IN-LAKE PRECIPITATION OF PHOSPHORUS BY LIME
TREATMENT.

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BY LIME TREATMENT

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<u>Recherche de la DEA sur la remise en état des lacs</u>

PRÉCIPITATION DU PHOSPHORE PAR LA CHAUX DANS LES LACS

T.P. Murphy, K.J. Hall, K.T. Ashley, A. Mudroch, M. Mawhinney et Hj. Fricker

Perspective-gestion

On a évalué la précipitation du phosphore par la chaux dans le lac Frisken pour déterminer si elle pouvait être une technique de remise en état des lacs. La zone expérimentale se situe dans la région intérieure semi-aride de la Colombie-Britannique. Le lac reçoit une forte charge de phosphore par la météorisation naturelle de l'apatite. Avant le traitement à la chaux, moins de la moitié du phosphore dans la couche épilimnique a été précipité par le carbonate de calcium.

En 1983, l'addition de 23 tonnes de $Ca(OH)_2$ a éliminé 80 p. 100 de la chlorophylle <u>a</u> et 97 p. 100 du phosphore réactif soluble (PRS) présents dans l'épilimnion. En 1984, l'addition de 16 tonnes de chaux a enlevé plus de 90 p. 100 de la chlorophylle <u>a</u> et 60 p. 100 du PRS présents dans l'épilimnion. Presque tout le phosphore précipité s'est dissout dans l'hypolimnion. Les divers changements chimiques enregistrés dans les eaux des lacs permettent d'avancer que le traitement à la chaux pourrait donner lieu à plus qu'une simple amélioration saisonnière de la qualité de l'eau.

RÉSUMÉ

Pour renverser le processus d'eutrophisation des lacs à alcalinité
élevée, on a évalué si l'addition de chaux pouvait être une technique
de remise en état des lacs. En 1983, l'addition de 23 tonnes de $Ca(OH)_2$
a enlevé 80 p. 100 de la chlorophylle a et 97 p. 100 du phosphore réactif
soluble (PRS) présents dans l'épilimnion. En 1984, l'addition de
16 tonnes de chaux a enlevé plus de 90 p. 100 de la chlorophylle a et
60 p. 100 du PRS présents dans l'épilimnion. Presque tout le phosphore
précipité s'est dissout dans l'hypolimnion.

Le traitement à la chaux peut favoriser davantage la précipitation
du carbonate de calcium et, à l'avenir, on devrait pouvoir ajouter de
moins en moins de chaux. Le titrage du pH de 1984 a été beaucoup plus
efficace que celui de 1983 (14 comparativement à 8 p. 100). En augmentant
la concentration de carbone inorganique, on devrait favoriser la
précipitation du carbonate de calcium.

On n'a pas observé de réactions du phosphore au contact du fer.
En grande partie, le fer ne semble pas être réactif. Ce dernier
pourrait être un oligo-élément qui empêche certaines réactions.

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Management Perspective

Lime induced precipitation of phosphorus in Frisken Lake was evaluated as a lake restoration technique. The experimental site is in the semi-arid interior of British Columbia. The lake receives a high phosphorus loading from the natural weathering of apatite. Before the lime treatment, less than half of the epilimnetic phosphorus precipitated with calcium carbonate.

In 1983, the addition of 23 tonnes of $Ca(OH)_2$ removed 80% of the chlorophyll a and 97% of the soluble reactive phosphorus (SRP) from the epilimnion. In 1984, the addition of 16 tonnes of lime removed more than 90% of the chlorophyll a and 60% of the SRP from the epilimnion. Most of the precipitated phosphorus dissolved in the hypolimnion. Several changes in lake chemistry indicated that lime treatment would produce more than a seasonal improvement in water quality.

ABSTRACT

To ameliorate eutrophication of hardwater lakes, lime addition was evaluated as a lake restoration technique. In 1983, the addition of 23 tonnes of $Ca(OH)_2$ removed 80% of the chlorophyll $\underline{a}$ and 97% of the soluble reactive phosphorus (SRP) from the epilimnion. In 1984, the addition of 16 tonnes of lime removed more than 90% of the chlorophyll $\underline{a}$ and 60% of the SRP of the epilimnion. Most of the precipitated phosphorus dissolved in the hypolimnion.

Lime treatment can enhance future calcium carbonate precipitation and future lime additions should require less lime. The efficiency of the 1984 pH titration was much higher than the 1983 titration (14 vs. 8%). The increase in the concentration of inorganic carbon should result in enhanced precipitation of calcium carbonate.

Phosphorus reactions with iron were not observed. Much of the iron appeared to be unreactive. Iron could be a limiting micronutrient

ACKNOWLEDGEMENTS

<u>Legend</u>

INTRODUCTION

Many studies of calcium carbonate precipitation have led to the hypothesis that an induction of calcium carbonate precipitation can reduce eutrophication. Calcium carbonate precipitation often occurs naturally in hardwater lakes during summer algal blooms (Kelts and Hsu 1978). This precipitation can significantly reduce the productivity of hardwater lakes (Wetzel 1975). Recent studies have reported the coprecipitation of phosphorus with calcium carbonate (Rossknecht 1980, Avnimelach 1983, Murphy et al. 1983a). A Polish group that used lime to reduce the humic content of a lake, observed a reduction of the phosphorus concentration (Hillbricht-Ilkowska et al. 1977). These carbonate precipitation reactions are also apparent in some lakes on the Thompson Plateau. Naturally eutrophic Wood Lake flows into oligotrophic Kalamalka Lake, a marl lake (Clemens et al. 1939). Kalamalka Lake receives a major groundwater flow of $CaSO_4$ rich water that must enhance calcium carbonate precipitation in Kalamalka Lake (Liebscher, Geological Survey of Canada, Vancouver).

In this study, lime was added to Frisken Lake, British Columbia, to induce the precipitation of calcium carbonate and phosphorus. This in-lake treatment may have particular utility in areas where diffuse nutrient loading cannot be controlled, where tertiary treatment of wastes is too expensive, and where treatment with toxins is not appropriate.

Frisken Lake, is primarily used for recreational fishing. Rainbow trout

grow very quickly, and their quality is excellent. However, intense algal blooms interfere with fishing and the fish frequently winterkill. The objectives of this study were to suppress the summer blue-green algal blooms, and to indirectly increase the lake oxygen concentration in winter. Several researchers have shown that the lake winter oxygen concentration is closely correlated to the lake summer chlorophyll concentration (Babin and Prepas 1985).

Although the first local concern was to enhance trout production, this research could have widespead use in suppressing eutrophication in hardwater lakes. Much of Western Canada is semi-arid and many conflicts exist over the use of the limited water supply. Many of the towns and cities in the interior of British Columbia have a shortage of good drinking water. Taste and odour problems are common in some of the reservoirs such as the Garnet Valley reservoir which supplies Summerland (McKean and Lim 1981) and Dragon Lake which supplies part of Quesnel (McKean 1981). A new lake restoration technique must be better than the current method, copper sulphate treatment.

Copper sulphate treatments have been used with limited success in restricting excessive algal growth (C. Bull, Fisheries Branch, Penticton, B.C.). The ephemeral response of copper sulphate treatments requires a large commitment of staff to lake monitoring (S. Withler, Fisheries Biologist, Kamloops, B.C.). Fisheries biologists in both British Columbia and Alberta are concerned about the potential adverse effects of copper sulphate treatment on fish. In Saskatchewan, lake managers often avoid conflicts

associated with dead fish by not stocking fish in lakes that receive repeated copper sulphate treatment. In much of western Alberta and the interior of British Columbia, lakes are highly valued for both sources of drinking water and fish. Similar concerns about copper sulphate treatment have been documented in the United States. Gentile and Mahoney (1969) reported fish kills in copper treatments. Taub and Kindig (1984) have reported that the killing of zooplankton by copper can result in algal blooms shortly after a copper treatment : the algae recover faster than the zooplankton. Moreover, copper treatments do not always reduce algal growth (Elder and Horne 1978). New management tools are needed.

1. Study Site

Frisken Lake is located upon the Thompson Plateau in south central British Columbia (Fig. 1), 30 km S.S.E. from the junction of the North and South Thompson Rivers (Kamloops, British Columbia; latitude 50° 27' 30", longitude 120° 8' 30"; Fig. 2). The plateau has a temperate continental climate that is moderated by the Pacific Ocean and protected from Arctic weather by the Rocky Mountains to the east. The plateau is in the rainshadow of the Cascade Mountains which borders the plateau on the west. Rainfall varies from 75 cm on the highlands to less than 35 cm in the semi-arid valleys (MacMillan-Bloedel 1972). More rainfall and less evaporation at higher elevation results in runoff that supplies much of the water requirements of the valleys. Frisken Lake has an area of 29 hectares, a maximum depth of 10 m, a mean depth of 4.3 m, and an elevation of 1138 m.

1.1 Geology

The geology of the Kamloops area is very complex. Two periods of marine volcanic arc formation with both shallow and deep marine sediment formation have been documented (Smith 1974). Microplates from the far east were also added to the local geology (Jones et al. 1982). Frisken Lake is situated upon a poorly defined volcanic intrusion called the Wildhorse Mountain Batholith. It formed during the Jurassic or the post-lower Cretaceous period (Cockfield 1947, 1948). This volcanic intrusion is classified geologically with the more intensively studied Iron Mask Batholith. Apatite is found in most of the rocks of the Iron Mask Batholith (Cockfield 1948) and veins of apatite are also found there (Cockfield 1947). These two batholiths are as close as three km on the surface. Since the phosphorus concentration of many lakes near Frisken Lake is unusually high (Fish and Wildlife Branch, unpublished data), and no significant anthropomorphic phosphorus sources exist, apatite deposits are likely present near Frisken Lake as well.

1.1.1 Relationship of Geological and Water Quality Variability

The random distribution of veins of apatite could be the main cause for the high variability in water quality that is found in the Frisken Lake area (Fish and Wildlife Branch, unpublished data). The three following geological processes could contribute to this spatial variability in water quality.

1) Later volcanic activity in the Miocene period formed Mount Bulman. This mountain is located south of Frisken Lake and within the watershed of

Roche Lake. In 1982, the spring runoff from the Mount Bulman area was much larger than that of the neighbouring streams. Mount Bulman is a 1000 m higher than Roche Lake and presumably the snowpack is substantial enough to result in a large runoff of low phosphate water into Roche Lake. The water flow of April 28, 1982 into John Franklin Lake (above Roche Lake) was at least 20 times the flow from Frisken Lake which was about 25 liters a second.

Volcanic ash deposits may have formed localized groundwater aquifers. Nesbitt (1974) has observed this type of aquifer in the Basque Lake area which is relatively near Frisken Lake.

2) Evaporite deposits could also effect water quality. Large deposits of sodium sulphate and calcium sulphate are found in the Iron Mask Batholith (Cockfield 1947). If these localized deposits were formed in old marine environments then similar but probably smaller deposits could be common on many of these old coastal batholiths. If these evaporite deposits formed recently, they would probably be found mainly in the valleys. Some of the large deposits around Beresford are found at intermediate elevation on the plateau.

3) The porosity and depth of glacial debris should influence the ratio of surface to groundwater flow. Most of the surface of the Wildhorse Mountain Batholith is covered with glacial debris. Northcote (1968) estimated that 97% of the Guichon Creek Batholith (another neighboring Jurassic coastal batholith) was covered by glacial debris. If the glacial debris was deep, the

buried apatite deposits should have much less of an effect upon surface water quality relative to a similar rock covered with less glacial debris. This glacial deposit is quite porous; thus, groundwater flow could be quite substantial. Some aquifers could be quite rich in phosphorus.

1.2 Water Residence

Although the hydraulic data for Frisken Lake is limited, one conclusion is obvious : treated lakewater would not be rapidly flushed. By assuming that the observed flow on April 28, 1982, of 25 liters a second into Frisken Lake lasted for two months, a water residence time of 84 years was calculated. The flow observed May 30, 1984 was 150 liters a second after a late snow melt and heavy rains. A similar calculation would produce a residence time of 24 years. These flow measurements were made in years of record rainfall. The surface flow into Frisken Lake normally only lasts for a short period in the spring (Fish and Wildlife unpublished data). Although groundwater flows could alter these estimates, a good potential for observing long term water quality changes exists.

METHODS

2.1 Chemical Treatments of Frisken Lake

Lime was added to Frisken Lake to induce precipitation of calcium carbonate. To distribute lime around the lake a slurry maker was mounted onto a barge. The barge was a 2.4 m by 6.1 m plywood deck fastened to two pontoons that was powered by a 10 hp outboard motor. The barge could carry 20, 20 kg bags of lime, two lime handlers, a driver, and the slurry maker. A large water pump (7.6 cm outlet port) directed jets of water into the base of the hopper of the slurry maker. The bags of lime were lifted to the top of the hopper where they were cut open. The lime fell into the turbulent water which then flowed over baffles within an enclosed 1.2 m long plywood mixing chamber before falling into the lake at the front of the barge. The slurry was applied evenly to all parts of the lake deeper than 3.0 m. To minimize corrosion and facilitate cleaning, all metal (especially aluminum) was protected with Interlux Viny-lux primewash, Vitaline aluminum primer, and an enamel finish.

For personal protection rubber gloves, raincoats, goggles, and dust masks with particle filters (Safeco model 1200) were used in 1983. The interface of the gloves and the raincoat were sealed with tape, but this precaution was not satisfactory unless long gloves were worn. Protective cream helped. The goggles provided insufficient protection. In 1984, dust hoods equipped with dust masks (Safety Supply Canada TD 1882) gave excellent

protection. A defogger solution kept the shield clear of condensation.

The lime was delivered by a truck that could load and unload 7.2 tonnes of lime with a small crane. In this way lime handling was minimized considerably and the delivery charge was less than gas would have cost for equivalent transportation in a smaller truck. Unfortunately, the unusually intense rains of 1983 made the road impassable. In 1983, lime was added to the lake during three relatively dry periods: June 16, 1.2 tonnes; June 17, 7.2 tonnes; July 26, 4.8 tonnes; July 27, 2.4 tonnes; August 16, 4.8 tonnes; and August 17, 2.4 tonnes (total 22.8 tonnes). In 1984, 16.0 tonnes of lime was added to Frisken Lake in one period: May 26, 2 tonnes; May 27, 10 tonnes; May 28, 4 tonnes. The lake received a total of 38.8 tonnes of $Ca(OH)_2$ over the two years.

2.2 Sample Collection

In 1983, water samples were collected with a three liter PVC Van Dorn bottle from two stations in Frisken Lake which were 100 m apart and both within the deepest part of the lake (9.8 m April 1982, 8.8 m Sept. 20, 1983). After the first lime addition, samples were collected at 1.0, 3.0, 5.0, and 8.0 m from the lake surface. Later lime additions were sampled at 1 m intervals from the lake surface to a depth of 8 m. In 1984 samples were collected from one station with a bilge pump.

Zooplankton samples were collected from 1 meter intervals with a

Schindler trap in 1983 and a bilge pump in 1984. All zooplankton samples were collected from one site. Samples were preserved with a sugar-formalin solution (Haney and Hall 1973).

All samples for bacterial enumeration were preserved with a 2% formaldehyde solution within two hours of sample collection. Samples were filtered later and counted with an acridine orange method (Daley 1979).

Water flow was measured with a Pygmy flow meter that was calibrated at the Canada Center for Inland Waters (CCIW).

2.3 Chemical Analyses Done at the Lake

All soluble reactive phosphorus analyses in 1982, 1983, and 1984 were done within 24 h with the ascorbic acid method (Strickland and Parsons 1972). Oxygen and temperature measurements were made with YSI oxygen meters. In 1982, 1983, and 1984 pH was measured at the lake with a Corning pH meter. Inorganic carbon was analyzed by a gas chromatography method (Stainton et al. 1977).

2.4 Laboratory Analysis of Field Samples
2.4.1 Chemical Analysis

Phosphorus was measured as total P after perchloric acid digestion. Calcium samples were preserved with HCl (1.0 ml of 50% HCl per 100 ml of sample) and analyzed later with atomic absorption spectrometry in CCIW with a

nitrous oxide/acetylene flame. The degree of calcium carbonate saturation was determined using calcium and carbonate data, corrected for ionic strength by the Debye-Huckel method (Berner 1971; Stumm and Morgan 1981). Samples for chlorophyll $\underline{a}$ analysis were filtered onto GF/C or GF/F (after 1982) filters at the lake within 2 h of collection, frozen, and analyzed by DMSO extraction (Burnison 1980) in the CCIW lab in Burlington. Iron content in the unenriched samples were measured by a modification of the bathophenanthroline method (Strickland and Parsons 1972). Samples for nitrate, ammonia, nitrite, dissolved organic nitrogen, particulate nitrogen, particulate carbon, dissolved inorganic carbon, total inorganic carbon, chloride, sulphate, calcium, magnesium, potassium and sodium were prepared in a field lab, and shipped on ice to the IWD Water Quality Lab in Vancouver, or Burlington (Environment Canada 1979).

2.4.2 Electron Microscopic Analysis

Water samples were filtered onto .2 μ Nuclepore membrane filters. They were dried and later analyzed for elemental composition at McMaster University on an electron microscope equipped with an EDAX microprobe for elemental analysis. With the larger calcite particles we were able to direct the beam at the center of the particle and obtain a semi-quantitative analysis.

2.4.3 Sediment Analysis

Sediment cores were taken from Roche, and Frisken Lakes with a Williams

lightweight corer (Williams and Pashley 1978). The cores were divided into 2.0 cm sections within an hour with the Williams extruder (Williams and Pashley unpublished data). The samples were then frozen and later freeze-dried at CCIW. Prior to analysis the samples were ground in a sediment grinder, pelletized and analyzed for major elements (Si, Al, Ca, Mg, Fe, Na, K, P, and Mn) by X-ray fluorescence spectrometry. Determination of minerals present in subsamples was carried out qualitatively by a Philips X-ray diffraction spectrometer.

Sediment samples were extracted with dilute acid to determine the relative solubility of phosphorus and calcium in Roche and Frisken Lake sediments. Freeze-dried and ground sediment (0.1 g) was shaken with 100 mls of 0.1 N HCl for 12 h. The samples were then centrifuged for 20 minutes at 1000 g and the supernatant was analyzed for soluble reactive phosphorus or calcium.

