20 cm of sediment), but is more costly and has been applied to only three lakes (see Cooke et al., 1986 and in press).

Macrophytes

There are no existing techniques that provide a reasonably long-term and ecologically sound (treats the cause) control that is similar to any of those described for algae.

a. **Dredging.** This technique has been used successfully to control macrophytes by deepening, thus decreasing the lake area that is less than the maximum depth of colonization, which is slightly greater than the depth of visibility. That depth in Lake Twelve is 4.1 m, so deepening much of the lake area less than 4 m to that depth or greater would very likely reduce macrophyte biomass over the deepened area.

In Lily Lake, Wisconsin, the most documented case of deepening to control macrophytes, the macrophyte biomass level actually increased and extended to the resulting greater depth. That occurred because water transparency increased (less P and algae) extending the depth of maximum colonization. An increase in transparency is not likely in Lake Twelve, because algal biomass in the epilimnion is apparently not controlled by P internal loading, which was reduced in Lily Lake. Although macrophte biomass was greater and extended to greater depths, the Lily Lake project was considered a success because deepening created a larger volume of the lake without visible macrophtes (they did not reach the surface), so fish habitat was improved and fish production increased (see Cooke et al., 1986 and in press).

Plant biomass near the shoreline would not be controlled by light limitation as in deeper water, but removing about 0.5 m from the 0-2 m contour could retard macrophyte (lily) recovery by removing a significant amount of vegetative parts (tubers) and organic matter. A dragline dredge was used around the shoreline of Lake Trummen, the world's classic case for dredging, to successfully and rather permanently remove near shore emergent and submergent vegetation (see Cooke et al., 1986 and in press; and cited references by Bjork). Deepening is really the only way to retard the process of increased sediment accrual by a future potential increase of milfoil.

A principal drawback to dreging Lake Twelve to reduce macrophyte distribution and associated sediment accrual/P internal loading is that most of the shallow area with the most macrophytes (i.e., milfoil) is at the east end of the lake, bordering the wetlands, which is relatively inaccessible for recreational use.

On the other hand, anoxic conditions would probably not be worsened by the dredging; the recommended 4 m depth would be in the epilimnion, so the hypolimnetic area and volume would not be increased. Also, the Osgood number $\left(\frac{\text{mean depth}}{\text{area}}\right)$ would be increased, lessening the potential of entrainment of sediment-released P and particulate matter. Finally, increased lake volume would increase sediment P retention by increasing water residence time, i.e., more incoming P would go to sediments rather than remain in the water for algal uptake. Thus, increased depth would improve water quality physically, especially in such a shallow-lake where one m in depth markedly increases volume.

b. **Sediment Covers.** Plant biomass in the recreationally-important shoreline could be controlled by sediment covers installed and manipulated by property owners/lake association members. These devices have a long life, so can be reused over and over. They can be moved to an untreated site after a month, thereby increasing cost effectiveness. Control is due to tactile response as much as to light. If left too long in one place, sediment will accumulate in pockets and plants will establish on the cover surface. For more detail, see Cook et al. (1986 and in press).

c. **Harvesting**. This technique is the most generally accepted, non-toxic, cost-effective macrophyte control. However, there are several limitations, among which are: 1) rapid grow back requiring multiple cutting, 2) slow rate of coverage by harvester, 3) disposal site required, 4) high capital cost, and 5) depth restriction (<1.5 m, 5 feet). On the other hand, the result is immediate, plants and their nutrients are removed and effectiveness can be long-lasting (with milfoil) if roots are cut below the sediment surface (see Cooke et al., 1986, and in press).

d. **Sterile Grass Carp**. These fish are very effective at reducing macrophtes. The problem is they are often too effective, and their numbers cannot be easily controlled once released into a lake. Potential problems due to nutrient recycling and interference with other fish populations are risks but they have not been demonstrated (see Cook et al., 1986 and in press).

e. **Lake Level Drawdown**. This technique has worked well at reducing rooted macrophytes in climates where winter temperatures remain below freezing long enough to desiccate the plants and roots. Hence, it is not recommended for Puget Sound lowland lakes (see Cooke et al., 1986, and in press).

Herbicides

Hebicides are not recommended because contrary to all other techniques, they provide no ecologically-sound advantages. Moreover, plants are left to decompose and release their nutrients, a purposefully toxic substance is added with incompletely understood side-effects (although toxicity of some herbicides to animals is low), and a tolerance selection process is invoked. Such a process interferes with any natural ecological selection/successional pressure, with the potential for greater nuisance conditions than existed before treatment.

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Appendix 1

Lake Twelve Water Quality Data

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^0C)	Chl *a* (µg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (µmhos)	T.P. (µg/l)	SRP (µg/l)	T.N. (µg/l)	NO_3^--N (µg/l)	NH_4^+-N (µg/l)
0 meters	3/29/91	4.3	6.6	12.4	8.9	6.8	NS	NS	3.5	3.3	NS	555.8	9.7
1 meter	3/29/91			12.4	8.5								
2 meters	3/29/91		6.9	12.2	8.4	3.8	NS	NS	8.7	3.6		553.0	20.3
3 meters	3/29/91			12.2	8.2								
4 meters	3/29/91		7.0	11.8	8.0	5.8	NS	NS	4.6	2.3		553.0	7.2
5 meters	3/29/91			11.7	8.0								
6 meters	3/29/91		7.0	11.4	8.0	1.8	NS	NS	7.9	2.6	NS	544.6	5.3
bottom	3/29/91		7.1	11.2	7.9	1.5	NS	NS	10.0	3.8		544.6	6.0
pond A (I-2)	3/29/91		NS	NS	NS		NS	NS	11.5	3.8	NS	342.8	6.6
pond A' (I-1)	3/29/91		NS	NS	NS		NS	NS	11.1	4.3	NS	673.5	5.3
outflow	3/29/91		NS	NS	NS		NS	NS	7.9	3.6	NS	513.7	6.6

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^{0}C)	Chl *a* (µg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (µmhos)	T.P. (µg/l)	SRP (µg/l)	T.N. (µg/l)	NO_3^--N (µg/l)	NH_4^+-N (µg/l)
0 meters	4/14/91	3.4	7.1	11.9	11.2	23.1	24.0	NS	7.1	3.3	1,413.1	510.9	11.6
1 meter	4/14/91			12.0	10.9								
2 meters	4/14/91		7.2	12.4	10.0	15.0	24.0	NS	12.7	4.1	1,094.3	516.5	41.5
3 meters	4/14/91			12.5	9.5								
4 meters	4/14/91		7.2	12.0	8.9	29.7	24.0	NS	14.3	4.1	982.5	499.7	5.3
5 meters	4/14/91			11.2	8.7								
6 meters	4/14/91		7.2	10.8	8.5	9.7	24.0	NS	16.0	4.1	957.7	510.9	16.6
bottom	4/14/91		7.1	9.8	8.4	8.4	24.0	NS	22.5	3.8	937.0	499.7	16.6
pond A (I-2)	4/14/91		NS	NS	NS		164.6	NS	35.8	3.1	862.5	438.1	15.3
pond A' (I-1)	4/14/91		NS	NS	NS		61.0	NS	40.3	5.8	1,529.1	1,183.4	13.4
outflow	4/14/91		NS	NS	NS		24.0	NS	22.0	3.3	1,177.1	477.3	18.4
0 meters	4/27/91	4.2	9.0	9.1	13.0	3.0	24.0	NS	5.9	2.3	630.6	365.2	7.2
1 meter	4/27/91		8.7	9.9	12.6								
2 meters	4/27/91		8.4	10.1	12.1	8.0	24.0	NS	14.7	4.8	601.6	365.2	8.5
3 meters	4/27/91		8.2	9.9	11.8								
4 meters	4/27/91		8.0	9.5	9.6	16.4	#N/A	NS	11.1	4.1	580.9	351.2	4.1
5 meters	4/27/91		7.8	7.9	8.9								
6 meters	4/27/91		7.5	6.5	8.8	6.7	#N/A	NS	9.5	2.8	630.6	432.5	7.8
bottom	4/27/91		7.2	5.6	8.6	5.3	24.0	NS	14.8	2.0	634.7	407.3	12.2
pond A (I-2)	4/27/91		NS	NS	NS		133.0	NS	15.6	3.1	481.5	152.3	12.8
pond A' (I-1)	4/27/91		NS	NS	NS		32.7	NS	21.6	5.6	705.1	471.7	12.8
outflow	4/27/91		NS	NS	NS		24.0	NS	9.1	3.6	535.4	334.4	7.2

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^{0}C)	Chl *a* (μg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (μmhos)	T.P. (μg/l)	SRP (μg/l)	T.N. (μg/l)	NO_3^--N (μg/l)	NH_4^+-N (μg/l)
0 meters	5/26/91	5.2	9.0	9.8	15.1	1.2	26.2	NS	8.8	4.8	465.0	158.6	42.7
1 meter	5/26/91			10.1	14.9								
2 meters	5/26/91		8.3	9.9	14.7	1.2	26.2	NS	8.3	2.8	415.3	149.8	35.9
3 meters	5/26/91			9.8	14.6								
4 meters	5/26/91		8.0	9.9	13.1	2.0	26.2	NS	10.3	2.6	431.9	148.7	25.3
5 meters	5/26/91			5.3	10.8								
6 meters	5/26/91		7.1	5.1	10.1	5.6	26.2	NS	14.7	1.8	431.9	150.9	42.1
bottom	5/26/91		7.2	1.0	9.9	6.8	26.2	NS	22.1	3.0	382.2	143.8	53.9
pond A (I-2)	5/26/91		NS	NS	NS		189.7	NS	44.7	2.6	415.3	64.9	21.5
pond A' (I-1)	5/26/91		NS	NS	NS		100.3	NS	14.7	4.1	423.6	156.4	16.6
outflow	5/26/91		NS	NS	NS		25.1	NS	7.0	4.1	394.6	127.8	24.0
0 meters	6/11/91	5.4	8.5	9.8	17.1	2.4	34.9	85.6	4.3	1.5	291.1	55.5	14.1
1 meter	6/11/91		8.1	10.2	17.0								
2 meters	6/11/91		7.9	9.7	16.7	0.8	30.5	83.3	8.5	1.6	291.1	54.4	15.3
3 meters	6/11/91		7.9	10.0	15.9								
4 meters	6/11/91		7.7	9.7	14.0	3.2	29.4	84.9	9.2	1.6	315.9	77.0	15.3
5 meters	6/11/91		7.6	5.1	11.2								
6 meters	6/11/91		7.4	4.5	10.0	3.2	30.5	86.5	11.0	2.1	332.5	98.5	14.1
bottom	6/11/91		7.0	3.8	10.0	0.8	29.4	88.5	15.5	6.5	274.5	103.5	7.8
pond A (I-2)	6/11/91		NS	NS	NS		237.6		54.3	2.1	431.9	68.2	15.9
pond A' (I-1)	6/11/91		NS	NS	NS		131.9		18.6	5.6	258.0	51.7	18.4
outflow	6/11/91		NS	NS	NS		30.5	98.5	7.8	2.1	299.4	32.9	24.0

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^{0}C)	Chl *a* (μg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (μmhos)	T.P. (μg/l)	SRP (μg/l)	T.N. (μg/l)	NO_3^--N (μg/l)	NH_4^+-N (μg/l)
0 meters	6/25/91	6.0	7.5	10.0	17.6	1.0	24.1	88.0	5.7	4.8	306.1	18.4	22.1
1 meter	6/25/91			10.2	17.5								
2 meters	6/25/91		7.6	10.2	17.4	3.5	26.0	90.0	6.8	2.7		13.3	
3 meters	6/25/91			10.0	17.2								
4 meters	6/25/91		7.5	9.0	15.3	0.5	26.5	88.9	6.4	2.0		9.4	
5 meters	6/25/91			4.9	12.2								
6 meters	6/25/91		7.3	3.9	10.8	6.0	28.4	94.2	8.5	3.3	315.0	17.2	19.7
bottom	6/25/91		7.0	3.9	10.8	0.5	27.9	94.0	11.0	2.8		36.8	
pond A (I-2)	6/25/91			8.7	12.8		193.7		130.1	2.0	553.8	52.5	22.1
pond A' (I-1)	6/25/91			8.1	13.4		137.5		14.1	2.8	306.1	15.0	8.8
outflow	6/25/91			7.7	17.7		22.7		7.8	2.7	368.1	5.5	19.7
0 meters	7/8/91	4.8	7.5	8.6	22.5	0.5	27.5	90.3	5.0	3.5	315.0	1.9	16.0
1 meter	7/8/91			8.5	22.3								
2 meters	7/8/91		7.8	8.3	22.0	2.0	26.5	90.6	5.7	4.0		1.9	
3 meters	7/8/91			9.5	20.0								
4 meters	7/8/91		7.5	8.8	17.5	10.5	24.5	90.7	SL	5.3		4.7	
5 meters	7/8/91			6.0	14.0								
6 meters	7/8/91		7.1	4.8	12.2	3.0	26.5	92.2	2.9	2.0	297.3	28.0	22.1
bottom	7/8/91		NS	5.0	12.2	1.5	26.5	92.5	6.8	AE		21.7	
pond A (I-2)	7/8/91		7.6	5.7	17.3		194.3	590.0	222.8	8.5	606.9	77.4	31.8
pond A' (I-1)	7/8/91		7.8	6.4	18.4		80.5	280.0	12.3	2.2	315.0	18.8	7.5
outflow	7/8/91		7.1	6.9	23.5		27.5	90.9	AE	6.6	430.0	90.8	46.4

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^0C)	Chl *a* (µg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (µmhos)	T.P. (µg/l)	SRP (µg/l)	T.N. (µg/l)	NO_3^--N (µg/l)	NH_4^+-N (µg/l)
0 meters	7/23/91	2.6	7.7	8.9	24.8	20.5	30.4	88.7	7.8	2.8	261.9	6.1	0.3
1 meter	7/23/91			8.8	23.4								
2 meters	7/23/91		7.8	8.7	22.5	3.0	28.5	84.3	4.3	2.7		4.7	
3 meters	7/23/91			7.8	21.3								
4 meters	7/23/91		7.6	5.5	18.2	9.5	25.5	85.4	5.0	2.0		5.4	
5 meters	7/23/91			1.9	14.8								
6 meters	7/23/91		7.3	0.3	13.6	AE	28.5	88.4	6.1	2.7	350.4	7.6	10.7
bottom	7/23/91		7.2	0.2	12.2	16.0	29.4		7.8	1.5		4.7	
outflow	7/23/91						28.5	85.9	4.7	2.0	385.7	1.9	4.9
0 meters	8/7/91	2.2	7.1	4.3	22.9	19.0	27.5	85.8	7.0	1.4	394.6	NA	4.9
1 meter	8/7/91			6.4	22.9								
2 meters	8/7/91		7.4	5.1	22.9	9.5	28.5	86.2	6.4	1.9		NA	
3 meters	8/7/91			4.4	21.9								
4 meters	8/7/91		7.3	1.4	19.0	1.0	28.5	86.5	4.7	1.2		NA	
5 meters	8/7/91			0.2	15.6								
6 meters	8/7/91		7.1	0.5	13.0	65.7	36.3	100.6	14.9	3.7	438.8	NA	46.7
bottom	8/7/91		6.8	0.7	12.2	18.7	33.4	97.7	11.5	2.7		NA	
outflow	8/7/91		7.1	NS	22.3		28.5	87.2	11.7	2.2	545.0	NA	2.6

