

THE EFFECT OF A WATER DIVERSION ON A
EUTROPHIC LAKE

T.P. Murphy

NWRI Contribution Series 85-168

THE EFFECT OF A WATER DIVERSION
ON A EUTROPHIC LAKE

T.P. Murphy

Aquatic Ecology Division, National Water Research
Institute, Box 5050, Burlington, Ontario, L7R 4A6

Recherche de la DEA sur la restauration des lacs

Effets de la dérivation des eaux sur l'eutrophisation d'un lac

T. P. Murphy, Division de l'écologie aquatique

PERSPECTIVE-GESTION

On a mis sous surveillance les débits d'eau et de nutriments qui se déversent dans le lac Chain, dans l'intérieur sec de la Colombie-Britannique, afin de découvrir pourquoi le système de dérivation des eaux du ruisseau Shinish ne parvenait pas à diminuer l'eutrophisation du lac. En effet, on s'est aperçu que les charges sédiments-phosphore en période d'étiage n'étaient pas modifiées outre mesure par l'effet de chasse du système de dérivation. À la lumière des résultats, on a conclu qu'il faudrait exercer une surveillance intensive durant tout un été avant de concevoir les systèmes de dérivation des eaux destinés à réduire l'eutrophisation. Avant de prendre des mesures en vue d'accroître la capacité du système de dérivation du ruisseau Shinish, il serait bon d'envisager d'autres solutions, notamment le dragage des sédiments dans le lac Chain.

Cette étude fait suite au programme de restauration des lacs de l'INRE; elle a été réalisée en collaboration avec les sections du Yukon et des Prairies de l'INRE ainsi que le ministère de l'Environnement de la Colombie-Britannique.

Recherche de la DEA sur la restauration des lacs

Effets de la dérivation des eaux sur l'eutrophisation d'un lac

T. P. Murphy, Division de l'écologie aquatique

PERSPECTIVE-GESTION

On a mis sous surveillance les débits d'eau et de nutriments qui se déversent dans le lac Chain, dans l'intérieur sec de la Colombie-Britannique, afin de découvrir pourquoi le système de dérivation des eaux du ruisseau Shinish ne parvenait pas à diminuer l'eutrophisation du lac. En effet, on s'est aperçu que les charges sédiments-phosphore en période d'étiage n'étaient pas modifiées outre mesure par l'effet de chasse du système de dérivation. À la lumière des résultats, on a conclu qu'il faudrait exercer une surveillance intensive durant tout un été avant de concevoir les systèmes de dérivation des eaux destinés à réduire l'eutrophisation. Avant de prendre des mesures en vue d'accroître la capacité du système de dérivation du ruisseau Shinish, il serait bon d'envisager d'autres solutions, notamment le dragage des sédiments dans le lac Chain.

Cette étude fait suite au programme de restauration des lacs de l'INRE; elle a été réalisée en collaboration avec les sections du Yukon et des Prairies de l'INRE ainsi que le ministère de l'Environnement de la Colombie-Britannique.

AED Lake Restoration Research

The Effect of a Water Diversion on a Eutrophic Lake

T.P. Murphy, Aquatic Ecology Division.

MANAGEMENT PERSPECTIVE

Water and nutrient fluxes into Chain Lake, in the dry interior of British Columbia, were monitored to determine why the Shinish Water Diversion was not reducing eutrophication. Sediment-phosphorus release during periods of low water flow was not changed effectively by the diversions's flushing effect. The results indicate that water diversions that are built to reduce eutrophication should be designed after an intensive summer monitoring program. Before enlarging the capacity of the Shinish Water Diversion, other options such as sediment dredging of Chain Lake should be evaluated.

This study was conducted as an extension of the NWRI Lake Restoration Project, in collaboration with the NWRI-P/Y Region and the British Columbia Ministry of the Environment.

ABSTRACT

Water and nutrient fluxes were monitored to determine why the Shinish Water Diversion was not reducing the eutrophication of Chain Lake, British Columbia. The 32 cottages and provincial campground accounted for less than 6% of the phosphorus loading in summer. About 62% of the external phosphorus loading entered the lake via groundwater. Groundwater phosphorus was not included in earlier budgets. A more important nutrient flux that earlier studies had not anticipated was internal loading. During periods of low flow, the decay of algal blooms resulted in the bottom water becoming anoxic. Iron (>1.5 mg/L) and phosphorus (>200 μ g/L) concentrations increased in the water column. Internal loading represented about 78% of the phosphorus supply to the lake in summer. The lake became oxidized in fall storms and two-thirds of the phosphorus precipitated back to the sediments. The asynchrony of water flow and phosphorus solubility must contribute to the inability of the diversion to control algal blooms.

LEGEND

1	Introduction.....	1
2	Methods.....	2
2.1	Water Sampling.....	4
2.1.1	Hydrology.....	4
2.1.2	Water Analysis.....	5
2.2	Limnocorral Experiments.....	5
2.3	Sediment Sampling.....	6
2.3.1	Sediment Analysis.....	6
3	Results and Discussion.....	8
3.1	Hydrology.....	8
3.2	Sources of Phosphorus.....	9
3.3	Lake Sediments.....	11
3.4	Seasonal Change in Phosphorus Solubility Reduces Effect of Water Diversion.....	14
3.5	Lake Flushing.....	15
3.6	Phosphorus Budget.....	17
3.7	Nitrogen and Iron Availability in Chain Lake.....	20

4	Lake Management Options and Research Needs.....	22
4.1	Control of Lake Level and the Swamp.....	22
4.2	Septic Wastes.....	23
4.3	Flushing of Chain Lake.....	24
4.4	Sediment Dredging.....	27
4.4.1	Use of Lake Sediments as a Soil Conditioner.....	30
4.4.1.1	Effect of Copper Algicide on Sediment Use.....	31
4.4.1.2	Fertilizer Value.....	31
4.4.2	Processing of Chain Lake as Diatomaceous Earth.....	32
4.4.3	Sediment Recovery.....	33
4.4.4	Additional Environmental Mitigation Measures.....	34
4.5	Partial Lake Dredging.....	34
	Summary.....	37
	References.....	38

INTRODUCTION

Chain Lake is situated 45 km N.E. of Princeton, British Columbia (longitude 120° 16', latitude 49° 42'; Fig. 1). The valley receives a mean rainfall of 52.7 cm a year (Envir. Can., 26 years of data). Rainfall is quite variable and severe water shortages have occurred. Chain Lake has been an important reservoir for irrigation water for downstream farms. To provide more water storage the lake level was raised 1.2 m by the construction of a dam on the outlet in 1957. Local residents observed that the lake turned from brown to green after the lake level was raised. Fish kills have occurred both during summer algal blooms and during winter.

This accelerated eutrophication of reservoirs has often occurred when soils are flooded (Baxter, 1977; Jackson and Hecky, 1980). Ennis (1972) stated that the main cause of eutrophication in Chain Lake was the swamp that had been created at the head of the lake when the lake level was raised. In 1977, the Water Investigations Branch of British Columbia (WIB) observed that the nutrient content of the stream increased as it passed through the swamp, but they did not believe that there was enough justification to isolate the swamp from the lake.

A water diversion from Shinish Creek to Chain Lake was constructed in 1968 to reduce eutrophication of the lake. The WIB (1977) estimated that in 1971 the diversion supplied 22% of the total outflow of Chain Lake and that Chain Lake had a flushing time of 2.8 months. Ennis (1972) found no significant reduction of eutrophication by the flushing. WIB confirmed this, but they believed that the diversion had not been operating long enough for the effect to be observed. Initially, the diversion was operated for only a small part of the summer.

The placement of the diversion was criticised for being too close to the outlet (Fig. 1). Bull (Penticton, Fisheries Branch) resolved this dispute by using a dye tracer. The diversion water was colder and denser than the lake water ;thus, the diversion water plunged to the lake bottom. This water then turned to the right and moved to the head of the lake. Thus, the diversion water did flush the lake water as well as the flow could permit.

The surface of Chain Lake is rectangular, 1.6 km long and 0.3 km wide. The lake is small (43.7 ha) and relatively shallow (6.1 mean and 7.9 m maximum depth (WIB, 1977)). The natural inlet, Hayes Creek is at the north end. The outlet flows from the southern end of the lake to the Similkameen River.

The lake lies at an elevation of 1006.5 m upon a volcanic intrusion that is primarily composed of granite and granodiorite (Rice, 1946). This Jurassic intrusion is similar in age and composition to intrusions south of Kamloops that are rich in the phosphate mineral apatite (Cockfield, 1947). The Chain Lake watershed is heavily forested with Lodgepole pine, Douglas fir, and Trembling aspen at lower elevations, and with Engelman spruce and subalpine fir at higher elevations. Only a few small hobby farms were observed in the watershed. The upstream lakes, Osprey and Link, are densely ringed with cottages. Chain Lake has thirty cottages, two permanent dwellings, and a provincial campground with about twenty campsites.

This study focused on the inability of the Shinish water diversion to prevent blue-green algal blooms. Initially, undocumented sources of phosphorus were sought. Later, the hypothesis that the sediment phosphorus release greatly exceeded the rate of phosphorus flushing was developed.

Acknowledgements

The Fish and Wildlife Branch of British Columbia, especially C. Bull, V. Marchand, and R. Dyer, helped by providing a field camp and field assistance. Dr. H. Fricker of EAWAG, Switzerland provided direction on hydrology measurements. A. Mudroch of NWRI provided guidance in preparation and analysis of sediment samples. Many residents of Chain Lake, especially G. Smith, provided accommodations, electricity, boats, and assistance many times. Dr. K. Hall and I. Yesaki of the University of British Columbia provided support. Technical Operations Division of NWRI, especially M. Mawhinney, were very helpful in collecting samples. Kathleen Kemp provided assistance in the field. Mrs. D. Urciuoli assisted with much of the laboratory studies. Drs. E. Prepas (U. Alberta), K.J. Hall (UBC, Vancouver), (T.G. Northcote, UBC, Vancouver), C.B.J. Gray (NWRI, Vancouver), R.J. Daley (NWRI, Vancouver), C.J.P. McKean (MOE, Victoria), C. Bull (MOE, Penticton), and B.K. Burnison (NWRI, Burlington) reviewed the paper.

2 METHODS

2.1 Water Sampling

Chain Lake water samples were collected in 1983 and 1984 with a three L PVC Van Dorn bottle. One station in the middle of the lake between the provincial campground and the diversion was the primary sampling site. On three days, other sites in the lake were analyzed for dissolved oxygen concentration or Secchi disk depth. The lake was monitored on 65 days. Before water samples were collected, oxygen and temperature profiles were measured with a YSI meter. Water samples were usually collected from 1.0, 3.0, and 5.0 m below the lake surface. Occasionally, when the lake was mixing quickly only two samples were collected. If the lake was highly stratified, samples were collected from every meter. Samples were filtered and preserved at the lake within two h of collection.

2.1.1 Hydrology

Water flow was measured with a Pygmy flow meter that was calibrated at the Canada Center for Inland Waters (CCIW). The natural inlet flow was measured in a culvert (triplicate measurements). The diversion flow was measured at three sites in a rectangular box that was built into the flume. The outlet flow was measured at nine sites across the concrete spillway of the dam. An echo sounder was used to measure water depth (20 transects). A mean depth of 6.1 m was used for calculations of water volume. The WIB has an automatic recorder of water height at Chain Lake. Environment Canada maintains a weather station at Chain Lake.

2.1.2 Water Analysis

Samples for chlorophyll a analysis were filtered through GF/C filters at the lake. The filters were frozen and later analyzed by DMSO extractions (Burnison, 1980). Soluble reactive phosphorus was measured the same day as samples were collected with the molybdate-ascorbic acid method (Strickland and Parsons, 1972). Some water samples were analyzed for ammonia at the field site with the indophenol method (Solorzano, 1969). Samples for total phosphorus were preserved with 0.2 mL of H_2SO_4 per 100 mL sample and analyzed later at CCIW by perchloric acid digestion and the molybdate-ascorbic acid method. Samples for total iron analysis were preserved with 2.0 mL of H_2SO_4 per liter and analysed in CCIW, Burlington with a modification of the bathophenanthroline method (Strickland and Parsons, 1972). The Inland Waters Directorate Water Quality lab in North Vancouver did all of the dissolved organic carbon, dissolved organic nitrogen, sulphate, chloride, nitrate, potassium, and sodium analysis; and some of the ammonia, calcium and magnesium analysis.

2.2 Limnocorral Experiments

The limnocorrals were clear woven polyethylene cylinders two m in diameter and five m deep with a sealed bottom. A collar that was resistant to ultraviolet light was sewn to the top of the limnocorrals and extruded styrofoam was inserted into the hollow collar for flotation. The eight limnocorrals were filled July 15 and enriched on July 18. The nitrogen limnocorrals were enriched with 8.3 g of NH_4Cl which should have produced a concentration of 200 μg N/L. Iron (3.42 g $FeCl_3 \cdot 6H_2O$ per limnocorral) was mixed with citric acid (96 g per limnocorral), the pH was adjusted to 7.0 with NaOH, and quickly the solutions were added to the limnocorrals. Two limnocorrals received 96 g of citric acid

that was also neutralized with NaOH. All limnocorrals, including what is referred to in the text as "untreated limnocorrals" received 1.98 g of KH_2PO_4 . All nutrient additions were made by mixing the nutrients in 50 L of water and then pumping the solution into the limnocorrals at five depths.

2.3 Sediment Sampling

Sediment samples were collected with a Williams light weight corer (Williams and Pashley, 1978) or a modified Moore's corer (CCIW, unpublished). The cores were immediately extruded and divided into 2.0 cm sections on the shore. Samples were then frozen, freeze-dried, and ground. Samples were pelletized and analyzed for major elements with X-ray fluorescence spectrometry at McMaster University in Hamilton. Determination of minerals present in subsamples was carried out with a Phillips X-ray diffraction spectrometer.

2.3.1 Sediment Analysis

For copper analysis, 1.0 g of freeze-dried and ground sediment was extracted with 10 mL of concentrated 1:1 $\text{HCl}:\text{HNO}_3$ for 16 h at room temperature. The extract was heated for 1.5 h in a waterbath at 90°C and filtered through a #44 Whatman filter. The filtrate was analyzed by atomic absorption spectrophotometry.

To determine "bioavailable" phosphorus, 10 mg of freeze-dried and ground sediment was extracted with 100 mLs of 0.5 N NaOH for 16 h at 16°C (Williams et al., 1980). The extract was filtered through GF/C filters, neutralized, and analyzed for soluble reactive phosphorus (Strickland and Parsons, 1972).