RESULTS

3.1 Source of Phosphorus in Frisken Lake

Surface runoff is not likely the major source of phosphorus. The stream entering Frisken Lake contained less phosphorus than the lake. Phosphorus may have concentrated in the lake as water evaporated, but this explanation is weak. The most soluble ions (K, Na, Cl, SO_4) were not significantly concentrated in the lakewater, relative to the stream (Table 1).

Groundwater dissolution of apatite may be the major source of phosphorus. A small spring in the northwest corner of Frisken Lake contained more than 300 µg P/L. In an area of similar geology, relief, and altitude, Chain lake receives about 60% of its external phosphorus loading from groundwater flow (Murphy 1985).

The combination of geology and hydrology probably accounts for much of the large difference between Frisken and Roche Lake (Table 1). Frisken Lake is rich in phosphorus and is presumably largely influenced by groundwater dissolution of an apatite outcropping. Roche Lake is downstream from Frisken Lake. Roche Lake is phosphorus poor and may be more influenced by surface runoff from Mount Bulman (section 1.1.1).

The hydraulic or apatite distribution hypotheses can not fully explain

the juxtaposition of clear Roche Lake to the hypertrophic Frisken Lake. The ion content of Roche Lake is much higher than the spring runoff it receives (Table 1). Ions, including phosphorus, may concentrate in the lake by evaporation or groundwater flow could be important. Moreover, the phosphorus loading from Frisken Lake into Roche Lake must be considerably higher than most lakes would receive, and the phosphorus concentration of Roche Lake is very low (Table 1).

A precipitation reaction probably removes phosphorus from Roche Lake water. Since both Frisken and Roche Lakes are hardwater lakes, calcium carbonate precipitation could precipitate phosphorus in both lakes. The higher phosphorus and organic carbon content in Frisken Lake should interfere with calcium carbonate precipitation (Reddy 1979, Waltor and Hanor 1979). Very different carbonate precipitation regimes should occur in each lake.

3.2 Sediment Chemistry

The carbonate chemistry of the sediments confirms that carbonate precipitation is substantial but quite different in each lake. The carbonate content of the sediments of Frisken Lake (Table 2) is much lower than that of Roche Lake (Table 2). In Frisken Lake the carbonate content decreases greatly with increasing depth while the deeper sediments of Roche Lake are still carbonate rich. A distinct peak of calcium and phosphorus at 10 cm in the Roche Lake sediments (Fig. 3) is not present in the Frisken Lake sediments (Fig. 4). These lakes are similar in that the correlation of total phosphorus

to calcium carbonate in each lake is strong only in the surface sediments (0-16 cm). In the surface sediments of both Roche and Frisken Lake the total phosphorus is positively correlated to the total calcium content and negatively correlated to the iron content (Roche r=0.80, n=6; r=-0.81, n=6: Frisken r=0.95, n=7; r=-0.86, n=7). Below 16 cm another process appears to control phosphorus solubility.

The change in correlation of calcium to phosphorus near 16 cm may reflect a diagenetic reaction that was not resolved. Chemical transformations (Berner 1971, 1980) and movement of ions with groundwater (Van Liere and Mur 1982) could alter the sediment. Anything that altered the production of organic acids or CO_2 in the sediment could have altered the solubility of phosphorus and calcium (Rhue and Hensel 1983). The greater productivity of Frisken Lake results in more organic matter in the Frisken Lake sediments (surface organic C=24%) than in Roche Lake (surface organic C=13%). Organic decay should cause more dissolution of precipitated carbonates and phosphates in Frisken Lake than in Roche Lake. Thus, the residual calcium and phosphorus in Frisken Lake should be more resistant to acid extraction.

More of the total phosphorus and calcium of the Roche Lake sediments was dissolved by acid than in Frisken Lake sediments (Table 3). This result indicates that less phosphorus and calcium is regenerated in the surface sediments of the less productive Roche Lake and that more apatite forms in Roche Lake. Apatite forms slowly (Stumm and Leckie 1970, Bremner 1980) ;thus,

dissolution of P-CaCO$_3$ must suppress the formation of apatite.

The lower efficiency of calcium extraction of the Frisken Lake sediments could also represent the complexation of calcium by organic matter. The higher organic content of Frisken Lake should result in an inhibition of calcite precipitation and perhaps apatite formation.

3.3 Pretreatment Water Chemistry

Total inorganic carbon (TIC) was used as an index of calcium carbonate solubility. Soluble reactive phosphorus (SRP) was used as an index of phosphorus solubility. Interference in SRP analysis by ferric precipitation as Nurnberg (1984) has observed in hypolimnetic samples was not observed in Frisken Lake. The SRP concentrations were close to the total phosphorus concentrations (93.4%, n=38; post-lime treatment values); moreover, the abundant particles in these samples presumably contained the remaining phosphorus. Frisken Lake appears to have very little reactive iron to interfere with SRP analysis.

A high correlation of phosphorus to inorganic carbon was found in the water column of Frisken Lake. In pretreatment samples significant correlations of TIC and SRP were observed in both the epilimnion/metalimnion (r=.735, n=21) and in the hypolimnion (r=.973, n=11). When these two pretreatment data sets were plotted two linear relationships were apparent (Fig. 5). The lowest TIC and P concentrations were in the epilimnion and the

highest TIC and P concentrations were in the hypolimnion (Fig. 6). The same relationship existed between total inorganic carbon and total phosphorus (epilimnion/metalimnion r=.753, n=21; hypolimnion r=.955, n=11). The similarity of the SRP/TIC and TP/TIC relationships was a reflection of the phosphorus chemistry. Most of the total phosphorus was in solution (epilimnion/metalimnion 70.6%; hypolimnion 73.8%) and the soluble phosphorus was highly correlated to the total phosphorus (epilimnion/metalimnion r=.912 n=21; hypolimnion r=.959, n=11).

Although geochemical precipitation of phosphorus with calcium carbonate is the most appropriate explanation of these observations, biological reactions seem, at first, to be as suitable. The "precipitation" of phosphorus could have been a coprecipitation of calcium carbonate with algal cells that were rich in phosphorus (Comeau 1984, Koschel et al. 1983, Shapiro 1967). This sedimentation would result in a good correlation of P to TIC in the epilimnion. The low pH in the hypolimnion was presumably produced by bacterial breakdown of algal cells. A very large complex community (filaments, clumps) that was not amenable to enumeration was observed only in the hypolimnion. With bacterial breakdown of organic-P in the hypolimnion, and subsequent dissolution of calcium carbonate, a high correlation of phosphorus and TIC would result.

This coincidence of biological and geochemical reactions weakens the common approach of interpreting field data by correlation analysis. Adjusting

the carbonate equilibria should provide better insight into the relative importance of geochemical and biological reactions. If this geochemical interpretation is appropriate, lime addition should control much of the biological productivity. Biological processes should modify the reactions in lime treatment, not control them.

3.4 CaO Treatment

In 1982, 0.72 tonnes of CaO were added to Frisken Lake by broadcasting quicklime onto an air-water plume created by a diesel air compressor. Although the turbulence caused by the lake aeration appeared to mix the lime quite well, the initial results of this aeration method were disappointing. Calculations with a pH titration curve of Frisken Lake water (Fig. 7), pH changes in the lake, and the volume of the lake, indicated that the titration was less than 10% efficient. The aerator-mixer appeared quite inefficient. However, the results of the 1983 study indicated that a low efficiency in the pH titration was caused by enhanced bacterial production of CO_2 (discussed later). The pH data were misinterpreted in 1982. The aeration experiment was cancelled too quickly.

Data obtained much later confirmed that our lime treatment strongly influenced the phosphorus concentrations. Independently of our sampling program, the Fish and Wildlife Branch had sampled the lake 5 days after our quicklime (CaO) addition in 1982. Although the sampling depths were slightly different, the effect of the quicklime was readily apparent. Within this 5

day period, the SRP decreased in the epilimnion from 260 (1 m) to 115 (0 m) µg/L, in the metalimnion from 276 (3 m) or 340 (5 m) to 196 (4 m) µg/L, and in the hypolimnion from 889 (9 m) to 591 (9 m) µg/L. Lime applications in 1983 and 1984 also took more than two days to change phosphorus concentrations.

The decision to use CaO rather than Ca(OH)$_2$ may seem inappropriate. The use of CaO requires additional safety precautions. CaO may be too dangerous for simple lime handling procedures. Two minor burns from Ca(OH)$_2$ were experienced in 1983 and similar accidents with the more caustic CaO could be serious. A poor seal between short gloves and the raincoat allowed lime to get under the tape and the lime was rubbed into the skin during the lime handling. Long gloves are a necessity. CaO may be more appropriate for more automated water treatment. CaO has more titration power and reactivity. At some sites, CaO is readily available by reprocessing lime wastes.

Other problems contributed to the termination of the quicklime treatment in 1982. Sampling of the treated water in the aeration plume was very awkward. The aeration plume brought hypolimnetic water to the surface, which then moved laterally 20 to 30 m, before sinking back down to the top of the hypolimnion (5 m from the lake surface). The shape of the plume was greatly affected by the wind. After five hours of aeration, the temperature and oxygen content of the hypolimnion a hundred meters from the aerator was unchanged. Since the oxygen and temperature of the aerated plume rose 2.0 mg

O_2/L and 3.5°C within five hours of aeration, a convection cycle must have been created with an unknown amount of entrained epilimnetic water, an unknown amount of untreated hypolimnetic water, and an unknown amount of recycled water.

It now seems that this combined aeration lime treatment would complement the surface additions of lime (presented next). The surface addition of lime results in a very nutrient rich anoxic hypolimnion. Treatment of the hypolimnion with a combined aeration-lime treatment, a month after surface lime treatment, would have these advantages.

1) Lime could react with the most phosphorus rich water.

2) A relatively small portion of lake volume could be treated.

3) The littoral zone of the lake could remain unaltered.

4) The pH could be titrated high enough to volatilize ammonia.

5) A high residual carbonate content could be left in the hypolimnion for future enhancement of carbonate precipitation.

6) The oxygen content of the lake could increase.

These aspects would be further enhanced by use of a hypolimnetic aeration design. Lime additions via a hypolimnetic aerator in lakes with anoxic hypolimnions would not expose fish to lime. This schedule of treatment might have many advantages but it offers less immediate treatment of the epilimnion than the slurry maker/barge lime addition method that was used in 1984.

3.4.1 Long-Term Effect of the CaO Treatment

Before the 1983 experiments began, the phosphorus concentration in the epilimnion was much lower than in earlier summers (Fig. 8). In June of 1983 the SRP concentrations in four samples from the epilimnion were only a relatively small percentage of the total phosphorus (14.3, 24.1 53.2, and 12.8%). The lowest earlier value observed in pretreatment samples was 43.5% (mean of all samples 70.6%, n=21). Earlier and later samples were less variable. These changes could have been a result of the quicklime addition in 1982. If over several months all of the quicklime reacted, the pH of the lake could have increased by 0.3 to 0.5 units, and the Ca concentration could have increased 0.42 mg/L. The stoichiometry of the microbial CO_2 processing or of atmospheric CO_2 exchange is unknown; thus, the final carbonate equilibrium cannot be calculated. The quicklime added through the aerator in 1982 probably was responsible for the low phosphorus concentrations observed in the spring of 1983. The 1983 and 1984 experiments also support the hypothesis that enhanced carbonate precipitation should occur in algal blooms that form after lime treatment.

3.5 The 1983 Lime Additions

A slurry maker/barge was used to improve the lime titration and to distribute lime around the lake. Any lime that did not react in the water column should have been evenly distributed over the sediments. Lime reactions in the sediments could reduce the amount of phosphorus released from the sediments into the water column, should result in a long term dissolution of

lime which would increase calcium and inorganic carbon concentrations in lake water, and thus maximize the long-term stimulation of calcium carbonate precipitation.

3.5.1 Short-Term Responses

The first obvious response was a colour change in the lake. At first a white plume appeared behind the boat that took about 15 min to disappear. Later the treated water become much browner than the untreated water. Streaks of brown water formed that later coalesced. The colour change to brown occurred faster in each subsequent lime treatment. The lake was clearer in July than in June (Fig. 9); however, the chlorophyll a content was unchanged (Table 4). The increase in clarity could have been caused by sedimentation of colloidal calcium carbonate. The surface calcium concentrations were reduced by 4 mg/L (Table 5).

The increased clarity after the second lime addition (Secchi depth of 4.0 m, Fig. 9) was associated with low chlorophyll a concentrations (Table 4). Observations of sediment traps suggested that enhanced settling of the algae was responsible for the greater clarity. The traps after the second lime application had 3 layers, a presumed lime layer, an algal layer and a greenish white layer. These observations can be further substantiated by simple comparisons of neighboring lakes. Algal blooms persisted through this period in both John Franklin and Bleeker Lakes.

3.5.1.1 Biotic Responses in 1983

The sedimentation of algae in the second lime application had a strong influence upon the hypolimnion. Within 48 h the pH of the hypolimnion decreased by as much as 0.3 (Fig. 10, Table 6). Some of this pH decrease could have been caused by microbial respiration such as heterotrophic utilization of decaying algae. In 1984, an increase in bacteria concentration in the deeper waters was observed (discussed later). Daphnia pulex appeared to move to the same water depth where was observed the greatest decrease in pH, presumably to feed upon the carbonate aggregates of bacteria and decaying algae (Fig. 11). Daphnia pulex was not killed by the lime and if the bacteria/zooplankton response is verified, zooplankton and thus fish may actually be enhanced by lime application. These results contrast strongly to the use of copper sulphate to suppress algal growth. Copper treatments can kill zooplankton (Taub and Kindig 1984).

3.5.1.2 pH Titration of Lake and Biological Buffering

Another implication of the enhanced bacterial activity in the hypolimnion is that an evaluation of the efficiency of lime dispersal can not be made by pH measurements. Calculations using pH data, a titration curve, and the lake volume indicated that the July and August lime additions in 1983 were only 8.7 and 10.3% efficient. Analyses of sediment from sediment traps and the total inorganic carbon (TIC) data demonstrated that these estimates of the reactivity of lime are much too low. A titration analysis with sediment from our sediment traps indicated that the efficiency of titration

was 58.9 %. If atmospheric CO_2 exchange is ignored, the increase in TIC after the second lime addition estimates a dissolution of 36.6% of the lime in the hypolimnion. Not enough data were collected in the first lime addition to use this approach.

The TIC did not increase after the third lime addition. Perhaps the lack of algae to be sedimented from the epilimnion in the third lime addition resulted in much less bacterial activity in the hypolimnion. Another observation supports this argument. Aggregates of bacteria and algae around lime crystals can be easily seen with a microscopic. The bacterial production of carbon dioxide must influence the dissolution of the lime. Unlike in the second lime treatment, the pH of the hypolimnion did not decrease after the third lime addition (Table 6). Since bacterial metabolic activity appeared to cause this pH decrease and TIC increase in the hypolimnion in the second period (Fig. 12), and pH and TIC did not change in the third period, the hypolimnetic bacterial metabolism in the last lime addition was probably unchanged.

The biological aspects of lime addition, algal sedimentation, and enhanced bacteria growth, strongly influence the short term and perhaps long term efficiency of lime treatment. Perhaps the optimal time to add lime to a lake is in an algal bloom. The last 1983 lime addition had little effect when algae were not present.

3.5.1.3 Effect of Lime Addition on Phosphorus Solubility

The SRP concentration of the water changed after lime addition but initially the response was not strong enough to distinguish geochemical reactions and biological phosphorus assimilation. The SRP concentration in the epilimnion was already low when the 1983 study began. The small initial changes in phosphorus concentrations during 1983 were a reflection of the new water chemistry which was probably a result of the quicklime treatment of 1982. Furthermore, the lime additions had to be split into three periods.

The SRP concentration in the epilimnion in the first treatment decreased (Table 7). The first lime addition resulted in a substantial decrease in particulate phosphorus and no decrease in chlorophyll a. The sedimentation of particulate-P (Table 8) could have been sedimentation of colloidal calcium phosphorus or calcium carbonate-phosphorus. At this time, the SRP fraction of total-P was unusually low for this lake. The only other time that such low ratios were observed was in hypolimnetic samples collected during the phosphorus and carbonate precipitation of Sept. 83. The first lime application removed calcium from the epilimnion (Table 5). The SRP concentration also decreased after the second lime addition in the 4.0 m samples (Table 7). The third treatment appeared to have little effect upon the soluble or total phosphorus distribution or concentration (Table 7, 8). The concentrations of sodium, potassium, and magnesium were relatively unaffected by our lime treatments (Tables 9, 10, 11).

3.5.1.4 Prediction of Effect of Lime Treatment on P Solubility

Perhaps the most significant observation is that the pretreatment data could predict the posttreatment response. The low phosphorus and TIC concentrations in the August 1983 epilimnetic samples agrees very well with the value predicted by an extrapolation of the pretreatment TIC-SRP relationship (Fig. 5, TIC of 30.6 mg/L and SRP of 0).

3.5.2 A Delayed Response to the 1983 Lime Treatment

A large precipitation of TIC and phosphorus occurred in the fall of 1983 when the lake started to destratify (Table 12) The top of the hypolimnion had mixed into the epilimnion (Fig. 13). The earlier dissolution of lime in the hypolimnion had increased the TIC (Table 13) and Ca (Table 5) content of the hypolimnion. The epilimnion had a high pH (Table 6). Mixing of these waters should have resulted in precipitation of calcium carbonate and phosphorus. The removal of 57% of the phosphorus (relative to Oct. 31, 1982, Tables 13, 7, 8) in the lake was probably caused by carbonate precipitation.

In spite of this initial enhancement of carbonate precipitation, much of this precipitated SRP redissolved in the winter. A simulation analysis with the computer program PHREEQE (Parkhurst et al. 1980) and the late summer data from 1983, indicated that initially calcium carbonate would precipitate when the lake mixed. However, complete mixing of the lake in the winter of 1983-84 should have resulted in complete redissolution of the calcium carbonate and the adsorbed phosphorus. Similar calculations indicate an enhanced carbonate

precipitation in future summers when the epilimnion warms.