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^0C)	Chl *a* (µg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (µmhos)	T.P. (µg/l)	SRP (µg/l)	T.N. (µg/l)	NO_3^--N (µg/l)	NH_4^+-N (µg/l)
0 meters	8/22/91	2.2	8.7	7.5	26.2	27.1	26.5	94.5	6.3	5.4	469.8	1.9	28.1
1 meter	8/22/91			6.3	25.0								
2 meters	8/22/91		8.2	6.5	23.8	5.0	29.4	86.5	8.2	3.3		1.9	
3 meters	8/22/91			3.6	21.8								
4 meters	8/22/91		7.4	0.6	18.6	12.0	27.5	85.6	3.9	3.3		1.9	
5 meters	8/22/91			0.8	16.0								
6 meters	8/22/91		6.8	0.1	12.9	4.0	35.3	96.3	6.9	4.0	456.5	1.9	42.1
bottom	8/22/91		6.7	0.0	11.6	15.0	36.3	99.7	11.1	5.1		1.9	
outflow	8/22/91		NS	NS	NS		27.5	91.3	10.5	3.3	527.3	1.9	11.9
0 meters	9/4/91	1.4	7.7	9.2	20.8	<0.5	31.4	91.1	6.9	AE	527.3	4.8	7.2
1 meter	9/4/91			9.2	19.0								
2 meters	9/4/91		7.7	8.3	18.5	16.0	33.4	88.7	6.5	3.0		4.8	
3 meters	9/4/91			7.2	18.0								
4 meters	9/4/91		7.7	4.9	17.8	12.0	31.4	88.8	7.5	2.7		3.7	
5 meters	9/4/91			2.1	14.9								
6 meters	9/4/91		7.2	1.9	12.9	AE	34.4		9.7	AE	456.5	2.7	65.3
bottom	9/4/91		6.9	1.9	12.2	16.0	#N/A		7.6	2.8		3.2	
outflow	9/4/91		7.2	9.2	23.0		37.3	90.1	7.4	4.6	500.8	6.9	16.5

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^0C)	Chl *a* (µg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (µmhos)	T.P. (µg/l)	SRP (µg/l)	T.N. (µg/l)	NO_3^--N (µg/l)	NH_4^+-N (µg/l)
0 meters	9/18/91	4.3	8.3	8.4	20.2	1.3	32.4	89.5	2.9	2.2	403.4	4.8	6.1
1 meter	9/18/91			8.7	18.8								
2 meters	9/18/91		7.9	8.4	18.1	2.7	33.4	88.9	5.5	3.0		4.8	
3 meters	9/18/91			7.6	17.8								
4 meters	9/18/91		7.6	6.4	17.3	2.7	32.4	93.0	6.8	3.6		3.7	
5 meters	9/18/91			1.1	16.1								
6 meters	9/18/91		7.2	1.1	13.6	18.7	42.2	102.2	8.2	2.7	465.4	4.8	75.7
outflow	9/18/91		7.9	8.6	21.8		32.4	90.4	5.7	3.6	474.2	3.7	13.6
0 meters	10/6/91	3.2	7.1	8.0	16.2	3.0	33.4	93.1	5.1	1.7	390.2	4.8	8.4
1 meter	10/6/91			7.8	15.9								
2 meters	10/6/91		7.4	7.6	15.8	8.0	33.4	95.0	4.4	1.6		16.0	
3 meters	10/6/91			7.6	15.5								
4 meters	10/6/91		7.2	6.3	15.2	1.0	34.4	92.6	4.4	2.0		8.5	
5 meters	10/6/91			6.6	15.2								
6 meters	10/6/91		7.4	4.6	15.2	AE	37.3	95.7	6.8	1.7	461.0	10.2	81.5
outflow	10/6/91		7.4	8.8	16.2		34.4	92.6	10.4	1.7	385.7	18.7	8.4
0 meters	10/19/91	3.0	7.1	7.8	15.8	5.5	34.4	77.7	4.8	1.5	425.6	3.7	14.2
1 meter	10/19/91			7.9	15.0								
2 meters	10/19/91		7.0	7.7	14.6	5.0	34.4	82.3	9.7	1.6		4.8	
3 meters	10/19/91			7.4	14.2								
4 meters	10/19/91		7.0	7.4	13.9	4.0	33.4	82.1	7.2	2.0		4.8	
5 meters	10/19/91			7.2	13.9								
6 meters	10/19/91		6.9	7.0	13.6	5.0	34.4	82.1	4.2	1.9	368.1	4.8	11.9
outflow	10/19/91		NS	7.8	19.2		34.4	83.0	15.0	4.3		6.9	3.8

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^0C)	Chl *a* (μg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (μmhos)	T.P. (μg/l)	SRP (μg/l)	T.N. (μg/l)	NO_3^--N (μg/l)	NH_4^+-N (μg/l)
0 meters	11/16/91	3.5	8.9	9.4	6.2	10.0	30.6	94.1	7.1	1.9	465.4	9.1	9.6
1 meter	11/16/91			9.1	7.6								
2 meters	11/16/91		7.8	9.1	7.8	12.7	29.6	95.1	11.5	1.9		11.2	
3 meters	11/16/91			9.3	7.8								
4 meters	11/16/91		7.6	9.4	7.8	8.7	30.6	93.2	11.0	1.8		6.9	
5 meters	11/16/91			9.4	7.9								
6 meters	11/16/91		7.6	9.4	7.9	15.4	30.6	89.6	14.1	1.5	421.1	8.5	6.1
pond A (I-2)	11/16/91		7.2	9.6	8.1		125.5	478.0	39.6	1.2	686.6	266.9	6.1
pond A' (I-1)	11/16/91		7.2	7.3	8.2		55.1	413.0	83.0	2.6	580.4	154.6	6.1
outflow	11/16/91		8.8	9.1	6.2		30.6	85.6	110.0	4.2	1,730.5	11.2	30.5
0 meters	12/21/91	4.2	8.3	11.8	5.2	3.0	33.7	102.1	10.6	5.1	571.5	225.2	44.4
1 meter	12/21/91			11.5	4.8								
2 meters	12/21/91		7.5	11.5	4.2	2.5	30.6	94.3	12.1	2.1		234.8	
3 meters	12/21/91			11.4	4.2								
4 meters	12/21/91		7.5	11.4	4.1	3.0	33.7	92.2	12.6	1.7		228.4	
5 meters	12/21/91			11.3	4.1								
6 meters	12/21/91		7.3	11.2	4.1	3.5	32.6	92.3	12.1	2.0	571.5	232.7	30.5
pond A (I-2)	12/21/91		7.1	11.8	7.0		120.4	517.0	21.1	2.1	668.9	309.7	14.2
pond A' (I-1)	12/21/91		7.3	11.9	6.8		73.4	228.0	12.1	3.8	739.6	480.9	6.1
outflow	12/21/91		7.4	11.8	7.0		24.5	91.5	12.4	2.3	615.8	239.1	98.3

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^{0}C)	Chl *a* (µg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (µmhos)	T.P. (µg/l)	SRP (µg/l)	T.N. (µg/l)	NO_3^--N (µg/l)	NH_4^+-N (µg/l)
0 meters	1/18/92	3.7	7.9	12.2	4.2	6.4	30.6	83.3	9.9	1.9	580.4	272.0	25.8
1 meter	1/18/92			12.2	4.2								
2 meters	1/18/92		7.7	12.6	4.1	5.6	28.6	81.3	15.3	1.8		262.0	
3 meters	1/18/92			12.6	4.1								
4 meters	1/18/92		7.7	12.8	4.1	6.0	29.6	83.3	10.9	1.9		364.3	
5 meters	1/18/92			13.4	4.1								
6 meters	1/18/92		7.6	13.4	4.1	2.4	27.5	83.9	15.5	2.7	580.4	257.0	25.8
bottom	1/18/92		7.6	13.6	4.0	6.0	28.6	87.4	11.8	2.7		683.7	
pond A (I-2)	1/18/92		7.4	11.9	6.2		151.0	509.0	14.6	1.9	810.4	252.0	16.5
pond A' (I-1)	1/18/92		7.5	10.9	6.1		49.0	161.0	13.5	3.2	721.9	456.6	6.1
outflow	1/18/92		7.9	11.8	4.2		60.2	81.7	11.0	2.1	651.2	SL	42.1
0 meters	2/22/92	3.6	7.3	11.9	7.2	8.0	26.5	83.6	10.8	2.9	775.1	447.4	NA
1 meter	2/22/92			12.1	7.2								
2 meters	2/22/92		7.0	12.2	7.2	9.5	26.5	81.4	12.7	4.2		453.2	
3 meters	2/22/92			12.2	7.0								
4 meters	2/22/92		6.9	12.6	6.9	7.0	27.5	83.5	10.3	1.3		464.9	
5 meters	2/22/92			12.8	6.8								
6 meters	2/22/92		6.8	13.8	6.8	6.5	27.5	86.8	9.2	1.2	686.6	459.1	NA
bottom	2/22/92		6.9	12.8	6.8	4.5	27.5	85.0	13.5	1.1		459.1	
pond A (I-2)	2/22/92		7.0	13.6	8.4		178.5	480.0	16.0	0.9	898.9	611.4	NA
pond A' (I-1)	2/22/92		7.0	12.2	8.2		43.9	129.0	14.9	4.0	1,235.1	1027.4	NA
outflow	2/22/92		7.2	11.0	7.3		21.4	71.7	21.1	2.8	969.7	470.8	NA

Depth	Date	Secchi (m)	pH	D. O. (mg/l)	Temp (^0C)	Chl *a* (µg/l)	Alk. $CaCO_3$ (mg/l)	Conduct (µmhos)	T.P. (µg/l)	SRP (µg/l)	T.N. (µg/l)	NO_3^--N (µg/l)	NH_4^+-N (µg/l)
0 meters	3/19/92	3.6	7.3	10.8	11.8	5.5	27.5	76.5	13.3	4.0	NA	822.3	NA
1 meter	3/19/92			10.6	10.7								
2 meters	3/19/92		7.3	9.2	10.2	9.5	28.6	75.9	12.0	1.2		880.9	
3 meters	3/19/92			6.4	9.9								
4 meters	3/19/92		7.2	4.4	9.4	14.0	27.5	76.7	11.0	0.8		1162.2	
5 meters	3/19/92			4.4	9.1								
6 meters	3/19/92		7.2	3.6	8.8	7.0	28.6	77.0	10.8	1.0	NA	359.5	NA
bottom	3/19/92		7.1	3.5	8.8	7.0	28.6	77.3	11.4	0.6		336.0	
pond A (I-2)	3/19/92		7.1	10.8	10.9		193.8	521.0	15.0	0.6	NA	259.9	NA
pond A' (I-1)	3/19/92		7.3	11.2	10.6		61.2	167.0	37.5	3.0	NA	640.7	NA
outflow	3/19/92		7.2	10.2	11.9		27.5	72.7	12.0	1.4	NA	382.9	NA

LAKE 12 WATER COLUMN (1991-1992)

Date	Depth	Concentration (mg/L) Na	K	Mg	Ca	Fe	Al	Mn	Cl	SO4
May-91	Surface	5.520	0.600	2.600	5.900	0.110	0.110	0.021		
	2m	5.420	0.410	2.550	5.820	0.110	0.100	0.021		
	4m	5.430	0.228	2.560	5.790	0.080	0.080	0.020		
	6m	5.480	0.510	2.600	5.940	0.069	0.100	0.053		
	Bottom	5.430	0.300	2.680	6.130	0.090	0.100	0.086		
Aug-91	Surface	6.080	0.490	2.860	9.750	0.460	0.150	0.050	2.02	6.17
	2m	5.380	0.350	2.530	6.060	0.430	0.090	0.055	3.38	5.76
	4m	6.230	0.510	2.770	8.640	0.210	0.110	0.156	2.04	6.54
	6m	5.610	0.540	3.230	9.650	2.330	0.130	1.886	2.20	5.56
	Bottom	4.300	0.280	2.490	6.830	1.370	0.100	1.446	3.08	4.94
Oct-91	Surface	6.000	0.480	2.860	6.690	0.360	0.110	0.106	1.86	5.18
	2m	5.990	1.018	2.900	6.820	0.370	0.150	0.126	1.98	5.08
	4m	5.850	0.600	2.890	6.930	0.580	0.130	0.266	3.34	4.83
	6m	6.070	0.700	3.140	7.750	1.450	0.140	0.746	2.36	4.74
	Bottom									
Jan-92	Surface	6.160	0.550	2.910	6.590	0.100	0.130	0.009	2.54	7.65
	2m	5.830	0.490	2.760	6.360	0.140	0.100	0.016	2.71	7.67
	4m	5.870	0.760	2.750	6.360	0.100	0.100	0.010	2.33	7.35
	6m	5.820	0.710	2.760	6.460	0.110	0.100	0.018	2.51	7.64
	Bottom	6.070	0.850	2.930	6.860	0.110	0.100	0.026	2.91	7.64
Mar-92	Surface								3.27	9.24
	2m								3.57	9.57
	4m								3.85	9.69
	6m								2.09	9.11
	6mr								2.10	9.06
	Bottom								2.08	9.14
	Inlet (003)								1.47	22.69
	Inlet (010)								1.70	125.25
	Outlet								2.03	7.89

Appendix 2

Lake Twelve Phytoplankton Data

Concentration Factor: 10
Magnification: 20X
# of Grids: 50

Sample Date	**29-Mar-91**		
Depth (m)	**2**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	6	150	0.04
Dinobryon	18	500	0.37
Melosira	12	1500	0.58
		total Volume (mm3 l-1)	0.98
		B-G Volume (mm3 l-1)	0.00
		percent B-G	0.00

Depth (m)	**6**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	198	150	1.21
Chroococcus	1	180	0.01
Rhodomonas	1	141	0.01
Tetrahaedron	1	220	0.01
		total Volume (mm3 l-1)	1.23
		B-G Volume (mm3 l-1)	0.01
		percent B-G	0.59

Sample Date	**14-Apr-91**		
Depth (m)	**2**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	115	150	0.70
Cryptomonas	5	3300	0.67
Melosira	4	1500	0.19
Pediastrum	1	5461	0.22
Rhodomonas	2	141	0.01
		total Volume (mm3 l-1)	1.80
		B-G Volume (mm3 l-1)	0.00
		percent B-G	0.00

Depth (m)	**6**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	102	150	0.62
Dinobryon	20	500	0.41
Navicula	1	1550	0.06
Rhodomonas	1	141	0.01
		total Volume (mm3 l-1)	1.10
		B-G Volume (mm3 l-1)	0.00
		percent B-G	0.00

Sample Date	**27-Apr-91**		
Depth (m)	**2**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	38	150	0.23
Dinobryon	16	500	0.33
Fragillaria	8	1000	0.33
Rhodomonas	1	141	0.01
Selenastrum	5	183	0.04
		total Volume (mm3 l-1)	0.93
		B-G Volume (mm3 l-1)	0.00
		percent B-G	0.00

Depth (m)	**6**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	68	150	0.42
Chroococcus	1	180	0.01
Dinobryon	158	500	3.22
Fragillaria	6	1000	0.24
Navicula	2	1550	0.13
Rhodomonas	2	141	0.01
Staurastrum	1	20000	0.82
		total Volume (mm3 l-1)	4.85
		B-G Volume (mm3 l-1)	0.01
		percent B-G	0.15

Sample Date	26-May-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	16	150	0.10
Chroococcus	2	180	0.01
Cosmarium	1	1000	0.04
Dinobryon	9	500	0.18
Pediastrum	1	5461	0.22
Rhodomonas	6	141	0.03
		total Volume (mm3 l-1)	0.59
		B-G Volume (mm3 l-1)	0.01
		percent B-G	2.47

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Ankistrodesmus	2	265	0.02
Aphanacapsa el.	6	8	0.00
Ceratium	9	55000	20.20
Dinobryon	7	500	0.14
Fragillaria	4	1000	0.16
Melosira	4	1500	0.19
		total Volume (mm3 l-1)	20.73
		B-G Volume (mm3 l-1)	0.00
		percent B-G	0.01

