# INTERIM REPORT ON THE ACIDIFICATION OF LAKE 223, EXPERIMENTAL LAKES AREA: BACKGROUND DATA, THE FIRST YEAR OF ACIDIFICATION (1976), AND PILOT EXPERIMENTS

by

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for

ALBERTA OIL SANDS ENVIRONMENTAL RESEARCH PROGRAM

Project AF 2.3.1

September 1979

The Hon. J.W. (Jack) Cookson
Minister of the Environment
222 Legislative Building
Edmonton, Alberta

and

The Hon. John Fraser
Minister of the Environment
Environment Canada
Ottawa, Ontario

Sirs:

Enclosed is the report "Interim Report on the Acidification of Lake 223, Experimental Lakes Area: Background Data, the First Year of Acidification (1976), and Pilot Experiments".

This report was prepared for the Alberta Oil Sands Environmental Research Program, through its Aquatic Fauna Technical Research Committee (now the Water System), under the Canada-Alberta Agreement of February 1975 (amended September 1977).

Respectfully,

W. Solodzuk, P.Eng.
Chairman, Steering Committee, AOSERP
Deputy Minister, Alberta Environment

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Member, Steering Committee, AOSERP
Regional Director-General
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# INTERIM REPORT ON THE ACIDIFICATION OF LAKE 223, EXPERIMENTAL LAKES AREA: BACKGROUND DATA, THE FIRST YEAR OF ACIDIFICATION (1976), AND PILOT EXPERIMENTS

## DESCRIPTIVE SUMMARY

### BACKGROUND

At the outset of AOSERP there was considerable concern that significant quantities of acid precipitation and heavy metals emanating from oil sands plants might adversely affect lakes near the oil sands area. Thus, the Aquatic Fauna Technical Research Committee initiated this project with the objectives to determine: 1) if acid fallout lowers the overall productivity of lakes; 2) if acidification directly affects the physiology of fishes, their eggs or fry; and 3) if heavy metals deposited in the lakes or mobilized from sediments are toxic to fishes or their food organisms. These objectives were addressed by the researchers in a whole lake experiment at the Experimental Lakes Area in northwestern Ontario because of the experimental control that could be effected at the facilities.

A complementary project directly concerning lakes adjacent to the oil sands region has been conducted. For further information the reader is referred to AOSERP Report No. 71, entitled "Lake Acidification Potential in the AOSERP Study Area".

### ASSESSMENT

The report has been reviewed by scientists at the Freshwater Institute and in AOSERP and there has been no argument over the quality of data presented. The reader should consider the interim nature of the report and it is anticipated that further reports on the project will be published. It is recommended that this interim report be made available to the public through distribution to selected Canadian libraries. Such recommendations, however, do not signify that the report's contents reflect the views and policies of Alberta Environment or Environment Canada nor does the mention of trade names or commerical products constitute an endorsement or recommendation for use.

The Alberta Oil Sands Environmental Research Program accepts the report, "Interim Report on the Acidification of Lake 223, Experimental Lakes Area: Background Data, the First Year of Acidification (1976), and Pilot Experiments" as a useful contribution and thanks the Freshwater Institute and the authors for their efforts.

S.B. Smith, Ph.D.
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## ABSTRACT

Data on the chemistry and biology of Lake 223 are presented for two background years, 1974 and 1975, and for 1976, when sulphuric acid was added to the lake. Four additional years of acidification are planned, eventually reaching a pH of about 5.

Background data are given for morphometry, substrate type, pH, sulphate, dissolved inorganic carbon, bicarbonate, dissolved organic carbon, suspended phosphorus, silica, calcium, magnesium, sodium, potassium, iron, manganese, zinc, copper, cobalt, lead, cadmium, suspended carbon, total dissolved nitrogen, suspended nitrogen, nitrate, ammonium, total dissolved phosphorous, chromium, aluminum, chlorophyll, phytoplankton, primary production, respiration, light, temperature, benthic emergence, crayfish, and fish populations.

The only observable chemical effect in 1976, the first year of acidification, was a halving of the dissolved inorganic carbon concentration. Short-term effects on the biota were not observable; acidification continues.

Mass-balance calculations allowed identification of two possible mechanisms in addition to dissolved bicarbonate which might buffer the lake system: lake sediments and dissolved colloids (humic substances).