3.5.2.1 Iron Chemistry

The iron in Frisken Lake may not be very reactive. The general trends in the concentration of iron in Frisken Lake are very similar to the two years of iron data collected from iron-limited Black Lake (Murphy et al. 1983b). The fall destratification of anoxic hypolimnetic water into the oxidized epilimnetic water had the potential to precipitate ferric phosphate. Particulate iron was quite high in the hypolimnion in midsummer; however, only 30% of this iron precipitated from the water column in September, 1983 (Table 15). Initially most of this iron was retained in the epilimnion. By spring most of the iron had precipitated. The slow precipitation suggests that much of the iron was not reactive. If all of the precipitating iron formed ferric phosphate (Sept. 83 depth integral of 49 µg/L decrease in iron), the lakewater would lose only an average of 27 µg P/L. In Frisken Lake, this reaction would be insignificant.

Recent studies in other lakes have shown that iron is rapidly coated by humic acids (Tipping 1981, Baccini et al. 1982). The dissolved organic content of Frisken Lake (23 mg/L) was quite high; thus, the iron in Frisken Lake could be complexed by organic compounds that limit its reactivity. Jackson and Hecky (1980) proposed that the reduction of primary production observed by humic acids was achieved by reducing the availability of iron to microbes. The 1984 data supports the hypothesis that Frisken Lake has very

little reactive iron (discussed in section 3.6.2). Although many geochemists have stated that iron usually dominates phosphorus geochemistry (Williams et al. 1971, Shukla et al. 1971), when iron is not reactive, calcium reactions will control phosphorus solubility (Bostrum and Petterson 1982, Stauffer 1985).

3.6 1984 Lime Treatment

In May 1984, Frisken Lake had a dense bloom of Oscillatoria. Much of the phosphorus that had precipitated in 1983 had redissolved. The lakewater was too cool for natural carbonate precipitation to have occurred yet.

In May 1984, lime was added via the slurry maker. Unlike in 1983, all of the lime was added during a brief period. Phosphorus concentrations in the epilimnion (Table 16, 17) and chlorophyll concentrations were rapidly reduced (Figure 14). By observing particles with the EDAX microprobe attachment of an electron microscope, particles of calcium carbonate with surface coatings of phosphorus were observed only after the lime treatment and only in the epilimnion.

3.6.1 Biotic Responses in 1984

The distribution of Daphnia pulex and Keratella quadrata became more variable after the lime treatment, but the population means were unchanged (Table 18, 19). No consistent change in the distribution with depth was apparent. After two weeks, both species had increased greatly in density.

The distribution of a _Polyarthra_ sp. became more variable after lime treatment (Table 20). Initially, its population mean was unaltered. After the lime treatment this species was much more concentrated in the bottom of the metalimnion (6.0 m). Two weeks later the density of _Polyarthra_ had decreased greatly.

Initially, the mean and the distribution of _Keratella cochlearis_ was unaltered by the lime treatment (Table 21). After two weeks the population had greatly decreased in density.

The concentration of _Polyarthra_ at six meters and the increased variance in the distribution _Daphnia pulex_ and _Keratella quadrata_ seemed to be caused by the lime treatment. The _Polyarthra_ concentration at six meters could have represented a response to an increased supply to food at this depth or settling decaying animals may have become neutrally buoyant at this depth.

The hypothesis that lime treatment enhances zooplankton feeding by aggregating small particles could not be tested (see section 3.5.1.1). The increase in bacteria density occurred in the anoxic hypolimnion (Table 22), and the zooplankton appeared to avoid anoxic water. The changing availability of food, and normal seasonal succession of zooplankton populations complicate detailed interpretation of the data. Regardless of the subtle effects of lime treatment, the availability of zooplankton food to fish was not radically

suppressed.

Although initially Gammarus seemed unaffected by the lime, between two weeks and a month after the lime treatment many of the pelagic Gammarus died. Perhaps the death of the Gammarus was caused by a precipitation of their food. The algal blooms were successfully suppressed throughout the summer (Fig. 14) and the bacteria numbers were significantly reduced in the epilimnion (Table 22). The collapse of the Gammarus was not apparently directly related to a pH shock. The Gammarus survived the lime application period and died when the pH was declining (Table 23).

This lime addition produced the most rapid pH change of the study. This result may have been a product of the increased efficiency of the pH titration. The second and third additions of lime in 1983 were 8.7 and 10.3% efficient. The 1984 lime addition was 14% efficient. The increased efficiency in the titration may have been related to the removal of 50% of the dissolved organic carbon in the lake by the 1983 lime treatment.

The decomposition of the algal bloom resulted in both the reduction of the initial high pH (Table 23) and a decreased oxygen concentration (Table 24). The low oxygen concentration in the deeper water and the high water temperatures of the surface water (Table 25) may have stressed the trout, but no dead fish were observed. Live trout were observed for two months after the lime treatment and in January of 1985. The pH values in these lime treatments

should not have killed trout (Wright 1984).

3.6.2 Algal Nutrient Limitation

Although the phosphorus concentration was greatly decreased the dissolved phosphorus concentration remained too high for the algae to have been phosphorus limited (Table 16). The suppression of algal growth may have been related to a lack of iron in the epilimnion. The epilimnetic concentrations of iron after the lime treatment remained lower than was found to limit algal growth in a similar lake, Black Lake (Murphy et al. 1983. The slight seasonal increase of epilimnetic iron in Frisken Lake (Table 26) was matched by a modest seasonal increase of chlorophyll $\underline{a}$ (Fig. 14).

Iron concentrations increased in the hypolimnion in midsummer, but the total hypolimnetic iron was only half as concentrated as in 1983 (Table 15, 24). The hypolimnion was anoxic prior to and after the iron release. Moreover, after the 1983 lime treatment the bottom two meters of Frisken Lake looked and smelled like the monimolimnion of meromictic Yellow Lake B.C. (clear and rich in H_2S). The production of a monimolimnion by lime treatment is consistent with the chemistry of biogenic meromictic lakes (Wetzel 1975). The enhancement of anoxia should enhance pyrite formation. Manning (unpublished data) has found that pyrite formation is rapid in Frisken Lake. Pyrite formation should reduce the entry of iron into the water column and enhance iron limitation.

3.7 1985 Results

The SRP concentrations in the epilimnion of Frisken Lake were much less than pretreatment concentrations (Fig. 8). This result confirms the hypothesis that the increase in TIC that lime treatment produced, can result in long-term enhancement of calcium carbonate precipitation and coprecipitation of phosphorus. The clarity of the lake was greater than pretreatment and the chlorophyll concentrations were lower than pretreatment (R. Grace, B.C. MOE, Kamloops, unpublished data). The long-term response was not as favourable as the immediate response.

Conclusion

The Frisken Lake studies demonstrated that calcite precipitation can control phosphorus chemistry. The affinity of phosphorus for calcite is weak and several processes regulate the adsorption of phosphorus to calcite. Iron in Frisken Lake is relatively unreactive and poorly characterized. The carbonate lakes of the Thompson Plateau appear to have too little reactive iron to influence phosphorus chemistry. Thus, oxygen can have little influence on iron reactions and phosphorus solubility. Lime addition may enhance iron limitation and decrease the availability of phosphorus.

MANAGEMENT IMPLICATIONS

To ameliorate eutrophication of many eutrophic lakes will require the use of _in situ_ lake restoration techniques. The immediate effects of lime treatment demonstrated that calcium carbonate precipitation can be enhanced. The long-term response of the lime treatment supported the hypothesis that calcium carbonate precipitation can regulate phosphorus reactions in hardwater lakes. Because carbonate precipitation in a lime-treated lake could be enhanced for several years and since carbonate precipitation is much faster in warm water, lime treatment could suppress summer blue-green algal blooms more than the spring diatom blooms. This hypothesis suggests that by appropriate management, a eutrophic lake could still sustain a good fishery without the problems that are commonly associated with blue-green algal blooms.

4.1.1 Lake Treatments

The substantial improvements in aesthetics and water quality of Frisken Lake for a chemical cost of less than $150.00 a hectare suggests that lime application could be a useful lake restoration tool. The water was clearer, and better tasting. The chlorophyll _a_ and phosphorus concentrations decreased. The cost is low enough that recreational lakes of high value could be treated. Moreover, some of the poor drinking water supplies in the interior of B.C. should respond well to lime treatment. This "in-lake" treatment is only suitable for hardwater lakes. Lime application in softwater lakes may be able to sediment algae and some phosphorus, but there would be little long term enhancement of natural precipitation reactions to in turn

suppress eutrophication.

"In-lake" treatments have some advantages over water treatment in a water treatment plant. Most lakes have long enough water residence times that fine carbonate precipitates can aggregate and sediment. To achieve equivalent precipitation in a fast flow through system requires five times as much lime and subsequent carbon dioxide injection to reduce the pH (Culp et al. 1978, Ferguson et al. 1973). Also preventing the formation in a lake of organic compounds may be less expensive than removing them in a water treatment plant with activated carbon. The activated carbon filters for Regina's drinking water cost $1,000,000, regeneration of the filters costs $400,000, and the building cost $15,000,000.

4.1.2 Local Calcium Sources For Lake Treatment

Future lake treatments should use whatever local materials appear most appropriate. Transportation charges will limit the utility of any product that is not locally available. Three local substrates warrant study; calcium sulphate, calcium carbonate, and waste lime. Calcium sulphate deposits are found near Kamloops (Cockfield 1947), but we are unaware of any commercial use of these deposits. Calcium carbonate is locally mined and converted into Portland cement. Calcium carbonate may not induce carbonate precipitation as readily as lime but if calcium carbonate is inexpensive, it could sediment algal cells, and promote long term precipitation of phosphorus; calcium carbonate might be a safe alternative for lake restoration. When available in large quantities as a waste, calcium carbonate could be a good sealant for

nutrient-rich sediments. Pulp mills occasionally have to dispose of large quantities of calcium carbonate (lime-mud). The lime-mud could be purified by incineration in a lime kiln.

Some resuspension of calcium carbonate may occur after lime treatment. Calcium carbonate resuspension can occur naturally (White and Wetzel 1975); however, the turbidity could be less opaque and should last for a shorter period than that previously caused by resuspended organic matter. Carbonates are much denser than organic matter. Resuspended carbonate should enhance the sedimentation of algae.

Waste lime is available from several sources such as fruit co-ops where it is used to regulate CO_2 levels during fruit storage. This lime is lumpy and might require grinding to improve its reactivity. About half of the calcium hydroxide has been converted into calcium carbonate. The production of acetylene is one of several industrial processes that can produce a relatively pure waste lime. Occasionally, direct discharge of waste material can be used effectively. For example, eutrophication of Onondaga Lake in New York State is suppressed by a discharge of calcium chloride from an alkali factory (Wodka et al. 1985). However, uncontrolled dumping of any lime into a lake should not be approved. Testing for contaminants and supervision of the disposal of the waste lime by technicians would be required.

4.1.3 Importance of Magnesium in Lime Treatment

Another local variable that should alter the precipitation of phosphorus is the magnesium concentration. In the very alkaline lakes of B.C., high magnesium concentrations may restrict calcium phosphate precipitation (Ferguson and McCarty 1971, Gulbrandson et al. 1984, and Wilson 1979). These studies suggest that certain agricultural dolomitic limes that have a high magnesium content are not suitable for lime application to a lake. Most industrial limes are very low in magnesium (Boynton 1966).

4.2 Future Research Needs

4.2.1 Formation of Apatite

The literature on phosphorus precipitation in lakes has many conflicting hypotheses. It is generally agreed that the initial phosphorus precipitate can not be apatite (Koutsoukos et al. 1980). However, Brown (1980) suggested that hydroxlapatite could form in aqueous calcitic limestone suspensions (ie. in lime treatments). Berner and Morse's (1974) model of phosphorus adsorption to kinks in the calcite crystal agrees very well with the Frisken Lake data. Other authors have proposed that the initial adsorption of phosphorus to calcite is followed by surface rearrangement of phosphorus into phosphate heteronuclei that ultimately form the mineral hydroxylapatite (Stumm and Leckie 1970, Griffin and Jurinak 1974).

The formation of calcium phosphate could interfere with the formation of apatite. The lime treatment produced an optimal pH for the precipitation of calcium phosphate (8.5-9.0, Witteman 1976). Tricalcium phosphate, octacalcium

phosphate and dicalcium phosphate could also precipitate in lake water (Koutsoukos et al. 1980, Avnimelach 1983).

Biological activity could strongly modify the geochemistry. The precipitation of apatite could develop via biological precipitation (Vogel and Ennever 1971). A resolution of the important factors influencing apatite formation could help guide lime treatments of lakes. If time for nucleation of apatite from initial precipitates is more important than biological precipitation, the treatment of the hypolimnion with lime should produce a long-term enhancement of apatite formation (see section 3.4).

4.2.2 Weathering of Apatite

A determination of the reactions of phosphorus minerals in the drainage basin could quickly lead to better lake management. For example, a study of Kamloops Lake showed that apatite in the lake sediments was unaltered detrital apatite derived from the drainage basin (St. John et al. 1975). The Kamloops Lake study showed that this form of apatite was resistant to weathering, that much of the phosphorus was not readily available to microbial assimilation, and that the total phosphorus loading to Kamloops Lake was not a good index of biological growth potential. A very similar argument has been made for apatite loading into Lake Erie (Burns et al. 1976).

Both the Lake Erie and Kamloops Lake work does not seem appropriate for other areas where apatite does appear to dissolve and is available for algal

growth (the Yellow Lake area, and the Wildhorse Mountain Batholith). Certainly everyone agrees that apatite in a stream will have much less effect upon a lake than the stream's soluble phosphorus. A disagreement exists over the rate of weathering of apatite in the drainage basin. Thomas (IJC, Windsor) has said that apatite in Ontario is completely insoluble (personal communication).

Perhaps the dissolution of apatite is hidden by adsorption of phosphorus to iron. Phosphorus is quickly immobilized in many soils in eastern North America by complexation with iron (Wood et al. 1984). Many of the rocks of around Kamloops are red from hematite ;thus, similar reactions could have influenced St. John's study. However, The Kamloops hematite deposit is not typical of much of western North America. Unlike eastern North American soils, many of the soils of arid and semi-arid western North America have insufficient iron to satisfy plant growth (Brown 1979). The soil science data indicates that there may be broad regional differences in iron reactivity and phosphorus mobility.

Perhaps apatite is too general a term. Apatite is a variety of calcium phosphate minerals (McConnell 1973) that vary in their solubility (Smith et al. 1978). As more ions substitute into the apatite crystal, apatite becomes more soluble (Brown 1966, Ferguson and McCarty 1971). Resolution of the mechanism responsible for regional differences in phosphorus mobility may provide lake managers with improved loading coefficients for phosphorus.

4.2.3 Management of Apatite Weathering in Lake Basins

At least three important management decisions require basic biogeochemical information on phosphate and calcite biogeochemistry:

1) What impact would acid rain have on the Thompson Plateau? Apparently acid rain has resulted in the alkalization and eutrophication of a hardwater lake in Michigan (Kilham 1982). The potential of enhanced weathering of volcanic-P by acid rain has not been studied.

2) How should the forests in important watersheds be managed? Any source of acid could dissolve apatite or calcite. The microbial production of organic acids and perhaps CO_2 by decaying vegetation should enhance the weathering of phosphorus minerals (Kuznetzov 1970). Forest management could influence the solubility of apatite. Wetzel and Manny (1978) demonstrated that logging led to enhanced weathering of calcite, enhanced carbonate precipitation in the lake, which in turn suppressed algal productivity. Likens et al. (1978) also observed a significant export of calcium from a logged forest. Wood et al. (1984) found that phosphorus was not exported from Likens study site in New Hampshire. Similar studies have apparently not been done in areas where phosphorus is mobile.

3) Which rock formations are the preferred sites for land disposal of treated sewage wastes? Certain areas with high concentrations of volcanic phosphorus bearing rock may not be good sites to dispose of treated sewage. Although conventional treatment removes 90% of the biological oxygen demand (BOD) of sewage, the remaining BOD can be important. Enhanced dissolution of volcanic-P by organic acids and CO_2 in the sewage could combine with the sewage-P to produce very phosphorus rich groundwater (Martinez et al. 1984).

Perhaps the organic wastes could stimulate phosphate solubilizing bacteria and fungi (Kucey 1983). Land disposal of sewage may be the optimal method if the land is low in soluble phosphorus; however, the geology of the Thompson Plateau is complex and both low and rich phosphorus rock are present.

4.2.4 Influence of Weather on Carbonate Precipitation

Wind is another process that influences carbonate precipitation. The amount of CO_2 exchange between the epilimnion and the atmosphere can regulate carbonate equilibria (Loewenthal and Marais 1978). Most lakes are not in equilibrium with the atmosphere and if the lake was brought into equilibrium by lake aeration, precipitation of calcium carbonate should occur. Carbon dioxide exchange is directly related to wind speed (Nishimura et al. 1984). Many of the eutrophic lakes in the interior of B.C. are very sheltered from wind aeration (e.g. Frisken, Corbett, and Yellow Lakes).

A determination of the effect of wind on carbonate precipitation would be difficult. The effects of wind on lake statification and nutrient cycling would make field experimentation difficult. This type of problem might be solved by using a computer model, and perhaps by building a model system to determine the appropriate coefficients of CO_2 exchange for various lakes found in B.C. Very few studies have measured CO_2 transfer coefficients, and they have almost all been conducted in marine environments (Kanwisher 1963, Broecker et al. 1978, and Nishimura et al. 1984).

4.2.5 Hydrology

Hydraulic data is required for any lake considered for lake restoration. Lakes with a short residence time will require much more treatment than a lake with a long residence time. Groundwater studies are needed in several areas. Groundwater is typically quite rich in CO_2 (Garrels and Christ 1965) and thus is capable of dissolving carbonates and phosphates. The high variability in geology discussed in section 2.1.2.1 results in a a highly variable groundwater flow and composition. The presence of springs indicates that much of the groundwater flow in the Thompson Plateau occurs along fractures in the bedrock. This type of flow complicates a study. Measurements of groundwater flow could be both expensive and difficult.

4.2.6 Biological Initiation and Control of Calcium Carbonate Precipitation

Several biological aspects of lime application and carbonate precipitation need clarification. The induction of carbonate precipitation is certainly catalyzed by algae, but the classical hypothesis that algae just influence the pH may be too simple. Darley (1974) claimed that certain algal divisions only precipitate calcite (eg. blue-green algae) and that other divisions only precipitate aragonite (another calcium carbonate mineral). Algae may act as a surface specific template for carbonate precipitation or they could direct the precipitation reaction. Energy is released in the precipitation reaction (Snoeyink and Jenkins 1980) and if certain algae could capture this energy, this algae would have an advantage over competing species.