Lake sediments were incubated with acetate to determine the effect of organic loading on sediment phosphorus release. The sediment samples had been collected with an Ekman dredge in the last week of October 1983, stored in the dark at 4°C, and then used in this experiment in March 1984. Before the incubations began, 120 gm of sediment was washed by shaking the sediment in phosphate free synthetic lake water and collecting the washed sediment by centrifuging. The synthetic lake water was made with CaCl_2 , MgSO_4 , Na_2CO_3 , and HCl so that the pH, ionic strength, and cation content was equivalent to the lake (Ca, 20; Mg, 4; K, 2; Na, 5 mg/L). Thirty-five g of sediment was placed into each erlenmeyer flask with 100 mL of phosphate free synthetic Chain Lake water. A control treatment and two concentrations of sodium acetate were used (5.6 and 22.4 mg C/L). Samples were incubated in the dark at 15°C on a shaker at 40 rpm. Ten mL samples were periodically withdrawn for filtration through GF/C filters. The filtrate was analyzed for soluble reactive phosphorus.

3 RESULTS and DISCUSSION

3.1 Hydrology

Samples were collected before and through the critical period of algal blooms. In 1983, the sampling began at the end of the spring runoff. The flow into Hayes Creek diminished to a low in August and then increased greatly in September (Table 1). The flow from Hayes Creek into Chain Lake was estimated to be about 6% of the lake volume or 7% of the lake outflow for the period from June 1 to October 1, 1983. In this period, the diversion flow was about 19% of the lake volume or 24% of the lake outflow. Over the study period, the lake level decreased by about 3 cm to a low in August that corresponded to a loss of <0.5% of the lake volume or 0.6% of the lake outflow. Rain falling directly on the lake represented about 4% of the lake outflow. No effort was made to measure evaporation. However, inclusion of an evaporation term would make the apparent discrepancy in the water budget worse. Only 36% of the lake outflow was accounted for in the water budget.

The water budget is reasonably accurate. The field camp was immediately adjacent to the diversion and lake outlet ;thus, frequent observations confirmed that the sampling was representative. Major pulses in flow were not observed. The upstream lakes and swamps may have moderated the rate of runoff into Chain Lake. Rainfall was unusually frequent ;thus, surges in flow would have been unlikely.

The discrepancy in the water budget could be resolved if groundwater flow is significant. Many springs emerge from the steep mountains and flow for short

distances before disappearing into the gravel that bordered the lake. Since most of the valley is lined with gravel, significant groundwater flow probably occurs both into and along the valley.

The indirect estimate of groundwater flow into Chain Lake agrees well with groundwater measurements at a nearby site, the Trapping Creek Basin (Lawson 1967). Trapping Creek is situated 49 km S.E. of Kelowna B.C. Trapping Creek and Chain Lake are both situated upon the Southern Interior Plateau. Both sites have similar rainfall, physical relief, and geology (granite and granodiorite). At Trapping Creek 12 of the 22 inches of rainfall flowed as groundwater.

3.2 Sources of Phosphorus

The increase of total and soluble reactive phosphorus in the lake in late summer of 1983 and 1984 occurred when the natural inflow had little or no surface flow (Fig. 2, Table 2). The inflowing Hayes Creek did not contain high concentrations of phosphorus. Most of the samples collected by WIB (1977) had about 50 µg/L of total phosphorus and less than 10 µg/L of soluble reactive phosphorus.

However, WIB collected one sample with more than 100 µg/L of total phosphorus in November of 1976. Thus, the potential that periodically the phosphorus flux from upstream is high into Chain Lake can not be dismissed. Nutrients could enter the lake at other times, precipitate, and then later be redissolved (Kennedy et al. 1985). Phosphorus can be trapped and accumulated in lakes (Cross and Rigler 1983, Dillon and Rigler 1975). Regardless of this uncertainty, the main cause of the summer peak of phosphorus, can not be a

direct effect of upstream nutrient loading.

Phosphorus increased in the lake in spite of the low phosphorus content of the diversion water. The diversion water had a much lower cation content than Chain Lake or Link Lake, the next lake upstream (Table 3). In spite of this strong difference, a cation budget could not be used to validate the water budget. The cation composition of the springs, the likely major source of water in summer, was variable. The ion content of the lake has not changed significantly since 1975 (WIB 1977). Only one qualitative deduction can be made from the cation data: the diversion water does not comprise a major part of the lake volume.

The direct loading of human wastes could not produce the summer peak of phosphorus. The number of people required to produce this increase of phosphorus would be an order of magnitude more than ever stayed near the lake. The average occupancy in summer is near thirty. The average phosphorus loading from sewage is $1 \text{ kg capita}^{-1} \text{ year}^{-1}$ (Allum et al. 1977). The August 1983 increase in phosphorus in the lake was about 4.4 kg/day (equivalent to 1600 people living on the lake).

Groundwater can also be dismissed as a direct cause of the phosphorus increase in summer. The phosphorus composition of the springs varied greatly, but they always contained considerably less phosphorus than the lake (Table 4). The calcium and total inorganic carbon (TIC) content of the springs also varied greatly. The variability of phosphorus, calcium, and TIC may have been a reflection of the random distribution of veins of apatite and calcite that often

are found in this type of volcanic rock (Cockfield 1948).

All but one of the springs and wells had very little ammonia or nitrate (Table 4). The well with ammonia may have received diluted septic wastes, lake water, or it may have contained groundwater that flowed along the valley. The other sites were close to the valley walls and presumably contained groundwater flowing into the valley. If the phosphorus had been derived from animal wastes the nitrogen content should have been higher in all of the springs and wells. The soluble reactive phosphorus content of three springs (66, 54, and 37 $\mu\text{g/L}$) and Lee Creek (37 $\mu\text{g/L}$) was natural.

3.3 Lake Sediments

The increased phosphorus concentration in the lake water in late summer was probably a result of rapid phosphorus release from lake sediments. The surface of the sediments is much richer in phosphorus than the deeper muds (Fig. 3). Some of this surface phosphorus was probably derived from deeper sediments. As ferric iron was reduced to ferrous iron, phosphorus would have been solubilized (Berner 1980). The surface sediments are slightly enriched with iron (5.2% of mineral content is Fe) relative to the subsurface sediments (4.5%). Below 10 cm most of the iron is pyrite (FeS), which can not complex phosphorus. Groundwater can then transport phosphorus and the little iron that did not form pyrite, to the sediment surface (Van Liere and Mur 1982). If the surface surface was oxidized, phosphorus would precipitate with ferric iron.

Microbial activity would also contribute to the surface accumulation of phosphorus in the sediment. The surface sediments (0-2 cm) contained a high

proportion of decaying filaments of blue-green algae. The deeper sediments were almost entirely comprised of diatoms. The decay of the blue-green algae should release phosphorus into solution. The solubility of this phosphorus would be controlled by the degree of oxidation of iron.

The surface peak of phosphorus in the sediments was very recent. The peak of copper in the sediments at about 5 cm (Fig. 3) was probably produced by recent copper sulphate treatments.

Much of the phosphorus in the surface sediments was capable of entering the water column. In laboratory incubations, phosphorus was readily released into solution from surface sediments (Table 5). The addition of 5.6 mg/L or 22.4 mg/L of sodium acetate to lake sediments initially stimulated microbial assimilation of phosphorus. Later, the assimilation of acetate stimulated the release of phosphorus into solution (Table 5). These incubations were validated in a similar experiment in which the flasks were not shaken. In this experiment the surface of the sediments (control flasks too) was colonized by a flocculant one cm microbial layer. These concentrations of acetate should mimic the effect of sewage on lake sediments. This observation suggests that organic wastes from sewage, the swamp, or algal blooms could enhance the sediment release of phosphorus.

Incubation without oxygen resulted in greater phosphorus release. At the end of the first sediment incubations, the sediment was placed into a one liter graduated flask and sealed the flask with Parafilm (27 cm of mud and 29 cm of water). Weeks later gas began rising from the sediments. A two cm black band of

presumed ferrous oxide had formed a cm under the mud surface. Oligochaetes were briskly stirring the water. The had 2.6 mg/L O_2 and no hydrogen sulphide odour. After two months the SRP concentration in the overlying water had increased to 3.1 mg/L. More elaborate experiments would be required to determine the relative importance of geochemical and biological processes in phosphorus release.

This laboratory study and observations of lake sediments during a period of low oxygen support a local belief. Residents have reported that the algal blooms start in a bay near the campground. This site was the only site sampled in February 1985 where black sediments were observed. These sediments had a distinct hydrogen sulphide odour. At five other sites the sediments were brown and free of hydrogen sulphide. Poor water circulation and accumulation of wind blown algae in this bay could produce more heterotrophic activity, more anoxia, and more phosphorus release from the sediments.

Sediment release of phosphorus has often been reported (Bostrum and Pettersson 1982; Holdren and Armstrong 1980; Jacoby et al. 1982; Kennedy et al., 1985; Larsen et al. 1981; Lazoff 1983; Riley and Prepas 1984, 1985; Ryding and Forsberg 1977; Stefan and Hanson 1981; Yoshida 1982). The hypothesis that organic wastes can stimulate the release of phosphorus has been supported by Banoub (1977), Bostrom and Pettersson (1982), and Riley and Prepas (1984). The classic mechanism of phosphorus release from lake sediments is the release of phosphorus into solution as ferric iron is reduced in anoxic water to ferrous iron (Mortimer 1941, 1942).

In periods of high phosphorus concentrations in lake water in 1973, 1976

(Fig. 3 and 7 in WIB 1977) and again in 1983, and 1984 (Fig. 4) the lower water column was anoxic. The iron data supports the hypothesis that in anoxic water iron is converted from insoluble ferric iron to the more soluble ferrous iron. The iron content of Chain Lake doubled from July 6, 1984 to September 14, 1984 (Table 6). The anoxic water contained more than 1.5 mg/L of iron.

3.4 Seasonal Change in Phosphorus Solubility Reduces Effect of Water Diversion

The most striking difference in Chain Lake and other phosphorus-rich lakes on the Thompson Plateau (Frisken and Black Lake) was in the seasonal change of phosphorus solubility (Fig. 5). In Chain Lake, the concentration of soluble reactive phosphorus was high only if the oxygen concentration was low (Fig. 4). Much of the variability in the relationship in Figure 4 is probably a reflection of the degree of water stratification. The data that are at the top of the graph are all from periods of rapid wind mixing. In Black or Frisken Lake, oxygen did not influence phosphorus solubility. Black Lake (Murphy et al. 1983a,b) and Frisken Lake (Murphy et al. 1985) were carbonate lakes with very little reactive iron. The only appropriate hypothesis for this difference was that ferric phosphate reactions were much more important in Chain Lake than they were in Black or Frisken Lake.

The precipitation of phosphorus in Nov. of 1983 in Chain Lake was probably controlled by ferric phosphate precipitation. The oxygen concentration had increased from 4.4 to 8.0 mg/L. Similar changes occurred in other years. In 1975, the soluble reactive phosphorus concentration decreased by 50% (p 55 in WIB 1977) when the oxygen concentration increased from about 4 to 8 mg/L (Fig. 3 in WIB, 1977). Similarly, the 5-fold decrease in total phosphorus from the fall

of 1973 to the spring of 1974 (p 34 in WIB, 1977) was associated with a tripling of the oxygen concentration (Fig. 3 in WIB, 1977). Presumably, the increased oxygen concentration oxidized ferrous iron to ferric iron, which then precipitated to the lake sediments with adsorbed phosphorus. The precipitation of ferric phosphate and the adsorption of phosphorus to ferric hydroxide are well established reactions (Birch 1976; Stumm and Morgan 1981).

The precipitation of phosphorus in oxidized water may prevent the spring runoff from flushing the phosphorus that is released from the sediments. Therefore, phosphorus may accumulate in the lake to a greater extent than would be the case in a system which was low in iron.

3.5 Lake Flushing

The main causes for inadequate flushing of Chain Lake by the diversion must be low water flow and the inverse relationship of phosphorus solubility and water flow. However, the structure of the outlet and algal bouyancy may have also contributed to poor flushing of phosphorus. The design of the main lake outlet only allows the very surface water to flow out of the lake. The control structure is eight m wide. The water flowing out of the lake is usually only about five cm deep in summer. The presence of cattails at the lake outlet may have further restricted physical mixing and thus enhanced laminar outflow.

For most of the day, blue-green algae avoid the surface of the lake. Healthy blue-green algae move deeper into the lake to avoid photooxidation (Walsby 1977). For most of the summer, phosphorus in the surface water was almost entirely associated with particles, presumably algal cells. Only during

the periods of low oxygen (Table 7) was there significant dissolved phosphorus (Table 2). The combination of low light, calm weather, and dense algal blooms led to occasional high concentrations of algae in the surface water. Only some of our chlorophyll a data were collected at the same time in both the lake and outlet; however, these data support the frequent observations of poor flushing of algae from the lake ($\mu\text{g/L}$ chlorophyll a, 1.0 m in lake/outlet: 40/5; 127/22; 2/2; 100/20; 54/6, Table 8). These samples were all collected near noon.

This restricted flushing hypothesis was confirmed in 1984. From July 28 to Aug. 21, water samples were collected with an Isco automatic water sampler. Samples were pumped at 2:00 and 14:00 into the same bottle ;thus, any diurnal sampling bias should have been minimal. The rapid increases in total phosphorus concentration in the lake outlet could only have been produced by algae losing their buoyancy control (i.e., scum formation, Fig. 6). Thus, most of the time algae are flushed ineffectively.

Future studies could attempt to improve the flushing of algae. Perhaps the simplest approach would be to accelerate the outflow of water when the algae are near the surface.

In Sept. 1983, the secondary outlet was opened so that a dense algal bloom could be flushed. At that time, the soluble reactive phosphorus was more than a $100 \mu\text{g/L}$ (Table 2), and the lake water had less than 4.4 ppm of oxygen. The lake was very close to a fish kill. More than 20% of the phosphorus, and considerably more of the algae was flushed. This action may have been responsible for the higher oxygen concentrations in 1984. The secondary outlet was not opened in

1984. The algal bloom that formed in August 1984 was observed Feb. 14, 1985 as a dense green layer on the surface of the sediments. The oxygen concentration in the lake was a mean of 2.0 mg/L.

3.6 Phosphorus Budget

Calculations of phosphorus budgets often utilize extrapolation, indirect measurements, and interpolation; but useful insights into the lake cycling of phosphorus can be found with even the roughest estimates. In this study the summer budget (Table 9) would be stronger if groundwater flow had been directly measured ;however, an indirect approach does not weaken three statements.

1) The sediment release of phosphorus was much larger than any other phosphorus flux in summer. The phosphorus release rate was calculated by subtracting the mean amount of phosphorus in the lake in July from the amount of total phosphorus in the lake in late summer. The three sets of phosphorus data in July were good replicates (Table 2). The calculation of water volume is accurate because the lake bottom is quite flat. Corrections for efflux and influx of phosphorus in streams and the change of water volume were minor. The natural inlet was dry and groundwater and the diversion were relatively minor sources of phosphorus. Groundwater flow was assumed to be constant. The greatest error may be the measurement of total phosphorus in the lake in late summer. Even an error as high as 20% would not change the general nature of the water budget. The estimate of phosphorus release from the anoxic sediments is probably too low because the rapid sediment release of phosphorus could have been accompanied by a precipitation of phosphorus from the surface water. In late July and August the sediment phosphorus release was 19.7 and 9.1 mg P m⁻² d⁻¹ respectively.