Pilot experiments were done in 10 m diameter tubes at pH 5 and 6, in order to anticipate changes which might take place in the lake as pH is lowered in future years. These experiments concentrated largely on chemistry, particularly trace metals. As pH dropped to 6 and below, aluminum, iron, manganese, zinc, thorium, and possibly cadmium were mobilized from sediments. There were no observable changes in arsenic, barium, calcium, cesium, chromium, cobalt, copper, iodine, lead, magnesium, mercury, potassium, selenium, sodium, and vanadium.

## ACKNOWLEDGEMENTS

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## 1. INTRODUCTION

The purpose of this research is to study the effects of acidifying a small soft-water lake on the biogeochemistry of the lake ecosystem.

The acidification of aquatic ecosystems by emissions of sulphur dioxide ($SO_2$) has been documented for several areas of the earth including Scandinavia (Gjessing et al. 1976; Leivestad and Muniz 1976), the eastern U.S.A. (Likens and Bormann 1974; Likens 1976), and the Sudbury area of Canada (Kramer 1976; Beamish 1974; Beamish and Harvey 1972). The pH of many surface waters in these areas has decreased to values of 5 or less, with attendant loss of fish and invertebrate populations, increase in heavy metals, and reduction in the production, populations, and species diversity of phytoplankton.

Despite the thorough documentation in the above studies, the mechanisms causing many of the observed effects are still unknown. For example, it is not known whether the disappearance of fish and some invertebrate species is because spawning ceases, because eggs are not fertilized, or because fertilized eggs do not develop. Disappearance of older individuals might be due to starvation as production decreases, or to disruption of physiological processes.

Likewise, the increase in heavy metals accompanying acidification has been attributed to atmospheric fallout from the same industries, or to metals leached from soils in the watersheds of the lakes. The possibility that metals are mobilized directly from lake sediments as pH decreases does not appear to have been considered, yet solubility products for many heavy metal complexes suggest that sediments should be an important source. These questions are of critical importance to the development of the Athabasca Oil Sands, because the high anticipated release of $SO_2$ to the atmosphere from oil sands processing activities could have serious effects on any softwater lakes of the area (for example, those identified by Hesslein 1979). Heavy metals released to the atmosphere could have harmful effects, even in lakes where the pH is unchanged.

In order to answer many of these questions, we decided to artifically acidify Lake 223, a small, natural lake in the Experimental Lakes Area (ELA) at the rate of 0.5 pH units per year, and to study the accompanying changes in chemical and biological mechanisms. Two years of background data were available for many parameters before acidification commenced. In addition, many parameters were studied simultaneously on Lake 224, a control lake in the same watershed, so that direct comparisons could be made in the years of the study.