This hypothesis is an old controversy in both geology and biology. Kuznetsov (1970) reviewed several studies of "calcitic" bacteria and dismissed the hypothesis because all of the responses could be explained chemically. Apparently only correlation analysis has been used to prove or refute the hypothesis. Morita (1980) also reviewed several studies of "calcitic" bacteria. Morita presented data to support the hypothesis of biological mediated calcite precipitation. He demonstrated that the ability to precipitate calcite was limited to 20% of his isolated marine bacteria. He suggested that bacteria initiated calcite precipitation by excreting calcium and by concentrating calcium from seawater. Novitsky (1981) disputed Morita's work but Danielli and Edington (1983) and Krumbein (1967) reported evidence that supports Morita's work.

4.2.7 Zooplankton Relationships to Calcite Precipitation

The effect of zooplankton on precipitating carbonates and the effect of lime treatment on zooplankton should be better resolved. Bacteria were associated with crystals of lime ;thus, bacteria may also be concentrated upon naturally occurring carbonate debris. Zooplankton could feed on these clumps. Many blue-green algae are thought to be a poor food item for zooplankton (Porter and Orcutt 1980, McNaught et al. 1980,). Some zooplankton may be able to selectively eat bacteria on carbonate clumps, but many zooplankton might feed nonselectively and still not be poisoned by these decomposing blue-green algae. Bacteria growing in association with blue-green algae are known to decrease the toxicity of blue-green algae (Carmichael and Gorham 1977).

REFERENCES

Avnimelech, Y. 1983. Phosphorus and calcium solubilities in Lake Kinneret. Limnol. Oceanogr. 28:640-646.

Babin, J. and E.E. Prepas. 1985. Modelling winter oxygen depletion rates in ice-covered temperate zone lakes in Canada. Can. J. Fish. Aquat. Sci. 42:239-249.

Baccini, P., E. Grieder, R. Stierli, and S. Goldberg. 1982. The influence of natural organic matter on the adsorption properties of mineral particles in lake water. Schweiz. Z. Hydrol. 44:99-116.

Berner, R.A. 1971. Principles of chemical sedimentology. McGraw-Hill.

Berner, R.A. and J.W. Morse. 1974. Dissolution kinetics of calcium carbonate in sea water IV. Theory of calcite dissolution. Am. Jour. Sci. 274:108-134.

Berner, R.A. 1980. Early diagenesis a theoretical approach. Princeton Univ. Press. 241p.

Bostrum, B. and K. Petterson. 1982. Different patterns of phosphorus release from lake sediments in laboratory experiments. Hydrobiologia 92:415-429.

Boynton, R.S. 1966. Chemistry and technology of lime and limestone. John Wiley and Sons. New York. 520p.

Bremmer, J.M. 1980. Concretionary phosphorite from SW Africa. J. Geol. Soc. Lond. 137:773-786.

Broecker, W.S., J. Peterman, and W. Siems. 1978. The influence of wind on CO_2 exchange in a wind-wave tunnel, including the effects of monolayers. J. Mar. Res. 36:595-610.

Brown, W.E. 1966. Solubilities of phosphates and other sparingly soluble

compounds. p. 203-239. In E.J. Griffith, B. Beeton, J.M. Spencer, and D.T. Mitchell (eds.) Environmental phosphorus handbook. John Wiley and Sons, New York.

Brown, J.C. 1979. Iron and Plants. In. Iron. pp 57-78. Medical and biological effects of environmental pollutants series. Univ. Park Press, Baltimore.

Brown, J.L. 1980. Calcium phosphate precipitation in aqueous calcitic limestone suspensions. J. Environ. Qual. 9:641-644.

Burnison, B.K. 1980. Modified dimethyl sulfoxide (DMSO) extraction for chlorophyll analysis of phytoplankton. Can. J. Fish. Aquat. Sci. 37:729-733.

Burns, N.M., J.D.H. Williams, J.M. Jaquet, A.L.W. Kemp, and D.C.L. Lam. 1976. A phosphorus budget for Lake Erie. J. Fish. Res. Board Can. 33:564-573.

Carmichael, W.W. and P.R. Gorham. 1977. Factors influencing the toxicity and anmimal susceptibility of Anabaena flos-aquae (Cyanophyta) blooms. J. Phycol. 13:97-101.

Clemens, W.A., D.S. Rawson, and J.L. McHugh. 1939. A biological survey of Okanagan Lake, British Columbia. Bull. Fish. Res. Bd. Can. 56.

Cockfield, W.E. 1947. Nicola, Kamloops, and Yale District Map. Geol Surv. Can. Sheet 92I (East Half).

Cockfield, W.E. 1948. Geology and mineral deposits of Nicola map-area, British Columbia. Can. Dept. Mines Resources Memoir 249. 42p.

Comeau, Y. 1984. Biochemical models for biological excess P removal from wastewater. MSc. Thesis, Civil Engineering, University of British Columbia.

Culp, R.L., G.M. Wesner, and G.L. Culp. 1978. Handbook of advanced

wastewater treatment. Van Nostrand Reinhold Company.

Daley, R.J. 1979. Direct epifluorescence enumeration of native aquatic bacteria: uses, limitations, and comparative accuracy. Sp. Tech. Pub. 695. Am. Soc. Test Mat.

Danielli, H.M.C. and M.A. Edington. 1983. Bacterial calcification in limestone caves. Geomicrobiol. J. 1:115.

Darley, W.M. 1974. Silification and calcification, p. 655-675. In. W.D.P. Stewart (ed.) Algal Physiology and Biochemistry. Univ. California Press. Berkley.

Elder, J.F. and A.J. Horne. 1978. Copper cycles and $CuSO_4$ algicidal capacity in two California lakes. Environ. Manage. 2:17-30.

Environment Canada. 1979. Analytical methods manual. Inland Water Directorate. Water Quality Branch, Ottawa, Can.

Ferguson, J.F. and P.L. McCarty. 1971. Effects of carbonate and magnesium on calcium phosphate precipitation. Envir. Sci. Tech. 5:534-540.

Ferguson, J.F., D. Jenkins, and J. Eastman. 1973. Calcium carbonate precipitation at slightly alkaline pH values. J. Water Poll. Control Fed. 45:620.

Garrels, R.M. and C.L. Christ. 1965. Solutions, minerals, and equilibria. Harper and Row. New York. 450p.

Gentile J.H. and T.E. Mahoney. 1969. Toxicity and environmental requirements of a strain of Aphanizomenon flos-aquae (L.) Ralfs. Can. J. Microbiol. 15:165-173.

Griffin, R.A., and J.j. Jurinak. 1974. Kinetics of phosphate interaction with calcite. Soil. Sci. Amer Res. J. of Canada. 15:59-72.

Gulbrandson, R.A., C.E. Roberson, and S.T. Neil. 1984. Time and the crystallization of apatite in seawater. Geochim. Cosmochim. Acta.

48:213:218.

Gunatilaka, 1982. Phosphate adsorption kinetics of resuspended sediments in a shallow lake. Hydrobiologia 91:293-298.

Hakanson, L., and M. Jansson. 1983. Principles of lake sedimentology. Springer-Verlag. New York. 316p.

Haney, J.F. and D.J. Hall. 1973. Sugarcoated Daphnia: a preservative technique for Cladocera. Limnol. Oceanogr. 18:331-332.

Harrison, M.J., R.E. Pacha, and R.Y. Morita. 1972. Solubilization of inorganic phosphates by bacteria isolated from upper Klamath Lake sediment. Limnol. Oceanogr. 17:50-57.

Hillbricht-Ilkowska, A., J.I. Rybak, Z. Kajak, K, Dusoge, J. Ejsmont-Karabin, I. Spodniewska, T. Weglenska, and W.A. Godlewska-Lipowa. 1977. Effect of liming on a humic lake. Ekologia Polska 26:379-420.

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. 37:2300-2317.

Jones, D.L., A. Cox, P. Coney, and M. Beck. 1982. The growth of Western North America. Sci. Amer. 247:70-84.

Kanwisher, L. 1963. Effect of wind on CO_2 exchange across the sea surface. J. Geophys. Res. 68:3921-3927.

Kelts, K. and K.J. Hsu. 1978. Freshwater carbonate sedimentation, p. 295-323. In A. Lerman (ed.) Lakes: Chemistry, Geology, Physics. Springer-Verlag. 363 p.

Kilham, P. 1982. Acid precipitation: its role in the alkalization of a lake in Michigan. Limnol. Oceanogr. 27:856-865.

Koschel R., J. Benndorf, G. Proft, and F. Recknagel. 1983. Calcite

precipitation as a natural control mechanism of eutrophication. Arch. Hydrobiol. 98:380-408.

Koutsoukos, P. Z. Amjad, M.B. Tomson, and G. H. Nancollas. 1980. Crystallization of calcium phosphates. A constant composition study. Am. Chem. Soc. 102:1553-1557.

Krumbein, W.E., 1968. Geomicrobiology and geochemistry of the "Nari-lime-crust" (Israel). p138-147. In G. Muller and G.M. Friedman (eds.) Recent developments in carbonate sedimentology in Central Europe. Springer-Verlag New York 255 p.

Kucey, R.M.N. 1983. Phosphate-solubilizing bacteria and fungi in various cultivated and virgin Alberta soils. Can. J. Soil Sci. 63:671-678.

Kuznetzov, S.J. 1970. The microflora of lakes and its geochemical activity. Univ. Texas Press.

Likens, G.E., F.H. Bormann, R.S. Pierce, and W.A. Reiners. 1978. Recovery of a deforested ecosystem. Science 199:492-496.

Kuznetzov, S.J. 1970. The microflora of lakes and its geochemical activity. Univ. Texas Press.

Likens, G.E., F.H. Bormann, R.S. Pierce, and W.A. Reiners. 1978. Recovery of a deforested ecosystem. Science 199:492-496.

Loewenthal, R.E. and G.V.R. Marais. 1978. Carbonate chemistry of aquatic systems, theory and practise. Ann Arbor Science. 433 p.

MacMillan-Bloedel Co. 1972. The Okanagan forest. Forestalk 1:13-15.

Marshall, K.C. 1976. Interfaces in microbial ecology. Harvard Univ. Press. 156 p.

Martinez, M.T. C. Romero, and J.M. Gavilan. 1984. Solubilization of phosphorus by humic acids from lignite. Soil Science 138:257-261.

McConnell, D. 1973. Apatite. Springer-Verlag, New York.

McNaught, D.C., D. Griesmer, and M. Kennedy. 1980. Resource characteritics modifying selective grazing by copepods. p 292-298. In W.C. Kerfoot (ed.) Evolution and ecology of zooplankton communities. New England Press. 793 p.

McKean, C.J.P. 1981. Dragon Lake General Water Quality and Nutrient Loading in 1980. British Columbia M.O.E. file 0273896-23.

McKean, C.J.P. and P.G. Lim. 1981. Proposed solutions to the taste and odour problems in Garnet reservoir. British Columbia M.O.E. file 0290862-general.

Morita, R.Y. 1980. Calcite precipitation by marine bacteria. Geomicrobiol. J. 2:63-82.

Murphy, T.P., K.J. Hall, and I. Yesaki. 1983. 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. Sci. Tot. Envir. 28:37-50.

Murphy, T.P. 1985. The effect of a water diversion on an eutrophic lake. Water. Poll. Res. J. Canada, submitted.

Nesbitt, H.W. 1974. The study of some mineral-aqueous solution interactions. PhD. thesis, Baltimore, Johns Hopkins University. 173p.

Nishimura, H., H. Nakajima, and M. Kumagai. 1984. Exchange of oxygen and carbon dioxide across the water surface during algal blooms in a pond. Water Res. 18:345-350.

Northcote, K.E. 1968. Geology and geochronology of the Guichon Creek Batholith, B.C. PhD. Thesis, Univ. of British Columbia.

Novitsky, J.A. 1981. Calcium carbonate precipitation by marine bacteria. Geomicrobiol. J. 2:375-388.

Nurnberg, G. 1984. Iron and hydrogen sulphide interference in the analysis

of soluble reactive phosphorus in anoxic waters. Water Res. 18:369-377.

Parkhurst, D.L., D.C. Thorstenton, and L.N. Plummer. 1980. PHREEQE a computer program for geochemical calculations. USGS Report WRI-80-96.

Porter, K.G. and J.D. Orcutt. 1980. Nutritional adequacy, manageability, and toxicity as factors that determine the food quality of green and blue-green algae for _Daphnia_. p 268-281. In W.C. Kerfoot (ed.) Evolution and ecology of zooplankton communities. New England Press. p 793.

Reddy, M.M. 1979. Kinetic inhibition of calcium carbonate formation by wastewater constituents, p. 32-58. In A.J. Rubin (ed.) Chemistry of wastewater technology. Ann Arbor Science.

Rhue, R.D. and D.R. Hensel. 1983. The effect of lime on the availability of residual phosphorus and its extractability by dilute acid. Soil Sci. Soc. Am. J. 47:266-270.

Rossknecht, V.H. 1980. Phosphate removal with calcium carbonate precipitation in the Lake of Constance (Obersee) (in German). Arch. Hydrobiol. 88:328-344.

Shapiro, J. 1967. Induced rapid release and uptake of phosphate by microorganisms. Science 155:1269-1271.

Shukla, S.S., J.K. Syers, J.D.H. Williams, D.E. Armstrong, and R.F. Harris. 1971. Sorption of inorganic phosphate by lake sediments. Soil Sci. Soc. Amer. Proc. 35:244-249.

Smith, R.B. 1974. Geology of the Harper Ranch Groupp (Carboniferous-Permian) and Nicola Group (Upper Triassic) northeast of Kamloops, British Columbia. M.Sc Thesis, Univ. of British Columbia.

Smith, E.A., C.I. Mayfield, and P.T.S. Wong. 1978. Naturally occurring apatite as a source of orthophosphate for growth of bacteria and algae. Micro. Ecol. 4:105-117.

Snoeyink, V.L. and D. Jenkins. 1980. Water Chemistry. John Wiley & Sons.

St. John, B.E., E.C. Carmack, R.J. Daley, C.B.J. Gray, and C.H. Pharo. 1976. The limnology of Kamloops Lake. Env. Canada Report.

Stainton, M.P., M.J. Capel, and F.A.J. Armstrong. 1974. Chemical analysis of fresh water. Can. Dept. Environ. Miscellaneous special publication No. 25. 119p.

Stauffer, R.E. Relationships between phosphorus loading and trophic state in calcareous lakes of southwest Wisconsin. Limnol. Oceanogr. 30:123-145.

Strickland, J.D.H., and T.R. Parsons. 1972. A practical manual of seawater analysis. Bull. Fish. Res. Board Can. 167p.

Stross, R.G. and A.D. Hasler. 1960. Some lime-induced changes in lake metabolism. Limnol. Oceanogr. 5:265-272.

Stumm, W., and J.O. Leckie. 1970. Phosphate exchange with sediments; its role in the productivity of surface waters. Proc. fifth international water pollution research conference. Pergamon Press.

Stumm, W. and J.J. Morgan. 1981. Aquatic Chemistry. Wiley-Interscience. 780p.

Taub, F.B., and A.c. Kindig. 1984. Can copper cause algal blooms? Abstract of paper for the 47th annual meeting of the American Society of Limnology and Oceanography

Tipping, E. 1981. The adsorption of aquatic humic substances by iron oxides. Geochim. Cosmochim. Acta. 45:191-199.

Van Liere, L., and L. R. Mur. 1982. The influence of simulated groundwater movement on the continuous flow system. Hydrobiologia 92:511-518.

Vogel, J.L. and J. Ennever. 1971. The role of lipoprotein in the intracellular hydroxyapatite formation in Bacterionema Matruchotii. Clin. Orthopedics. 78:218-222.

Waltor, L.M., and J. S. Hanor. 1979. Effect of orthophosphate on the dissolution kinetics of biogenic magnesium calcites. Geochim. Cosmochim. Acta 43:1377-1385.

Wetzel, R.G. 1975. Limnology. W.B. Saunders Co. 743p.

Wetzel, R.G. and B.A. Manny. 1978. Postglacial rates of sedimentation, nutrient and fossil pigment deposition in a hardwater marl lake of Michigan. Pol. Arch. Hydrobiol. 25:453-469.

White, W.S. and R.G. Wetzel. 1975. Nitrogen, phosphorus, particulate and colloidal carbon content of sedimenting seston of a hard-water lake. Int. Ver. Theor. Angew. Limnol. Verh. 19:330-339.

Williams, J.D.H., J.K. Syers, R.F. Harris, and D.E. Armstrong. 1971. Fractionation of inorganic phosphate in calcareous lake sediments. Soil. Sci. Amer. Proc. 35:250-255.

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.

Wilson, K.M. 1979. Diagensis of phosphorus in carbonate sediments from Bermuda. Msc. thesis. Univ. of New Hampshire.

Witteman, J.P. 1976. The computer simulation of phosphate removal from wastewater using lime. MSc. thesis. McMaster Univ. 120 p.

Wright, K.J. 1984. Lethal concentrations of reagent grade and limemud calcium oxide for rainbow trout (Salmo gairdneri) in Sooke Lake and Dragon Lake waters. B.C. Envir. File 50:60:06.

Wodka, M.C., S.W. Effler, and C.T. Driscoll. 1985. Phosphorus deposition from the epilimnion of Onondaga Lake. Limnol. Oceanogr. in press.