2) The sediments had a very large reserve of "bioavailable" phosphorus that could be flushed into the water column by groundwater. To estimate the amount of phosphorus that could enter the water column the difference in the amount of "bioavailable" phosphorus (sodium hydroxide extractable) between the surface and subsurface sediments was used. Most studies of this type usually assume all of the sodium hydroxide extracted phosphorus is available (Williams et al., 1980).

This large reservoir of "bioavailable" phosphorus may override any attempt to control eutrophication of Chain Lake by reducing external phosphorus loading. Other investigations found that the suppression of external loading was unable to control eutrophication: phosphorus from sediments sustained algal blooms (Bjork 1972; Larsen and Malueg 1981).

3) The lake was losing much more phosphorus in summer than it received. The diversion may seem responsible for an enhanced flushing. However, Chain Lake may have always lost more phosphorus in summer than it received. In the estimate of a yearly phosphorus budget, Chain Lake retains more phosphorus than it loses (Table 9). Perhaps Chain Lake traps phosphorus in the spring runoff. The precipitation of ferric phosphate in spring could be an appropriate mechanism.

Extrapolations to a yearly budget are more speculative; however, that budget also leads to interesting observations. The yearly estimated flux of phosphorus into Chain Lake is much higher than would be predicted from Dillon and Rigler's model (1975). From estimates of cottage use in Ontario (Department of Tourism and Information 1971) and sewage data (Allum et al. 1977), the phosphorus load from cottages at Chain Lake is estimated to be about 29.3

kg/year. The Ontario data probably overestimate cottage use at Chain Lake. Upstream cottage phosphorus loading would be largely retained in those lakes (Dillon and Rigler 1975), and would not likely exceed the flux from Chain Lake. With the highest natural loading coefficient in Dillon and Rigler's paper, and hydraulic data from WIB (1977), the natural external phosphorus loading is estimated to be 96 kg of phosphorus a year. Evaporation was ignored.

A yearly estimate of natural phosphorus loading for Chain Lake is about 568 kg. This estimate is derived with the mean phosphorus inflow concentration of this study (29.7 $\mu\text{g P/L}$), the annual hydrology from WIB (1977), and the assumption that groundwater flow is relatively constant. The yearly estimate is only approximate, but nevertheless the difference from Dillon and Rigler's estimate of natural phosphorus loading is great. Natural weathering of the phosphate mineral apatite must be quite important in this valley. This calculation suggests that present cottage loading of phosphorus is insignificant. Related studies suggest that nitrogen may be a limiting nutrient and that any effect of cottages on eutrophication might be via nitrogen loading (section 3.7).

Another useful comparison can be made of the natural loading to the flushing imposed by the diversion. The diversion can only flush about a tenth of the external phosphorus loading. Certainly, earlier fears that unrestricted operation of the diversion would impair the productivity of the fishery in this lake were unwarranted. Unfortunately, an accurate estimate of how large the diversion should be to flush more phosphorus from the lake than enters the lake cannot be made from this data set. A ten-fold increase in the diversion flow

would dramatically improve the lake. The effect of a two or three-fold increase in flow cannot be resolved from this P budget.

Options such as accelerating the flushing of algal blooms are even more difficult to include into a model. Changing the hydrology must change the organic carbon load to the sediments and should influence the sediment release of phosphorus. Also, there appears to be a high degree of spatial variability in the development of anoxia in the lake sediments. Factors that initiate and control the sediment release of phosphorus need to be better quantified.

3.7 Nitrogen and Iron Availability in Chain Lake

Limnocoerrals were used to determine if algal oxygen production was suppressed by a lack of nitrogen or available iron.

The addition of ammonia greatly stimulated algal production of oxygen (Fig. 5). The Aphanizomenon density appeared to much higher in the ammonia limnocoerrals.

Although Chain Lake has high iron concentrations, much of the iron might be complexed with humic matter and this form of iron is not well characterized. Jackson and Hecky (1980) found humic iron unavailable to microorganisms. They proposed that iron was both strongly covalently bound and covered by organic layers. Hence the iron should be incapable of further reactions. However, in other lakes humic iron has been shown to react with phosphorus (Eisenreich and Armstrong 1980).

The addition of ferric citrate can stimulate both bacteria and algae. Bacteria utilize citrate as a carbon source; thus, some depletion of oxygen occurs. To resolve the effect of iron from the citrate effect, citric acid treated limnocorrals were used as control treatments. If iron enrichment stimulated algal oxygen production, the oxygen content of the iron citrate limnocorrals would be greater than the sodium citrate limnocorrals. Two untreated limnocorrals were also used as a standard to gauge the effect of iron stimulation.

The limnocorral additions of ferric citrate did not result in a significant stimulation of algal oxygen production (Fig. 7). After the third day, the oxygen content of the sodium citrate limnocorrals increased more than the ferric citrate limnocorrals. Presumably algae were not limited by a lack of iron. The iron response was much different from that observed in identical experiments conducted in Black Lake (Murphy 1985). After citrate was utilized in Black Lake, iron enriched limnocorrals produced more oxygen than the citric acid limnocorrals. The Chain Lake limnocorral experiments indicate that iron in Chain Lake was available to algae and presumably reactive. The relationship between oxygen concentration and phosphorus solubility also supports this hypothesis (section 3.5).

4. LAKE MANAGEMENT OPTIONS AND RESEARCH NEEDS

4.1 Control of Lake Level and the Swamp

The effect of the flooding of the swamp above Chain Lake upon Chain Lake can not be resolved with the present data. Ennis (1972) and WIB (1977) observed negative effects of the swamp upon water quality, but WIB is correct in recommending that not enough data exists to justify isolating the swamp from the lake.

The swamp could have reached a new equilibrium with the new water level imposed in 1957. Jackson and Hecky (1980) stated that flooded soils release a pulse of nutrients for up to several years. The swamp could have quite a different effect on the lake now. A critical reaction would be the formation of pyrite (FeS_2) and the subsequent solubilization of phosphorus. Any treatment that influenced the oxygenation of the swamp would change the rate of pyrite formation and rate of phosphorus release. An improvement in flow by the construction of channels should reduce phosphorus mobility, but the effect on eutrophication is uncertain.

The effects of channel construction on nitrogen reactions could be adverse. An intensive study in southern Ontario has shown that vegetation that significantly retards water drainage results in a loss of nitrogen via denitrification (Robinson et al. 1977). If the productivity of Chain Lake is restricted by low nitrogen loading, as our limnecorral work suggested (Fig. 54), the swamp could reduce eutrophication of Chain Lake.

Some lake restoration projects have built swamps to intercept nutrients before they reached the lake (Barten 1983, Hartland-Rowe and Wright 1974, Mudroch 1981, Seidel 1976). Before any major action is taken to isolate the swamp from the lake someone should demonstrate that the swamp discharges a nutrient that significantly effects the lake.

A study of the swamp would improve nutrient loading budgets for the lake. A high percentage of nutrient loading can occur during peaks of high flow (Fricker 1981; Kennedy et al. 1985). Fricker (1981) has shown that nutrient analysis must be done in conjunction with flow measurements and this approach has only been used at Chain Lake for a summer.

The option of lowering the water level back to 1956 levels so that the swamp drains presents two risks. 1) The lake would then be shallow enough that rooted macrophytes could grow in most of the lake. 2) The risk of winter fish kills would increase (Barica and Mathias 1979).

4.2 Septic Wastes

The soils at Chain Lake are composed of very porous gravel ;thus, soil retention of nutrients would be minimal (Dillon and Rigler 1975; Otis 1979). At Chain Lake, the effect of nitrogen in sewage could be more important than the phosphorus or organic content of sewage. The phosphorus content of the sewage appears insignificant relative to the larger sediment release of phosphorus and to the groundwater phosphorus supply. The organic carbon load from sewage might be less significant than the carbon flux from algal blooms. With the use of data from five northern eutrophic lakes (mean carbon production of $427 \text{ g M}^{-2} \text{ y}^{-1}$, Wetzel 1975) it can be estimated that Chain Lake algae would produce as much

carbon as the sewage of 2000 people.

The effect of nitrogenous wastes on Chain Lake is uncertain. The limnocorral experiment in this study demonstrated significant stimulation from ammonia enrichment ;thus, nitrogen could be limiting the production of Chain Lake. A calculation of the degree of importance cannot be made. Blue-green algae might be supplying significant nitrogen loading via nitrogen fixation. The bioavailability of dissolved organic nitrogen in the lake and springs is unknown. The availability and microbial demand for nitrogen in the sediments is not known. How much the cottage nitrogenous wastes influence the sediment phosphorus release is not known.

The concept that cottage wastes can result in eutrophic lakes is well established in Ontario (Dillon and Rigler 1975; Michalski et al. 1973). The model that Dillon and Rigler developed to estimate the effect of cottage development on eutrophication of Ontario lakes has many limitations for use in B.C. Empirical models require data that represent the local site.

4.3 Flushing of Chain Lake

Modifications to the lake outlet could be made to improve the flushing of algae from Chain Lake. Accelerating the flushing of algae could remove phosphorus from the lake and perhaps suppress phosphorus release from the sediments. This procedure would require constant attention from a responsible resident. If enough water is available, manipulation of the outlet could be effective.

In 1983, enough water was flowing to flush one bloom in July, another bloom

in August, and two or three in September. The June, July, and August rainfall of 1983 was slightly greater than normal (218 mm vs a 9 year mean of 203.4 mm at Chain Lake or a 202.5 mm mean of 16 years of data from Osprey Lake). Presumably, the outflow was only slightly more than normal. Many opportunities for flushing algal blooms occurred in 1983.

The algal bloom on July 16, 1984 could have been flushed. A dense bloom developed on the southern half of the lake. The chlorophyll a (169 µg/L) and total phosphorus concentrations (355 µg/L) were exceptionally high in the surface water (0.2 m). The northern end of the lake was much less turbid than the southern end (1.8 m vs. 0.6 m Secchi disk depth). With an ideal flush, about 20% of the phosphorus and over 50% of the algae could have been flushed by losing only 5% of the lake water. The 1984 example represents an infrequent opportunity. July 16, 1984 was the only day in which the primary sampling site did not represent the lake well.

Because of the need to prevent coarse fish movement into Chain Lake, the main outlet structure probably cannot be modified. The secondary outlet could be modified so that algae could be flushed from the lake before they settled to the sediments. The secondary outlet is a 12" pipe that exits the lake through the earthen dam. This could be modified to maximize a laminar flow of the top 0.5 m of water. The optimal design would minimize turbulent upwelling of deeper water into the inlet. To allow rapid flow and minimal water mixing the effective diameter of the outlet would have to be about 30 meters (from Boyce et al. 1983). A training structure could produce the effect of an outlet with a 30 m diameter. Woven polyethylene could be installed from the lake bottom to 0.5 m from the lake surface in a semicircle with a 15 m radius around the pipe. The

pipe size should be increased so that greater flows could be generated at critical times.

The timing of the flush would be very important. Flushing too soon would waste water. If the flush was too late or too slow most of the algae would settle to the sediments. The flush must be made with a fast flow to avoid a potential problem of coarse fish entry into Chain Lake through this outlet. Perhaps by only operating this outlet with high flows, coarse fish would not be able to enter the lake. Other preventive measures should also be taken to ensure that coarse fish could not enter the lake.

Successful implementation of this accelerated flushing would lead to a reevaluation of the diversion. The diversion adds enough water for a flush in summer. If more water was available in the critical period of July and August at least the cosmetic effect of clearing away the blooms would be enhanced.

With the present flow regime the diversion flow would have to be increased several fold to overcome the nutrient flux from the sediments. If the sediment release of phosphorus could be reduced, the effectiveness of the diversion could be increased. Although the hypothesis is speculative, it is quite possible that a doubling of the diversion, with an accelerated flushing, could significantly reduce eutrophication of Chain Lake. This accelerated flushing scenario is highly speculative and it requires that a local "keeper of the dam" make decisions regularly. Acceptance of the lake residents to a fluctuating lake level would also be required (as much as a 0.5 m drop).

In the future, the diversion water could be very useful if sediment dredging is utilized as a lake restoration procedure. A significant cost of most sediment dredging treatments is associated with the cost of treating the water pumped with sediment for return to the lake. At Chain Lake, that water could be used for irrigation of downstream farms at much less cost.

4.4 Sediment Dredging

Sediment dredging is the only effective long-term lake restoration treatment for eutrophic shallow lakes. Sediment dredging has potential applications at many sites in B.C. (eg. Dragon Lake) and other sites in western Canada (eg. Pike Lake, near Saskatoon). Riley and Prepas (1985) have shown that shallow lakes in western Canada are more productive per unit of spring phosphorus than are deep lakes. Sediment release of phosphorus in shallow lakes seems to produce this effect (Riley and Prepas 1984).

Sediment dredging has successfully reduced eutrophication in several lakes in Sweden (Andersson et al. 1973; Bjork 1972; Ryding 1982), Japan (Yoshida 1982), and the U.S.A. (Breithaupt et al. 1983; Peterson 1979; Wisconsin Department of Natural Resources 1981). Dredging of carp ponds is often done to remove aquatic weeds and silt, albeit the reduced productivity created by dredging in those aquaculture systems was not desirable (Hejny and Husak 1978). Other shallow lakes have been dredged and the projects are not fully reported yet (Dunst 1980; Worth 1981). Seven lakes are currently being dredged via funding from the U.S.A. Clean Lakes program (NALMS 1985). Many reports on dredging are not widely distributed (Tillotson, Mudcat, Detroit).

The procedure is relatively expensive ;however, two major expenses would not apply at Chain Lake. A major cost of dredging has been the treatment of the dredged water with chemicals to remove nutrients so that the water can be added back to the lake. The Shinish Diversion water would make this treatment unnecessary. Another major cost of dredging has been land acquisition for disposal of dredged mud. At Chain Lake and much of Canada, land is readily available. The mechanical and engineering costs of dredging are high when expressed per site, but like other long-term treatments (water diversions, sewer lines etc.) the cost can be amortized over many years.

Sediment dredging is a mechanism for removing nutrients from a lake. Since the surface sediments are richer in phosphorus and this phosphorus is more mobile than the phosphorus in deeper muds, removal of the surface muds should reduce the release of phosphorus into the water column. The removal of the top 10 or 80 cm of sediment from Chain Lake would reduce the bioavailable phosphorus content of the surface sediments by 50 or 75% respectively (Fig. 3). The only study that appears to support this hypothesis is the Lake Trummen study (Andersson et al. 1973; Bjork 1972). They increased the maximum depth from two m to three m by removing the top metre of nutrient rich black mud on a one km² lake to expose a brown gyttja. This lake restoration project was successful and the effect was probably caused by nutrient removal. That amount of dredging would not greatly alter the physical stratification of that size of lake (Stefan and Hansen 1981a).