Sample Date	11-Jun-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Ceratium	2	55000	4.49
Chlamydomonas	5	150	0.03
Cosmarium	2	1000	0.08
Dinobryon	11	500	0.22
Fragillaria	2	1000	0.08
Pediastrum	1	5461	0.22
Rhodomonas	2	141	0.01
Scenedesmus	4	500	0.08

total Volume (mm3 l-1)	5.22
B-G Volume (mm3 l-1)	0.00
percent B-G	0.00

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Ceratium	4	55000	8.98
Chlamydomonas	9	150	0.06
Cosmarium	3	1000	0.12
Dinobryon	26	500	0.53
Rhodomonas	2	141	0.01
Selenastrum	1	183	0.01

total Volume (mm3 l-1)	9.71
B-G Volume (mm3 l-1)	0.00
percent B-G	0.00

Sample Date	25-Jun-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Anabaena II	54	392	0.86
Chlamydomonas	34	150	0.21
Chroococcus	1	180	0.01
Closterium	26	1413	1.50
Coelosphaerium	7	3143	0.90
Fragillaria	1	1000	0.04
Pediastrum	1	5461	0.22
Rhodomonas	3	141	0.02
Scenedesmus	9	500	0.18
		total Volume (mm3 l-1)	3.94
		B-G Volume (mm3 l-1)	1.77
		percent B-G	44.92

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	39	150	0.24
Cryptomonas	1	3300	0.13
Dinobryon	9	500	0.18
Pediastrum	1	5461	0.22
		total Volume (mm3 l-1)	0.78
		B-G Volume (mm3 l-1)	0.00
		percent B-G	0.00

Sample Date	8-Jul-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Anabaena II	12	392	0.19
Chlamydomonas	18	150	0.11
Cosmarium	4	1000	0.16
Dinobryon	199	500	4.06
Gleocapsa	5	268	0.05
Gloeocystis	10	120	0.05
Pediastrum	1	5461	0.22
Scenedesmus	5	500	0.10

total Volume (mm3 l-1)	4.95
B-G Volume (mm3 l-1)	0.25
percent B-G	5.05

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Anabaena II	23	392	0.37
Aphanacapsa el.	2	8	0.00
Chlamydomonas	40	150	0.24
Crucigenia	7	224	0.06
Dinobryon	24	500	0.49
Gleocapsa	15	268	0.16
Gloeocystis	75	120	0.37
Pediastrum	1	5461	0.22
Scenedesmus	4	500	0.08
Stephanodiscus	3	100000	12.24

total Volume (mm3 l-1)	14.25
B-G Volume (mm3 l-1)	0.54
percent B-G	3.79

Sample Date	24-Jul-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Anabaena II	15	392	0.24
Chlamydomonas	16	150	0.10
Cosmarium	1	1000	0.04
Dinobryon	642	500	13.10
Gleocapsa	2	268	0.02
Gloeocystis	4	120	0.02
Pediastrum	4	5461	0.89
Rhodomonas	4	141	0.02
		total Volume (mm3 l-1)	14.44
		B-G Volume (mm3 l-1)	0.26
		percent B-G	1.80

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Anabaena II	10	392	0.16
Chlamydomonas	81	150	0.50
Cosmarium	4	1000	0.16
Cryptomonas	3	3300	0.40
Dinobryon	427	500	8.71
Gleocapsa	8	268	0.09
Gloeocystis	20	120	0.10
Navicula	1	1550	0.06
		total Volume (mm3 l-1)	10.18
		B-G Volume (mm3 l-1)	0.24
		percent B-G	2.36

Sample Date	7-Aug-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Cosmarium	11	1000	0.45
Dinobryon	1420	500	28.98
Gloeocystis	55	120	0.27
Pediastrum	1	5461	0.22
Rhodomonas	13	141	0.07
Scenedesmus	9	500	0.18

total Volume (mm3 l-1)	30.18
B-G Volume (mm3 l-1)	0.00
percent B-G	0.00

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Ankistrodesmus	29	265	0.31
Aphanizomenon	113	990	3.59
Chlamydomonas	98	150	0.60
Cosmarium	15	1000	0.61
Dinobryon	5434	500	110.90
Gleocapsa	50	268	0.55
Gloeocystis	328	120	1.61

total Volume (mm3 l-1)	118.17
B-G Volume (mm3 l-1)	4.14
percent B-G	3.50

Sample Date	4-Sep-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Ankistrodesmus	6	265	0.06
Cryptomonas	3	3300	0.40
Dinobryon	461	500	9.41
Gleocapsa	12	268	0.13
Gloeocystis	149	120	0.73
Melosira	28	1500	1.35
Navicula	6	1550	0.38
Pediastrum	1	5461	0.22
Rhodomonas	4	141	0.02
Scenedesmus	18	500	0.37
		total Volume (mm3 l-1)	13.08
		B-G Volume (mm3 l-1)	0.13
		percent B-G	1.00

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Aphanizomenon	26	990	0.83
Chlamydomonas	323	150	1.98
Cosmarium	1	1000	0.04
Dinobryon	3324	500	67.84
Gleocapsa	155	268	1.70
Gloeocystis	385	120	1.89
Melosira	155	1500	7.46
Navicula	1	1550	0.06
		total Volume (mm3 l-1)	81.78
		B-G Volume (mm3 l-1)	2.52
		percent B-G	3.08

Sample Date	18-Sep-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	212	150	1.30
Dinobryon	362	500	7.39
Fragillaria	3	1000	0.12
Gleocapsa	36	268	0.39
Gloeocystis	4	120	0.02
Melosira	18	1500	0.87
Scenedesmus	94	500	1.92

total Volume (mm3 l-1)	12.01
B-G Volume (mm3 l-1)	0.39
percent B-G	3.28

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Aphanizomenon	116	990	3.68
Chlamydomonas	464	150	2.84
Cosmarium	1	1000	0.04
Dinobryon	3386	500	69.10
Gleocapsa	554	268	6.06
Melosira	103	1500	4.96
Navicula	2	1550	0.13
Scenedesmus	1	500	0.02

total Volume (mm3 l-1)	86.83
B-G Volume (mm3 l-1)	9.75
percent B-G	11.22

Sample Date	6-Oct-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	70	150	0.43
Coelosphaerium	296	3143	37.97
Cosmarium	1	1000	0.04
Dinobryon	494	500	10.08
Gloeocystis	55	120	0.27
Melosira	24	1500	1.15
Navicula	3	1550	0.19
Rhodomonas	1	141	0.01
Selenastrum	17	183	0.13
		total Volume (mm3 l-1)	50.27
		B-G Volume (mm3 l-1)	37.97
		percent B-G	75.54

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Ankistrodesmus	23	265	0.25
Asterionella	51	1000	2.08
Chlamydomonas	118	150	0.72
Cosmarium	7	1000	0.29
Dinobryon	1779	500	36.31
Gleocapsa	125	268	1.37
Gloeocystis	277	120	1.36
Melosira	177	1500	8.52
Navicula	37	1550	2.34
Scenedesmus	40	500	0.82
Stephanodiscus	39	100000	159.18
		total Volume (mm3 l-1)	213.23
		B-G Volume (mm3 l-1)	1.37
		percent B-G	0.64

Sample Date	19-Oct-91		
Depth (m)	**2**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	32	150	0.20
Cosmarium	29	1000	1.18
Cryptomonas	23	3300	3.10
Dinobryon	107	500	2.18
Fragillaria	1	1000	0.04
Navicula	3	1550	0.19
		total Volume (mm3 l-1)	6.89
		B-G Volume (mm3 l-1)	0.00
		percent B-G	0.00

Depth (m)	**6**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Ankistrodesmus	1	265	0.01
Chlamydomonas	54	150	0.33
Closterium	8	1413	0.46
Coelosphaerium	1	3143	0.13
Cryptomonas	16	3300	2.16
Dinobryon	140	500	2.86
Fragillaria	5	1000	0.20
Gleocapsa	36	268	0.39
Gloeocystis	28	120	0.14
Lyngbyia	2	6000	0.49
Melosira	13	1500	0.63
Navicula	4	1550	0.25
Scenedesmus	2	500	0.04
Stephanodiscus	3	100000	12.24
		total Volume (mm3 l-1)	20.33
		B-G Volume (mm3 l-1)	1.01
		percent B-G	4.98

Sample Date	16-Nov-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	26	150	0.16
Cryptomonas	14	3300	1.89
Cyclotella	5	2497	0.51
Dinobryon	339	500	6.92
Fragillaria	1	1000	0.04
Gleocapsa	31	268	0.34
Gloeocystis	18	120	0.09
Navicula	8	1550	0.51
Scenedesmus	3	500	0.06
		total Volume (mm3 l-1)	10.51
		B-G Volume (mm3 l-1)	0.34
		percent B-G	3.23

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Aphanizomenon	6	990	0.19
Chlamydomonas	36	150	0.22
Cryptomonas	6	3300	0.81
Dinobryon	112	500	2.29
Fragillaria	3	1000	0.12
Gleocapsa	36	268	0.39
Gloeocystis	36	120	0.18
Melosira	10	1500	0.48
Navicula	1	1550	0.06
Scenedesmus	6	500	0.12
Staurastrum	4	20000	3.27
		total Volume (mm3 l-1)	8.13
		B-G Volume (mm3 l-1)	0.58
		percent B-G	7.19

Sample Date	21-Dec-91		
Depth (m)	2		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Ankistrodesmus	1	265	0.01
Chlamydomonas	9	150	0.06
Cosmarium	1	1000	0.04
Cryptomonas	9	3300	1.21
Cyclotella	3	2497	0.31
Dinobryon	65	500	1.33
Gleocapsa	7	268	0.08
Scenedesmus	2	500	0.04
Staurastrum	2	20000	1.63
		total Volume (mm3 l-1)	4.70
		B-G Volume (mm3 l-1)	0.08
		percent B-G	1.63

Depth (m)	6		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Chlamydomonas	26	150	0.16
Cosmarium	4	1000	0.16
Cryptomonas	19	3300	2.56
Dinobryon	79	500	1.61
Fragillaria	6	1000	0.24
Gleocapsa	23	268	0.25
Melosira	7	1500	0.34
Navicula	5	1550	0.32
Rhodomonas	1	141	0.01
Scenedesmus	6	500	0.12
Staurastrum	3	20000	2.45
		total Volume (mm3 l-1)	8.22
		B-G Volume (mm3 l-1)	0.25
		percent B-G	3.06

Sample Date	18-Jan-92		
Depth (m)	**2**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Coelosphaerium	20	3143	2.57
Dinobryon	8	500	0.16
Navicula	12	1550	0.76
Pediastrum	1	5461	0.22
Rhodomonas	4	5461	0.89
Staurastrum	2	20000	1.63

total Volume (mm3 l-1)	6.24
B-G Volume (mm3 l-1)	2.57
percent B-G	41.15

Depth (m)	**6**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Pediastrum	1	5461	0.22
Rhodomonas	2	141	0.01
Scenedesmus	4	500	0.08

total Volume (mm3 l-1)	0.32
B-G Volume (mm3 l-1)	0.00
percent B-G	0.00

Sample Date	**22-Feb-92**		
Depth (m)	**2**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Gloeocystis	12	120	0.06
Pediastrum	2	5461	0.45
		total Volume (mm3 l-1)	0.50
		B-G Volume (mm3 l-1)	0.00
		percent B-G	0.00

Depth (m)	**6**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Coelosphaerium	7	3143	0.90
Gleocapsa	1	268	0.01
Gloeocystis	7	120	0.03
Navicula	10	1550	0.63
		total Volume (mm3 l-1)	1.58
		B-G Volume (mm3 l-1)	0.91
		percent B-G	57.68

Sample Date	19-Mar-92		
Depth (m)	**2**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Coelosphaerium	21	3143	2.69
Dinobryon	13	500	0.27
Gloeocystis	26	120	0.13
Pediastrum	4	5461	0.89
Rhodomonas	1	141	0.01
		total Volume (mm3 l-1)	3.98
		B-G Volume (mm3 l-1)	2.69
		percent B-G	67.62

Depth (m)	**6**		
ALGA	# counted	Cell Volume	Volume (mm^3/L)
Coelosphaerium	21	3143	2.69
Gloeocystis	23	120	0.11
Pediastrum	11	5461	2.45
		total Volume (mm3 l-1)	5.26
		B-G Volume (mm3 l-1)	2.69
		percent B-G	51.23

Appendix 3

Lake Twelve Zooplankton Data (number/liter)

Date	Daphnia	Bosmina	Ceriodaphina	Eurycercus	Chydorus	Calanoida	Cyclopoida	Nauplia	Ostracoda
3/29/91	6.6	0.0	0.7	0.1	0.0	6.0	7.7	19.9	0.8
4/14/91	8.4	0.2	1.1	1.1	0.1	6.6	27.4	27.7	1.4
4/27/91	5.6	0.3	1.0	1.2	0.2	8.5	30.1	9.2	0.9
5/26/91	37.5	0.2	0.3	0.0	0.0	33.0	5.2	13.7	0.8
6/11/91	16.5	0.5	3.0	0.3	0.1	36.3	6.1	13.7	1.3
6/25/91	3.0	0.7	8.8	1.5	0.4	14.0	6.2	2.9	0.3
7/8/91	2.9	1.2	27.7	1.0	0.0	12.1	9.1	14.3	0.9
7/24/91	7.9	11.4	17.0	0.8	0.0	7.4	7.8	24.4	0.7
8/7/91	0.2	1.2	2.6	0.2	0.0	2.1	5.1	18.0	0.7
8/22/91	0.0	3.4	2.1	0.1	0.0	4.2	7.2	8.7	0.8
9/4/91	0.0	0.8	2.6	0.1	0.0	2.0	1.6	4.4	0.4
9/18/91	0.1	0.2	2.5	0.1	0.0	5.0	8.8	19.9	0.0
10/6/91	0.5	2.1	44.1	0.0	0.2	9.3	6.6	15.1	0.0
10/19/91	0.4	4.0	12.7	0.3	0.0	5.4	4.2	3.8	0.2
11/16/91	4.1	31.0	8.7	0.4	0.0	6.0	11.3	14.2	1.1
12/21/91	2.0	19.8	4.1	0.1	0.1	3.8	7.0	6.4	0.0
1/18/92	1.3	11.7	0.1	0.1	0.0	5.1	8.1	1.7	0.1
2/22/92	0.1	2.3	0.4	0.3	0.0	0.9	37.5	31.4	0.0
3/19/92	0.0	2.4	0.0	0.3	0.0	1.4	28.1	18.4	0.0
Average	5.1	4.9	7.3	0.4	0.1	8.9	11.8	14.1	0.5

Appendix 3

Lake Twelve Zooplankton Data (number/liter)