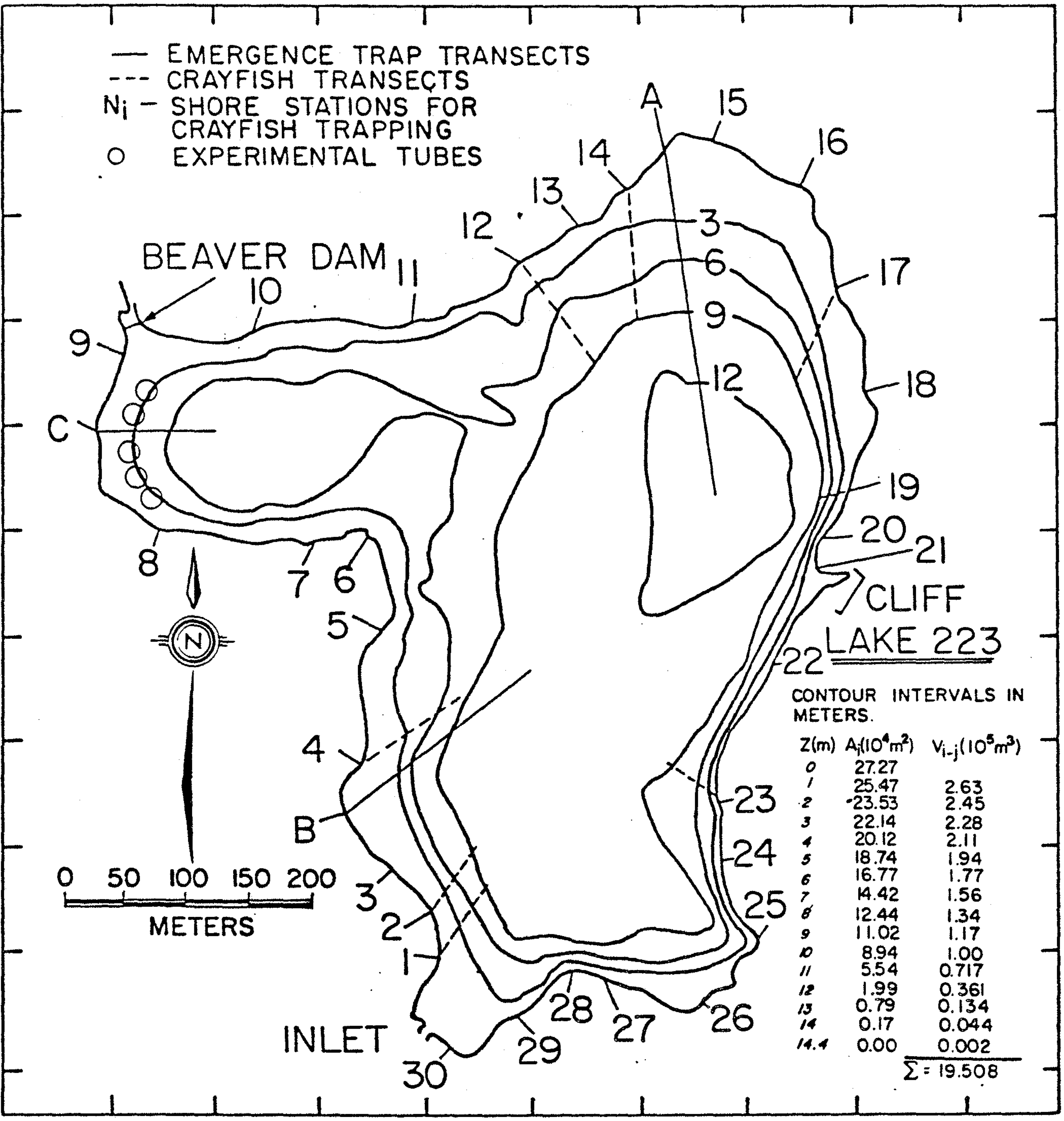
Lakes 223 and 224 are typical small Precambrian Shield lakes, and one should be able to extrapolate the results of our investigations to similar soft-water lakes throughout temperate regions of Canada.

This report summarizes background data on the system and the first year of acidification, 1976, so that this information may serve as a reference point for changes observed when the pH is much lower than normal in future years. As a result, it is rather long and unwieldy.

## 1.1 DESCRIPTION OF THE LAKE

Morphometric data for Lake 223 and a morphometric map are shown in Figure 1. The lake, located in the Experimental Lakes Area of northwestern Ontario, is typical for lakes of this size in the Precambrian Shield. A single inlet and outlet, each monitored by a V-notch weir serviced by Water Survey of Canada, supply and drain the lake. There are no areas of significant input or loss of groundwater. The watershed of the lake is of Precambrian gneisses and granities, forested by typical boreal pine-spruce forest. Brunskill and Schindler (1971) give a more detailed description.

We believed that sediments would prove to be of major significance in buffering the lake against acidification. Different benthic organisms of interest to this study also occupy different bottom types. A preliminary bottom map was therefore constructed, using information obtained from SCUBA divers doing experiments on Lake 223 (Figure 2). A more accurate and detailed map will be constructed in 1977, using diver-surface communications to relay detailed data on bottom type.