Wood, T., F.H. Bormann, and G.K. Voigt. 1984. Phosphorus cycling in a northern hardwood forest: biological and chemical control. Science
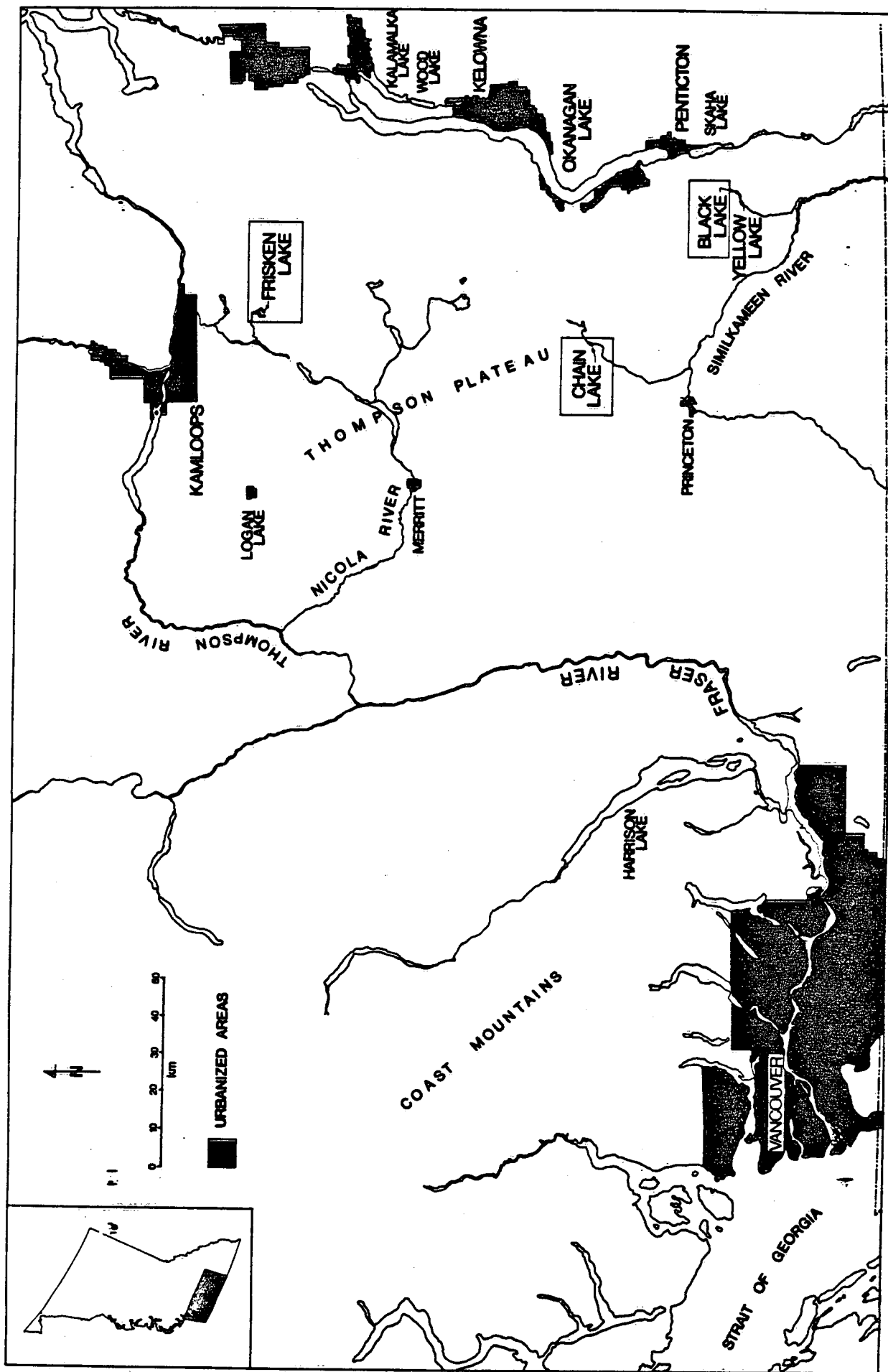
Figure 1 Map of southern British Columbia.

WILD HORSE

MOUNTAIN
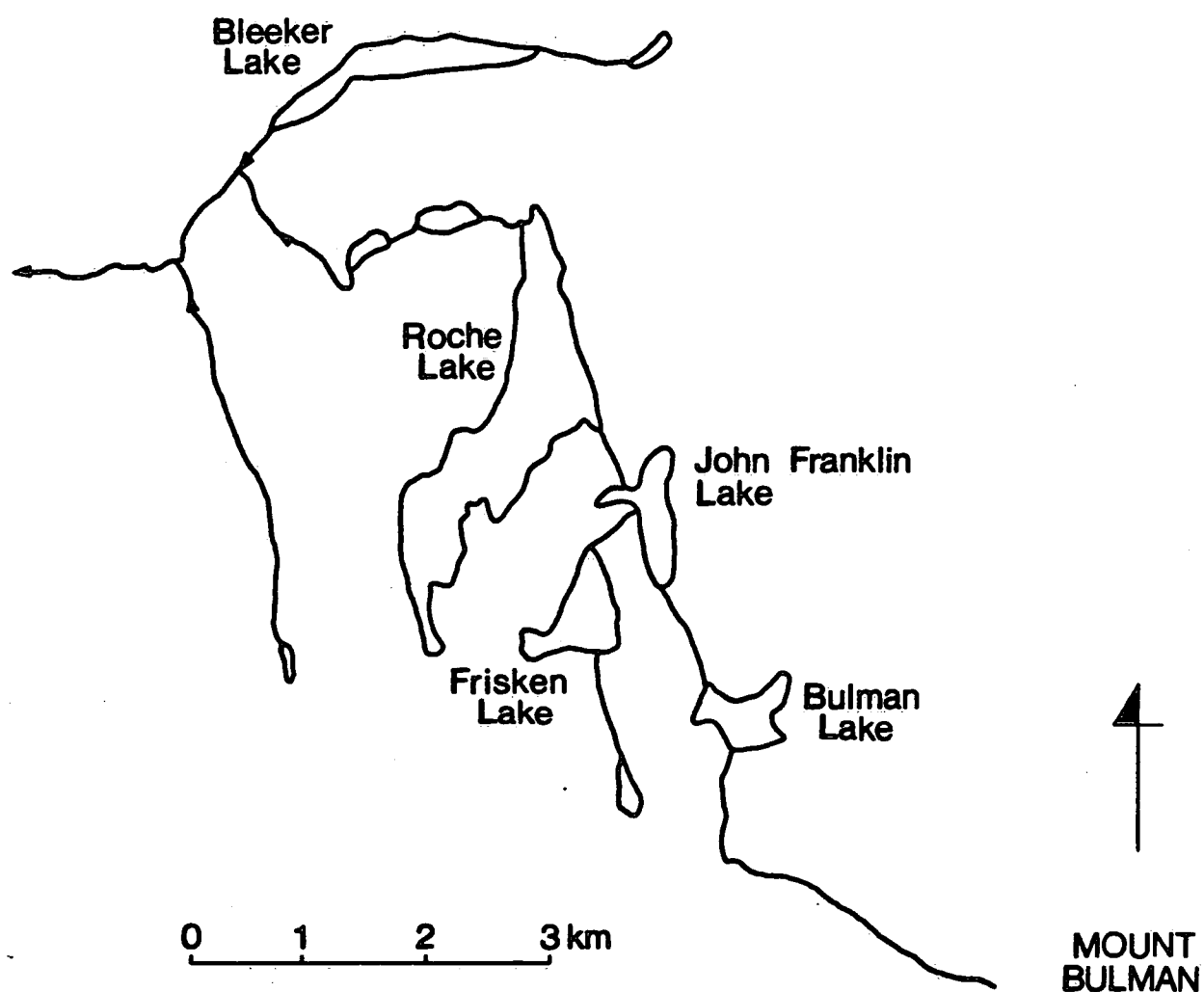
Figure 2 Map of the Wild Horse Mountain Batholith.
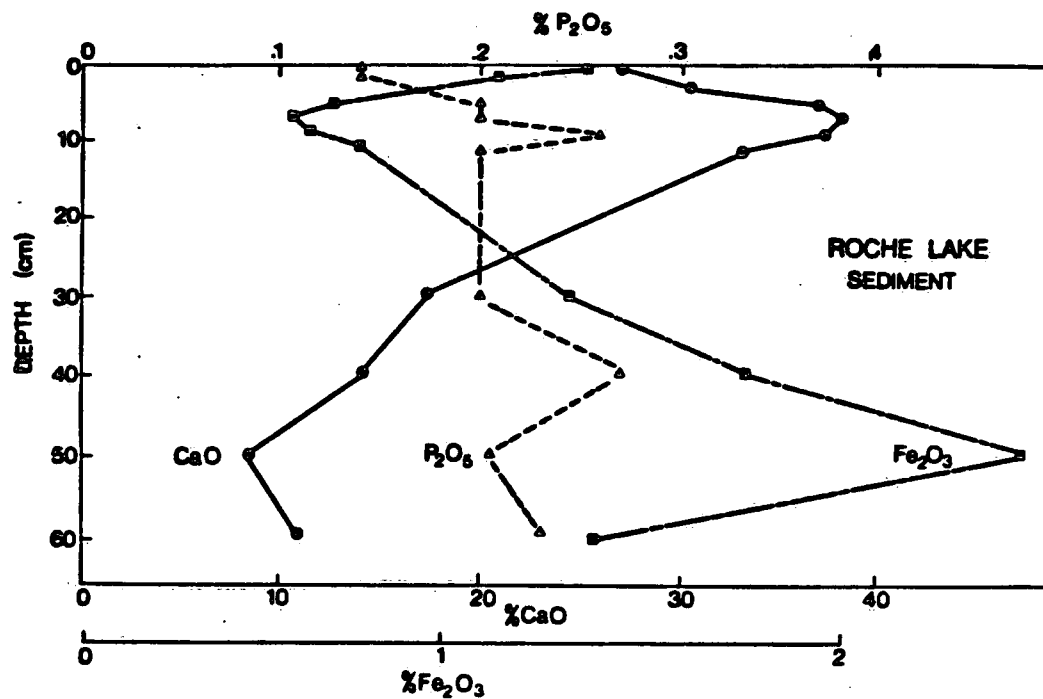
Figure 3 Roche Lake sediment chemistry.
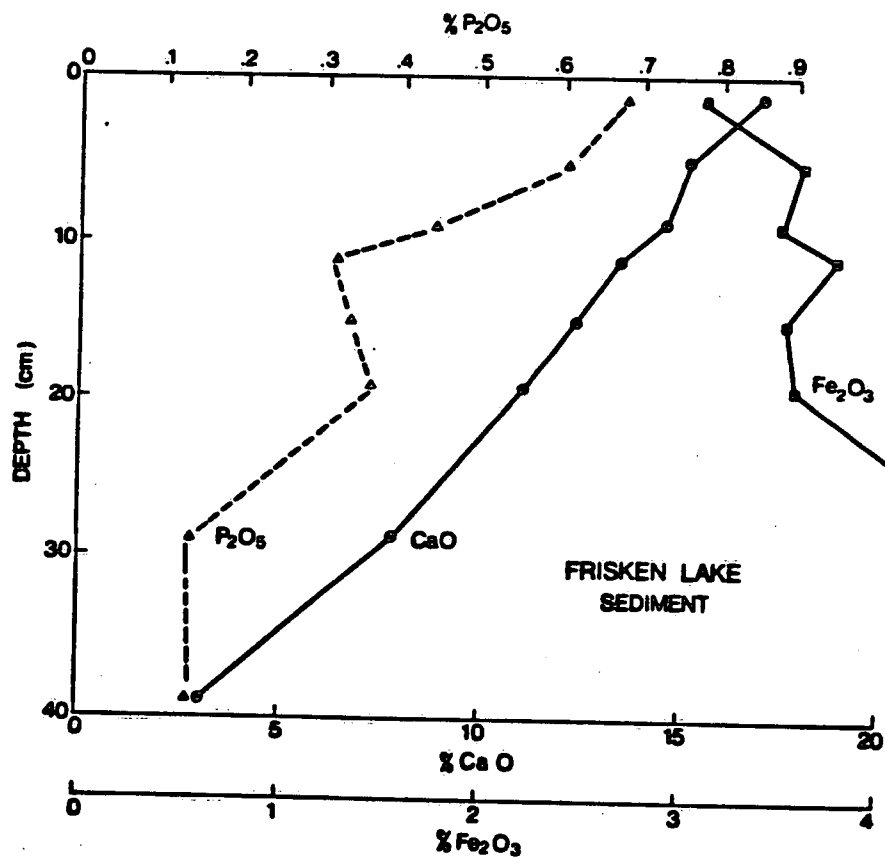
Figure 4 Frisken Lake sediment chemistry.
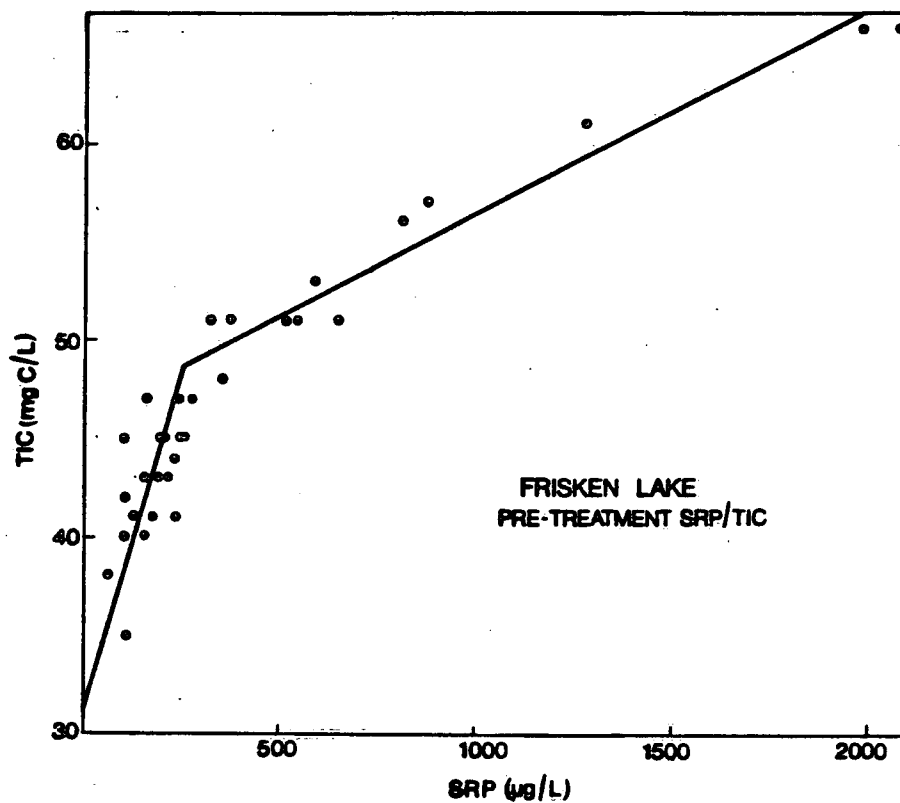
Figure 5   Frisken Lake pretreatment SRP/TIC.
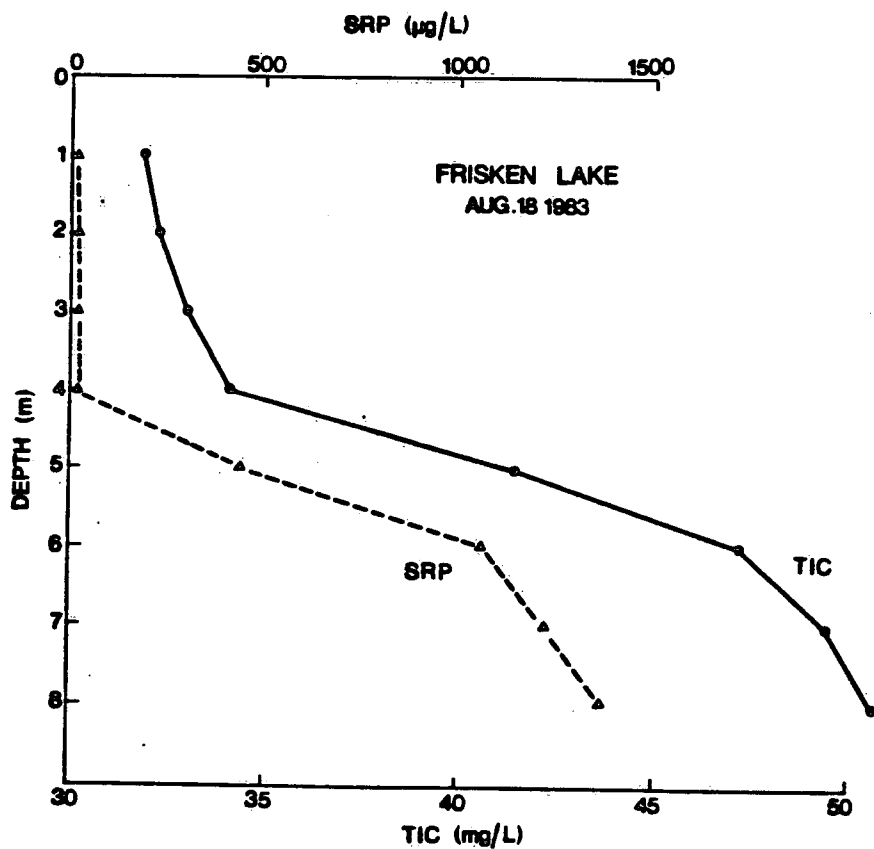
Figure 6   A depth distribution of SRP and TIC in Frisken
          Lake.
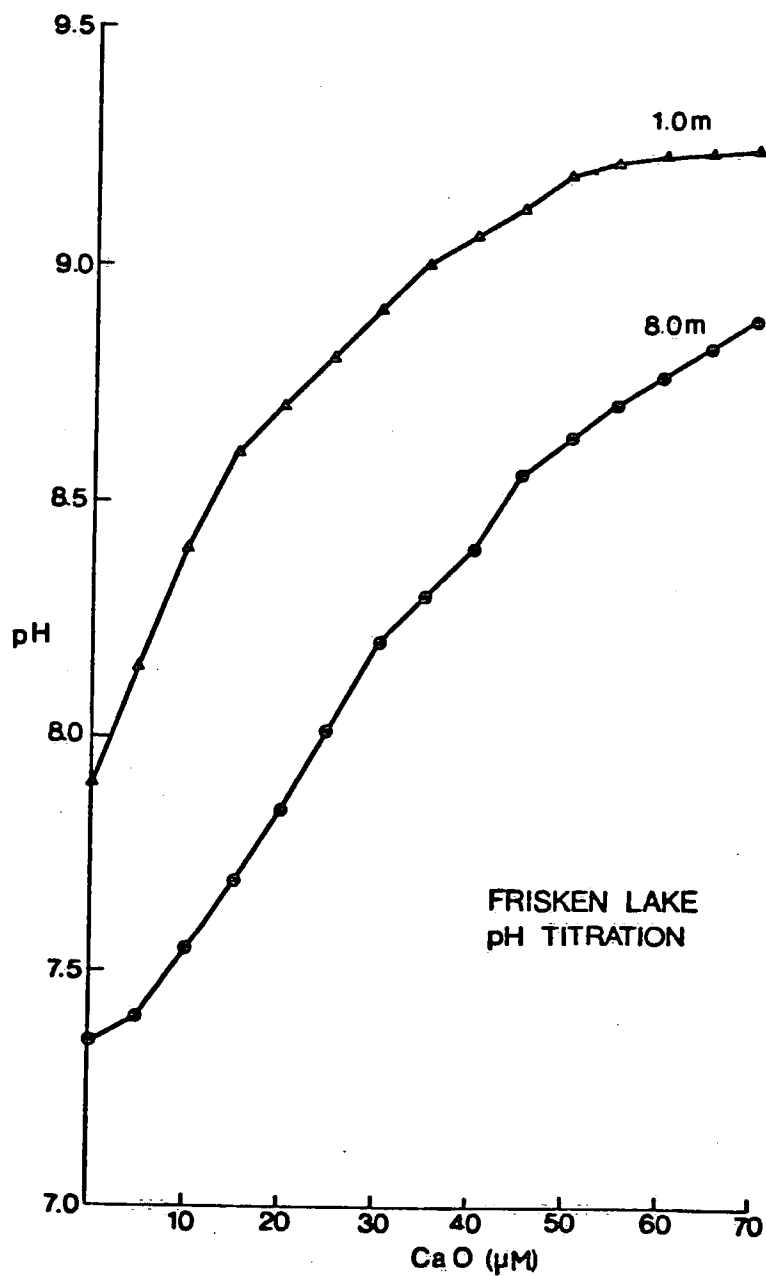
Figure 7 Titration curves of epilimnetic and hypolimnetic
Frisken Lake water.
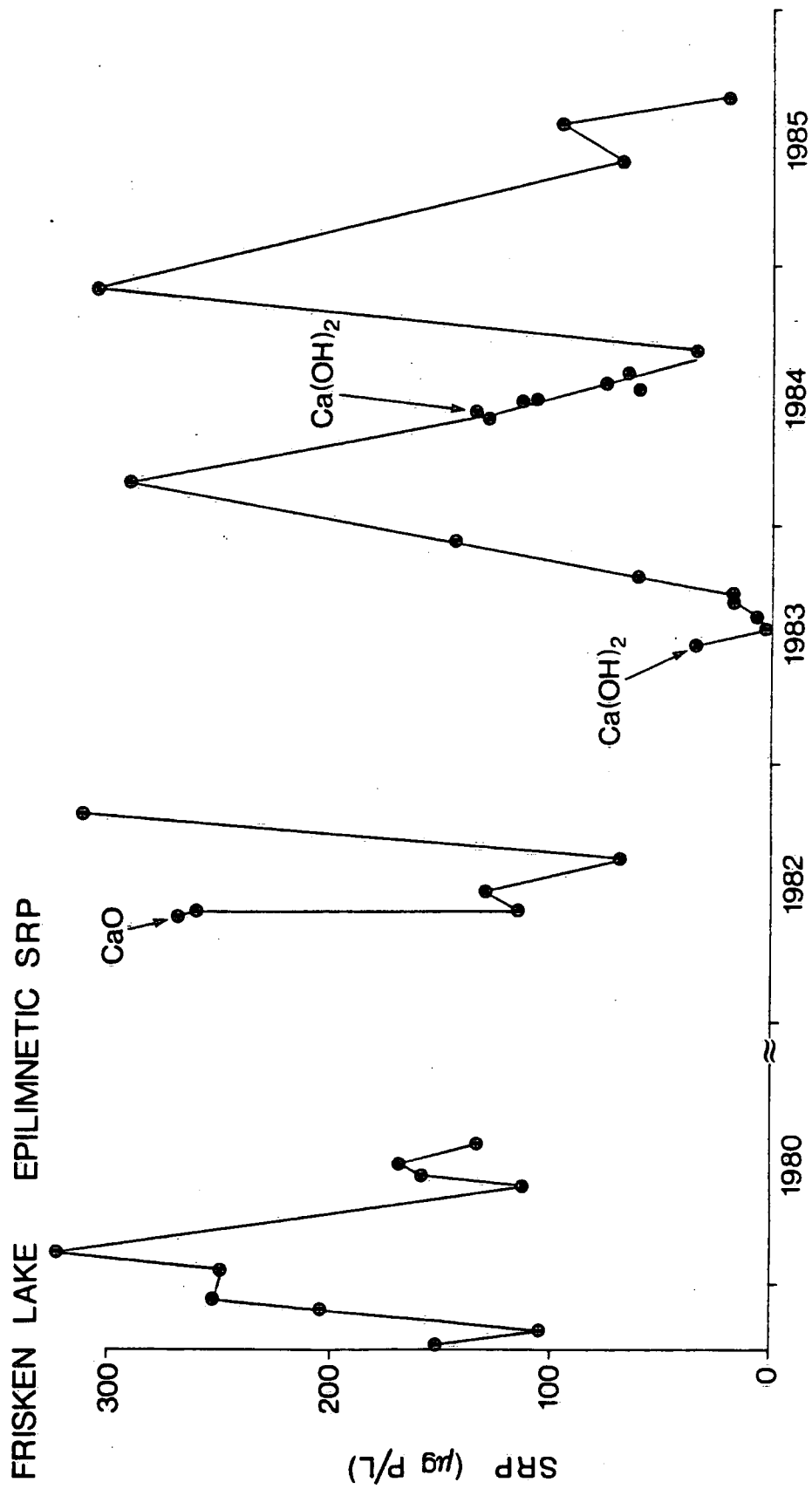
Figure 8 Frisken Lake epilimnetic SRP.