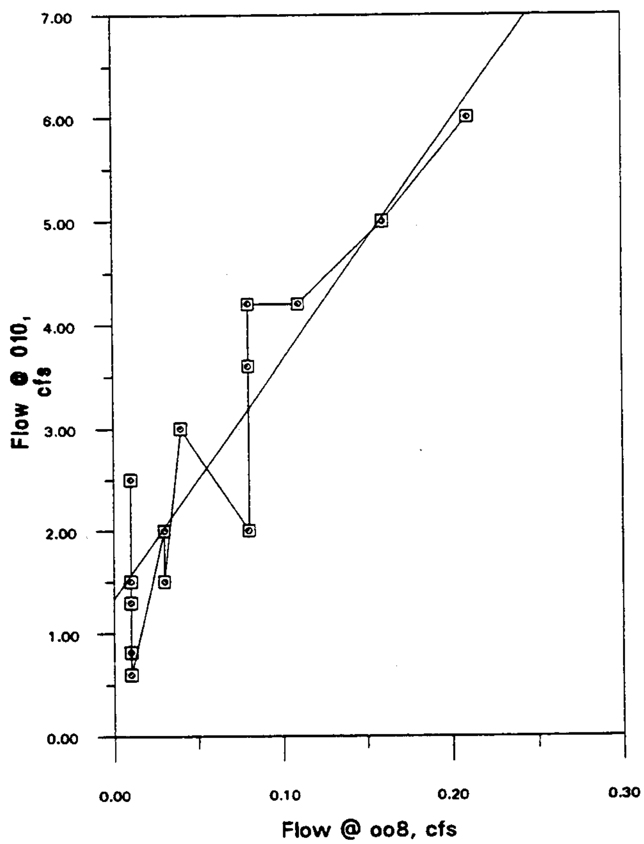
Date	Daphnia	Bosmina	Ceriodaphina	Eurycercus	Chydorus	Calanoida	Cyclopoida	Ostracoda
3/29/91	6.6	0.0	0.7	0.1	0.0	6.0	7.7	0.8
4/14/91	8.4	0.2	1.1	1.1	0.1	6.6	27.4	1.4
4/27/91	5.6	0.3	1.0	1.2	0.2	8.5	30.1	0.9
5/26/91	37.5	0.2	0.3	0.0	0.0	33.0	5.2	0.8
6/11/91	16.5	0.5	3.0	0.3	0.1	36.3	6.1	1.3
6/25/91	3.0	0.7	8.8	1.5	0.4	14.0	6.2	0.3
7/8/91	2.9	1.2	27.7	1.0	0.0	12.1	9.1	0.9
7/24/91	7.9	11.4	17.0	0.8	0.0	7.4	7.8	0.7
8/7/91	0.2	1.2	2.6	0.2	0.0	2.1	5.1	0.7
8/22/91	0.0	3.4	2.1	0.1	0.0	4.2	7.2	0.8
9/4/91	0.0	0.8	2.6	0.1	0.0	2.0	1.6	0.4
9/18/91	0.1	0.2	2.5	0.1	0.0	5.0	8.8	0.0
10/6/91	0.5	2.1	44.1	0.0	0.2	9.3	6.6	0.0
10/19/91	0.4	4.0	12.7	0.3	0.0	5.4	4.2	0.2
11/16/91	4.1	31.0	8.7	0.4	0.0	6.0	11.3	1.1
12/21/91	2.0	19.8	4.1	0.1	0.1	3.8	7.0	0.0
1/18/92	1.3	11.7	0.1	0.1	0.0	5.1	8.1	0.1
2/22/92	0.1	2.3	0.4	0.3	0.0	0.9	37.5	0.0
3/19/92	0.0	2.4	0.0	0.3	0.0	1.4	28.1	0.0
Average	5.1	4.9	7.3	0.4	0.1	8.9	11.8	0.5

Appendix 4

Lake Twelve Macrophyte Data

Trans	Sample	Depth meters	Nymphaea odorata g/m^2	Brasenia entapetla g/m^2	Myriophyllum spicatum g/m^2	Elodea canadensis g/m^2	No ID (grass like) g/m^2	Potamogeton nodosus g/m^2
1	1	0.50	80.8					
1	2	1.00	197.3					
1	3	1.50	125.1				16.9	
1	4	1.50	57.6					
1	5	2.00			18.0			
1	6	2.50			39.2			
1	7	2.25			9.4			
1	8	2.25			31.4	10.6		
1	9	3.50	44.7		5.1			
1	10	4.25					25.9	
2	11	1.50	70.6					
2	12	1.75	109.8					
2	13	2.25			13.3			80.4
2	14	2.75					30.2	
2	15	2.75			73.7			
2	16	3.25			32.5			
2	17	4.00			4.3			
2	18	5.00						
3	19	1.75			11.0	2.7		
3	20	1.75			30.2	8.2		
3	21	1.25	26.7	26.7	20.0			
3	22	1.25	64.3					
3	23	1.75			41.6	0.8		
3	24	1.75	149.0					
3	25	1.00	104.3					
3	26	1.00	63.9					
3	27	1.50		148.6				
3	28	1.50		3.9			2.0	6.7
3	29	1.50		98.8				
3	30	1.75	103.5	46.7				
4	31	3.50			42.0			
4	32	3.75			10.2			
4	33	3.00					12.5	
4	34	3.00			40.8			
4	35	3.25					5.9	
4	36	5.25					44.3	
4	37	1.50	44.3				3.1	
4	38	2.25			12.2			
4	39	6.25						
4	40	4.75					4.3	
average			88.7	64.9	25.6	5.6	16.1	43.5

Appendix 5
Flow Correlation Equation 008/010



$y = 1.33 + 23.20x \quad r^2 = 0.86$

Date	008 Flow (cfs)	010 Flow (cfs)	Date	008 Flow (cfs)	010 Flow (cfs)
1/16/92	.08	2.0	2/1/92	.08	3.6
1/17/92	.01	1.3	2/2/92	.08	4.2
1/23/92	.08	3.6	2/3/92	.04	3.0
1/24/92	.03	2.0	2/5/92	.01	1.5
1/25/92	.01	2.5	2/6/92	.01	1.3
1/26/92	.01	2.5	2/7/92	.01	.82
1/28/92	.21	6.0	2/8/92	.03	1.5
1/29/92	.21	6.0	2/9/92	.01	1.3
1/30/92	.16	5.0	2/10/92	.01	.60
1/31/92	.11	4.2			

Appendix 6

Lake Twelve Ground Water Data

Lake Twelve ground water flow calculations

All elevations relative to staff gauge on Esko Cate's dock

Ericksons	MW-1 Well Head Elev, m	Depth to Water, m	Water Elev, m	MW-2 Well Head Elev, m	Depth to Water, m	Water Elev, m	dH (m)	dL (m)	k (cm/sec)	v (cm/sec)	v (m/day)	Aquifer Thickness, meters	Area of Dsicharge, m^2	Flow, m^3/day	Flow, m^3/month
10/13/91	2.78	1.59	1.19	10.15	2.46	7.69	6.50	30.45	1.67E-05	3.57E-06	3.08E-03	3.76	542	1.7	51.7
11/16/91	2.78	0.95	1.83	10.15	1.82	8.33	6.50	30.45	1.67E-05	3.56E-06	3.08E-03	4.40	634	2.0	58.5
12/21/91	2.78	0.83	1.95	10.15	1.85	8.30	6.34	30.45	1.67E-05	3.48E-06	3.01E-03	4.37	629	1.9	58.6
1/18/92	2.78	0.80	1.97	10.15	1.80	8.34	6.37	30.45	1.67E-05	3.49E-06	3.02E-03	4.42	636	1.9	59.6
2/22/92	2.78	0.74	2.04	10.15	1.75	8.40	6.36	30.45	1.67E-05	3.49E-06	3.01E-03	4.47	644	1.9	56.2
3/19/92	2.78	1.26	1.51	10.15	1.87	8.28	6.77	30.45	1.67E-05	3.71E-06	3.21E-03	4.35	627	2.0	62.3

Area of discharge=length of west shore/2 * depth of ground water=144m*depth of aquifer (m)
Aquifer thickness= depth of well - depth to water=6.22m-depth to water in MW-1

Rods	MW-4 Well Head Elev, m	Depth to Water, m	Water Elev, m	MW-3 Well Head Elev, m	Depth to Water, m	Water Elev, m	dH (m)	dL (m)	k (cm/sec)	v (cm/sec)	v (m/day)	Aquifer Thickness, meters	Area of Dsicharge, m^2	Flow, m^3/day	Flow, m^3/month
10/13/91	7.06	2.11	4.95	3.77	2.61	1.16	3.79	48.83	1.67E-05	1.30E-06	1.12E-03	2.94	423	0.5	14.7
11/16/91	7.06	1.96	5.10	3.77	2.63	1.14	3.96	48.83	1.67E-05	1.36E-06	1.17E-03	2.92	420	0.5	14.8
12/21/91	7.06	1.68	5.39	3.77	2.24	1.53	3.86	48.83	1.67E-05	1.32E-06	1.14E-03	3.31	476	0.5	16.8
1/18/92	7.06	1.58	5.48	3.77	2.04	1.72	3.76	48.83	1.67E-05	1.29E-06	1.11E-03	3.51	505	0.6	17.4
2/22/92	7.06	1.37	5.69	3.77	1.93	1.84	3.85	48.83	1.67E-05	1.32E-06	1.14E-03	3.62	521	0.6	17.2
3/19/92	7.06	1.75	5.31	3.77	2.16	1.61	3.71	48.83	1.67E-05	1.27E-06	1.09E-03	3.39	488	0.5	16.6

Area of discharge=length of west shore/2 * depth of ground water=144m*depth of aquifer (m)
Aquifer thickness= depth of well - depth to water=5.55m-depth to water in MW-3

Esko	MW-5 Well Head Elev, m	Depth to Water, m	Water Elev, m	MW-6 Well Head Elev, m	Depth to Water, m	Water Elev, m	dH (m)	dL (m)	k (cm/sec)	v (cm/sec)	v (m/day)	Aquifer Thickness, meters	Area of Dsicharge, m^2	Flow, m^3/day	Flow, m^3/month
10/13/91	7.87	3.20	4.66	3.03	1.69	1.35	3.32	36.52	8.73E-06	7.93E-07	6.85E-04	1.51	1163	0.8	24.7
11/16/91	7.87	2.05	5.82	3.03	1.40	1.63	4.18	36.52	8.73E-06	1.00E-06	8.64E-04	1.80	1382	1.2	35.8
12/21/91	7.87	1.88	5.99	3.03	1.33	1.70	4.28	36.52	8.73E-06	1.02E-06	8.85E-04	1.87	1436	1.3	39.4
1/18/92	7.87	1.52	6.35	3.03	1.62	1.41	4.94	36.52	8.73E-06	1.18E-06	1.02E-03	1.58	1214	1.2	38.4
2/22/92	7.87	1.43	6.44	3.03	1.12	1.91	4.53	36.52	8.73E-06	1.08E-06	9.35E-04	2.08	1597	1.5	43.3
3/19/92	7.87	1.57	6.30	3.03	1.20	1.83	4.47	36.52	8.73E-06	1.07E-06	9.22E-04	2.00	1536	1.4	43.9

Area of discharge=length of south shore * depth of ground water=768m*depth of aquifer (m)
Aquifer thickness= depth of well - depth to water=3.2m-depth to water in MW-6

Wetland WP-6
K=8.86 m/day, 1E-2 cm/sec
A=471 m^2

Date	Lake Level, m	Well Head Elev, m	Depth to Water, m	Water Elev, m	dH	dL	Q=KA (dH/dL), m^3/day	Flow, m^3/month
10/13/91	0.36	1.31	0.95	0.36	0.00	8.84	0.00	0.0
12/21/91	0.67	1.31	0.64	0.67	0.00	8.84	0.94	29.3
2/22/92	0.69	1.31	0.60	0.71	0.02	8.84	9.44	273.8
6/2/92	0.53	1.31	0.73	0.58	0.05	8.84	23.60	708.1

Total Discharge to Lake = 2* South Shore (MW-6)+Ericksons(MW-1)+Rods(MW-3)+ Wetland Discharge (WP-6)

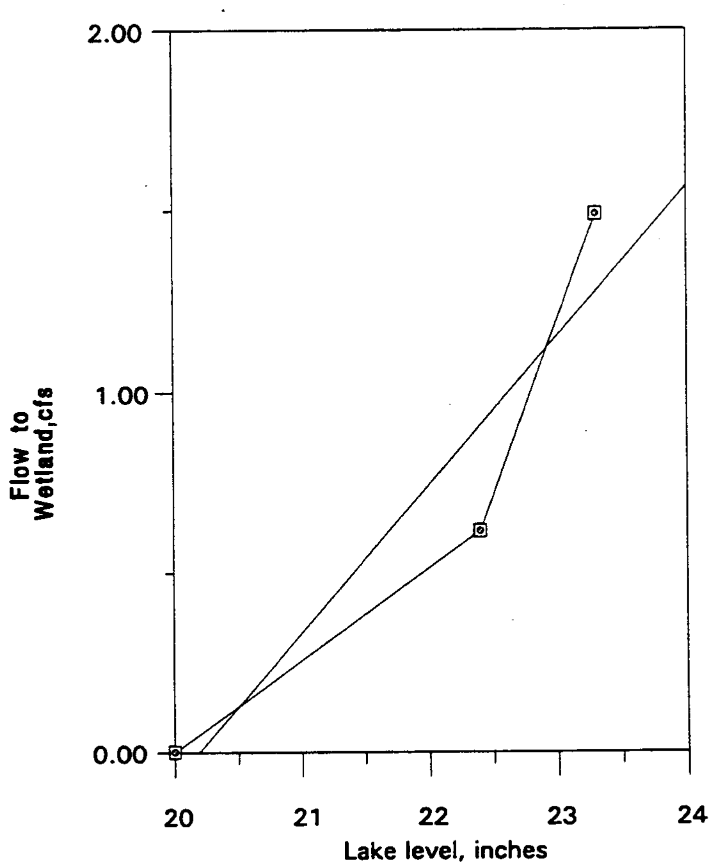
Date	South Shore	North Shore	Ericksons	Rods	Wet Land	Total GW
Oct-91	25	25	52	15	0	116
Nov-91	36	36	59	15	15	160
Dec-91	39	39	59	17	29	184
Jan-92	38	38	60	17	152	305
Feb-92	43	43	56	17	274	434
Mar-92	44	44	62	17	274	441

LAKE 12 WELL ANALYSIS: Cl, SO4, mg/L (1991-1992)

	October		November		December		January		February		March	
Well	Cl	SO4	Cl	SO4	Cl	SO4	Cl	SO4	Cl	SO4	Cl	SO4
MW 1	2.08	1.31	2.39	0.63	3.49	2.95	4.48	5.92	2.68	0.96	2.19	3.30
MW 1 Rep									5.13	0.93	4.01	5.28
MW 2	5.04	2.53	2.06	0.84	2.16	1.15	2.92	1.77	6.13	2.26	4.03	6.18
MW 3	6.04	3.03	4.25	1.07	2.53	4.31	2.39	5.93	10.21	2.97	2.45	2.70
MW 3 Rep							2.20	5.70				
MW 4	4.97	2.45	3.70	0.69	2.44	0.28	2.06	0.41	2.44	0.28	2.31	0.30
MW 4 Rep					2.27	0.23						
MW 5	3.14	34.80	3.78	19.21	3.30	10.81	5.44	9.74	2.98	8.75	2.85	8.57
MW 6	12.99	13.14	13.68	15.87	13.30	14.42	10.02	12.53	9.15	6.95	5.41	8.69
WP 1	2.60	3.87	2.21	1.47	1.74	0.66	2.20	1.18	2.66	1.30	1.79	0.57
WP 2	3.31	4.88	2.03	1.38	1.77	0.90	1.74	0.47	1.79	0.46	1.57	0.43
WP 3	1.24	1.58	3.80	2.87	1.89	2.57	1.73	0.50	1.98	0.55	1.75	0.46
WP 4	41.08	0.38							26.95	0.21		
WP 5	18.52	0.05										
WP 6	2.19	0.16	1.71	0.37	0.98	0.12			3.17	0.15		
WP 7	4.64	3.66										

Appendix 7

Wetland Discharge Flow Regression



$y = -8.30 + 0.41x \quad r^2 = 0.88$

Date	Lake Level (in)	Wetland Flow (cfs)
4/23/92	23.3	1.49
5/7/92	22.4	0.62
6/15/92	20.0	0

APPENDIX 8

Lake Twelve ground water nutrient concentrations.