CONTOUR INTERVALS IN METERS.

| Z(m) | $A_i(10^4 m^2)$ | $V_{i-j}(10^5 m^3)$ |
|---|---|---|
| 0 | 27.27 | |
| 1 | 25.47 | 2.63 |
| 2 | 23.53 | 2.45 |
| 3 | 22.14 | 2.28 |
| 4 | 20.12 | 2.11 |
| 5 | 18.74 | 1.94 |
| 6 | 16.77 | 1.77 |
| 7 | 14.42 | 1.56 |
| 8 | 12.44 | 1.34 |
| 9 | 11.02 | 1.17 |
| 10 | 8.94 | 1.00 |
| 11 | 5.54 | 0.717 |
| 12 | 1.99 | 0.361 |
| 13 | 0.79 | 0.134 |
| 14 | 0.17 | 0.044 |
| 14.4 | 0.00 | 0.002 |
| | | $\Sigma = 19.508$ |

Figure 1. A morphometric map of Lake 223, showing the location of emergence trap transects, transects for SCUBA estimation of crayfish, stations for trapping of crayfish with minnow traps, and large tubes for pH and heavy metal experiments.

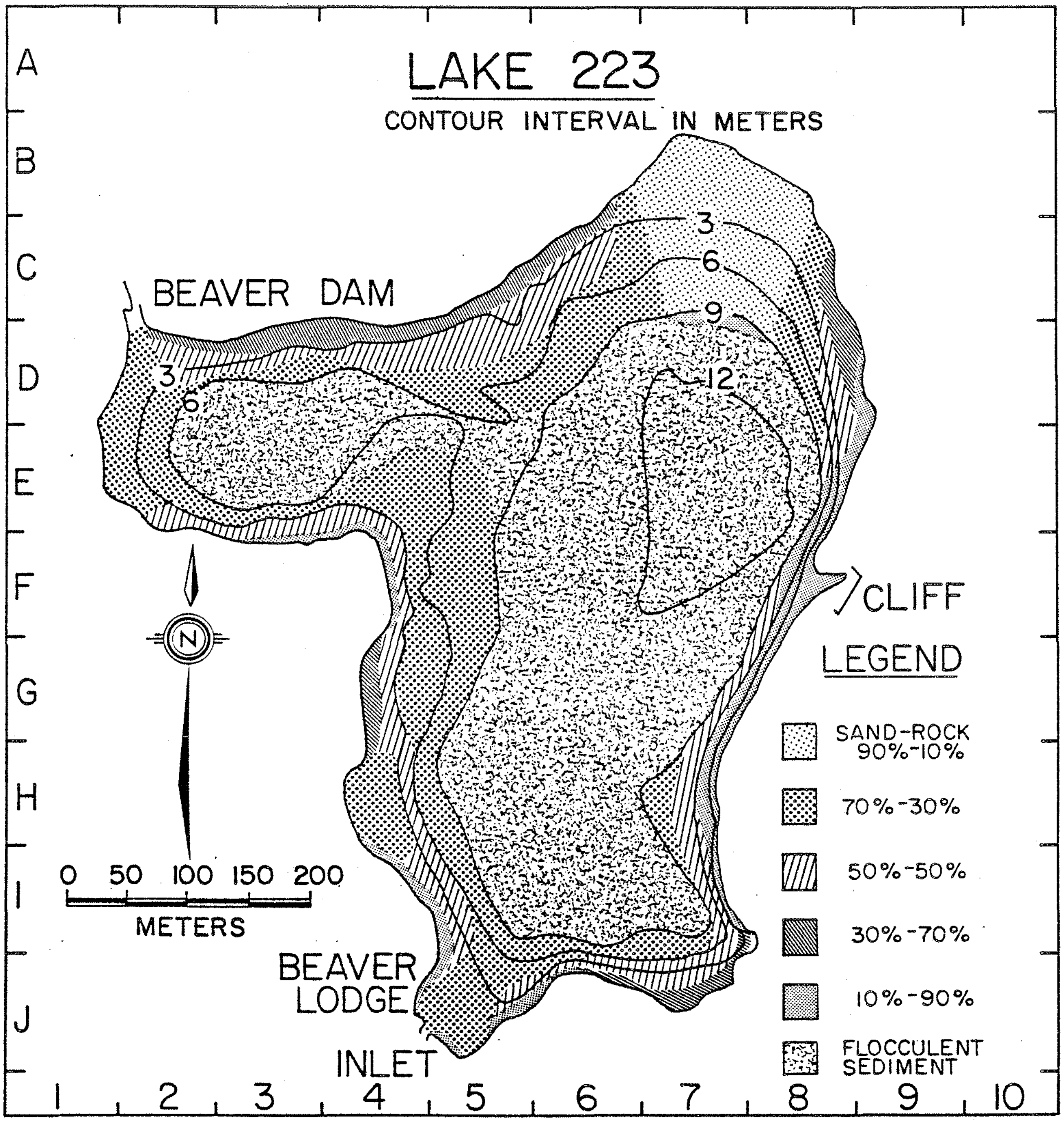


Figure 2. A preliminary map of bottom surface types in Lake 223.