FRISKEN LAKE SECCHI DEPTH 1983

Lime Addition
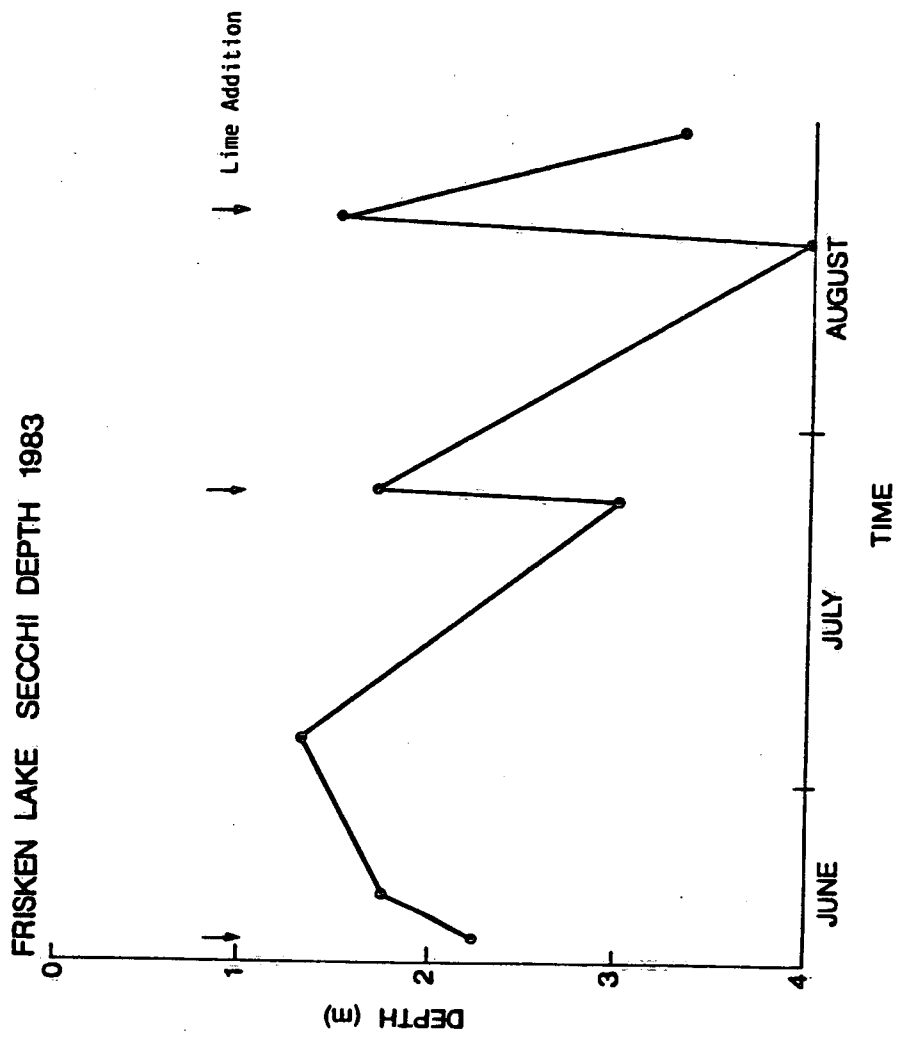
Figure 9   Secchi Disc Visibility in Frisken Lake – 1983

ΔpH JULY 25-27 1983

FRISKEN LAKE

pH   ΔpH

pH JULY 27 1983
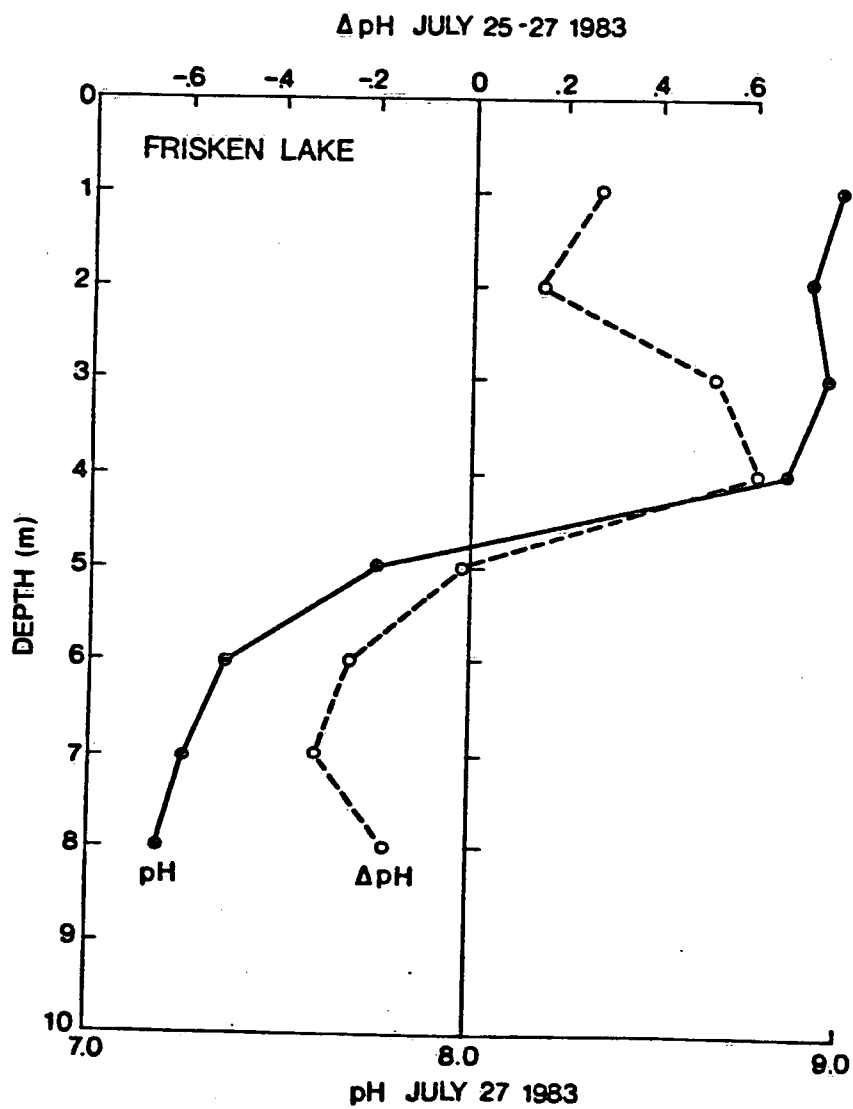
·Figure 10   Biotic modification to lime titration.
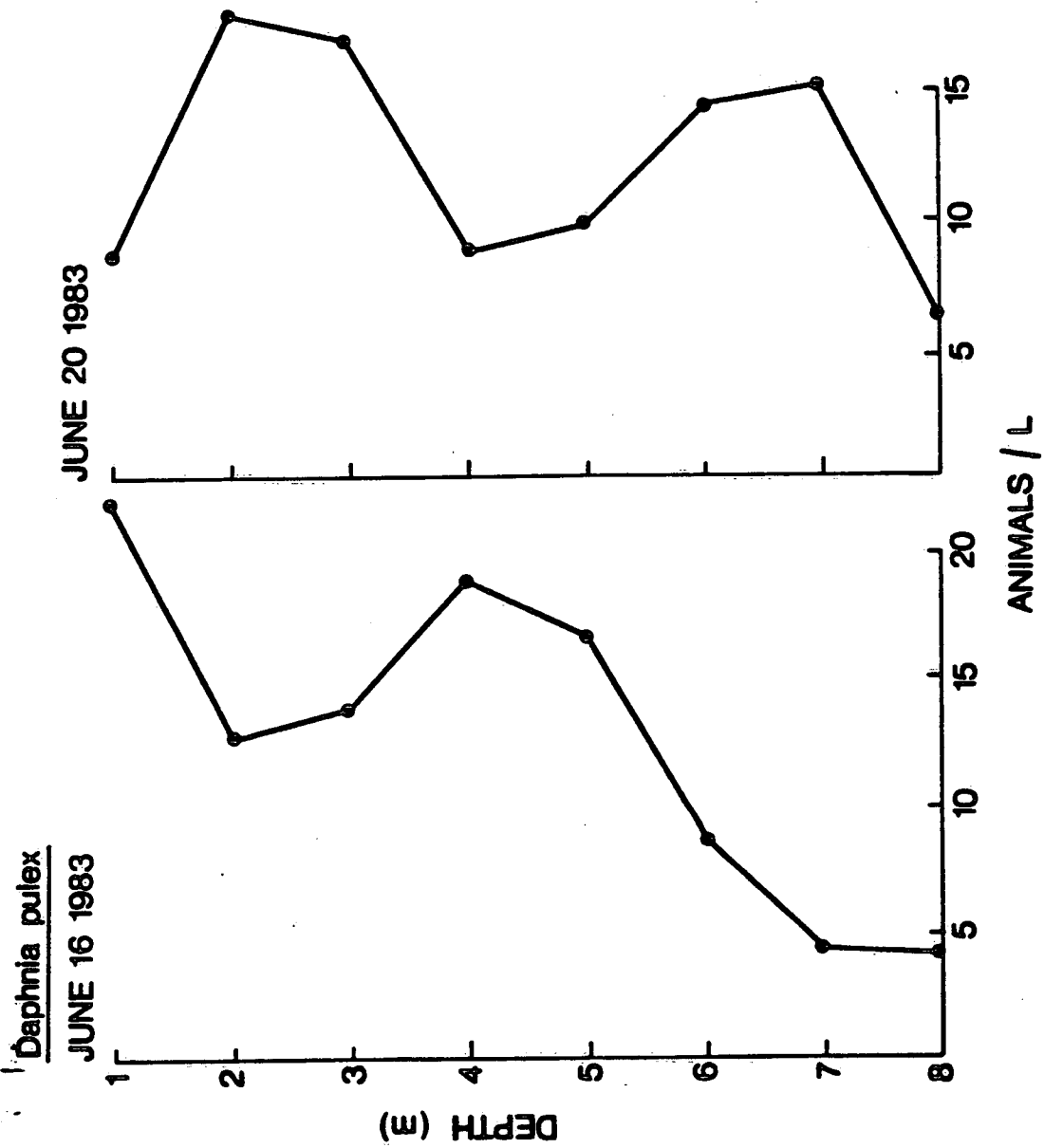
Figure 11. _Daphnia pulex_ Response to Lime Treatment.
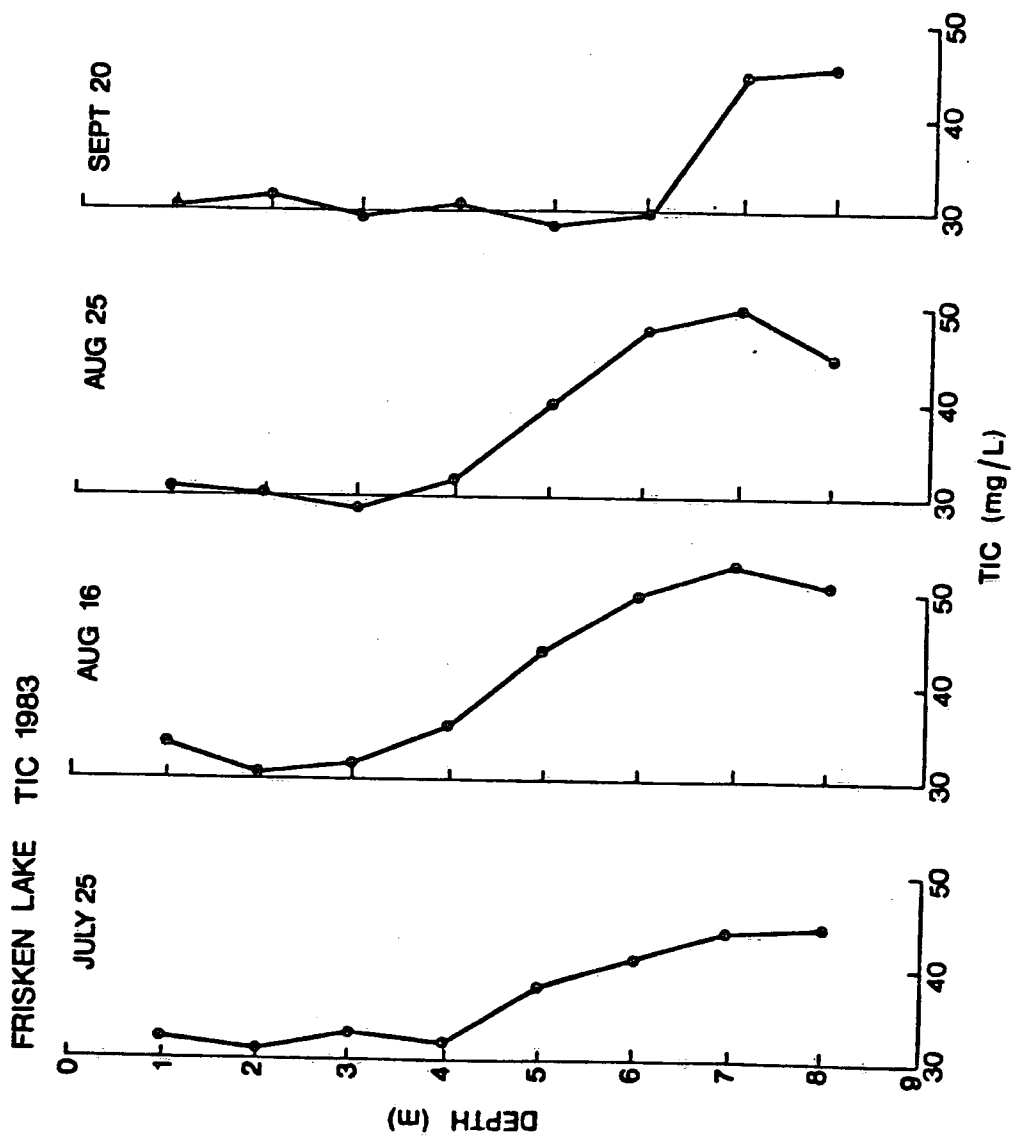
Figure 12 Frisken Lake Total Inorganic Carbon (TIC) – 1983.
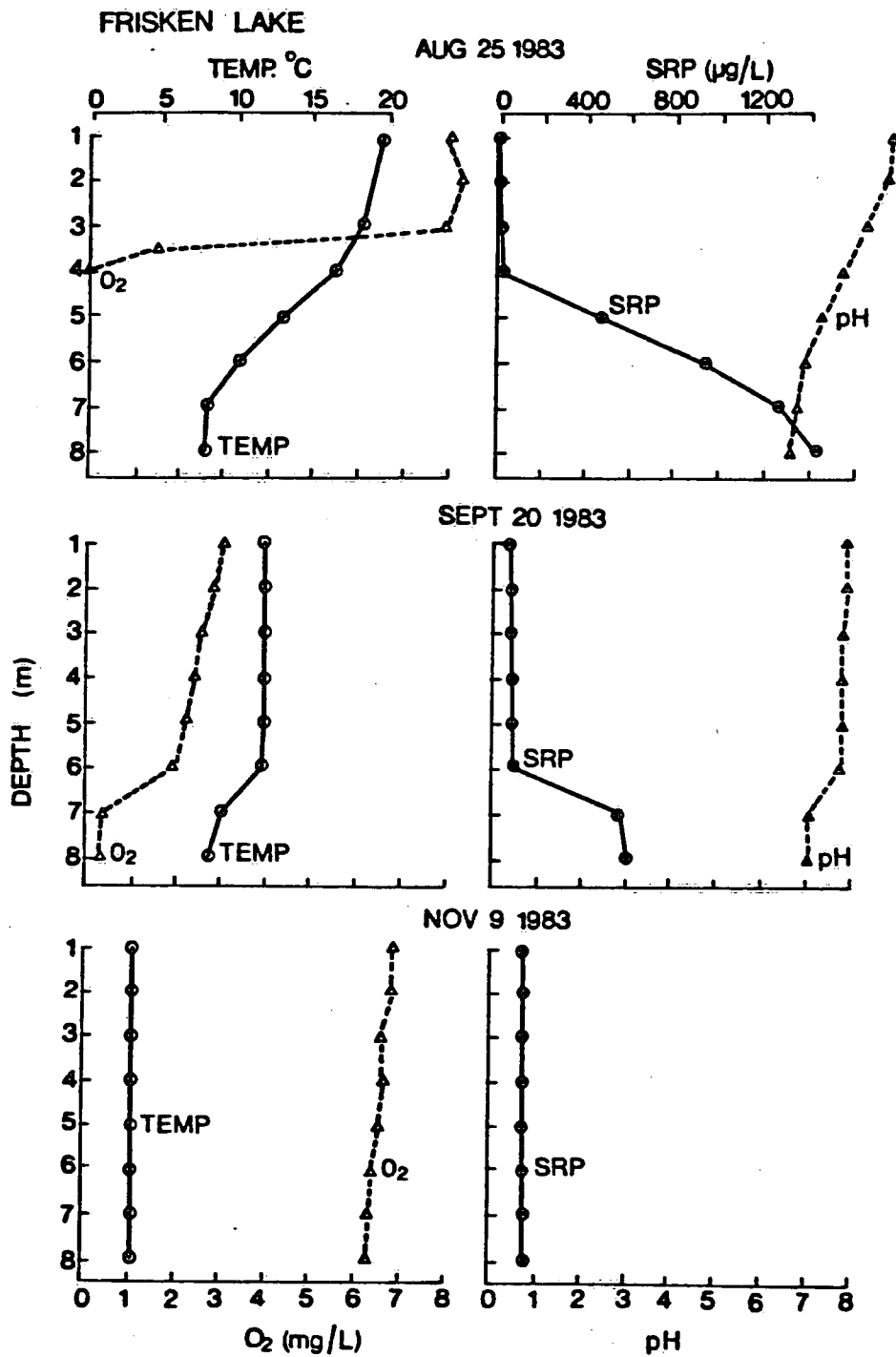
Figure 13 Frisken Lake Fall Destratification
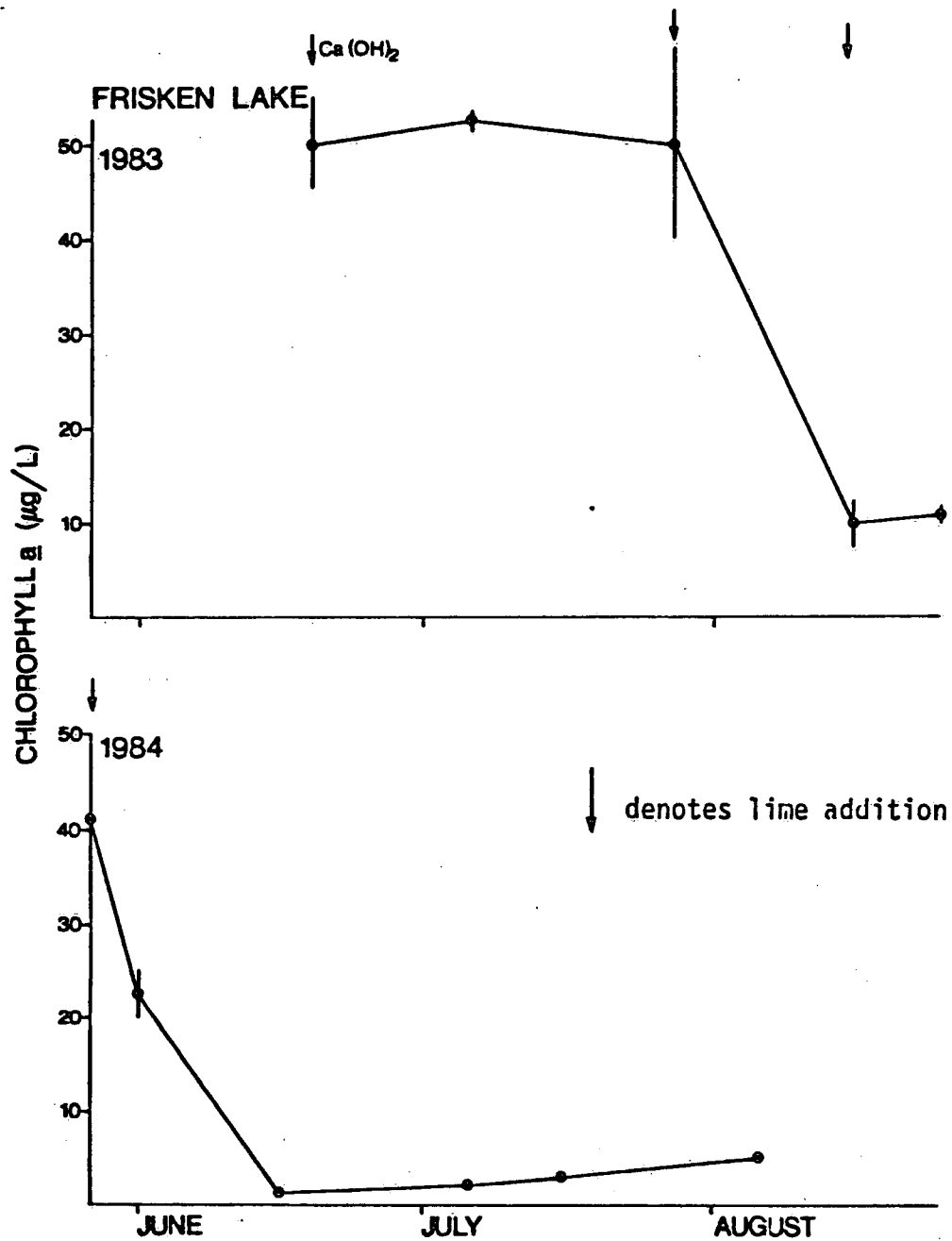
Figure 14 Suppression of algal biomass by lime application.