Physical deepening of a lake may be more important than nutrient removal. Stefan and Hanson (1981a, 1981b) state that lake dredging only reduces eutrophication by physical processes related to a deeper water column. Also,

dredges are not designed for shallow dredging. They may have overstated the physical aspects, but any argument is academic. A dredge moves around a lake very slowly and removing the very surface mud (ie. 10 cm) would be inefficient. If the dredge head was not well immersed into the sediment, more lake water than sediment would be pumped.

Deeper dredging would maximize the probability of reducing eutrophication. Deeper lakes are more stratified and sediment processes have less influence on the epilimnion. Eventually the sediment release of phosphorus would be reestablished. The properties associated with a deeper lake would last longer than the effects of removing the surface sediments.

Sediment dredging would reduce eutrophication by reducing sediment resuspension. Sediment mixing by wind can rapidly enhance the flux of nutrients from the sediments. With the equations of Carper and Bachman (1984) and a fetch of 1500 m, it can be estimated that a wind greater than 60 km/h could resuspend the sediments from the deepest parts of the lake. If the lake was two meters deeper, sediment resuspension could only occur with winds greater than 80 km/h.

Sediment dredging would reduce eutrophication in other ways. The greater water volume contains a greater reserve of oxygen; thus, the rate of reduction of ferric iron to ferrous iron and related phosphorus release would be decreased. Increasing the oxygen content would greatly reduce the potential of winter fish kills (Barica and Mathias 1979). Establishing thermal stratification in a lake is good in two ways. Nutrients in the hypolimnion would be much less available for algal growth (Schindler 1971). Thus, if significant sediment release of phosphorus was reestablished most of the nutrients would be less

available for algal growth. Also, the sediments would remain much cooler. Cooler sediments would have less microbial productivity, less oxygen consumption, less reduction of ferric iron to ferrous iron, and less phosphorus release (Kamp-Neilsen 1974).

The amount of sediment removal needed to establish stratification is dependent upon the size of the lake and the maximum wind speed. Data from Lake Sisseton in Minnesota could be used as a guide (Stefan and Hansen 1981). Lake Sisseton is 54 hectares in size and had a maximum and mean depth of 6.0 and 3.5 metres. Stefan and Hansen estimated, that dredging to a maximum and mean depth of 8.0 and 5.8 metres would produce stratification. In Chain Lake, a similar morphology could be established by dredging an average of a metre of sediment.

Stefan and Hansen's data indicate that the complete dredging of Chain Lake would cost \$315,000. A pilot study is required to 1) better estimate the cost for dredging Chain Lake, 2) assess the potential sale of the dredged material, 3) to predict the effect of extensive or moderate dredging on Chain Lake, and 4) to design procedures for dredging small lakes in Canada.

4.4.1 Use of Lake Sediments as a Soil Conditioner

When lake sediments can be sold the cost of dredging can be considerably reduced. Sediment from dredged lakes in Massachusetts and Holland are sold (Worth 1979, Kleinbloesem and Gast 1980). Because the interior of British Columbia has little soil, the farmland in the Okanagan is protected from development by legislation. Thus, the demand for topsoil is high. A dry lake bed near Carmi is mined and sold for topsoil in the south Okanagan (Von Neisen, Naramata). Transportation to the north Okanagan from Chain Lake via Peachland is

the same distance as from Carmi to the southern Okanagan.

4.4.1.1 Effect of Copper Algicide On Sediment Use

Copper sulphate had been used for a short period as an algicide. The copper content of the Chain Lake sediments is not high enough to prevent their use as a soil fertilizer. The mean concentration in the surface sediments is about 90 $\mu\text{g/g}$ and if dredging is done to two meters the mean copper concentration should be about 50 $\mu\text{g/g}$. These concentration are not unusually high, and are low for an area that is naturally rich in copper (such as much of the Thompson Plateau) (Thornton 1979).

The copper in the Chain Lake sediments is much less concentrated than sewage sludge. The mean concentration of copper in Ontario sludge is 810 $\mu\text{g/g}$ (Campbell and Bridle, Canada Center for Inland Waters, Burlington, unpublished data). With very few restrictions the Ontario sludge is freely available for application to farms. The mean concentration of copper in uncontaminated sediments of the Lower Great Lakes exceeds that found in the surface of Chain Lake (Nriagu et al. 1981) and sediments from uncontaminated areas is applied to fields.

Nevertheless, copper in high concentrations is toxic (Nriagu 1979, Spear and Pierce 1979) and caution is appropriate. Continued use of copper to suppress algal growth in lakes that are used for sediment recovery is questionable.

4.4.1.2 Fertilizer Value

The use of Chain Lake sediments as a fertilizer in calcareous soils has considerable promise. Iron limitation is commonly found in alkaline soils (Brown

1979). Iron can be present in limiting quantities in soils in the Okanagan and Similkameen areas (J. Dawson, farmer in Summerland and Cawston, B.C.). The highly reactive iron in the Chain Lake mud should be an excellent iron source. The high organic content of the Chain Lake sediments (50%) would enhance the solubility of iron.

The organic matter could dissolve phosphate minerals in the soil and thus enhance the availability of phosphorus (Martinez et al. 1983). The high cation exchange capacity of the organic matter could reduce the leaching of nutrients. The nitrogen content of the sediment (2%) could make the sediments competitive with commercial nitrogen fertilizers (equivalent value of \$40.00 a tonne). Calcium is present as calcium carbonate (1%). The water retention of soils treated with lake sediment should be improved by the organic matter.

The organic or trace metal contents may determine if the sale of Chain Lake sediments could become commercially viable to farmers. If no organic dissolution of residual phosphorus occurred and farmers had to add enough phosphorus to equal inorganic fertilizer application (Fixen and Ludwick 1983), farmers would require a layer of dried mud a cm deep. Since transportation charges are high the cost of using this much mud would probably restrict purchase of sediment to small gardeners. Since lake sediments are free of weed seeds, sediments would be an excellent top soil for seeding lawns, flowerbeds, etc. Test gardens should be used to determine the optimal sediment application rate.

4.4.2 Processing of Chain Lakes as Diatomaceous Earth

The sediments of Chain Lake are a relatively pure deposit of diatomaceous earth. The phosphorus and silica rich volcanic rock has resulted in the

deposition of at least a three metre deep deposit that appear to cover the 43 hectare lake bottom. The lake sediments are over nine metres deep; however, the composition of the deeper sediments is unknown. The swamp at the inlet to the lake restricted entry of soil into the lake.

Diatomaceous earth has many industrial uses. It is used as filtration media, filler, adsorbant, mild abrasive, anticaking agent, catalyst, and a soil conditioner (Meisinger 1980, Kadey 1983). Lake deposits are mined in Iceland and England (Meisinger 1980). Lake deposits used to be mined in eastern Canada (Eardley-Wilmot 1928) and new deposits have been recently sought under Ontario Lakes by the Ontario government (Story, Ministry of Natural Resources, Kenora).

The world market for diatomaceous earth is controlled by a few large American firms. The sale of certain diatomaceous earth products might be difficult. However, some products such as kitty litter, insulating bricks, oil adsorbant, and some water filtration aids, would require little product development. If the products could be produced in a cottage-scale industry, the firm could be competitive in at least local markets.

4.4.3 Sediment Recovery

Sedimentation basins would need to be constructed on land south of the lake. Mr. G. Smith would allow the use of his land free. The basins would have to be large to allow dispersal of sediment in a layer no deeper than 1.5 metre. This type of basin is used in the Netherlands to obtain dry soil after a year (Kleinbloesem and Gast 1980). The aqueous supernatant in the basins could be pumped onto the farm immediately below the basins with no land disposal fee. No discharge to the stream would occur.

4.4.4 Additional Environmental Mitigation Measures

Although no barriers are usually placed around dredges working in uncontaminated areas, a bubble barrier could be used to isolate a pilot study in Chain Lake. The southern bay could be isolated by first lowering the lake level so that no silt laden water entered the stream, and by second placing a bubble barrier across the points of land (van Oostrum et al. 1980).

4.5 Partial Lake Dredging

For several reasons a pilot project is needed. Lake dredging has not been done in Canada in the fashion that would be needed for lake restoration. Some experience with operating problems is required. An evaluation of a dredge requires data on the pumping rate of mud, the % water content of the mud/water mixture, how the dredge responds to debris like sunken logs, how much resuspension of mud into the water column occurs, how long any turbidity lasts, how quickly the mud dries on shore, and how quickly the chironomids recolonize the sediments (Andersson et al. 1975).

A pilot project could test the utility of dredging part of the lake. The critical test is whether a hole two metres in depth and two hectares in area in Chain Lake would remain stratified. Data is needed on how the sediments move in the lake. The potential movement of neighbouring muds into the dredged hole by either slumping or sediment focusing would determine the objectives that partial dredging of a lake could achieve. Sediment focusing is the movement of fine particles like algal cells into the water column during storms and the resettlement into deeper and less turbulent water (Hargrave and Neilsen 1979). Sediment focusing may be important at Chain Lake. Chain Lake can be very windy and the lake bottom is very flat. Sediment focusing could have filled in any

holes that were originally present. If a hole was dug at the southern end of the lake, fine particles would eventually fill that hole. The rate at which that occurred would determine what options were available for a limited or partial dredging of Chain Lake.

If the hole filled in very quickly (within a year), then an attractive scenario might be to redredge that hole every fall, until most of the fine organic matter was removed from the lake. Harbours that require dredging each year often use siltation basins to trap the finer silt. The siltation basins can be emptied later with simple pumps with a 70% reduction in cost (van Ostrum 1980, Berger et al. 1985). Chain Lake muds have a high water content (Fig. 8). This option might achieve the removal of the surface nutrient-rich layer from all of the lake without moving a dredge over the whole lake.

If sediment focusing was moderate, then the hole would become anoxic and rich in phosphorus. Hypolimnetic withdrawal (the pumping of bottom water from the lake) would then become an attractive option. This procedure has been used in many reservoirs (Martin and Arneson 1978; Mackie et al. 1983; Kortmann et al. 1983; Wisconsin Department of Natural Resources 1974) and is considered for Dragon Lake, B.C. This procedure greatly enhances the flushing of nutrients from a lake or reservoir.

If sediment focusing and slumping were insignificant, then a third option might be considered. A deep hole could provide a suitable site for lake aeration. Aeration is more efficient in deeper water because the greater hydrostatic pressure enhances the diffusion of oxygen (Ashley 1984). Chain Lake has a high iron concentration and if the water column could be kept oxidized,

phosphorus would be precipitated with ferric iron and made unavailable for algal growth. Lake aeration in this type of lake would be much more effective than in hypertrophic carbonate lakes that typically have much less reactive iron. If lake aeration could maintain oxygen values above 5 mg/L, phosphorus should be precipitated in Chain Lake. Chain Lake has about nine metres of sediment that could be dredged and the B.C. government has several staff that are trained in lake aeration. If aeration basins could be established in shallow lakes, many lakes with winter and fish summer kills might be greatly improved.

All of these partial dredging options are presented to illustrate that short-term objectives within a pilot study are valid. Too many uncertainties exist to consider the optimal dredging program now. A pilot-scale dredging program that contained extensive environmental mitigation measures could be done by a government for about \$50,000. A pilot study would establish the cost for a private firm to carry out future dredging programs. Guidelines for environmental mitigative measures and dredging procedures in small lakes could be developed.

Summary

The original calculations that were used to predict the effect of the water diversion underestimated the phosphorus loading into Chain Lake. Groundwater discharge of phosphorus appears to be high and must be a result of weathering of apatite in the volcanic rock. Phosphorus is concentrated in the surface sediments and released into the overlying water during periods of anoxia. The anoxia occurs during periods of low flow ;thus, the diversion can not flush phosphorus from the lake fast enough to suppress eutrophication.

Some improvement in water quality can be achieved by rebuilding the lake outlet so that algal blooms can be flushed more effectively. To achieve a major reduction in eutrophication would require either diversion of all of the flow of Shinish Creek from July 1 to Oct. 1 into Chain Lake or sediment dredging of much of the lake.