## 2. MATERIALS AND METHODS

### 2.1 ACIDIFICATION OF LAKE 223

Electrolyte grade concentrated sulphuric acid was used for all additions. It was analyzed for trace contaminants by the analytical unit at the Freshwater Institute and found to have none of consequence.

One hundred and fifty 75 lb. (34.1 kg) carboys were flown to Lake 223 by helicopter at the beginning of April. From 6 May onward, six carboys per week were added to the lake by pouring them slowly out of the back of a moving outboard boat, directly into the propwash of an outboard motor. Personnel handling the acid wore complete neoprene rubber suits, gloves, and goggles.

By mid-June, when the rates of acid addition were judged insufficient, the addition rate was increased to 12 carboys per week. This made a second purchase of acid necessary. Due to severe forest fire problems in the area, it was not possible to fly acid to the lake in time to maintain the 12 carboy per week schedule. As a result, only six carboys were added on 26 August, but 18 added on 16 September to compensate. Figure 3 gives the amount of acid added on each date of addition.

### 2.2 HYDROLOGY

Inflow from Lake 224 and outflow from Lake 223 were monitored using V-notch weirs equipped for continuous recording of water level. Weirs were installed and serviced by Water Survey of Canada. Precipitation was measured at the Rawson Lake meteorological station 4 km to the south.

### 2.3 CHEMICAL SAMPLING

Profiles of samples were taken monthly from both Lakes 223 and 224 using a peristaltic pump and specially cleaned polyethylene tubing. Sampling depths were 1, 4, 8, 12, and 14 m in the former and 1, 5, 10, 20, and 25 m in the latter. Analysis for pH, dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), $O_2$, total dissolved phosphorous (TDP), $NO_3 + NO_2$, $NH_3$, total dissolved nitrogen (TDN),