TABLE 1. Major Ions: Lakes, Streams in the Frisken Lake Area [1]

Depth	TDN	DON	DOC	DIC	TIC	NO3	NO2	NH3	PON	POC	Ca	Mg	K	Na	Cl	SO4	SRP	TP
Frisken Lake Water Chemistry November 3, 1983																		
1 m	2.04	1.27	19	37	40	0.076	0.007	0.688	0.297	1.26	50	11	2.0	7.7	1.0	0.5	0.311	0.395
5 m	1.94	1.21	20	30	40	0.071	0.056	0.60	0.378	1.83	50	11	2.0	7.6	1.0	0.5	0.332	0.415
8 m	1.79	1.04	19		41	0.077	0.002	0.875	0.283	1.29	29	11	2.0	7.7	1.0	0.5	0.305	0.383
Roche Lake Water Chemistry November 3, 1983																		
2 m	1.00	0.86	18	38	40	0.016	0.007	0.127	0.042	0.315	38	15	3.4	1.0	1.4	3.8	0.023	0.018
10 m	0.968	0.84	19	37	40	0.007	0.002	0.124	0.076	0.295	38	15	3.4	1.0	1.4	3.5	0.015	0.018
18 m	0.926	0.82	17	38	41	0.007	0.002	0.105	0.039	0.280	33	15	3.4	1.0	1.5	3.5	0.013	0.021
Frisken Lake Main Inlet April 28, 1982 (1), July 20, 1982 (2) and June 25, 1981 (3)																		
0 m		0.71			36		0.004				35	5.2	2.6	6.2	1.3	0.5	0.016	0.065
0 m		0.63	29		48	0.008	0.002	0.018	0.046	0.453	32	28	1.2	6.3	0.9	0.7	0.039	0.054
0 m		0.78	31		87		0.002	0.016	0.059	0.598	85	28	6.1	13	2.0	2.9	0.301	0.310
Frisken Lake Secondary Inlet April 28, 1982																		
0 m		0.95			38	0.004	0.008	0.014			45	5.2	3.7	13	4.5	4.4	0.280	0.345
Outflow from Provos Lake April 28, 1982																		
0 m		0.81			54	0.005	0.003	0.027			61	5.2	6.4	21	6.7	18	0.070	0.140
Main Inlet into John Franklin Lake April 28, 1982*																		
0 m		0.65			19	0.009	0.002	0.032			21	5.1	1.7	3.9	0.9	0.7	0.005	0.038
Shumway Lake April 28, 1982																		
0 m		1.39			96	0.150	0.014	0.185	1.03	3.89	41	5.3	16	14	14	95	0.007	0.180

SRP analysis was done the same day as samples were collected. Other analyses were done at the Inland Waters Directorate Laboratory in North Vancouver.

*John Franklin is a small lake upstream of Roche Lake (see Figure 2).

[1]All values as mg/L.

Table 2   Sediment Chemistry of Frisken and Roche Lakes

A) FRISKEN

Depth	SiO_2	Al_2O_3	Fe_2O_3	MgO	CaO	Na_2O	K_2O	TiO_2	MnO	P_2O_5	$CaCO_3$	Org-C	Nit
0-2	41.10	2.53	3.13	0.68	17.07	0.36	0.38	0.20	0.18	0.68	31.7	23.8	2.16
4-6	44.60	3.22	3.49	0.79	15.23	0.50	0.51	0.25	0.18	0.61			
8-10	47.86	3.99	3.67	0.86	14.70	0.55	0.62	0.31	0.17	0.44			
10-12	50.71	4.29	3.79	0.96	13.53	0.58	0.68	0.34	0.18	0.31	32.75	17.62	1.71
14-16	54.60	4.64	3.52	0.94	12.39	0.68	0.71	0.37	0.18	0.36			
18-20	56.08	4.75	3.58	0.98	11.13	0.62	0.72	0.38	0.18	0.36			
28-30	58.35	4.54	4.67	0.91	7.99	0.59	0.74	0.41	0.20	0.14	6.92	23.72	2.45
38-40	59.42	3.13	4.16	0.61	3.12	0.45	0.41	0.33	0.17	0.14			

B) ROCHE

Depth	SiO_2	Al_2O_3	Fe_2O_3	MgO	CaO	Na_2O	K_2O	TiO_2	MnO	P_2O_5	$CaCO_3$	Org-C	Nit
0-2	48.88	1.22	1.37	0.66	27.01	0.36	0.08	0.07	0.10	0.14	43.0	12.74	0.97
2-4	48.84	1.12	1.09	0.63	30.50	0.38	0.08	0.06	0.11	0.14			
4-6	39.79	1.00	0.71	0.86	36.86	0.21	0.07	0.05	0.09	0.20			
6-8	38.61	0.89	0.62	0.58	38.14	0.29	0.07	0.04	0.09	0.20			
8-10	39.57	1.22	0.65	0.75	37.28	0.35	0.07	0.05	0.08	0.27			
10-12	42.13	0.95	0.79	0.68	33.89	0.39	0.07	0.05	0.08	0.20			
28-30	60.62	1.04	1.31	0.50	17.43	0.27	0.06	0.08	0.10	0.20	47.75	11.9	0.97
26-40	58.48	1.24	1.76	0.54	15.06	0.29	0.08	0.10	0.11	0.27			
48-50	54.80	1.42	3.02	0.49	8.42	0.26	0.09	0.14	0.14	0.21			
58-60	66.55	0.99	1.37	0.39	10.38	0.21	0.05	0.08	0.08	0.23	25.0	16.4	1.42

All values are expressed as % dry weight.
The depth of sample is in cm.
This XRF (X-Ray fluorescence) analysis reports the total concentration as if the element was present as a simple oxide. Please note that the total concentration here can not equal 100%. We do not know the exact composition of the individual minerals (ie. we know that much of the aluminum, silicon, and iron are part of the same clay minerals, but we don't know the exact composition of each mineral).
The calcium carbonate concentration was determined by comparing the carbon content of an acid treated sample to a control sample. The organic carbon (Org-C) and nitrogen (Nit) values were obtained by chromatographic analysis of the gases generated by combusted samples. The presence of calcite (calcium carbonate) was confirmed by XRD (X-Ray diffraction) analysis.

Table 3 . Sediment P and Ca Extraction[#]

Depth (cm)	P (μg/gm)	Ca (mg/gm)	% P*	% Ca*
Frisken Lake				
0 - 2	971	94	28.8	77.1
0 - 2	991	101	29.4	82.8
4 - 6	927	83.6	34.8	76.8
10 - 12	655	80	48.4	82.7
38 - 40	364	10.4	59.6	44.6
Roche Lake				
0 - 2	373	191	61	99.2
0 - 2	372	184	60.8	95.6
4 - 6	374	285	42.8	108.3
10 - 12	429	237	49.4	99.7
28 - 30	869	101	99.5	81.0

* % P and % Ca are the percent extracted of the total ion as measured by X-ray fluorescence.

0.1 N HCl extract.

Table 4. Chlorophyll a # in Frisken Lake - 1983

Depth (m)	June 16	June 18	July 5	July 25	July 27	Aug. 16	Aug. 18	Aug. 25	Sept. 20*
1	24±5	50±9	52±1	41±5	58±15	10±1	13±1	12±1	40
2						5±4	17±8	13±3	38
3	14±6	5	17±6			2	15±4	11±1	34
4						24±16	15±1	21±6	32
5	55±32	35±2	16±5	4±2	3±2	12±8	17±2	11±6	33
6						21±16	5±2	13±12	31
7						4±3	8±1	17±12	13
8	35±29	18±10	9	3	3	3±3	13±6	6±1	72

* These samples were from the aerator station. All other samples were the mean ±1 standard deviation of the two sampling stations.

(μg/L)

Table 5. Calcium# in Frisken Lake - 1983

Depth (m)	June 16	June 18	July 5	July 25	July 27	Aug. 16	Aug. 18	Aug. 25	Sept. 20*	Nov. 9*
1	36.6±1	42.9±2.2	41.7±0.5	32.4±0.1	34.7	37.7±2.7	35.9	37.5±0.7	38.5	46.7
2						38.3±1.7	6.6	35.5±0.2	39.3	44
3	36.9±1.1	42.3±0.5	41.3±0.2			37.8	36.8±0.5	38.0±0.5		44
4						41.1±1.5	38.7±0.6	40.8±.8	39.8	44
5	44.1±1.6	47.8±0.2	48.8±0.1	41.8±0.2	41.1	45.3±0.4	48.5±0.1	46.7±0.2	41.1	44.9
6						50.9±1.2	51.7±0.5	49.7±0.4	.	42.2
7						51.8±0.6	52.0±0.2	53.8±0.2	56.3	43.4
8	45.4±3.2	48.6±0.8	52.8±2.5	46.9±3.6	46.5	52.7±0.8	54.1±1.1	54.2	77.2	48.5

* These samples were from the aerator station. All other samples were the mean ±1 standard deviation of the two sampling stations.