References

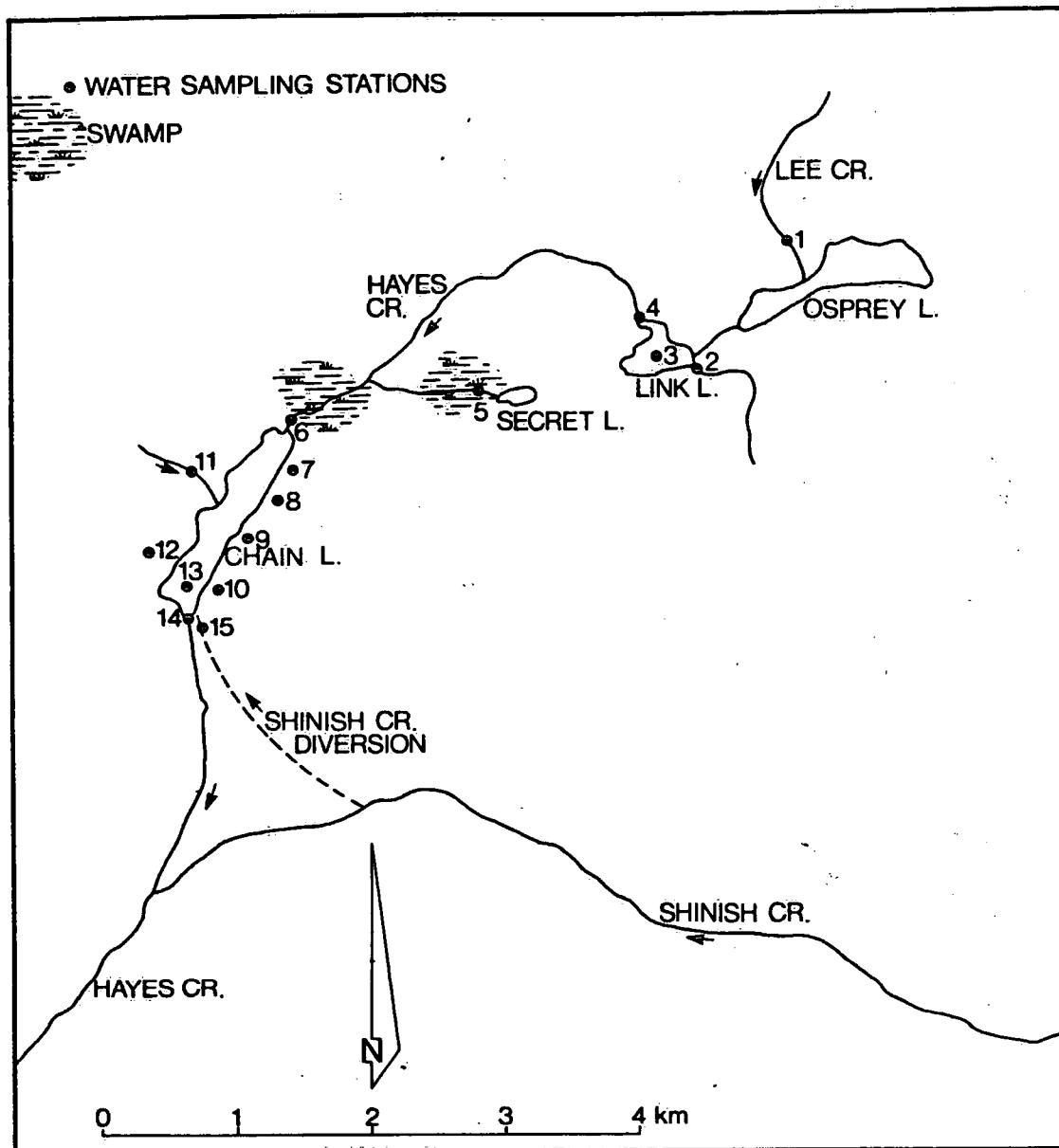
- Allum, M.O., R.E. Glessner, and J.H. Gakstatter. 1977. "An evaluation of the National Eutrophication Survey Data-Working Paper No. 900," National Eutrophication Survey, Corvallis Environmental Research Laboratory, Corvallis, Oregon, U.S. EPA, GPO 699-440.
- Andersson, G., G. Cronberg, and C. Gelin. 1973. Planktonic changes following the restoration of Lake Trummen, Sweden. *Ambio* 2:44-47.
- Andersson, G., H. Berggren, and S. Hamrin. 1975. Lake Trummen restoration project III. Zooplankton macrobenthos and fish. *Verh. Int. Verein. Limnol.* 19:1097-1106.
- Ashley, K.I. 1984. Hypolimnetic aeration: Practical design and application. *Water Research* 19:735-740.
- Banoub, M.W. 1977. Experimental investigation on the release of phosphorus in relation to iron in freshwater/mud system, p 324-330. In H.L. Golterman (ed.), *Interactions Between Sediments and Freshwater*. Dr. W. Junk, The Hague.
- Barica, J., and J.A. Mathias. 1979. Oxygen depletion and winterkill risk in small prairie lakes. *J. Fish. Res. Board Can.* 36:980-986.
- Barten, J. 1983. Nutrient removal from urban stormwater by wetland filtration: the Clear Lake restoration project, p. 23-30. In *Lake Restoration, Protection, and Management*. EPA 440/5-83-001.
- Baxter, R.M. 1977. The environmental effects of dams and impoundments. *Annu. Rev. Ecol. Syst.* 8:255-283.
- Berger, R.C., S.B. Hertzfel, R.F. Athow, D.R. Richards, and M.J. Trawle. 1985. Norfolk Harbour and channels deepening study. Department of Army, Vicksburg, Mississippi, USA, 39180-0631.
- Berner, R.A. 1980. *Early Diagenesis*. Princeton University Press, Princeton, New Jersey.
- Birch, P.B. 1976. The relationship of sedimentation and nutrient cycling to the trophic status of four lakes in the Lake Washington drainage. PhD. Dissertation. Univ. Washington, Seattle.
- Bjork, S. 1972. Swedish lake restoration program gets results. *Ambio* 1:153-165.
- Bostrom, B., and K. Pettersson. 1982. Different patterns of phosphorus release from lake sediments in laboratory experiments. *Hydrobiologia* 92: 415-429.
- Boyce, F.M., D.G. Robertson, and G.N. Ivey. 1983. Summer thermal structure of Lake Ontario off Toronto: Cooling the big city. *Atmosphere-Ocean* 21:397-417.
- Breithaupt, S.A. 1983. The dredging of liberty lake and its immediate environmental impact, p. 279-283. In *Lake Restoration, Protection, and Management*. EPA 440/5-83-001.
- Brown, J.C. 1979. Iron and Plants, p. 57-78. In *Iron. (Medical and biological effects of environmental pollutants series)*. Univ. Park Press, Baltimore.
- Burnison, B.K. 1980. Modified dimethyl sulfoxide (DMSO) extraction for chlorophyll analysis of phytoplankton. *Can. J. Fish. Aquat. Sci.* 37:729-733.
- Carper, G.L., and R.W. Bachmann. 1984. Wind resuspension of sediments in a prairie lake. *Can. J. Fish. Aquat. Sci.* 41:1763-1767.
- Cockfield, W.E. 1947. Map 886A Nicola, Kamloops and Yale Districts. Geological Survey of Canada.

- Cockfield, W.E. 1948. Geology and mineral deposits of Nicola map-area, British Columbia. Can. Dept. Mines Resources Memoir 249. 42p.
- Cross, P.M., and F.H. Rigler. 1983. Phosphorus and iron retention in sediments measured by mass budget calculations and directly. Can. J. Fish. Aquat. Sci. 40:1589-1597.
- Department of Tourism and Information. 1971. Analysis of Ontario cottage survey. Travel Research Report. No.55:114p.
- Dillon, P.J., and F.H. Rigler. 1975. A simple method for predicting the capacity of a lake for development based on lake trophic status. J. Fish. Res. Board Can. 32:1519-1531.
- Dunst, R.C. 1981. Dredging activities in Wisconsin's lake renewal program, p. 86-88. In Lake Restoration of Lakes and Inland Waters. EPA 440/5-81-010.
- Eardley-Wilmot, V.L. 1928. Diatomite, its occurrence, preparation and uses. Can. Depart. Mines No. 691.
- Eisenreich, S.J. and D.E. Armstrong. 1980. Association of organic matter, iron and inorganic phosphorus in lake waters. Environment International 3:485-490.
- Ennis, G.L. 1972. Effects of low nutrient water addition on summer phytoplankton diversity in a small eutrophic lake. B.Sc. Thesis, University of British Columbia.
- Fixen, P.E., and A.E. Ludwick. 1983. Phosphorus and potassium fertilization of irrigated alfalfa on calcareous soils: 1. Soil test maintenance requirements. Soil Sci. Soc. Am. J. 47:107-112.
- Fricker, H. 1981. Critical evaluation of the application of statistical phosphorus loading models to alpine lakes. PhD Thesis Swiss Federal Institute of Technology, Zurich.
- Hargrave, B.T. and L.K. Neilsen. 1979. Accumulation of sedimentary organic matter at the base of steep bottom gradients p. 168-173. In H.L. Golterman (ed.), Interactions Between Sediments and Freshwater. Dr. W. Junk, The Hague.
- Hartland-Rowe, R.C.B., and P.B. Wright. 1974. Swamplands for sewage effluents. Report No. 74-4 of Task Force on Northern Oil Development. QS-1553-000-EE-A1.
- Hejny, S., and S. Husak. 1978. Ecological effects of fishpond amelioration. p. 409-415. In D. Dykyjova and J. Kvet (ed.), Pond Littoral Ecosystems. Springer-Verlag, New York.
- Holdren, G.C., and D.E. Armstrong. 1980. Factors affecting phosphorus release from intact lake sediment cores. Environ. Sci. Technol. 14:79-87.
- Jackson, T.A., and R.E. Hecky. 1980. Depression of primary productivity by humic matter in lake and reservoir waters of the boreal forest zone. Can. J. Fish. Aquat. Sci. 37:2300-2316.
- Jacoby, J.M., D.D. Lynch, E.B. Welch, and M.A. Perkins. 1982. Internal phosphorus loading in a shallow eutrophic lake. Water Res. 16:911-919.
- Kadey, F.L. 1983. Diatomite. In S.J. Lefond (ed.) Industrial minerals and rocks. Am. Inst. Mining Metal. Pet. Eng., New York
- Kamp-Nielson, L. 1974. Mud-water exchange of phosphate and other ions in undisturbed sediment cores and factors affecting the exchange rates. Arch. Hydrobiol. 73:218-237.
- Kennedy, R.H., W.F. James, R.H. Montgomery, and J. Mix. 1985. The influence of sediments on the nutrient status of Degray Lake, Arkansas. Proceedings of the Third International Symposium of Interactions Between Sediments and Water. Geneva. Aug. 1984, in press.

- Kleinbloessem, W.C.H., and J. de Gast. 1980. Dredged silt in Rotterdam looks valuable. Proceedings of WODCON IX, Symcon Pub. Long Beach.
- Kortmann, R.W., E. Davis, C.R. Frink, and D.D. Henry. 1983. Hypolimnetic withdrawal: Restoration of Lake Wononscopomuc, Connecticut, p. 46-55. In Lake Restoration, Protection and Management. EPA 449/5-83-001.
- Lawson, D.W. 1967. A groundwater investigation of the Trapping Creek representative basin. Master Science Thesis, University of Guelph, Ontario.
- Larsen, D.P., K.W. Malueg, D.W. Schults, and R.M. Brice. 1975. Response of eutrophic Shagawa Lake, Minnesota, U.S.A., to point-source, phosphorus reduction. Verh. Int. Ver. Limnol. 19:884-892.
- Larsen, D.P., D.W. Schults, and K.W. Malueg. 1981. Summer internal phosphorus supplies in Shagawa Lake, Minnesota. Limnol. Oceanogr. 26:740-753.
- Larsen, D.P., and K.W. Malueg. 1981. Whatever became of Shagawa Lake? p. 67-72. In Restoration of Lakes and Inland Waters. EPA 440/5-81-010.
- Lazoff, S.B. 1983. Evaluation of internal phosphorus loading from anaerobic sediments, p. 123-126 In Lake Restoration, Protection and Management. EPA 440/5-83-001.
- Mackie, G.L., J.B. Rooke, J.C. Roff, and J.F. Gerrath. 1983. Effects of changes in discharge level on temperature and oxygen regimes in a new reservoir and downstream. Hydrobiologia 101:179-188.
- Martin, D.B., and R.D. Arneson. 1978. Comparative limnology of a deep-discharge reservoir and a surface-discharge lake on the Madison River, Montana. Freshwater Biology 8:33-42.
- Martinez, M.T., C. Romero, and J.M. Gavilan. 1984. Solubilization of phosphorus by humic acids from lignite. Soil Science 138:257-261.
- Meisinger, A.C. 1980. Diatomite. In. Mineral Facts and problems. US Bureau of Mines Bull. 671.
- Michalski, M.F.P., M.G. Johnson, and D.M. Veal. 1973. Muskoka Lakes water quality evaluation. Report No. 3. Ministry of Environment, May 1973.
- Mortimer, C.H. 1941. The exchange of dissolved substances between mud and water in lakes: I and II. J. Ecol. 29:280-329.
- Mortimer, C.H. 1942. The exchange of dissolved substances between mud and water in lakes: III and IV. J. Ecol. 30:147-201.
- Mudroch, A. 1981. A study of selected Great Lakes Coastal Marshes. Scientific Series No. 122. National Water Research Institute, Burlington, Ont.
- Murphy, T.P. 1985. The regulation of blue-green algal blooms by iron availability and calcite precipitation in hardwater lakes. PhD thesis, Zoology Dep. U.B.C. Vancouver, Canada.
- Murphy, T.P., K.J. Hall, and I. Yesaki. 1983a. Coprecipitation of phosphate with calcite in a naturally eutrophic lake. Limnol. Oceanogr. 28:58-69.
- Murphy, T.P., K.J. Hall, and I. Yesaki. 1983b. Biogenic regulation of iron availability in a eutrophic hardwater lake. The Science of the Total Environment 28:37-50.
- Murphy, T.P., K.J. Hall, K.I. Ashley, A. Mudroch, M. Mawhinney, and H.J. Fricker. 1985. In-lake precipitation of phosphorus by lime treatment NWRI contribution series 85-167.
- NALMS, 1985. Clean Lakes: status report, Lake Line 5:32.
- Nriagu, J.O. 1979. Copper in the Environment. Part 1: Ecological Cycling. John Wiley & Sons, Inc., Toronto.
- Nriagu, J.O., H.K.T. Wong, and R.D. Coker. 1981. Particulate and dissolved trace metals in Lake Ontario. Water Res. 15:91-96.

- Otis, R.J. 1979. Treatment of domestic wastes in lakeshore developments, p. 95-102. In *Lake Restoration*. EPA 440/5-79-001.
- Peterson, S.A. 1979. Dredging and lake restoration, p. 105-114. In *Lake Restoration*. EPA 440/5-79-001.
- Rice, H.M.A. 1946. Princeton map area, Sheet 92H (east half), Geol. Surv. Can.
- Riley, E.T., and E.E. Prepas. 1984. Role of internal phosphorus loading in two shallow, productive lakes in Alberta, Canada. *Can. J. Fish. Aquat. Sci.* 41:845-855.
- Riley, E.T. and E.E. Prepas. 1985. Role of internal phosphorus loading in two shallow, productive lakes in Alberta, Canada. *Can. J. Fish. Aquat. Sci.* 41:845-855.
- Robinson, J.B., J.A. Cherry, D. Draper, N.K. Kaushik, C.K. Lee, R. Mason, P. Sain, W.N. Stammers, W.H. Stiebel, and H.R. Whitely. 1977. Modelling nitrogen transport and cycling in a natural flow system. Final Report to Inland Waters Directorate. 267p.
- Ryding, S.-O., and C. Forsberg. 1977. Sediments as a nutrient source in shallow polluted lakes, p. 227-234. In H.L. Golterman (ed.), *Interactions Between Sediments and Freshwater*. Dr W. Junk, The Hague.
- Ryding, S.-O. 1982. Lake Trehorningen restoration project. Changes in water quality after sediment dredging. *Hydrobiologia* 92:549-558.
- Schindler, D.W. 1971. A hypothesis to explain differences and similarities among lakes in the experimental lakes area, Northwestern Ontario. *J. Fish. Res. Bd. Canada* 28:295-301.
- Seidel, K. 1976. Macrophytes and water pollution, p. 109-122. In J. Tourbier and R.W. Pierson (eds.), *Biological Control of Water Pollution*. University of Pennsylvania Press.
- Solorzano, L. 1969. Determination of ammonia in natural waters by the phenylhypochlorite method. *Limnol. Oceanogr.* 14:799-801.
- Spear, P.A., and R.C. Pierce. 1979. Copper in the Aquatic Environment: Chemistry, Distribution, and Toxicology. Pub. No. NRCC 16454, Envir. Secretariat, Canada.
- Stefan, H.G., and M.J. Hanson. 1981a. Phosphorus recycling in five shallow lakes. *ASCE J. Environ. Eng. Div.* 107:713-730.
- Stefan, H.G., and M.J. Hanson. 1981b. Predicting dredging depths to minimise internal nutrient recycling in shallow lakes, p. 79-85. In *Restoration of Lakes and Inland Waters*. EPA 440/5-81-010.
- Strickland, J.D.H., and T.R. Parsons. 1972. A practical manual of seawater analysis. *Bull. Fish. Res. Board Can.* 167.
- Stumm, W., and J.J. Morgan. 1981. *Aquatic Chemistry: An Introduction Emphasizing Chemical Equilibria in Natural Waters*, 2nd edition. John Wiley & Sons, Inc. Toronto.
- Thornton, I. 1979. Copper in soils and sediments, p. 171-216. In J.O. Nriagu (ed.) *Copper in the Environment Part 1: Ecological Cycling*. John Wiley & Sons, Inc., Toronto.
- Van Liere, L., and L. R. Mur. 1982. The influence of simulated groundwater movement on the continuous flow system. *Hydrobiologia* 92:511-518.
- Van Oostrum, W.H.A., W.R. Parker, and R. Kirby. 1980. Maintenance dredging in fluid mud areas. *Third Int. Symp. Dredging Technology*, Bordeaux, France. ISBN 0-906085-09-08.
- Walsby, A.E. 1977. The gas vacuoles of blue-green algae. *Scientific American* 237:90-97

- Water Investigations Branch of B.C. 1977. Observations on the water quality in the Chain - Link - Osprey lakes system. File 0290862-General.
- Wetzel, R.G. 1975. Limnology. W.B. Saunders Toronto. 743p.
- Williams, J.D.H., and A.E. Pashley. 1978. Lightweight corer designed for sampling very soft sediments. J. Fish. Res. Board Can. 36:241-246.
- Williams, J.D.H., H. Shear, and R.L. Thomas. 1980. Availability to Scenedesmus quadricauda of different forms of phosphorus in sedimentary materials from the Great Lakes. Limnol. Oceanogr. 25:1-12.
- Wisconsin Department of Natural Resources. 1974. Survey of lake rehabilitation techniques and experiences. Technical Report No. 75.
- Wisconsin Department of Natural Resources. 1981. Lilly lake protection and rehabilitation project Final report. U.S. Environmental Protection Agency Grant No. S005687.
- Worth, D.D.Jr. 1981. Nutting lake restoration project: A case study, p. 89-92. In Restoration of lakes and inland waters. EPA 440/5-81-010.
- Yoshida, T. 1982. On the summer peak of nutrient concentrations in lake water. Hydrobiologia 92:571-578.