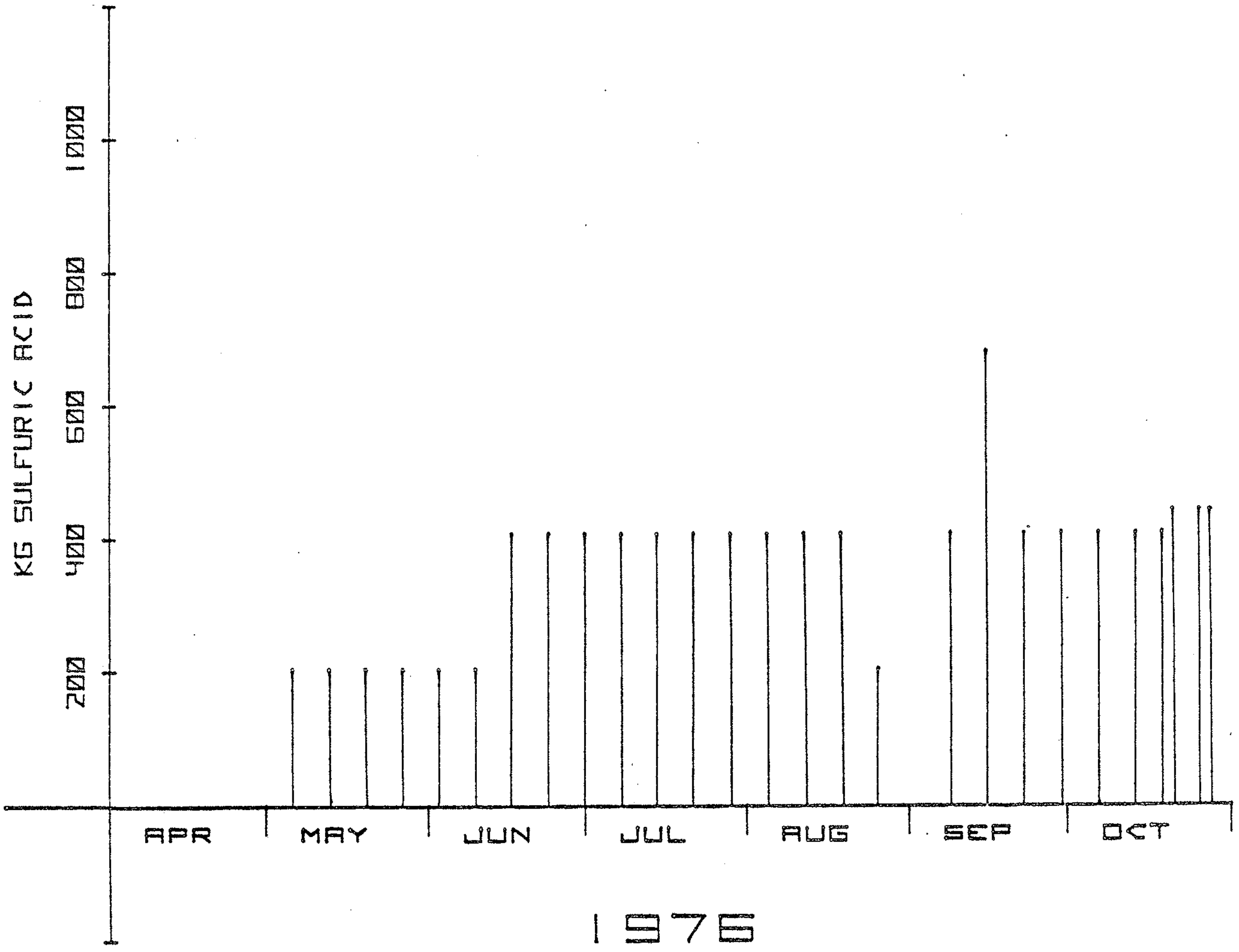


Figure 3. Dates and amounts of $H_2SO_4$ added to the lake.

conductivity, chlorophyll, and dissolved colour were performed at the ELA camp, within one hour after collection. Analyses for suspended P, N, and C, reactive silicate, $SO_4$, Cl, Ca, Mg, Na, K, Fe, and Mn were performed at the Freshwater Institute in Winnipeg. Methods for the above were those of Stainton et al. (1977).

An additional set of profile samples for trace metal analysis was also taken each month. One litre of water was taken in a specially cleaned linear polyethylene bottle at each depth and preserved with 1 cc of redistilled concentrated $HNO_3$. A second litre of water was kept at 4-6°C and transported to Winnipeg within 24 hours, where it was filtered through a millipore membrane of 0.22 μm pore size. Samples were analyzed by flameless atomic absorption for Al, Cu, Zn, Mn, Pb, Cd, Fe, Co, and Cr using a 403 Perkin-Elmer Atomic Absorption Spectrophotometer with an HGA-2100 Power Supply.

In addition to the above, weekly samples were taken from the inlets and outlets of both lakes and analyzed for all of the above chemicals. Bulk precipitation samples were collected in a polyethylene container and analyzed as well.

## 2.4 PHYSICAL AND HYDROLOGICAL MEASUREMENTS

Temperature, secchi depth, and light extinction were measured at monthly intervals, usually more frequently. Inflow and outflow were measured continuously with V-notch weirs and continuous recording float levels, installed and maintained by the Kenora office of the Inland Waters Branch, Water Survey of Canada.

## 2.5 PHYTOPLANKTON STANDING CROP AND PRODUCTION

Integrated phytoplankton samples were taken from the epilimnion, metalimnion, and hypolimnion at two-week intervals. An aliquot of each was preserved in Lugol's solution. Identification and counting were done as outlined by Kling and Holmgren (1972) and Findlay and Kling (1975). Other aliquots were used for measuring chlorophyll a and suspended carbon concentrations.

The remainder of the sample was used for primary production measurements in a light incubator (Fee 1973) and phytoplankton respiration. Details are given by DeClercq et al. (1977). In situ continuous depth profiles of chlorophyll concentrations were taken on three occasions using a field fluorometer (Fee 1976).