Date 10/13/9
1

	TP, ug/l	TN, ug/l
MW-1	AE	123.9
MW-2	AE	123.9
MW-3	AE	93.1
MW-4	AE	302.7
MW-5	AE	352.0
MW-6	AE	234.9
WP-1	AE	197.9
WP-2	AE	130.0
WP-3	AE	1104.2
WP-4	AE	1350.8
WP-5	AE	SL
WP-6	AE	1067.2
WP-7	AE	1289.2
Browns	AE	167.0

Date 11/16/9
1

	TP, ug/l	TN, ug/l
MW-1	AE	142.4
MW-2	AE	154.7
MW-3	AE	160.9
MW-4	AE	154.7
MW-5	AE	709.6
MW-6	AE	204.0
WP-1	AE	191.7
WP-2	AE	1301.5
WP-3	AE	500.0
WP-4	AE	685.0
WP-5	AE	SL
WP-6	AE	86.9
WP-7	AE	SL
Browns	AE	160.9

APPENDIX 8

Date 12/21/91	TP, ug/l	TN, ug/l
MW-1	11.1	154.7
MW-2	69.8	167.0
MW-3	87.8	216.4
MW-4	105.7	259.5
MW-5	155.5	185.5
MW-6	134.8	367.7
WP-1	41.4	493.8
WP-2	30.3	247.2
WP-3	34.0	376.7
WP-4	NS	NS
WP-5	NS	NS
WP-6	133.8	493.8
WP-7	NS	NS
Browns	20.7	231.8

Date 1/18/92	TP, ug/l	TN, ug/l
MW-1	64.9	117.7
MW-2	56.6	111.5
MW-3	75.4	139.3
MW-4	51.6	265.7
MW-5	100.8	537.0
MW-6	50.4	185.5
WP-1	16.4	179.4
WP-2	26.6	413.7
WP-3	20.7	130.0
WP-4	NS	NS
WP-5	NS	NS
WP-6	NS	NS
WP-7	NS	NS
Browns	11.1	142.4

Date 2/22/92	TP, ug/l	TN, ug/l
MW-1	67.4	127.0
MW-2	171.5	93.1
MW-3	55.0	93.1
MW-4	37.7	173.2
MW-5	159.8	345.8
MW-6	25.6	543.1
WP-1	96.1	222.5
WP-2	3.7	197.9
WP-3	7.7	463.0
WP-4	20.1	438.3
WP-5	NS	NS
WP-6	108.2	555.5
WP-7	NS	NS
Browns	8.6	250.3

Date 3/19/92	TP, ug/l	TN, ug/l
MW-1	86.4	93.1
MW-2	21.0	68.4
MW-3	38.7	99.2
MW-4	84.7	99.2
MW-5	64.0	228.7
MW-6	31.9	648.0
WP-1	26.1	241.0
WP-2	18.7	SL
WP-3	17.4	456.8
WP-4	NS	NS
WP-5	NS	NS
WP-6	NS	NS
WP-7	NS	NS
Browns	41.8	136.2

Storm water nutrient concentration data

Date	Location	TP (µg/l)	SRP (µg/l)
1/30/92	1	17.4	2.1
	2	15.3	4.2
	3	24.7	2.7
	4	9.9	2.9
	5	28.9	7.0
3/17/92	1	24.7	1.7
	2	20.1	2.4
	3	39.7	2.8
	4	11.8	1.9
	5	43.1	8.4

Composite Precipitation Data

Location	TP (µg/l)
1	26.3
2	11.7

Non Steady State Phosphorus, Chla and Secchi Disk Model for Lake 12

APPENDIX 9

Results Summary					
Maximum TP, ug/l	14.61	Maximum Chla, ug/l	3.50	Minimum SD, m	5.11
Summer Ave TP, ug/l	6.81	Summer Ave Chla, ug/l	1.60	Summer Ave SD, m	8.58
Annual Ave TP, ug/l	8.51	Annual Ave Chla, ug/l	1.49	Annual Ave SD, m	8.61

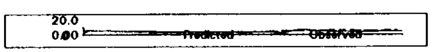
These values may be changed	
Dilution Water TP ug/l:	0
Dilution water input, mgd:	0
Percent Internal Loading:	100%
Initial P concentration, ug/l:	11.3

Fixed values	
Lake Volume, m^3:	5.98E+05
Lake Area, m^2:	1.75E+05
Average depth, m:	3.4

	4/7/91	4/14/91	4/21/91	4/28/91	5/5/91	5/12/91	5/19/91	5/26/91	6/2/91	6/9/91	6/16/91	6/23/91	6/30/91	7/7/91
1-Week Period	1	2	3	4	5	6	7	8	9	10	11	12	13	14
External Loadings														
Ground Water														
Q-m^3	76	76	76	76	76	76	76	76	61	61	61	61	61	29
TP-ug/l	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3
TP Loading-g	6	6	6	6	6	6	6	6	5	5	5	5	5	2
Precipitation														
Q-m^3	23535	5728	1998	4885	4396	4885	4352	4396	1021	1332	5508	4485	666	0
TP-ug/l	19	19	19	19	19	19	19	19	19	19	19	19	19	19
TP Loading-g	447	109	38	93	84	93	83	84	19	25	105	85	13	0
Pond A (003)														
Q-m^3	5554	1737	367	440	343	220	73	294	171	0	49	171	24	0
TP-ug/l	15	39	22	22	37	24	19	28	29	47	12	33	33	0
TP Loading-g	83	68	8	10	13	5	1	8	5	0	1	6	1	0
Pond A' (008)														
Q-m^3	1982	1199	196	122	49	0	0	0	0	0	0	0	0	0
TP-ug/l	14	14	13	20	14	0	0	0	0	0	0	0	0	0
TP Loading-g	28	17	3	2	1	0	0	0	0	0	0	0	0	0
Location 010														
Q-m^3	60264	49392	23869	18986	7594	0	0	0	0	0	0	0	0	0
TP-ug/l	11.1	40.3	21.6	21.6	21.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TP Loading-g	669	1990	516	410	164	0	0	0	0	0	0	0	0	0
Ungaged Inflow														
Q-m^3	0	0	0	0	33252	33252	33252	33252	20672	20672	20672	20672	20672	15656
TP-ug/l	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6
TP Loading-g	0	0	0	0	785	785	785	785	488	488	488	488	488	369
Total External Loading														
Q-m^3	91,411	58,132	26,506	24,509	45,710	38,433	37,753	38,018	21,926	22,065	26,288	25,389	21,424	15,685
Increased load factor 1 Toal P Ldg-g	1233	2190	570	521	1051	889	875	882	517	518	598	583	506	372
Ldg/V-ug/l	2.06	3.66	0.95	0.87	1.76	1.49	1.46	1.48	0.86	0.87	1.00	0.98	0.85	0.62
Internal Loading														
Calib. Ldg-mg/m^2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Calib. Ldg-mg	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ldg/V -ug/l	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total P LDG -ug/l	2.06	3.66	0.95	0.87	1.76	1.49	1.46	1.48	0.86	0.87	1.00	0.98	0.85	0.62
Phosphorus Outflow														
Total Q -m^3	91,411	58,132	26,506	24,509	45,710	38,433	37,753	38,018	21,926	22,065	26,288	25,389	21,424	15,685
Total Q/V (rho)	0.1529	0.0972	0.0443	0.0410	0.0764	0.0643	0.0631	0.0636	0.0367	0.0369	0.0440	0.0425	0.0358	0.0262
P Outflow -ug/l		1.10	0.53	0.45	0.80	0.63	0.59	0.58	0.32	0.32	0.37	0.35	0.29	0.21
Sedimentation														
Sigma	0.249	0.178	0.100	0.094	0.149	0.131	0.129	0.130	0.087	0.087	0.099	0.097	0.085	0.068
P Sedimented -ug/l		2.01	1.18	1.04	1.56	1.29	1.22	1.18	0.76	0.75	0.83	0.79	0.68	0.53
Mass Balance														
Delta P -ug/l	2.06	0.55	0.75	-0.63	-0.60	-0.44	-0.35	-0.28	-0.22	-0.20	-0.20	-0.16	-0.12	-0.12
Total P in lake -ug/l — Predicted	11.30	11.85	11.10	10.47	9.87	9.42	9.07	8.79	8.57	8.37	8.18	8.01	7.89	7.77
Actual Conc in lake - ug/l — Observed	5.8	11.3	#N/A	10.2	#N/A	#N/A	#N/A	9.6	#N/A	7.4	#N/A	6.5	#N/A	5.1
(pred-obs)^2	30.3	0.3		0.1				0.7		0.9		2.3		7.1
Chla, ug/l	2.31	2.49	2.24	2.04	1.85	1.72	1.62	1.54	1.47	1.42	1.37	1.32	1.29	1.26
Secchi Depth, m	6.45	6.18	6.56	6.91	7.30	7.61	7.87	8.10	8.29	8.47	8.65	8.81	8.93	9.06

	7/14/91	7/21/91	7/28/91	8/4/91	8/11/91	8/18/91	8/25/91	9/1/91	9/8/91	9/15/91	9/22/91	9/29/91	10/6/91
1-Week Period	15	16	17	18	19	20	21	22	23	24	25	26	27
External Loadings													
Ground Water													
Q-m^3	29	29	29	29	29	29	29	23	23	23	23	23	29
TP-ug/l	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3
TP Loading-g	2	2	2	2	2	2	2	2	2	2	2	2	2
Precipitation													
Q-m^3	0	710	1510	0	710	0	0	8037	222	222	0	0	0
TP-ug/l	19	19	19	19	19	19	19	19	19	19	19	19	19
TP Loading-g	0	13	29	0	13	0	0	153	4	4	0	0	0
Pond A (003)													
Q-m^3	0	0	0	0	0	0	0	0	0	0	0	0	0
TP-ug/l	0	0	0	0	0	0	0	0	0	0	0	0	0
TP Loading-g	0	0	0	0	0	0	0	0	0	0	0	0	0
Pond A' (008)													
Q-m^3	0	0	0	0	0	0	0	0	0	0	0	0	0
TP-ug/l	0	0	0	0	0	0	0	0	0	0	0	0	0
TP Loading-g	0	0	0	0	0	0	0	0	0	0	0	0	0
Location 010													
Q-m^3	0	0	0	0	0	0	0	0	0	0	0	0	0
TP-ug/l	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TP Loading-g	0	0	0	0	0	0	0	0	0	0	0	0	0
Ungaged Inflow													
Q-m^3	15656	15656	15656	5139	5139	5139	5139	4097	4097	4097	4097	4097	0
TP-ug/l	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	0
TP Loading-g	369	369	369	121	121	121	121	97	97	97	97	97	0
Total External Loading													
Q-m^3	15,685	16,395	17,195	5,168	5,878	5,168	5,168	12,157	4,342	4,342	4,120	4,120	29
Increased load factor 1 Toal P Ldg-g	372	385	400	123	137	123	123	251	103	103	98	98	2
Ldg/V-ug/l	0.62	0.64	0.67	0.21	0.23	0.21	0.21	0.42	0.17	0.17	0.16	0.16	0.00
Internal Loading													
Calib. Ldg-mg/m^2	0	0	0	0	0	0	0	0	0	0	0	0	0
Calib. Ldg-mg	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Ldg/V -ug/l	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total P LDG -ug/l	0.62	0.64	0.67	0.21	0.23	0.21	0.21	0.42	0.17	0.17	0.16	0.16	0.00
Phosphorus Outflow													
Total Q -m^3	15,685	16,395	17,195	5,168	5,878	5,168	5,168	12,157	4,342	4,342	4,120	4,120	29
Total Q/V (rho)	0.0262	0.0274	0.0288	0.0086	0.0098	0.0086	0.0086	0.0203	0.0073	0.0073	0.0069	0.0069	0.0000
P Outflow -ug/l	0.20	0.21	0.22	0.06	0.07	0.06	0.06	0.15	0.05	0.05	0.05	0.05	0.00
Sedimentation													
Sigma	0.068	0.070	0.072	0.030	0.033	0.030	0.030	0.056	0.026	0.026	0.025	0.025	0.001
P Sedimented -ug/l	0.53	0.54	0.55	0.22	0.24	0.22	0.21	0.40	0.18	0.18	0.17	0.17	0.00
Mass Balance													
Delta P -ug/l	0.11	-0.10	-0.10	-0.08	-0.09	-0.07	-0.07	-0.13	-0.06	-0.06	-0.06	-0.05	0.00
Total P in lake -ug/l — Predicted	7.68	7.58	7.47	7.39	7.30	7.23	7.16	7.03	6.97	6.91	6.85	6.80	6.80
Actual Conc in lake - ug/l — Observed	#N/A	5.9	#N/A	7.0	#N/A	#N/A	6.5	#N/A	7.1	5.0	#N/A	#N/A	4.9
(pred-obs)^2		2.8		0.2			0.4		0.0	3.6			3.6
Chla, ug/l	1.23	1.20	1.18	1.16	1.14	1.12	1.10	1.07	1.05	1.04	1.03	1.01	1.01
Secchi Depth, m	9.17	9.28	9.39	9.48	9.58	9.67	9.76	9.92	10.00	10.08	10.16	10.23	10.23

sigma=rho^.74 0.74



10/13/91	10/20/91	10/27/91	11/3/91	11/10/91	11/17/91	11/24/91	12/1/91	12/8/91	12/15/91	12/22/91	12/29/91	1/5/92	1/12/92	1/19/92	1/26/92	2/2/92	2/9/92	2/16/92	2/23/92	3/1/92	3/8/92	3/15/92	3/22/92	3/29/92	
28	29	30	31	32	33	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48	49	50	51	52	Yearly Total
29	29	29	40	40	40	40	37	37	37	37	37	76	76	76	76	109	109	109	109	88	88	88	88	88	2,901
75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	75.3	
2	2	2	3	3	3	3	3	3	3	3	3	6	6	6	6	8	8	8	8	7	7	7	7	7	
0	1155	8748	1843	8304	10746	10169	9148	8082	3597	3686	1110	4485	1332	5062	9103	17007	1687	3464	9148	1110	2931	2265	3952	1687	208,220
19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	
0	22	166	31	158	204	193	174	154	68	70	21	85	25	96	173	323	32	66	174	21	56	43	75	32	
0	0	0	0	0	0	1199	2129	1125	881	294	269	391	220	1028	1125	3866	563	514	1713	587	269	147	685	98	26,545
0	0	0	0	0	0	32	19	23	34	28	21	19	13.2	13.9	19.7	11.4	10.5	20	16	26	300	93	13	20	
0	0	0	0	0	0	38	40	26	30	8	6	7	3	14	22	44	6	10	27	15	81	14	9	2	
0	0	0	0	0	0	391	783	391	538	147	73	0	0	220	318	2080	343	196	758	416	98	73	245	49	10,667
0	0	0	0	0	0	24	25	33	19	28	16	12	9.5	11.7	15.1	11.4	10.5	20	16	21	16	59	20	21	
0	0	0	0	0	0	9	20	13	10	4	1	0	0	3	5	24	4	4	12	9	2	4	5	1	
0	0	0	0	0	0	24960	40158	31468	34727	22783	11391	0	0	11396	20077	65691	30382	17361	39615	32011	15188	4883	21701	7594	591492
0.0	0.0	0.0	0.0	0.0	0.0	83.0	83.0	83.0	12.1	12.1	12.1	13.5	13.5	13.5	13.5	14.9	14.9	14.9	14.9	37.5	37.5	37.5	37.5	37.5	
0	0	0	0	0	0	2072	3333	2612	420	276	138	0	0	154	271	979	453	259	590	1200	570	183	814	285	
0	0	0	0	0	0	0	22035	22035	22035	22035	22035	41021	41021	41021	41021	42005	42005	42005	42005	23943	23943	23943	23943	23943	902,027
0	0	0	0	0	0	0	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	23.6	
0	0	0	0	0	0	0	520	520	520	520	520	968	968	968	968	991	991	991	991	565	565	565	565	565	
																									1,741,853
29	1,184	8,777	1,683	8,344	10,786	36,759	74,289	63,139	61,815	48,981	34,916	45,973	42,649	58,803	71,720	130,758	75,088	63,648	93,348	58,155	42,517	31,399	50,614	33,459	1,741,853
2	24	166	34	161	207	2316	4090	3327	1052	881	689	1066	1002	1241	1445	2369	1494	1338	1803	1817	1279	816	1474	892	44,609
0.00	0.04	0.28	0.06	0.27	0.35	3.87	6.84	5.56	1.76	1.47	1.15	1.78	1.68	2.07	2.42	3.96	2.50	2.24	3.02	3.04	2.14	1.36	2.47	1.49	
0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	
0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	
0.00	0.04	0.28	0.06	0.27	0.35	3.87	6.84	5.56	1.76	1.47	1.15	1.78	1.68	2.07	2.42	3.96	2.50	2.24	3.02	3.04	2.14	1.36	2.47	1.49	
29	1,184	8,777	1,683	8,344	10,786	36,759	74,289	63,139	61,815	48,981	34,916	45,973	42,649	58,803	71,720	130,758	75,088	63,648	93,348	58,155	42,517	31,399	50,614	33,459	
0.0000	0.0020	0.0147	0.0028	0.0140	0.0180	0.0615	0.1242	0.1056	0.1034	0.0819	0.0584	0.0769	0.0713	0.0983	0.1199	0.2187	0.1258	0.1064	0.1561	0.0972	0.0711	0.0525	0.0846	0.0560	3
0.00	0.01	0.10	0.02	0.09	0.12	0.39	1.12	1.36	1.51	0.89	0.63	0.76	0.67	0.90	1.04	1.80	0.97	0.81	1.18	0.73	0.60	0.46	0.74	0.51	25
0.001	0.010	0.044	0.013	0.042	0.051	0.127	0.214	0.189	0.186	0.157	0.122	0.150	0.142	0.180	0.208	0.325	0.215	0.191	0.253	0.178	0.141	0.113	0.161	0.118	
0.00	0.07	0.30	0.09	0.28	0.33	0.81	1.93	2.43	2.72	1.90	1.31	1.49	1.34	1.64	1.80	2.67	1.66	1.45	1.92	1.34	1.20	0.99	1.40	1.07	
0.00	-0.04	-0.11	-0.05	-0.10	-0.10	2.67	3.78	1.78	-2.48	-1.43	-0.78	-0.47	-0.34	-0.48	-0.42	-0.51	-0.13	-0.02	-0.08	0.98	0.34	-0.09	0.33	-0.09	
6.79	6.75	6.64	6.59	6.49	6.39	9.05	12.83	14.61	12.13	10.71	9.93	9.48	9.12	8.66	8.23	7.72	7.59	7.57	7.49	8.46	8.81	8.71	9.04	8.95	
#N/A	6.9	#N/A	#N/A	#N/A	10.0	#N/A	#N/A	#N/A	#N/A	11.6	#N/A	#N/A	#N/A	12.3	#N/A	#N/A	#N/A	#N/A	11.2	#N/A	#N/A	#N/A	12.2	#N/A	
	0.0				13.1					0.8				13.3					13.8				10.0		103
1.01	1.00	0.97	0.96	0.94	0.92	1.61	2.84	3.50	2.59	2.11	1.87	1.73	1.63	1.50	1.38	1.25	1.21	1.21	1.18	1.44	1.54	1.51	1.61	1.58	
10.23	10.29	10.45	10.52	10.67	10.82	7.66	5.75	5.11	6.05	6.77	7.25	7.58	7.83	8.21	8.60	9.11	9.26	9.27	9.37	8.38	8.09	8.17	7.90	7.97	