THE CHAIN LINK OSPREY LAKES SYSTEM AND
THE LOCATIONS OF THE WATER SAMPLING STATIONS

Figure 1 Map of the Chain Lake area

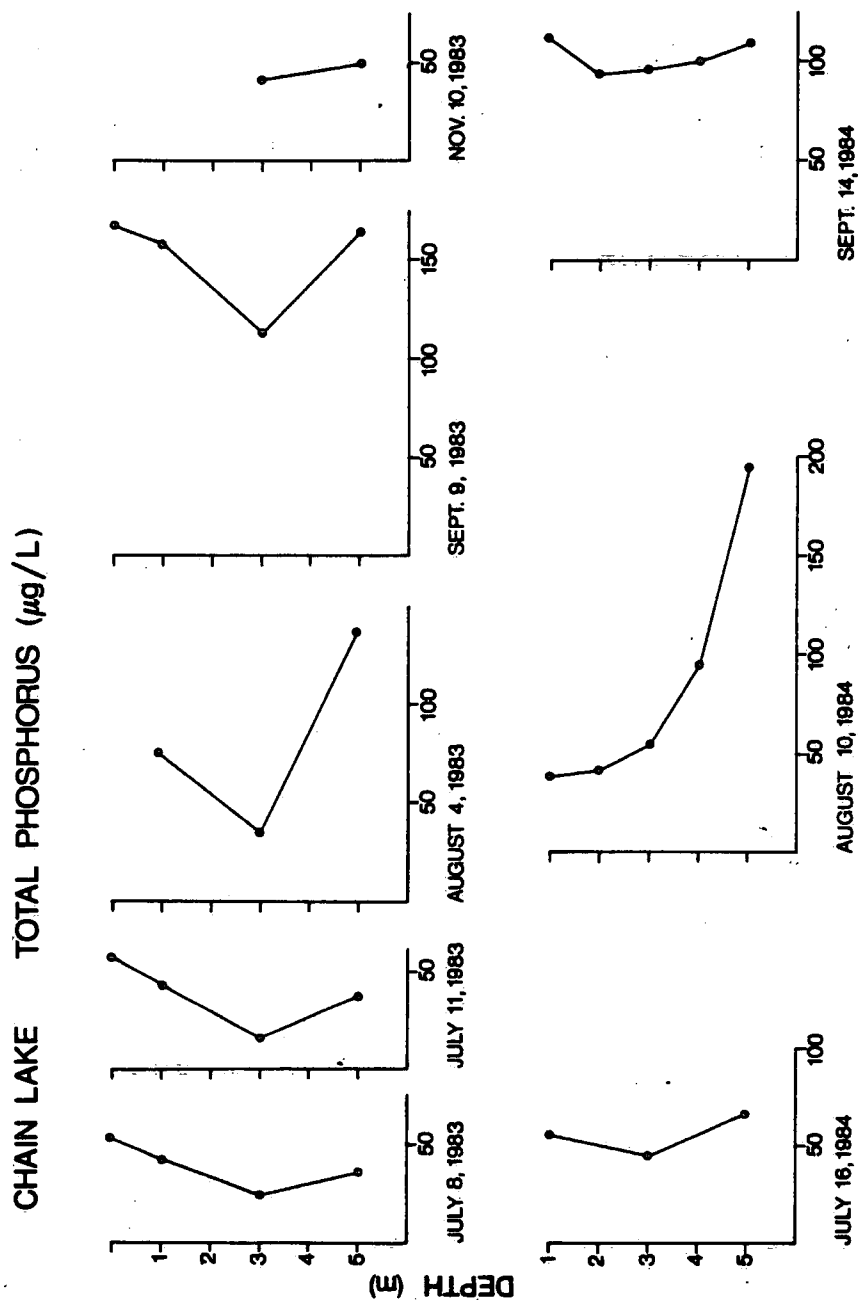


Figure 2 Total phosphorus concentration in Chain Lake

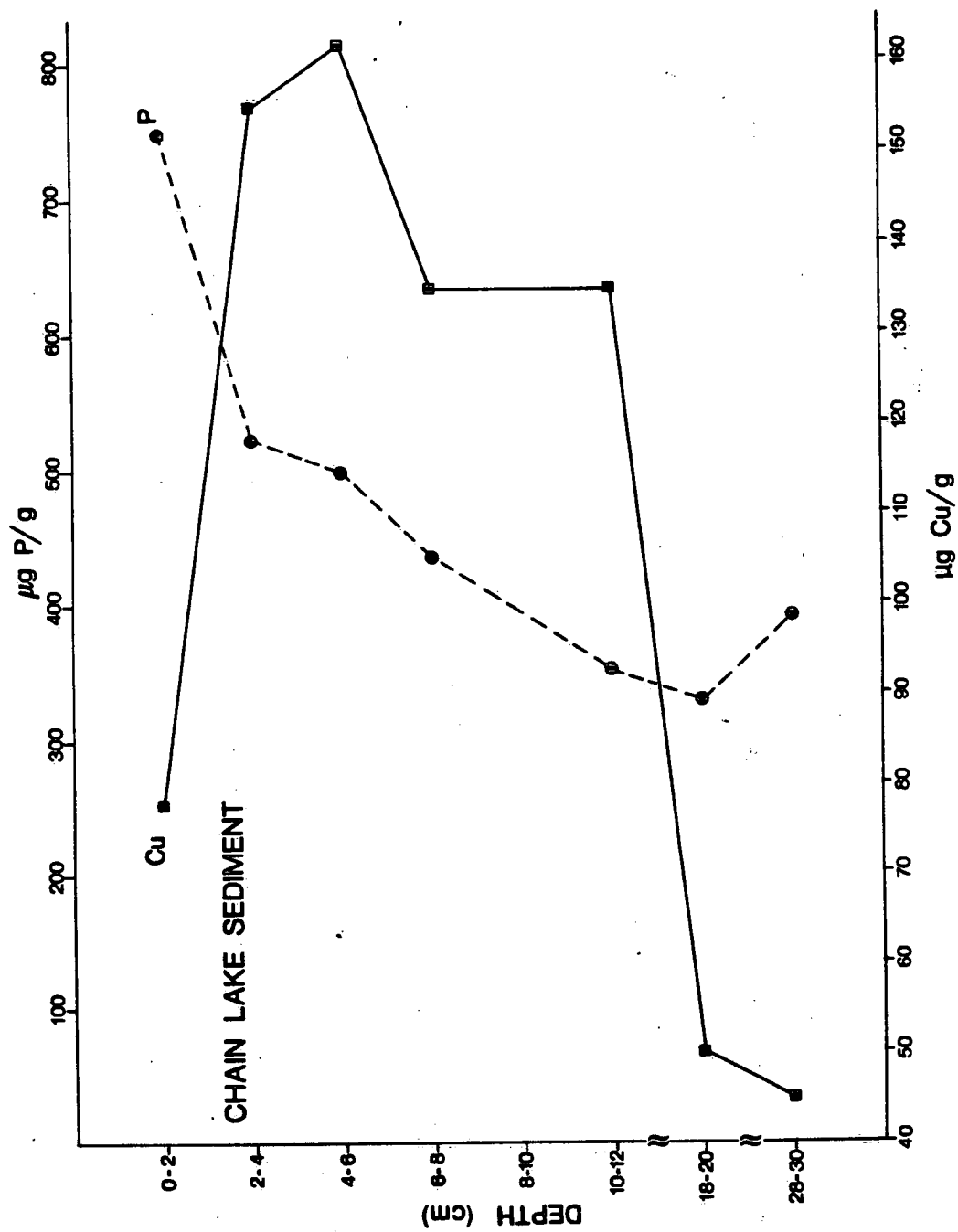


Figure 3 Bioavailable phosphorus and copper in Chain Lake sediments

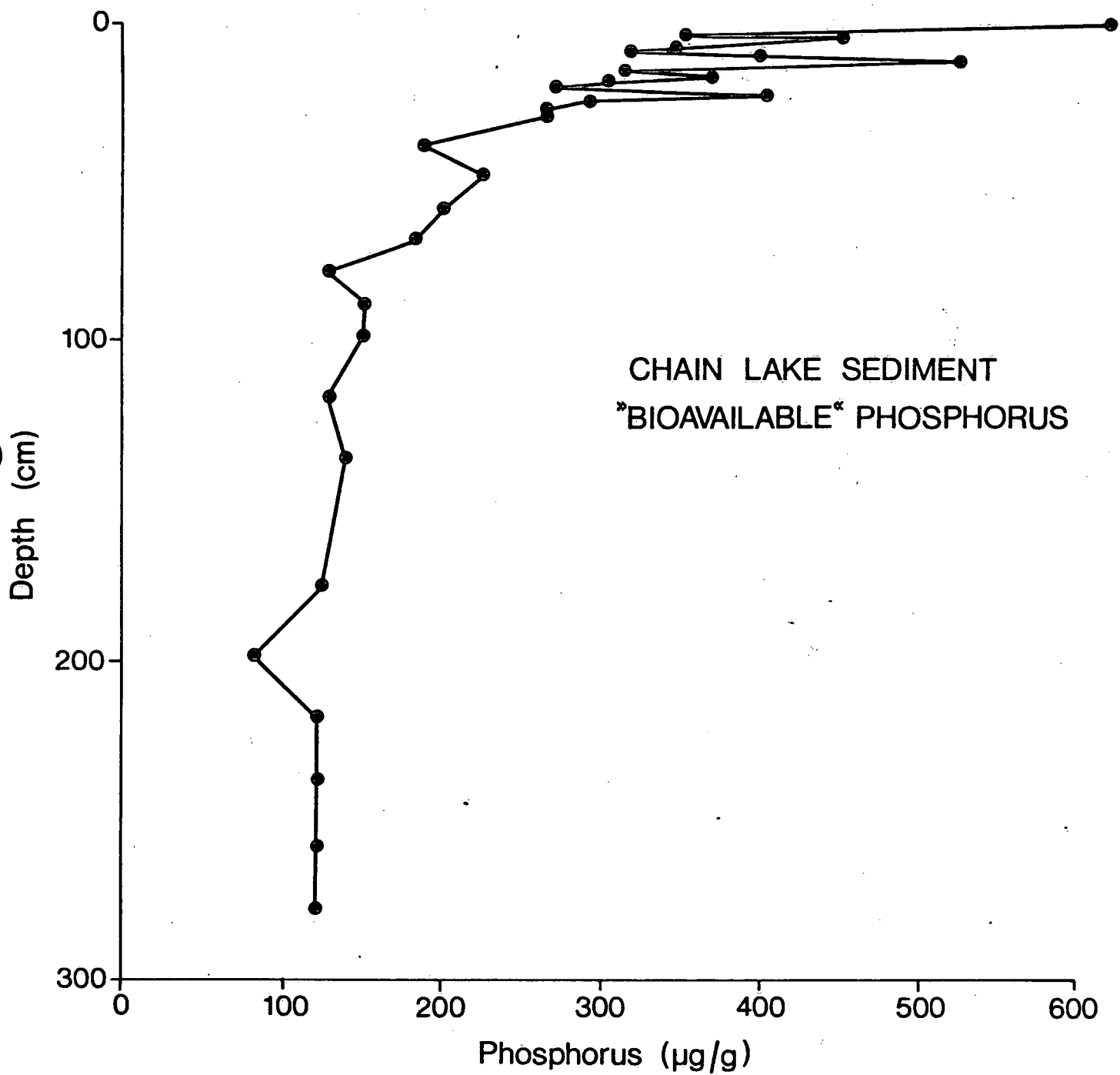
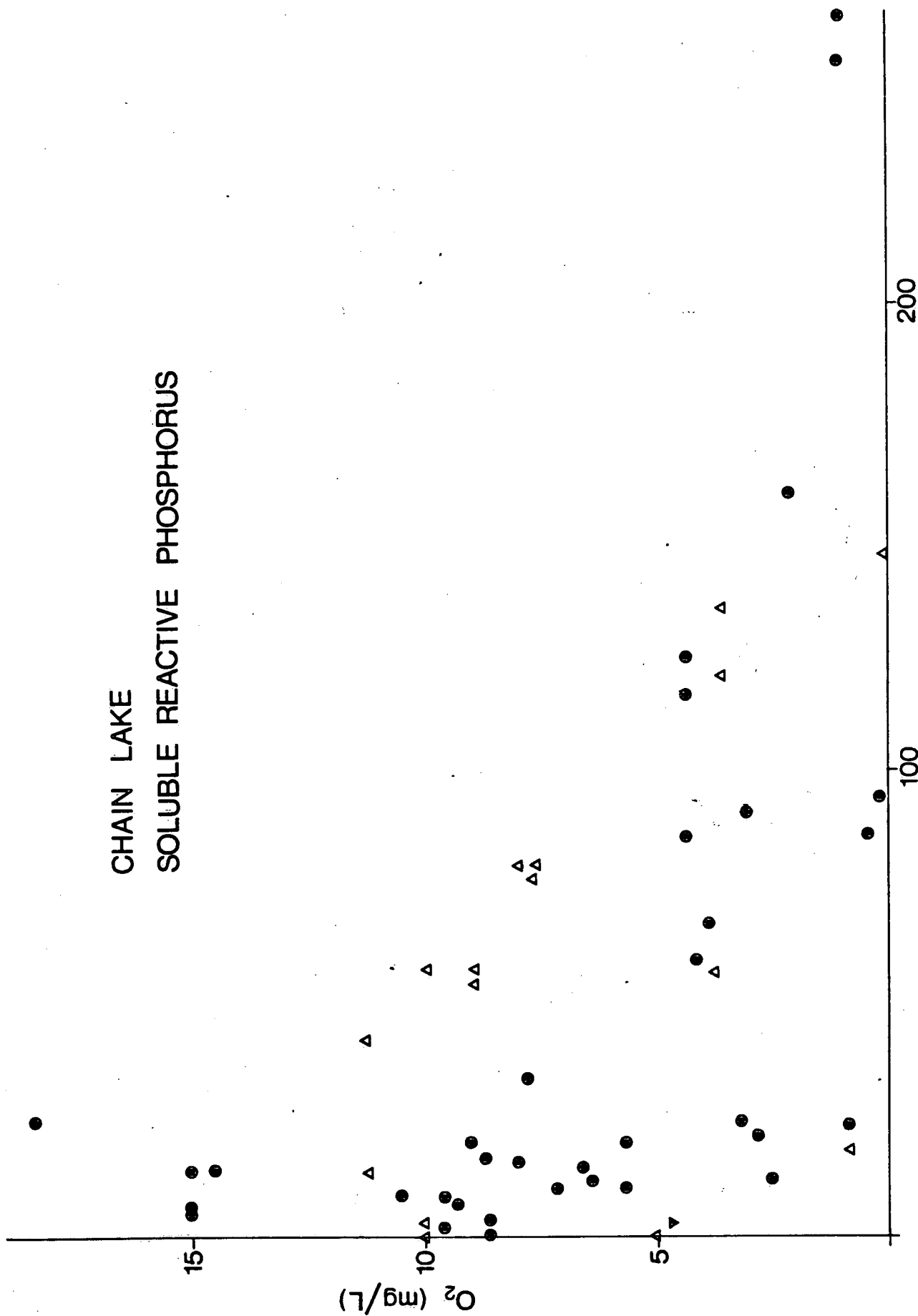