(mg/L)

TABLE 6.   pH in Frisken Lake - 1983

Depth (m)	June 8 1982	June 10 1982	July 17 1982	June 16 1983*	June 18 1983*	July 5 1983*	July 25 1983*	July 27 1983*	Aug. 16 1983*	Aug. 18 1983*	Aug. 25 1983*	Sept. 20 1983
1	8.4	8.3	8.2	8.8	8.8	8.7	8.75±0.05	9.0	8.65±0.05	8.65±0.05	8.85±0.05	7.9
2							8.8	8.9	8.65±0.05	8.7±0.1	8.8	7.9
3	8.3	8.3		8.4	9.05±0.15	8.7	8.7±0.2	8.95±0.05	8.5±0.1	8.55±0.05	8.4±0.02	7.8
4							8.35±0.15	8.9	8.15±0.05	8.5±0.1	7.65±0.15	7.8
5	7.8	7.6	7.4	7.9±0.1	8.3±0.1	7.25±0.05	7.8	7.75±0.05	7.2	8.2±0.2	7.35±0.35	7.8
6							7.65±0.05	7.35±0.05	6.85±0.05	6.9	6.85±0.05	7.8
7	7.0	7.2					7.45±0.05	7.25±0.05	6.8	6.8	6.7	7.1
8				7.45±0.05	7.6±0.1	6.95±0.05	7.35±0.05	7.2±0.1	6.7±0.1	6.7	6.6	7.1
9	6.9	7.0	7.3									

* Values for these dates are the mean ±1 standard deviation of the two sampling stations. Other sampling dates are for the aerator station.

TABLE 7.   SRP* in Frisken Lake 1982-83

Depth (m)	June 8 1982	June 10 1982	July 17 1982	Oct. 31 1982	June 16 1983	June 18 1983	July 5 1983	July 25 1983	July 27 1983	Aug. 16 1983	Aug. 18 1983	Sept. 20 1983	Nov. 9 1983
1	268	260	130	311	37±9	74±50	6.5±0.5	12	22±16	15±3	18±2	82	167
2								9±1	10±2	16±2	23±2	102	170
3	276	276			57±9	88±58	23±10	22±14	10	17±7	31±16	94	165
4								68±31	48±1	23±7	43±3	98	172
5	337	340	387	332	465±9	492±29	412±46	560±23	504±74	445±25	483±81	98	165
6								714±30	641±6	1067±27	930±6	104	181
7	664	651						855±3	962±12	1232±7	1293±3	594	-
8					838±2	930±70	917±260	962±62	981±111	1378±29	1410±18	606	186
9	920	889	808	305									

± Values for these dates are the mean ±1 standard deviation of the two sampling stations. Other sampling dates are for the aerator station.

* (ug P/L)

Table 8. Total Phosphorus# in Frisken Lake - 1983.

Depth (m)	June 16*	June 18*	July 5*	July 25	July 27	Aug. 16*	Aug. 18*	Aug. 25*	Sept. 20	Nov. 9
1	219±103	183±130	75±11	40	38	36	51±19	47±13	103	219
2				44		36±10	60±10	67±11	111	248
3	249±125	132±71	59±4	62	48	55±24	64±15	49±2	112	232
4				74	68	104±39	69±20	102±33	72	225
5	596±5	696±32	511±90	582	530	433±99	380±12	537±100	164	212
6				744	660	877±20	982±75	1001±36	147	219
7				930	1020	1038±41	1080±2	1393±110	1291	
8	1112±90	1192±9	1362±162	980	1200	1401±45	1278±35	1474	1508	219

* Values for these dates are the mean ±1 standard deviation of the two sampling stations. Other sampling dates are for the aerator station.

(μg P/L)

Table 9. SODIUM[#] in Frisken Lake - 1983

Depth (m)	June 16	June 18	July 25	July 27	Aug. 16	Aug. 18	Aug. 25	Sept. 20	Nov. 9
1	9.0±0.1	9.7±0.4	5.4±0.3	5.3	6.3±0.4	6.1	7.4±0.1	7.4	9.4
2					6.2±0.1	6.3±0.3	7.7±0.1		9.3
3	9.0±0.1	8.7±0.4		5.2	6.3±0.2	5.9±0.1	7.7±0.1		9.3
4					6.4±0.3	5.9	7.7±0.3		9.4
5	9.8±0.2	10.2±0.1	5.5		6.1±0.1	6.5±0.2	7.9±0.1	7.3	9.6
6					7.1±0.3	7.5±0.5	8.0±0.2		9.3
7					6.8±0.2	7.3	8.6±1.0		
8	9.9±0.1	10.±1.0	6.6	5.8	7.0±0.2	7.3	8.6	7.7	9.5

* These samples were from the aeration station. All other values are the mean ±1 standard deviation of the 2 stations.

(mg/L)

TABLE 10 Potassium # in Frisken Lake 1983

Depth (m)	June 16	July 25	July 27	Aug. 16	Aug. 18	Nov. 9
1	2.9 0.2	3.0 0.1	2.8	3.6 0.3	3.5 0.3	3.3
2				3.9 0.1	3.7 0.4	3.3
3	2.8 0.1		3.9	3.9 0.3	3.2 1	3.3
4				3.7 1	3.2 1	3.6
5	3.8 0.1	3.9 0.1		4.0 0.2	3.6 1	3.8
6				5.4 0.1	4.7 0.3	3.3
7				4.4 1	4.7 0.1	
8	4.2 0.1	4.7 0.3	4.6	4.5 0.1	4.9 0.1	3.3

When two stations were sapled the mean 1 standard deviation is displayed. Single values were from the aeration station.

(mg/L)

TABLE 11 Magnesium # in Frisken Lake 1983

Depth (m)	June 16	July 5	July 25	July 27	Aug. 16	Aug. 18	Nov. 9*
1	9.6	10 0.1	8.2 0.2	8.4	10.6 0.1	10.3 0.1	10.5
2					10.6	10.8 0.3	10.7
3	9.7 0.1	10.6 0.2			10.5 0.1	10.4 0.1	10.6
4					10.6 0.2	10.5 0.1	10.5
5	10.5 0.1	12.0 0.1	8.7 0.2	8.7	11.1 1.0	11.0 0.1	10.7
6					12.0 0.2	11.9 0.2	10.7
7					11.8 0.2	12.0	
8	10.9	12.7	9.9 0.2	8.9	11.9 0.1	12.1 0.1	11.0

* Aeration station, other data is mean 1 standard deviation of 2 stations

(mg/L)

TABLE 12.   Temperature in Frisken Lake 1982-83

Depth (m)	June 8 1982	June 10 1982	July 16 1982	Oct. 31 1982	June 16 1983	June 18 1983	June 20 1983	July 5 1983	Aug. 18 1983	Aug. 25 1983	Sept. 20 1983	Nov. 9 1983
1	14.0	14.8	16.0	5.5	15.5	14.5	14.5	16.0	20.0	19.5	12.0	3.3
2	13.5	13.5	16.0	5.5	15.2	14.5	14.3	16.0	19.5	19.0	12.0	3.3
3	11.5	12.2	15.5	5.5	14.5	14.2	14.3	14.5	19.0	18.5	12.0	3.3
4	8.0	8.5	12.5	5.5	12.0	13.6	12.8	12.0	17.5	16.5	12.0	3.3
5	5.5	5.5	9.0	5.5	7.5	7.3	7.4	9.0	13.5	12.8	12.0	3.2
6	4.9	4.8	7.0	5.5	6.0	5.9	6.0	6.8	9.5	9.9	11.9	3.2
7	4.5	4.5	6.0	5.4	5.0	5.1	5.0	5.8	8.5	8.0	9.1	3.2
8	4.5	4.5	5.0	5.3	4.9	5.0	4.9	5.2	8.5	7.8	8.1	3.2
9	4.4	4.5	5.0	5.3	4.9	4.9	4.9	5.0		7.8		3.1

Table 13.   TOTAL INORGANIC CARBON[#] IN FRISKEN LAKE, 1983

Depth (m)	July 25	Aug. 16	Aug. 18	Aug. 25	Sept. 20
1	32.3	33.0	31.9	30.5	30.6
2	30.7	30.5	32.3	29.7	31.6
3	32.9	31.4	33.0	28.5	29.3
4	31.8	35.5	34.2	31.1	30.5
5	37.9	43.6	41.4	39.7	28.9
6	40.5	49.8	47.2	47.7	29.6
7	43.5	52.7	49.4	49.7	44.5
8	43.7	50.1	50.6	44.7	45.6

[#] (mg/L)

TABLE 14. Oxygen # in Frisken Lake

Depth (m)	June 8 1982	June 10 1982	July 16 1982	Oct. 31 1982	June 16 1983	June 18 1983	June 20 1983	July 5 1983	Aug. 18 1983	Aug. 25 1983	Sept. 20 1983	Nov. 9 1983
1	15.2	16.0	5.9	4.7	11.2	11.0	10.2	12.0	7.5	8.1	3.4	7.2
2	11.8	15.0	5.8	4.5	10.9	10.8	10.2	11.9	7.3	8.0	3.1	7.0
3	16.0	14.8	5.6	4.4	6.7	10.2	9.8	11.8	7.6	8.3	2.9	6.9
4	20.0	18.6	5.4	4.3	1.0	9.7	9.8	5.0	4.1	8.0	2.6	6.7
5	3.0	0.6	1.3	4.3	0.5	1.6	0.8	0.4	0.2	0.1	2.5	6.6
6	0.6	0.6	0.4	4.3	0.5	0.3	0.4	0.4	0.1	0.1	2.3	6.4
7	0.6	0.6	0.4	4.3	0.4	0.3	0.4	0.4	0.0	0.0	2.0	6.4
8	0.6	0.6	0.4	4.3	0.4	0.3	0.3	0.4	0.0	0.0	0.4	6.3
9	0.6	0.6	0.4	4.2	0.4	0.2	0.3	0.4	0.0	0.0	0.4	6.0

(mg/L)

Table 15. Frisken Lake Total Iron (µg Fe/L) – 1983

Depth (m)	June 16	July 23	July 27	Aug. 18	Aug. 25	Sept. 20
1	27±4	18±1	26±5	24	36	83
2				22	31	137
3	18±2			34	30	134
4				32±2	35	129
5	68±3	39±10	45±12	131	126	43
6				559	536	80
7				791	923	39
8	94±11	226±102	219±245	1065±224	1253	444

When two stations were sampled the mean ± 1 standard deviation is displayed.

Single values were from the aerator station.

TABLE 16. SRP in Frisken Lake – 1984*

Depth	May 24	May 25	May 26	May 29	May 31	June 15	July 5	July 17	Aug. 5
1	125	152	108	97	97	68	98	72	36
2	135	170	116	99	99	102	100	86	34
3	135	177	124	99	102	104	102	104	60
4	135	177	130	101	116	130	150	154	221
5	143	202	148	132	120	150	238	300	500
6	162	265		165	156	264	422	490	768
7	243	453	292	298	356	338	480	680	885
8	291	718	419	414	386	446	560	610	966
Time	1:30	6:00	11:00	6:00	6:00	17:00	15:00	18:00	10:00

Table 17

TOTAL PHOSPHORUS IN FRISKEN LAKE 1984*

Depth	July 17	August 5
1	100	80
2	106	77
3	118	90
4	165	208
5	290	440
6	465	590
7	630	830
8	800	910

*µg/L

Table 18

SPECIES: <u>Daphnia pulex</u>   Response to Lime Treatment   1984

Date

Depth (m)	May 24* 6:00	11:00	1:30	19:00	May 25* 6:00	11:00	19:30	May 29# 6:00	11:00	19:00	May 31# 6:00	11:00	18:00	June 15# 18:00
1		11		45	3	18	8	10	25	85	30	44	57	550
2	5	18	92	24	22	21	8	20	2	7	44	20	65	255
3	0	4	30	50	12	10	0	9	10	1	81	50	15	73
4	2	5	18	18	18	44	0	6	20	1	10	5	5	315
5	0	9	20	10	18	10	6	7	10	2	8	0	0	235
6	4	9	10	6	12	8	4	2	0	0	0	5	0	0
7	2	0	0	0	0	2	0	0	0	0	0	0	0	0
8	1	0	0	0	1	2	1	0	0	2	0	1	0	0

*Pretreatment   animals/litre   #Posttreatment

Table 19

SPECIES: <u>Keratella quadrata</u>   Response to Lime Treatment   1984

Date

Depth (m)	May 24* 6:00	11:00	1:30	19:00	May 25* 6:00	11:00	19:30	May 29# 6:00	11:00	19:00	May 31# 6:00	11:00	18:00	June 15# 18:00
1	2500	1200		9900	3250	2700	4000	10950	7750	6000	6300	7600	6800	9750
2	1800	1625	2900	4500	4750	2550	2000	3500	1000	3375	5000	4000	6400	9625
3	125	200	1600	1500	2475	2800	750	750	2000	250	7050	1800	800	5750
4		1500	1650	1575	900	1600	500	0	1625	200	900	400	-	5500
5		96	1000	480	300	600	500	0	600	0	400	200	400	8750
6		54	225				0	0	0	0	200	100	-	1775
7	165	42	10	40	75	60	40	25	50	45	-	20	60	2500
8	225	62	60	25	50	105	35	5	100	40	60	128	10	1825

*Pretreatment   animals/litre   #Posttreatment

Table 20

SPECIES: <u>Polyarthra</u>   Response to Lime Treatment   1984

Depth (m)	**May 24***				**May 25***			**May 29#**			**May 31#**			**June 15#**
	6:00	11:00	1:30	19:00	6:00	11:00	19:30	6:00	11:00	19:00	6:00	11:00	18:00	18:00
1	9750	32000		27750	28750	12750	25000	4300	25750	5250	6900	6800	6200	750
2	24300	23500	20500	26400	32750	24300	26500	17000	26250	6125	8200	7200	10400	375
3	19375	45400	16900	25575	36550	21600	32000	33250	29000	20125	9900	11000	10200	4900
4	29500	18525	17925	36750	26550	19600	69750	26000	28250	10200	37350	20800	25600	1200
5	33500	19850	20650	22880	30600	33600	34750	46000	40700	21500	38600	18400	53600	10125
6	39750	8360	31200	38850	15075	19760	38750	30125	42750	92750	79600	91900	59000	25
7	136	686	22560	200	1035	19260	125	210	1250	175	1335	1040	0	100
8	63	204	836	50	48	945	50	25	160	175	264	68	300	25

*Pretreatment   animals/litre   #Posttreatment

Table 21

SPECIES: <u>Keratella cochlearis</u>   Response to Lime Treatment   1984

Depth (m)	**May 24***				**May 25***			**May 29#**			**May 31#**			**June 15#**
	6:00	11:00	1:30	19:00	6:00	11:00	19:30	6:00	11:00	19:00	6:00	11:00	18:00	18:00
1	181500	100200		227400	155250	92100	155750	108050	203500	372000	180000	192000	214800	24250
2	305700	68000	134300	156600	172500	91200	164000	213500	122500	271675	210600	207200	266600	38875
3	161875	138500	79900	111975	154800		140000	231250	135875	165875	225900	326800	244000	53700
4	189500	97650	105750	156975	141150	70300	199250	178875	109250	67600	283200	263200	146600	47500
5	290250	83500	87800	124800	159900	164800	190500	250500	231500	208600	163200	179800	188100	67375
6	304500	80760	94800	302100	308850	70560	255000	165250	103000	61375	86400	84800	4800	16800
7	1359	742	48720	370	4155	45240	685	530	3475	950	1035	2200		20400
8	1548	928	2258	305	686	2220	520	200	970	1935	2470	1665	1190	7500

*Pretreatment   animals/litre   #Posttreatment

Table 22

BACTERIA IN FRISKEN LAKE

Depth	BACTERIAL NUMBERS (10^6 cells.ml^{-1})				
	May 24	May 25	May 29	May 31	June 15
1m	2.44	3.00	2.40	2.63	2.18
2m	3.22	2.68	2.64	2.57	2.17
3m	2.76	2.88	2.27	2.66	1.77
4m	2.90	2.46	2.40	2.85	1.95
5m	2.80	4.18	2.05	2.31	1.59
6m	2.42	3.48	2.43	3.34	3.60
7m	2.59	2.84	3.33	3.58	4.57
8m	3.13	2.30	4.18	3.80	4.67
$\overline{X}$ (integral)	2.75	2.96	2.53	2.77	2.35

TABLE 23. pH in Frisken – 1984

Depth	May 25	May 29	May 31	June 15	July 5	July 17	Aug. 5
				Date			
1	8.6	8.3	9.1	7.9	7.6	8.8	9.0
2	7.4	9.0	9.1	8.75	9.0	8.8	9.0
3	7.3	7.7	9.0	8.8	8.85	8.6	8.8
4	6.9	8.0	9.0	8.8	8.55	8.3	8.8
5	7.0	8.0	8.9	8.7	7.8	8.0	8.6
6	8.0	7.3	8.8	8.3	7.65	7.7	8.4
7	7.3	7.2	8.2	8.3	7.3	7.6	8.3
8	7.0	7.1	8.05	8.1	7.1	7.5	8.25
Time	6:00	6:00	6:00	17:00	15:00	18:00	10:00

TABLE 24. Oxygen in Frisken Lake – 1984*

Depth (m)	Date				
	May 26	June 15	July 5	July 16	Aug. 5
S	15.9	10.8	7.2	10.2	9.4
1	16.1	10.7	7.2	10.8	10.2
2	14.9	10.7	6.5	8.1	7.7
3	13.2	10.7	6.3	5.1	3.9
4	10.2	7.9	2.2	1.3	1.3
5	8.2	3.8	0.4	0.6	1.0
6	1.8	0.5	0.4	0.5	1.0
7	0	0.3	0.4	0.4	1.0
8	0	0.2	0.4	0.4	1.0
9		0.2	0.4	0.3	1.0
Time	13:30	15:00	17:00	6:00	10:30

* mg/L

Table 25

TEMPERATURE IN FRISKEN LAKE 1984

Depth (m)	May 26	May 30	June 15	July 5	July 16	August 5
S	2.6	9.9	17.0	16.0	20.0	21.0
1	7.8	9.9	16.0	16.0	20.0	21.0
2	7.0	9.7	14.5	15.0	18.8	20.6
3	7.0	8.0	12.5	14.0	17.0	19.0
4	7.0	7.0	10.0	13.0	14.6	15.0
5	7.0	5.1	9.0	10.0	11.9	12.0
6	4.2	4.1	8.0	8.0	9.0	9.0
7	4.1	4.0	7.0	7.0	7.2	8.0
8	4.0	4.1	6.5	7.0	7.0	7.1
			6.5	6.5	6.8	7.0

TABLE 26. Total Iron* in Frisken Lake – 1984

Depth (m)	June 15	July 16	Aug. 5
1	15	18	41
2	19	24	41
3	25	17	48
4	16	23	49
5	18	38	52
6	65	84	233
7	186	104	311
8	185	194	625

*µg/L