## 2.6 ZOOPLANKTON

Zooplankton samples were taken once each month, using a 30 L transparent trap (Schindler 1969) with a 50 µm mesh. They were preserved in 4% formaldehyde for later counting and identification.

## 2.7 ZOOBENTHOS

Dipteran emergence was monitored at 27 stations in both 1975 and 1976. Stations were located at 1 m depth intervals from 1 to 8 m, then at 2 m depth intervals to maximum depth in the lake (Figure 1). Plastic cone-traps were submerged at a depth of 1 m at each station, and used to capture insects for one 24-hour period each week. Yearly integrated totals for each depth were calculated and multiplied by the area of the lake bottom at each depth to calculate total emergence.

Populations of crayfish *Orconectes virilis* were estimated by mark and recapture, using both baited wire minnow traps and SCUBA diving along premarked transects to capture speciments. Specimens were marked with a telson clip. Stations used for crayfish trapping are shown in Figure 1.

## 2.8 FISH

Fish were captured in a large trap net (Beamish 1973) from early May until late October. Nets were checked five times per week until mid-August, after which they were checked three times per week. In May and October, gill nets were set for several hours per night and checked every half hour. Fish were anesthetized in a solution of MS 222. Fork length, total length, and weight were recorded, two or three rays were clipped from one fin and two or three scales were removed for ageing. Fish greater than 0.1 kg were tagged with small numbered plastic cylinders sewn beneath the dorsal fin with nylon monofilament. Fish were held in a recovery tub until the effects of the anesthetic wore off, after which they were released. A few fish were held overnight in a holding pen in the lake to check that survival was good. Recaptured fish which were first marked in 1975 had a second fin clipped, while fish first marked in 1976 were measured and released without a second fin clip when recaptured.

In early August, two groups of fathead minnow, totalling over 4500 individuals, were collected in the trap net and preserved for later examination. No attempt was made to estimate the population size, because of the difficulty in marking and recapturing such small fish.

Fin-rays were air dried, mounted in epoxy, sectioned with a jewellers' saw, and aged under a microscope by counting annuli.

## 2.9 TUBE EXPERIMENTS

Tubes of 10 m diameter and 2 to 2.5 m depth were constructed as follows: circular wooden rings were sawed from 1" x 8" (2.5 x 20 cm) spruce, then nailed and glued together to form a circular ring 3 inches thick and 100 feet in circumference. These were supported by 6" x 6" x 4' (15 x 15 x 120 cm) styrofoam blocks, banded to the ring in a continuous circle with plastic banding material. The inside of the ring was padded with used fire hose. The tube wall was constructed of 6 mil crosswoven polyethylene, treated to be resistant to ultraviolet light. The material was sewn where necessary with double-lapped seams or fastened with POLY-ZIP, constructed of rigid UV resistant polyethylene.

Tests performed at the Freshwater Institute proved that this material was low in leachable trace contaminants, and would not become brittle even at temperatures of -60°C. Tubes were installed in a protected bay of the lake (Figure 1).

Tubes were anchored with six 10 m lengths of 1/4" (63 cm) cable, attached to concrete anchors which were in turn buried in piles of rocks. SCUBA divers sealed the bottom of the tube wall to the sediments by piling a continuous ring of rocks on the material. Figure 4 is a sketch showing details of construction.

Four tubes were used in the first experiment. They were located in 2 to 2.5 m of water near the outlet of the lake (Figure 1). The bottom sediment was of medium-grain sand, with an occasional large boulder, submerged log, or ferromanganese deposit.

Two tubes, B and C, were titrated to pH 5 and 6, respectively, using $H_2SO_4$ ten days before radiotracer experiments began. Sufficient acid was added twice weekly to hold the pH's near these values throughout the ice-free season. Tubes D and E were left at their natural pH.

Tubes, B, C, and D were stocked with six suckers each for metal uptake experiments (J. Klaverkamp unpublished). Tube 3 was kept in a fishless condition.

Before addition of radioactivity, several short cores were taken with plexiglass cylinders and placed on the bottom of the tubes by SCUBA divers. A line to the rim of the tube from each core cylinder allowed cores to be retrieved from the surface as desired. Sedimentation traps, 22 cm in diameter, constructed of plastic bags were rigged in the same manner, to collect sedimenting matter.