Figure 3 continued. Bioavailable phosphorus in Chain Lake sediments.

CHAIN LAKE SOLUBLE REACTIVE PHOSPHORUS



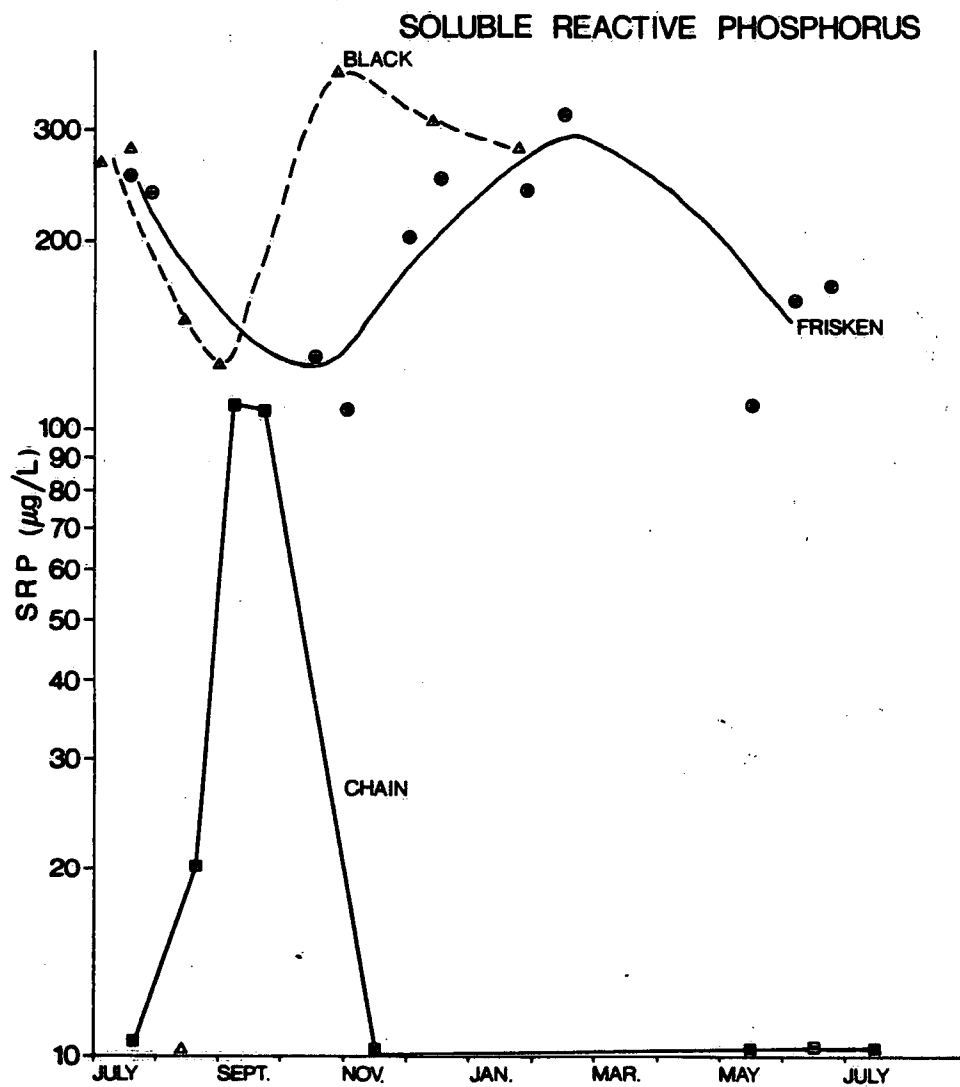


Figure 5

The soluble reactive phosphorus concentration (SRP) at 1.0 m from the surface of Frisken Lake (●) before lime treatment, Chain Lake (■), and Black Lake (▲). (Δ) is one sample from Black Lake (1979) collected during a calcium carbonate precipitation period.

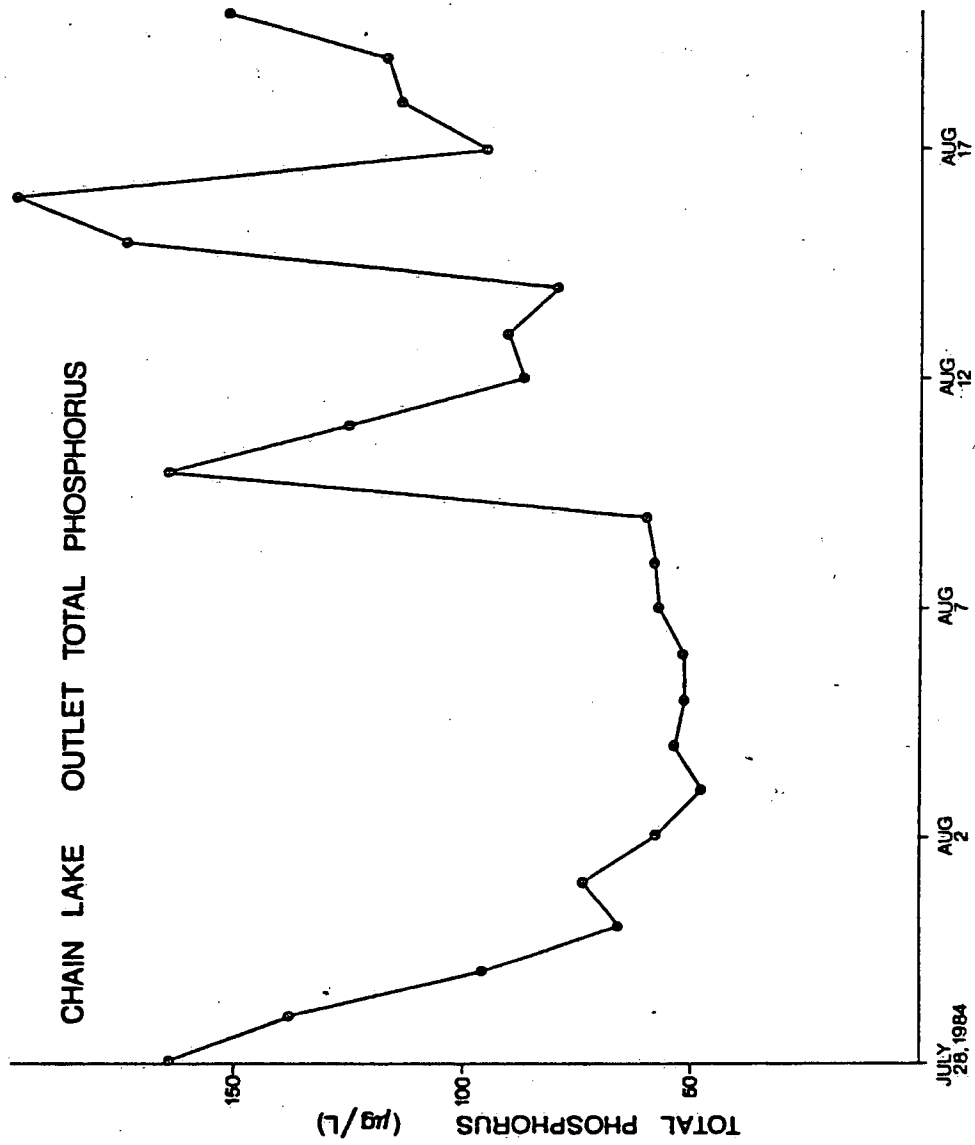


Figure 6 The total phosphorus concentration in the outlet of Chain Lake.

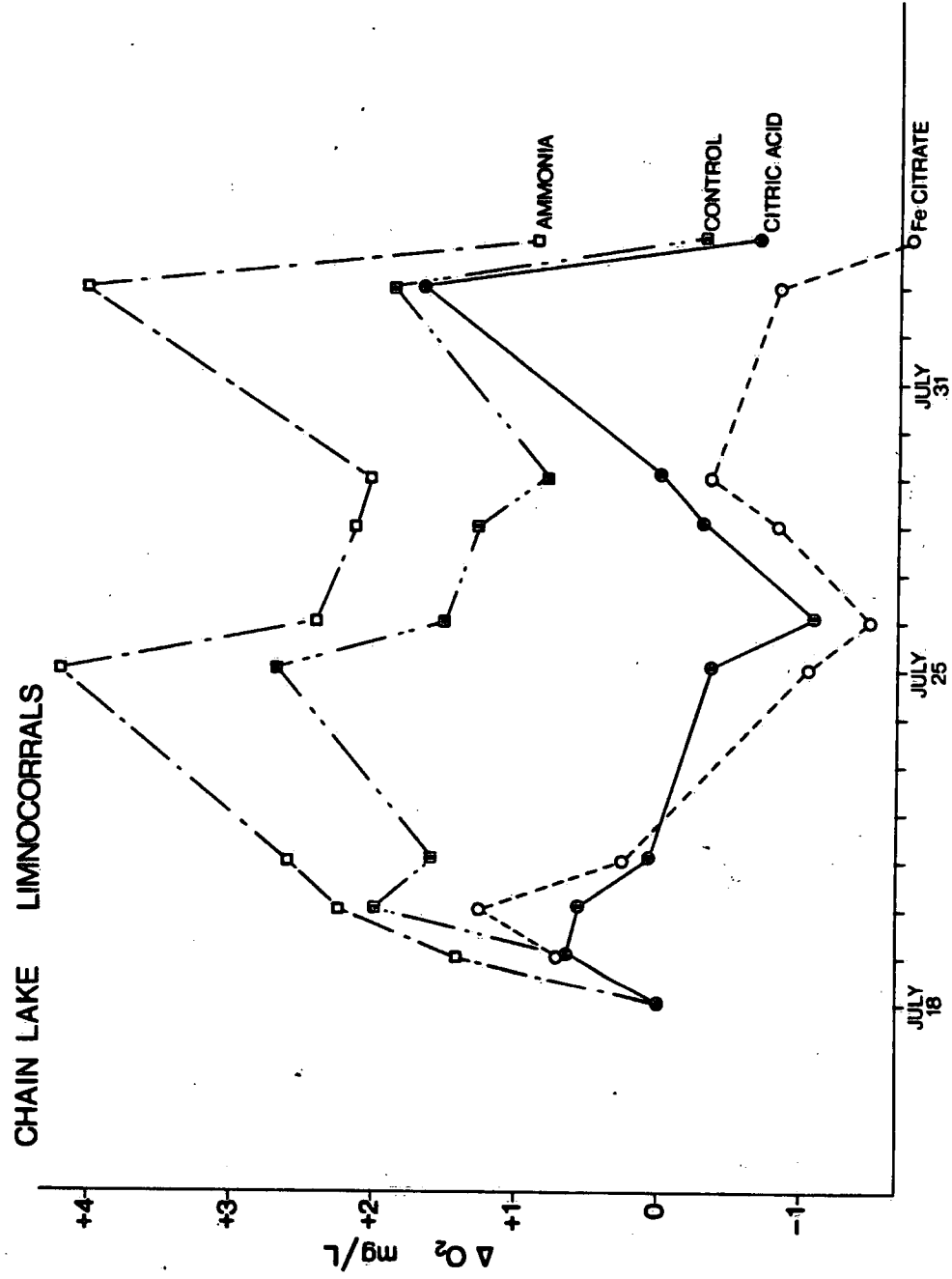


Figure 7 Changes in oxygen concentration in limnocorrals enriched with ammonia, citric acid or ferric citrate.