Appendix 10

Lake Twelve Benthic Invertebrate Data

10/19/91

Taxon			Sample 1*	Sample 2	Sample 3*
Insecta					
	Diptera				
		Chironomidae	344		
		Culicidae			645
		Tricyphona			
	Odonata				
		Zygoptera			
	Coleptera				
		Philopotamos			

* Numerous empty Trichoptera cases and tube worm cases

3/21/92

Taxon			Sample 1	Sample 2	Sample 3*
Insecta					
	Diptera				
		Chironomidae	2239	517	258
		Culicidae			
		Tricyphona	1033		
	Odonata				
		Zygoptera	172		
	Coleptera				
		Philopotamos	344		

* Numerous empty Trichoptera cases and tube worm cases

Table 1. Analytical precision of sediment analyses. Precision is expressed as percent relative standard deviation (% RSD) for subsamples of sediment and sludge.[a]

	P	Pb	Zn	Cu	Fe	Mn	Al
SOLID SUBSAMPLE							
Lake Twelve							
Core A-10 cm (n=2)	0.13	1.62	9.25	1.85	0.12	0.75	1.62
22 cm (n=2)	1.42	1.57	2.41	3.81	0.10	0.80	0.32
44 cm (n=2)	3.11	19.27	7.74	1.03	6.88	2.95	0.11
Core C- 9 cm (n=2)	2.16	11.35	7.80	0.28	5.16	0.35	3.23
27 cm (n=2)	3.11	9.95	5.70	22.13	12.77	1.65	1.06
37 cm (n=2)	2.10	12.81	16.57	8.35	0.06	0.81	0.09
41 cm (n=2)	1.54	—	7.71	—	10.65	—	—
EPA Sample							
Municipal Sludge (n=2)	7.80	13.4	10.10	2.30	1.80	17.70	12.40
DIGESTRATE							
Replicated analyses	0.2-5.4	0.9-13.4	0.4-8.7	0.8-12.1	0.1-3.7	0.0-2.4	0.7-5.4

a. Precision for weight lost on ignition ranged from 0.1 to 5.4 % RSD.

Table 2. Accuracy of sediment analyses as indicated by percent recovery of constituent concentrations of an EPA quality control sample (dry municipal sludge).

Constituent	Mean Value (mg/kg)	95 96 CI (mg/kg)	Conc. Founded (mg/kg)	% Recovery
Aluminum	4,560	2,010 - 7,110	4,630	101.5
Copper	1,080	882 - 1,280	882	82.9
Iron	16,500	11,200 - 21,700	14,035	85.1
Manganese	202	182 - 223	197	97.5
Lead	526	372 - 680	505	96.0
Phosphorus	15,660	7,500 - 15,600	1,238	79.1
Zinc	1,320	1,190 - 1,450	1,114	84.5

Table 3. Physical and chemical characteristics of Lake Twelve sediments, Core A.

Core No.	Wet/Dry Ratio	% Loss Ignition	%P	% Al	% Fe	% Mn	Pb PPM	Zn PPM	Cu PPM
1	56.14	26.23	0.1928	1.5664	4.6956	0.1412	110.93	195.11	28.87
2	31.25	23.00	0.1840	1.6109	3.0370	0.0797	112.02	133.63	30.11
3	29.98	41.12	0.1814	1.9996	2.6686	0.0651	134.06	175.13	34.23
4	27.27	41.84	0.1702	1.4571	2.2469	0.0455	108.50	111.58	24.10
5	23.19	40.30	0.1799	1.8229	2.0420	0.0384	103.24	112.43	27.68
6	22.37	42.81	0.1725	1.5405	1.7589	0.0286	111.19	105.81	27.78
7	23.06	42.70	0.1870	2.2857	1.8301	0.0265	125.67	141.90	34.29
8	22.69	45.40	0.1644	1.9247	1.6301	0.0228	120.96	120.78	28.25
9	22.56	42.42	0.1564	1.5247	1.3985	0.0180	104.39	98.03	27.14
10	22.72	43.71	0.1667	1.6749	1.4816	0.0189	123.36	124.32	26.34
11	22.15	45.61	0.1567	1.7822	1.4524	0.0183	108.41	106.10	29.74
12	23.06	43.15	0.1589	1.8495	1.4445	0.0182	108.65	107.85	29.50
13	22.04	42.34	0.1580	1.8310	1.4599	0.0183	109.65	112.19	32.23
14	22.00	43.01	0.1587	1.8250	1.4563	0.0178	108.90	136.57	31.70
15	21.29	44.37	0.1613	1.8405	1.5047	0.0182	107.86	107.71	30.10
16	20.20	44.35	0.1632	1.8489	1.5538	0.0186	113.65	113.61	30.20
17	18.66	43.06	0.1613	1.8243	1.5207	0.0181	110.72	109.84	29.78
18	18.71	42.35	0.1608	1.8551	1.5196	0.0181	107.58	106.82	29.73
19	18.25	40.56	0.1624	1.8691	1.5254	0.0181	108.42	108.20	30.66
20	16.75	41.82	-	1.3264	1.1542	0.0137	90.08	84.73	-
21	17.01	41.34	0.1580	1.7433	1.4641	0.0174	108.04	110.00	28.65
22	16.67	41.03	0.1644	1.8295	1.4591	0.0176	111.34	121.28	29.71
23	16.99	42.87	0.1712	1.9272	1.4676	0.0177	112.78	117.03	30.73
24	16.11	-	0.1589	1.8147	1.4271	0.0173	108.28	109.81	30.71
25	16.70	41.62	0.1537	1.7525	1.3683	0.0164	107.87	110.35	29.16
26	17.01	41.90	0.1579	1.7880	1.3387	0.0163	111.62	112.49	29.25
27	16.55	45.71	0.1548	1.6922	1.2593	0.0155	105.06	107.28	29.13
28	16.27	42.40	0.1586	1.5440	1.2111	0.0149	108.57	109.71	34.79
29	15.32	41.53	0.1555	1.5832	1.2252	0.0147	99.59	101.43	27.82
30	15.57	44.20	0.1346	1.3191	1.2209	0.0134	56.60	72.23	23.01
31	15.78	45.95	0.1376	1.3167	1.1958	0.0133	55.59	85.32	21.04
32	15.83	44.44	0.1373	1.3269	1.1743	0.0130	57.06	94.62	22.00
33	15.68	47.25	0.1414	1.2556	1.1273	0.0124	54.04	79.71	24.77
34	15.86	46.35	0.1398	1.2696	1.2060	0.0124	52.84	72.82	19.65
35	15.74	48.10	0.1412	1.2702	1.2295	0.0126	53.85	72.51	19.49
36	15.96	47.25	0.1421	1.2689	1.0508	0.0120	46.21	60.24	22.19
37	15.91	49.40	0.1453	1.0594	0.9018	0.0096	26.91	46.69	14.70
38	16.27	-	0.1363	1.0661	0.8228	0.0095	24.44	37.84	14.77
39	16.93	46.62	0.1403	1.0925	0.8596	0.0097	24.29	45.28	15.20
40	16.24	46.70	0.1430	1.0883	0.8861	0.0097	26.10	56.66	21.63
41	15.76	49.54	0.1396	1.1012	0.8533	0.0098	26.08	55.61	16.47
42	15.88	45.80	0.1430	1.1024	0.8543	0.0098	24.00	40.81	21.08
43	15.63	48.80	0.1467	1.1285	0.7612	0.0096	25.27	42.82	17.43
44	16.10	43.86	0.1456	1.1178	0.8022	0.0096	26.38	45.48	16.47
45	15.13	44.50	0.1473	1.1262	0.8570	0.0095	24.34	37.47	15.93
46	15.20	47.14	0.1578	0.7664	0.6208	0.0060	14.97	26.38	12.48
47	14.98	47.33	0.1526	0.7379	0.5222	0.0053	4.83	24.20	11.81
48	14.65	47.00	0.1527	0.7106	0.6276	0.0049	2.21	26.10	11.83
49	14.61	47.95	0.1509	0.7679	0.5022	0.0056	9.55	27.13	12.40
50	14.72	44.24	0.1500	0.8041	0.5269	0.0061	22.74	33.37	12.77
51	14.54	44.88	0.1506	0.7295	0.4476	0.0092	8.04	25.34	12.79
52	15.77	46.90	0.1528	0.7895	0.4567	0.0055	9.19	34.14	12.20
53	15.36	50.75	0.1505	0.8178	0.5055	0.0061	20.98	40.08	13.41
54	14.80	47.24	0.1486	0.8043	0.4656	0.0057	15.69	36.95	13.26
55	14.00	46.44	0.1503	0.8787	0.5198	0.0064	15.29	46.64	13.29

Table 4. Physical and chemical characteristics of Lake Twelve sediments, Core C.

Core No.	Wet/Dry Ratio	% Loss Ignition	% P	% Al	% Fe	% Mn	Pb PPM	Zn PPM	Cu PPM
1	36.95	39.27	0.2075	1.8846	4.3170	0.1337	106.65	262.53	30.99
2	33.32	38.76	0.1960	1.7815	3.6614	0.1089	102.61	213.88	31.64
3	28.39	41.04	0.1813	1.9730	2.5541	0.0664	99.73	189.64	27.63
4	26.64	44.09	0.1769	2.0276	2.1578	0.0506	99.65	120.71	27.65
5	22.53	44.00	0.1802	1.9315	1.8926	0.0402	92.17	165.58	25.64
6	21.42	41.67	0.1720	1.9946	1.6262	0.0279	101.66	205.77	25.96
7	22.05	43.38	0.1752	1.9849	1.5429	0.0215	100.88	168.41	27.30
8	21.15	43.51	0.1613	1.9661	1.5024	0.0192	99.51	142.24	26.99
9	-	-	0.1572	1.6982	1.5576	0.0200	128.43	109.19	30.23
10	-	-	0.1518	1.9522	1.4508	0.0168	93.06	101.77	26.62
11	-	-	0.1543	1.7195	1.3904	0.0161	82.24	126.40	24.76
12	16.41	42.73	0.1486	1.9944	1.4897	0.0176	108.80	160.08	26.55
13	16.64	40.72	0.1511	2.0092	1.4951	0.0177	100.32	125.73	27.24
14	16.57	42.76	0.1539	1.9294	1.4232	0.0168	95.91	118.44	25.77
15	16.37	43.50	0.1547	1.9121	1.4629	0.0172	96.51	131.52	25.84
16	16.44	40.82	0.1483	1.7680	1.3733	0.0161	96.96	126.00	25.11
17	16.27	42.92	0.1446	1.9320	1.4193	0.0168	99.37	111.39	26.46
18	15.96	42.16	0.1595	1.7652	1.3282	0.0158	92.04	113.14	23.36
19	16.56	43.07	0.1600	1.9573	1.3294	0.0161	101.92	156.65	26.49
20	16.19	43.20	0.1551	1.2937	1.0833	0.0129	80.65	135.97	23.10
21	16.49	46.05	0.1600	1.6715	1.2295	0.0149	89.81	148.75	26.92
22	16.17	40.48	0.1449	1.3550	1.0794	0.0116	84.16	165.87	32.01
23	15.95	39.11	0.1566	1.6337	1.1458	0.0139	93.87	126.38	26.92
24	15.43	41.40	0.1529	1.6198	1.1522	0.0135	93.48	136.48	27.71
25	15.40	41.58	0.1525	1.6347	1.1897	0.0138	100.86	143.87	26.31
26	15.12	39.15	0.1501	1.4675	1.0477	0.0121	83.51	102.09	23.61
27	15.12	35.96	0.1443	1.4814	1.0535	0.0128	79.99	114.10	23.35
28	15.15	41.91	0.1453	1.1691	0.9353	0.0101	58.01	112.33	21.37
29	14.40	43.17	0.1452	0.9447	0.8356	0.0091	42.86	86.37	14.97
30	15.03	40.81	0.1397	0.9086	0.9598	0.0088	29.10	97.76	14.78
31	16.11	40.14	0.1338	1.1354	1.0478	0.0096	26.68	126.09	15.27
32	15.63	43.26	0.1347	1.1469	1.1347	0.0095	27.64	89.64	15.15
33	15.44	40.27	0.1323	1.0297	0.9840	0.0089	21.54	78.88	14.48
34	14.21	41.03	0.1381	1.0741	0.9766	0.0084	18.76	105.92	14.77
35	13.97	42.56	0.1420	1.2072	1.0545	0.0098	26.78	104.43	19.91
36	13.37	43.75	0.1487	1.0312	0.8453	0.0084	25.33	85.86	14.40
37	13.17	45.51	0.1515	1.0907	0.8026	0.0087	23.05	89.59	15.51
38	12.75	45.58	0.1535	0.7761	0.5744	0.0062	16.03	122.67	11.27
39	12.72	44.26	0.1480	-	1.0901	0.0131		90.04	12.20
40	12.67	43.92	0.1507	1.0796	0.7713	0.0083	23.72	128.94	14.86
41	13.12	43.94	0.1503	-	1.1373	0.0123		110.57	-
42	13.29	42.98	0.1444	0.9787	0.6434	0.0072	13.50	48.90	12.59
43	13.20	45.27	0.1404	1.1574	0.7077	0.0086	16.02	54.79	16.08
44	13.26	43.01	0.1497	1.0399	0.6826	0.0076	14.39	79.64	13.87
45	13.79	43.83	0.1589	1.0974	0.7600	0.0084	17.12	38.95	15.70
46	13.46	44.34	0.1544	1.1066	0.7099	0.0080	18.13	36.64	13.98
47	13.50	44.99	0.1613	0.9874	0.6567	0.0073	14.39	29.76	13.15
48	13.56	44.15	0.1471	1.0200	0.7377	0.0083	21.95	50.23	15.95
50	13.73	44.35	0.1589	0.9039	0.6606	0.0074	18.18	32.54	13.86

Table 5. Wet to dry ratios of Lake Twelve sediments, Core L.