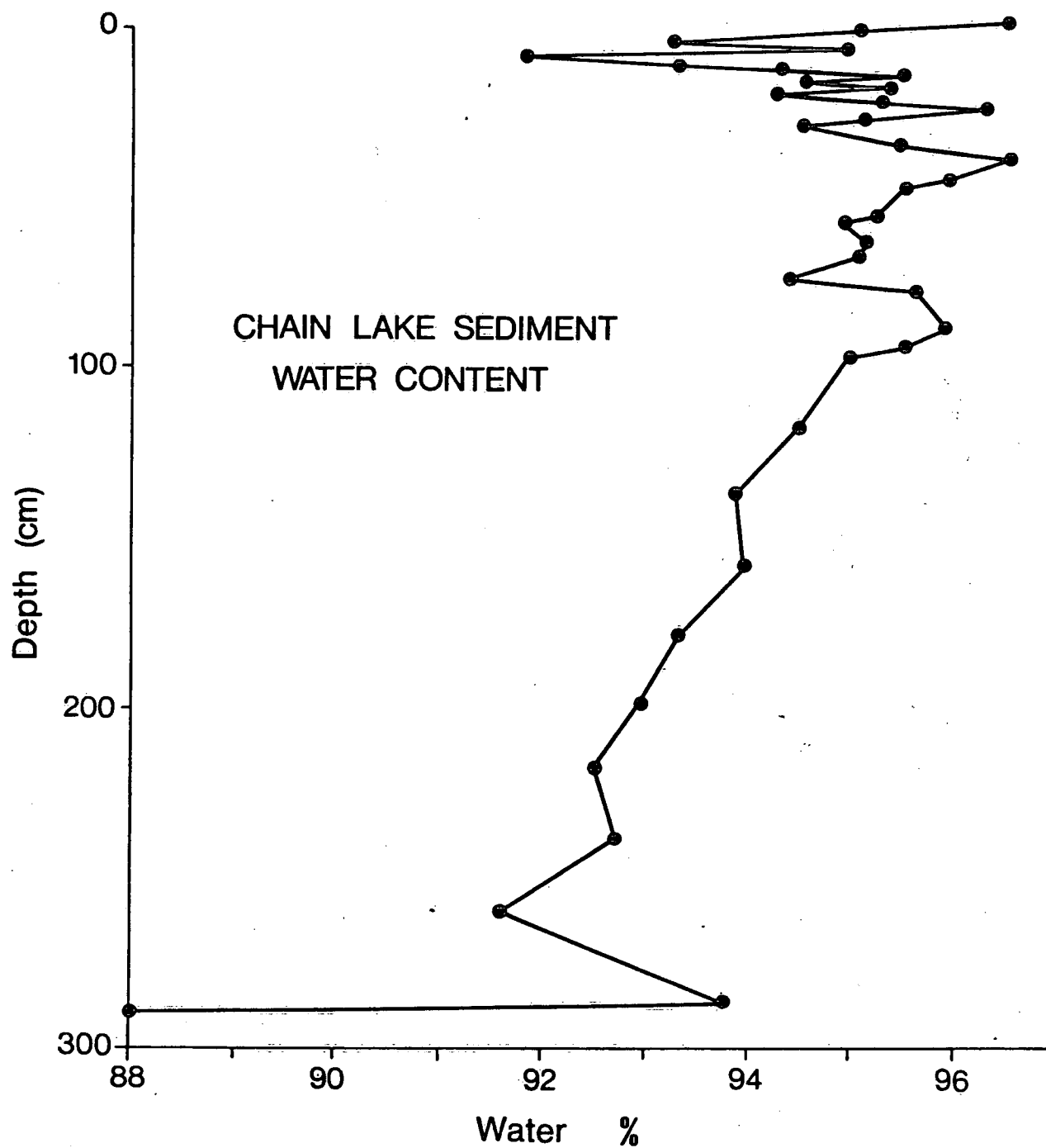


Figure 8 Water content of Chain Lake sediments.

TABLE 1: Hydrology - 1983

Date	Flow (L/sec)	TP ($\mu\text{g/L}$)	\bar{X} - Flow (L/sec)	$\bar{X}P$ - Flow (mg/sec)	H ₂ O - Loss ($\text{m}^3 \times 10^3/\text{month}$)	P Loss (Kg/month)
<u>Outflow</u>						
June 10	132	30	121	6.5	314	16.9
15	102	47				
23	130	83				
July 8	117	38	108	4.2	289	11.3
21	102	45				
26	111	33				
Aug 23	72	60	66	16.5	177	44.2
31	60	478 ⁺ 80				
Sept 19	487	92 ⁺ 16	487	44.8	1304	116
28	814*	108				
					<u>2084</u>	<u>188</u>
<u>Diversion</u>						
June 10	33.8	12	31	0.39	80	1.0
15	30.7	12				
23	28.5	14				
July 8	41.7	13	33	0.71	89	2.4
21	50.3	28				
26	7.8	24				
Aug 31	37	22	37	0.81	99	2.8
Sept 19	91.5	18	91.5	1.65	237	4.3
					<u>505</u>	<u>10.5</u>
<u>Natural Inlet</u>						
June 10	46	19	30	0.67	77	1.8
15	21	24 ⁺ 6				
23	23	28				
July 8	19	21	24	0.89	65	2.4
21	24	52				
26	30	34				
Aug 23	no flow					
31	no flow					
Sept 1	no flow					
19	no flow					
21	no flow					
					<u>142</u>	<u>4.2</u>

* secondary outlet was opened. This discharge was not included in the water budget.

P - Phosphorus

\bar{X} - Arithmetic mean

TP - Total phosphorus

TABLE 2 Chain Lake Total Phosphorus #

Date	Surface	1.0	2.0	Depth (m)	4.0	5.0	Diversion	Inlet	Outlet
Jun11/83							12	19	30
Jun15/83							12	25	47
Jun23/83							14	28	83
Jul 8/83	56	42		25		40	13	20	38
Jul11/83	55	45		17		37			
Jul15/83									
Jul21/83		31		17		43	28	52	26
Jul26/83							24	34	45
Aug 4/83		75		35		135			33
Aug31/83									
Sep 9/83	177	162		112		167			450
Sep20/83									126
Nov10/83				43		49			88
Jul16/84	355	56		44		65			
Aug 6/84		57		38		380			
Aug10/84		38	42	53	95	195			164
Sep14/84		112	91	95	98	112			

(µg P/L)

TABLE 2 (Continued) Chain Lake soluble reactive phosphorus #

Date	Surf	Depth				Div	Inlet	Outlet
		1.0	2.0	3.0	5.0			
Jun10/83						8	10	18
Jun15/83						8	18	8
Jun22/83					20		18	20
Jul 8/83		18		20	20	8	20	14
Jul15/83		90		14			19	
Jul21/83		14		7	17	5	4	12
Jul26/83		9				8	25	4
Aug 4/83		14		16	86			
Aug15/83		10		12	94			
Sep 9/83		124		86	116			190
Jul 5/84		12		10	28		20	
Jul17/84	10	26		20	24			
Aug 6/84		0		8	281			
Aug10/84		3	8	34	202			
				67				

(µg/L)

TABLE 3 : Major ions in the Chain Lake area - 1983

Site	Site #	Major ions mg/L						TIC	DON	DOC
		Ca	Mg	K	Na	Cl	SO ₄			
Lee Creek*	1	22	5	2.1	4.2	0.6	0.5	17	0.138	4.9
Secret Lake Outlet*	5	15	2.8					10		
Stream S.E. Link*	2	9	2.0	1.7	3.6	0.4	0.5	7	0.176	8.2
Link Lake July 15	3	16.7	3.3	2.2	4.7	1.0	0.5	19.5		
Chain Lake Inlet	6	23.6	5.6	2.3	5.7	1.4	0.5	22.	0.231	9.4
Chain Lake Outlet	14	17.8	3.9	1.9	5.1	1.4	0.5	20.2	0.274	9.4
Chain Lake June 15	13	20.7	4.1					21.9		
June 26	13	17.7	4.3					20.7		
July 11	13	18.0	4.0	2.1	5.1	1.4	0.5	19.8	0.229	9.9
Aug. 3	13	17.7	3.9	1.9	5.1	1.4	0.5	19.3	0.340	8.3
Diversion	15	5.1	0.6	0.4	2.2	0.4	1.5	3	0.106	8.2

* These are derived from 1 sample. All other data are mean values of at least 3 samples

Fig. 1

Table 4 Chemistry of Springs

Chain lake	Ca	Mg	Na	K	SRP	TIC	NH ₃	NO ₃	Cl	SO ₄
Sites- #										
G. Smith-12	37.2	6.3	5.6	2.5	4	10				
Suggets-7	23.5	4.3			66	24	4	3	1.0	0.5
Suggets Jr	54.7	8.8	7.8	1.9	8	28	158	9	1.4	0.5
Wellburn-8	41	9.8	6.9	4.0	37	26	6	2	0.9	0.5
Banks-11	17	4.0	10.2	2.6	54	14	2	3	1.4	3.6
Maurer-10	25	5.1	11.3	2.6	28	20	2	4	0.8	7.9
D. Smith-9										

Soluble reactive phosphorus (SRP) dissolved organic nitrogen (DON), ammonia (NH₃), and nitrate (NO₃) are in units of $\mu\text{g L}^{-1}$. All other data are in units of mg L^{-1} . The pH of the springs was 6.9.

Refers to numbers on Fig.1

TABLE 5: Iron concentrations in Chain Lake - 1984

Depth (m)	Iron Concentration ($\mu\text{g/L}$)			
	July 6	July 16	Aug 10	Sept 14
1	426	377	236	252
2			322	633
3	264	379	812	1525
4			784	777
5	385	654	1967	741
Inlet	893	219		

TABLE 6: Sediment Incubation

Sample	Concentration P ($\mu\text{g P/L}$)				
	Day 1	Day 2	Day 6	Day 13	Day 19
Control	51	32	12	36	56
5.6mg C/L	13	29	9	38	75
22.4mg C/L	26	24	12	13	91

Carbon source used was acetate

Table 7

CHAIN LAKE OXYGEN DATA - 1983

DEPTH	JUN22/83	JUL14/83	JUL18/83	JUL20/83	JUL21/83	JUL22/83	JUL25/83	JUL26/83	JUL28/83
Surf	7.7	9.2	11.6	14.4	14.2	16.1	16.8	12.2	14.8
1	7.6	8.9	11.5	14.5	12.4	16.4	16.8	12.4	14.8
2	7.5	9.0	11.3	12.3	10.2	16.0	15.8	12.4	13.9
3	7.4	9.0	10.8	9.0	8.7	14.5	8.3	11.8	13.7
4	7.1	8.6	9.1	8.1	5.1	10.1	5.6	9.6	11.1
5		5.3	7.9	5.3	2.5	4.0	1.1	1.7	6.3
5.5				1.5	1.5	0.6	0.5	0.7	1.2
time	14.00	15.00		16.00		10.00	11.00	15.00	

CHAIN LAKE OXYGEN DATA - 1983 (Continued)

DEPTH	JUL29/83	AUG2/83	AUG3/83	AUG3/83	AUG15/83	AUG23/83	AUG31/83	SEP18/83	NOV10/83
Surf	16.8	16.1	12.2	15.0	5.9	8.1	6.0	4.7	9.4
1	16.5	16.1	12.2	15.4	5.7	7.6	5.9	4.4	9.2
2	13.8	13.2	12.0	12.8	4.5	7.6	5.5	4.4	9.1
3	12.0	8.5	4.5	8.0	2.5	4.8	5.5	4.4	9.0
4	8.2	7.5	2.1	6.4	0.2	1.2	5.0	4.4	9.0
5	1.8	2.4	0.2	0.4	0.1	0.3	4.4	4.4	9.0
5.5	0.4	0.4	0.2		0.0	0.2	4.0	4.4	
time	16.00		9.00	16.20	13.30	16.00	16.00	15.00	13.00

CHAIN LAKE OXYGEN DATA - 1984

DEPTH	JUN 9/84	JUL5/84	JUL17/84	AUG10/84
Surf	8.8	9.5	20.0	8.2
1	8.6	9.5	18.5	9.6
2	8.6	9.4	15.4	9.6
3	8.6	9.4	9.0	7.8
4	7.6	8.4	6.4	3.9
5	6.4	6.4	3.2	1.0
time	10.00	11.00	10.00	9.00

Table 7 continued. CHAIN LAKE TEMPERATURE DATA - 1983

DEPTH	JUN22/83	JUL14/83	JUL18/83	JUL20/83	JUL21/83	JUL22/83	JUL25/83	JUL26/83	JUL28/83
Surf	14.8	15.2	16.0	18.5	18.0	20.5	17.2	16.0	17.0
1	14.8	15.2	16.0	17.5	16.0	18.5	17.2	16.0	16.3
2	14.8	15.2	16.0	16.8	15.5	17.1	17.0	16.0	16.3
3	14.8	15.2	15.8	15.5	15.2	16.8	15.9	16.0	16.0
4	14.5	15.2	15.1	15.0	14.9	15.5	15.1	15.5	16.0
5		14.8	15.0	15.0	14.8	15.0	14.5	14.9	15.3
5.5					14.8	14.8	14.5	14.8	14.8

CHAIN LAKE TEMPERATURE DATA-1983 (CONTINUED)

DEPTH	JUL29/83	AUG2/83	AUG3/83	AUG3/83	AUG15/83	AUG23/83	AUG31/83	SEP18/83	NOV10/83
Surf	20.0	19.5	19.0	20.0	20.0	20.8		13.7	3.5
1	16.8	19.0	18.8	19.4	20.0	20.0		13.7	3.4
2	16.5	18.2	18.2	18.0	20.0	19.0		13.7	3.3
3	16.0	17.0	16.2	17.0	19.0	18.5		13.7	3.2
4	15.5	16.7	15.5	15.5	18.0	18.2		13.7	3.2
5	15.0	15.6	15.0	15.1	16.5	17.9		13.7	3.2
5.5	14.8	15.0	14.9		16.5	17.0			

CHAIN LAKE TEMPERATURE DATA-1984

DEPTH	JUN 9/84	JUL5/84	JUL17/84	AUG10/84
Surf	13.5	15.0	21.0	19.9
1	13.5	15.0	18.1	19.8
2	13.0	15.0	17.0	19.7
3	12.0	15.0	15.5	18.8
4	11.5	15.0	15.2	18.0
5	10.9	14.5	15.0	16.3

Table 8 Chain Lake Chlorophyll a[#]

Date	Surface	Depth (m)					Outlet
		1.0	2.0	3.0	4.0	5.0	
June 22/83		2.0		1.6		1.9	2.0
July 2/83	4.0	22.0		5.0		2.0	
July 8/83		40.0		20.0		2.0	5.0
July 20/83		127.0		19.0		2.0	22.0
July 26/83							20.0
Aug 3/83		99.0		30.0		21.0	20.0
Aug 15/83		2.0		2.0		11.0	
Sept 19/83		54.0		41.0		22.0	6.1
May 30/84		7.2		7.8		13.3	
June 16/84		1.6		1.6		1.3	
July 5/84		16.0		13.5		4.0	
July 17/84	170.0	26.0		9.6		5.2	
Aug 10/84		11.1	5.6	0.3	1.4	2.0	

[#] (µg/L)

Table 9 Phosphorus Budget for Chain Lake

Summer of 1983		Annual
<u>Inflow</u>		
Diversiion	10.5	10.5*
Hayes Creek	4.2	278.**
Springs [#]	70.9	319.***
Internal	406	406
Loading ^{##}		
Sediment ^{###}	1966	
<u>Outflow</u>	188	438.****

All values are kilograms of phosphorus.

Springs apparently contributed 64.2% of the flow. Mean SRP of the springs was 30 µg/L.

Internal loading was calculated from the increase in phosphorus in the lake plus the outflow.

The pool of phosphorus available for exchange with the surface water was calculated from the difference in the bioavailable phosphorus in the surface 2 cm from the amount of bioavailable phosphorus in the sediment at 2-4 cm.

* The diversion only flows for the summer period.

** We assumed that the rest of the inflow had the same mean phosphorus as summer values (29.7 µg/L) and we used the flow from WIB (1977).

*** We assumed that groundwater had the same flow yearly as it had in the summer.

**** We assumed that the outflow had the same phosphorus concentration as the lake had prior to the algal blooms and we used the WIB (1977) hydrology.