Core L Section	Wet Weight (gram)	Dry Weight (gram)	Wet/Dry Ratio
1	8.7407	0.2596	33.67
2	10.5353	0.3715	28.36
3	8.0435	0.3406	23.62
4	8.0725	0.3560	22.68
5	8.2062	0.3919	20.94
6	9.1195	0.4142	22.02
7	10.7782	0.4529	23.80
8	8.8654	0.4498	19.71
9	9.2622	0.4615	20.07
10	8.5133	0.4346	19.59
11	9.2629	0.4635	19.98
12	9.4911	0.4853	19.56
13	9.1135	0.4795	19.01
14	9.4182	0.5075	18.56
15	9.1766	0.4867	18.85
16	9.3733	0.5059	18.53
17	9.3308	0.5171	18.04
18	9.3971	0.5223	17.99
19	9.2282	0.5281	17.47
20	9.2505	0.5531	16.72
21	9.3203	0.5663	16.46
22	10.1302	0.6309	16.06
23	8.6337	0.5584	15.46
24	9.0316	0.5905	15.29
25	9.7509	0.6690	14.58
26	9.5244	0.6345	15.01
27	9.3663	0.6188	15.14
28	8.4279	0.6090	13.84
29	9.6243	0.6263	15.37
30	8.7232	0.5765	15.13
31	9.2225	0.6160	14.97
32	8.8185	0.5995	14.71
33	8.1790	0.5590	14.63
34	8.5959	0.5686	15.12
35	9.5972	0.5787	16.58
36	9.4092	0.5881	16.00
37	9.1411	0.5599	16.33
38	8.9607	0.5447	16.45
39	9.2409	0.5474	16.88
40	10.0524	0.5368	18.73
41	8.7459	0.4919	17.78
42	8.5379	0.5105	16.72
43	7.9936	0.4778	16.73
44	10.0503	0.5522	18.20
45	8.9747	0.5400	16.62
46	9.8178	0.5829	16.84
47	8.4046	0.4984	16.86
48	9.7143	0.5689	17.08
49	9.1686	0.5306	17.28
50	8.7876	0.5136	17.11

Table 6. ^{210}Pb dating data for Lake Twelve sediments, Core L.

Lead-210 dating of Lake 12

Section (cm)	Weight (g)	Po-209	(C210/C209) x 3.7 (Pci)	Pb-210 * (Pci/g)	Unsupported Pb-210 (Pci/g) * *
2	0.361	250 l = 3.7 Pci	459 /476 * 3.7 = 3.568	9.883 +/- 1.0	9.471 +/- 1.0
7	0.449	"	229 /160 * 3.7 = 5.3	11.794 +/- 1.2	11.382 +/- 1.2
15	0.425	"	/ * 3.7 = 4.58	10.78 +/- 1.0	10.364 +/- 1.0
18	0.483	"	43 / 45 * 3.7 = 3.575	7.40 +/- 0.74	6.990 +/- 0.74
20	0.511	"	/ * 3.7 = 5.76	11.27 +/- 1.1	10.860 +/- 1.1
21	0.566	"	/ * 3.7 = 3.70	6.54 +/- 0.7	6.125 +/- 0.7
22	0.610	"	/ * 3.7 = 5.89	9.66 +/- 1.0	9.244 +/- 1.0
25	0.667	"	/ * 3.7 = 5.71	8.56 +/- 0.9	8.149 +/- 0.9
28	0.603	"	165 /316 * 3.7 = 1.932	3.204 +/- 0.32	2.792 +/- 0.32
31	0.471	"	105 /375 * 3.7 = 1.036	2.20 +/- 0.22	1.788 +/- 0.22
33	0.417	"	11 / 67 * 3.7 = 0.607	1.457 +/- 0.15	1.045 +/- 0.15
35	0.458	"	28 /195 * 3.7 = 0.53	1.160 +/- 0.11	0.748 +/- 0.11
37	0.410	"	4 /113 * 3.7 = 0.131	0.319 +/- 0.03	* * *
40	0.386	"	13 /250 * 3.7 = 0.192	0.498 +/- 0.05	* * *
48	0.376	"	4 / 94 * 3.7 = 0.157	0.419 +/- 0.04	* * *

C210/C209 = A210/A209 { A210 = A209 x C210/C209 = C210/C209 x 3.7 Pci }

* C210 x 3.7 / C209 / Weight

* * Supported Pb-210 value subtracted

* * * Supported = avg of deepest three sections = 0.412 Pci

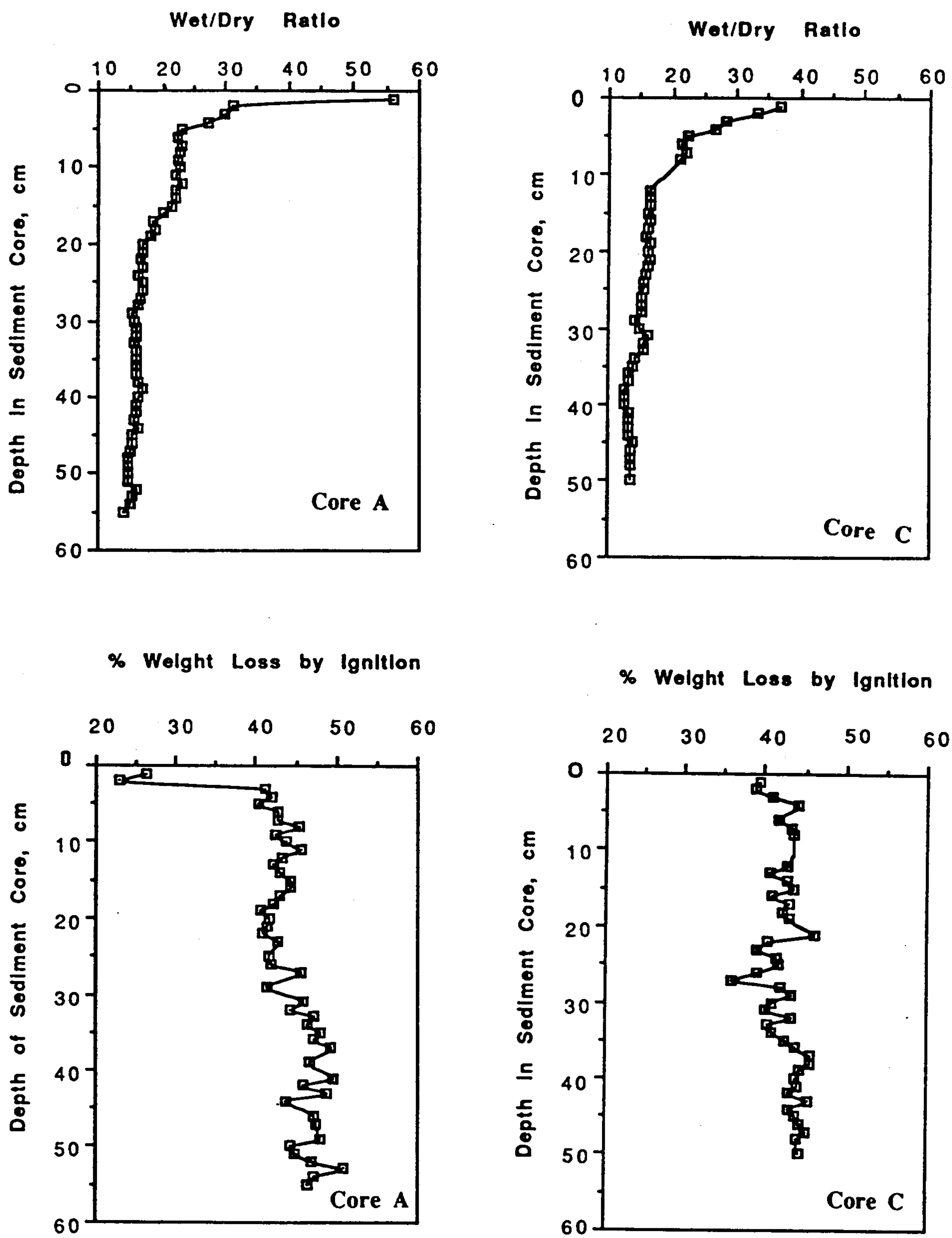


Figure 1. Wet to dry ratios and weight lost on ignition for Cores A and C from Lake Twelve.

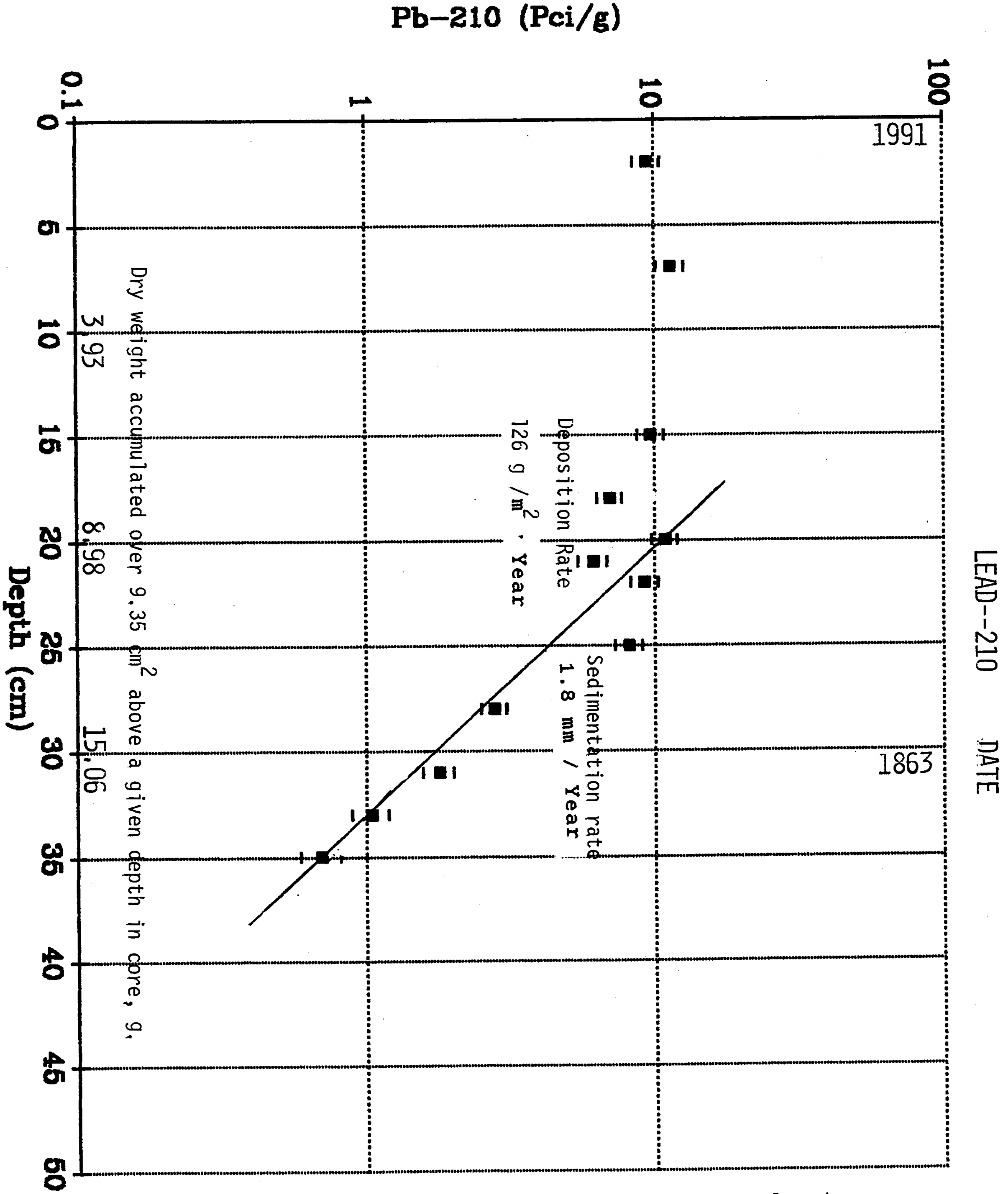


Figure 2. Lead--210 Chronology of Lake Twelve sediments. Based on a deposition rate of 126 g dry sediment/$m^2 \cdot y$, the 30 cm in core L corresponds to about 1863.

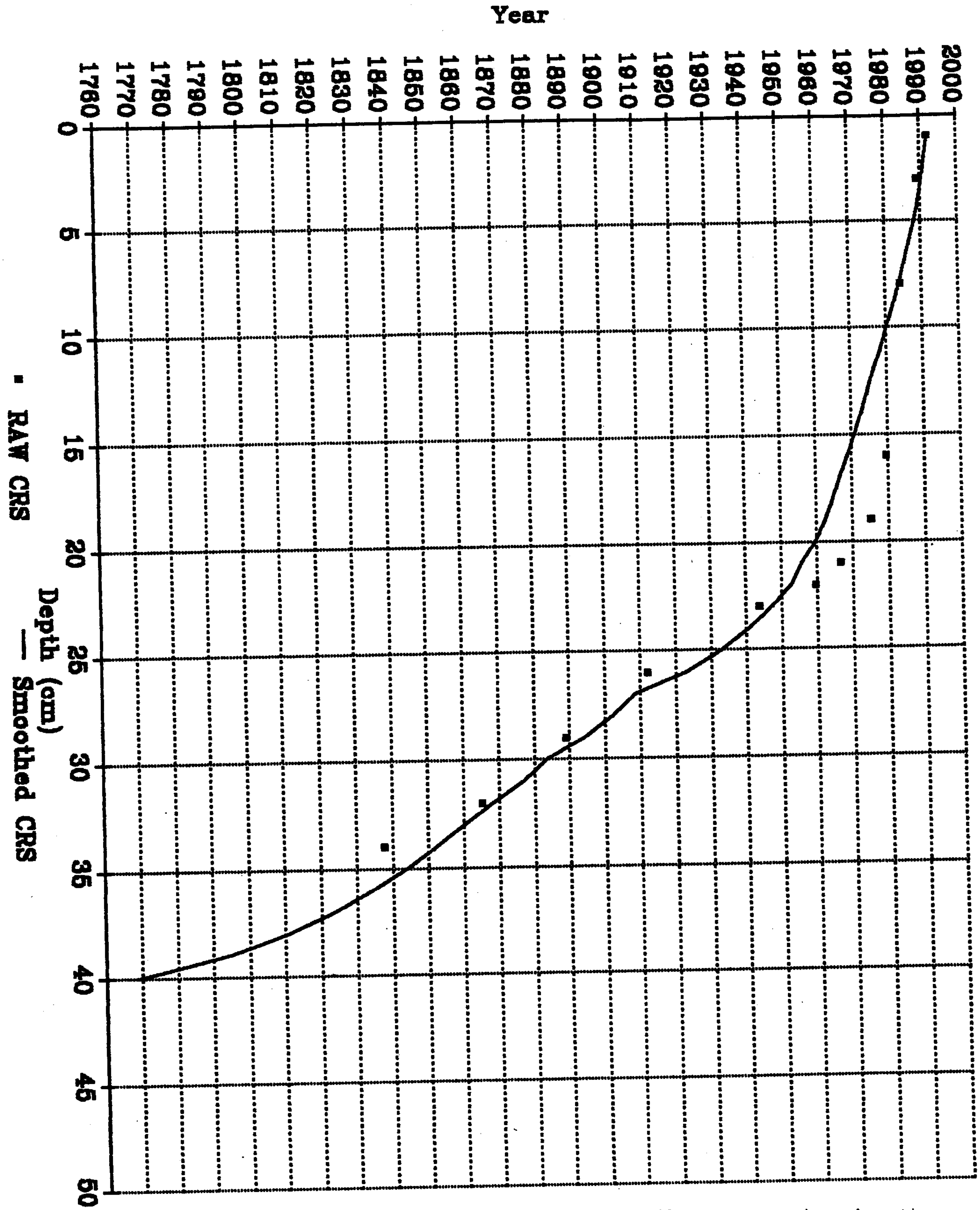


Figure 3. Lead--210 dating of Lake Twelve sediments, core L, using the constant rate supply (CRS) model.

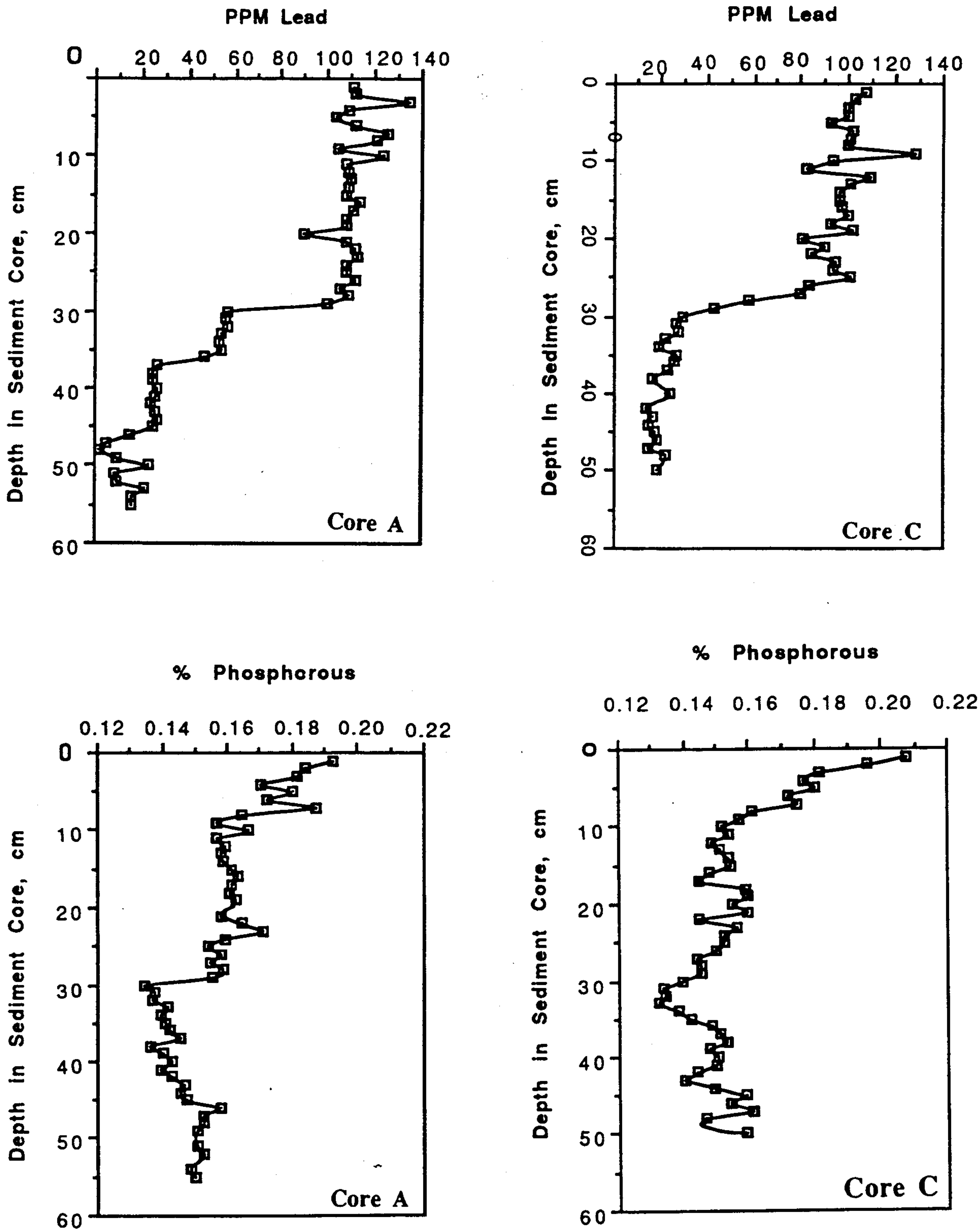


Figure 4. Pb and P sediment profiles in Cores A and C from Lake Twelve.

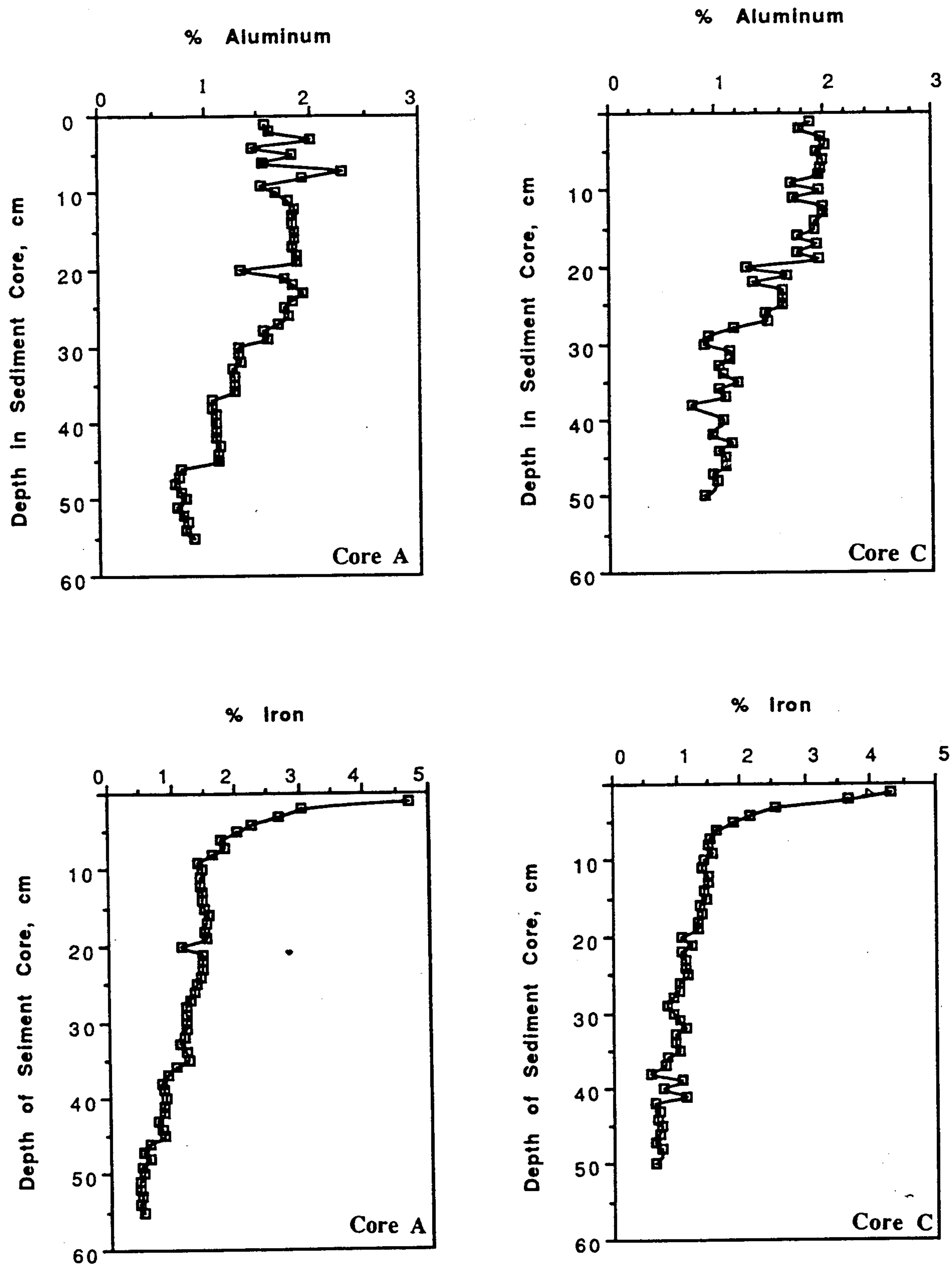


Figure 5. Al and Fe sediment profiles in Cores A and C from Lake Twelve.

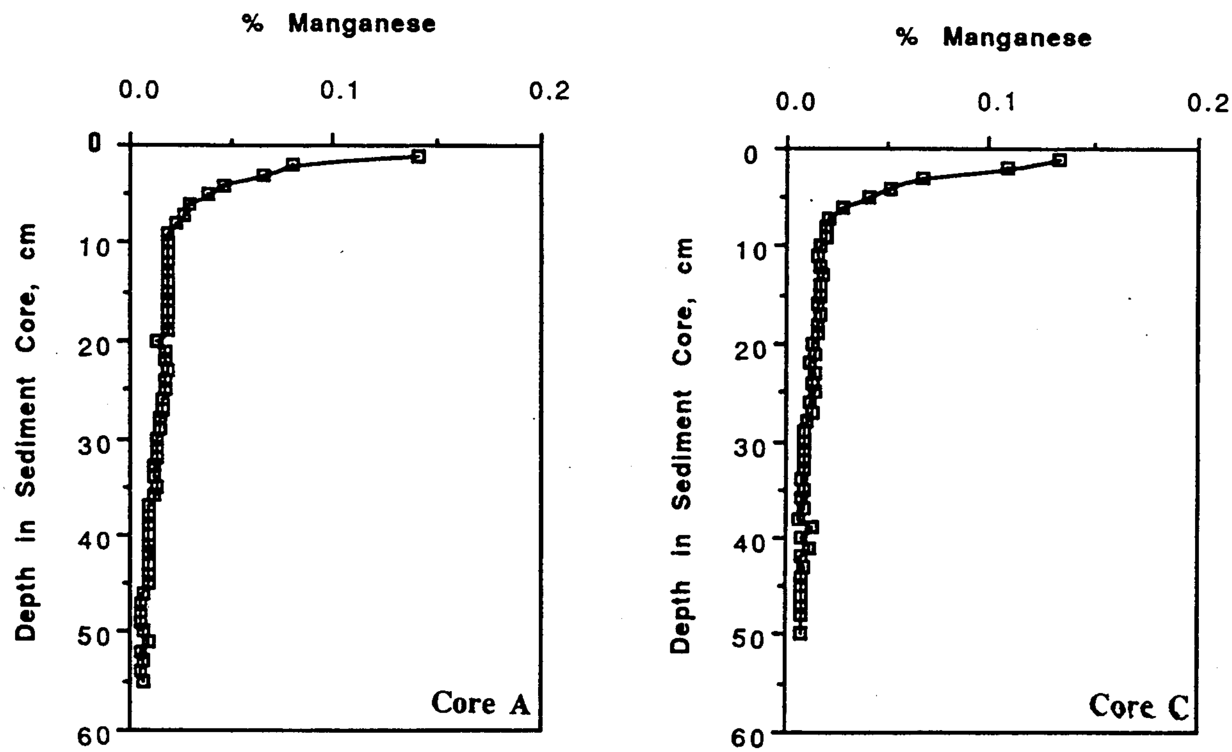


Figure 6. Mn sediment profiles in Cores A and C from Lake Twelve.

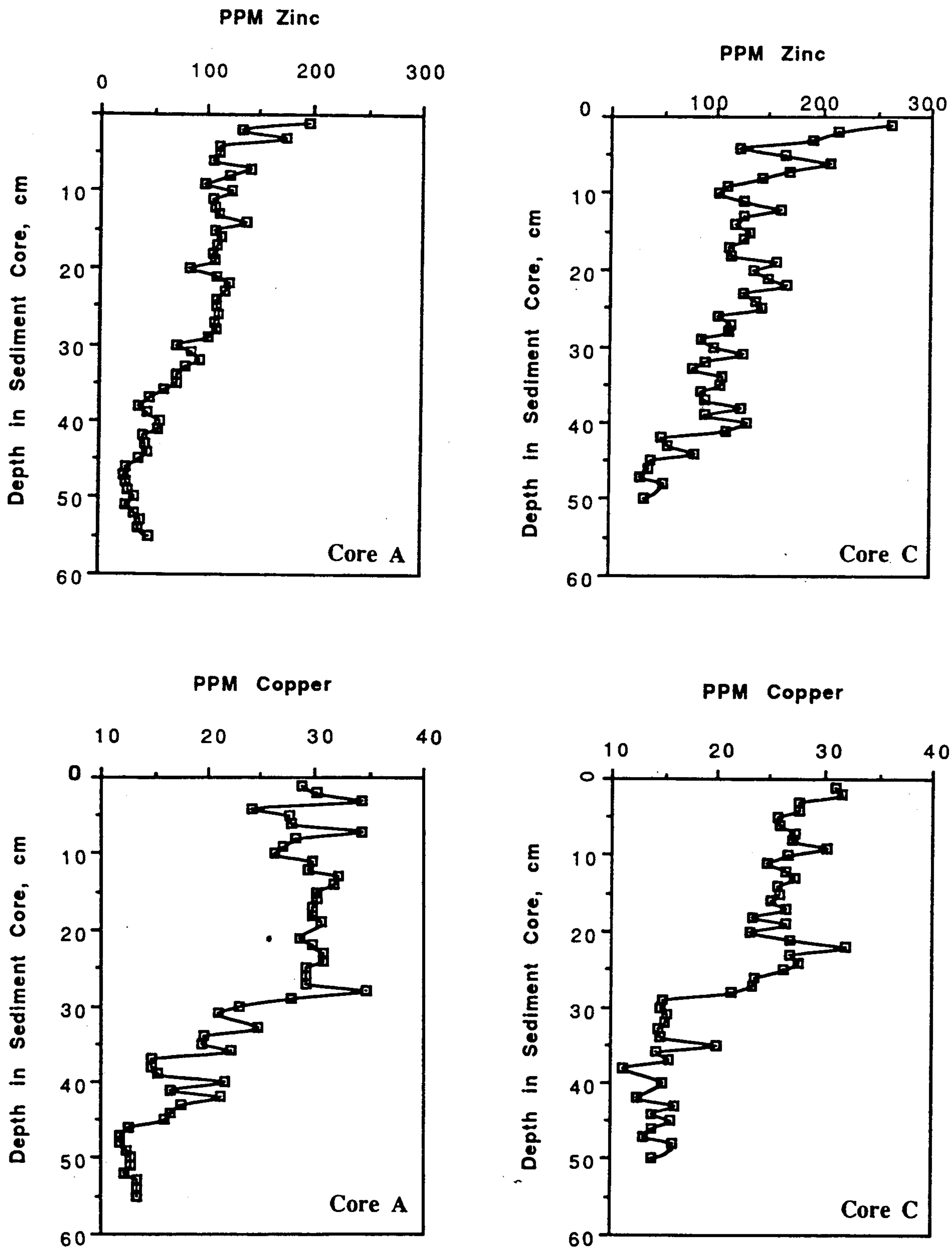


Figure 7. Cu and Zn sediment profiles from Cores A and C from Lake Twelve.

APPENDIX 12

Length of milfoil growth (not including roots) over a 30 day period.

LOCATION:	GREEN LAKE		UNION BAY		LAKE TWELVE	
REPLICATE:	1	2	1	2	1	2
Individual plant length (cm)	42	25	35	37	39	36
	28	25	24	37	34	34
	34	35	29	33	35	37
	32	30	23	46	39	37
	30	27	22	38	27	35
	32	27	34	34	31	31
	34	23	32	43	37	19
	30	33	39	28	39	27
	32	38	40	16	18	25
		33	33		17	37
		22	10		11	
		26				
MEAN	32.7	28.7	29.2	34.7	29.7	31.8
STDEV	4.0	5.1	8.8	8.8	10.1	6.2
	Green Lake		Union Bay		Lake Twelve	
Pooled Mean	30.4		31.7		30.7	
Pooled stdev	9.9		11.0		11.2	