Thirteen gamma-emitting nuclides were added to each tube. These were As-74, Ba-133, Co-60, Cr-51, Cs-134, Fe-59, Hg-203, I-133, Mn-54, Se-75, Th-228, V-48, and Zn-65. A mixed spike containing 0.5 mCi of each of the above except I-131 was added in 60 mL of 0.1N HCl. Because of its volatility, I-131 was added separately in 5 mL of 0.1N NaOH. In order to keep track of losses due to leakage, about 1 mCi of tritiated water was added to each spike at the same time.

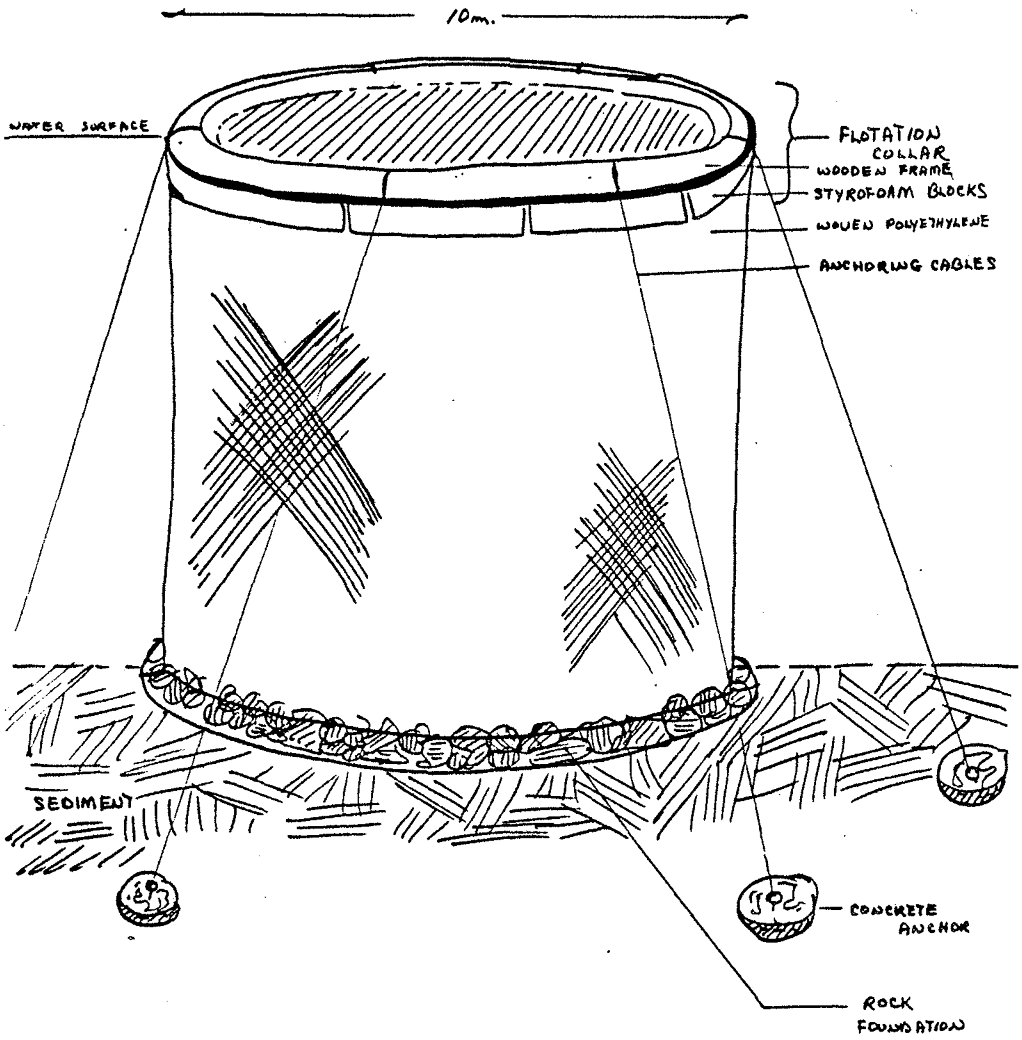


Figure 4. Details of construction of the large tubes used for pH and heavy metal experiments.

Radionuclide spikes were added as follows: a small electric outboard attached to the tube rim was used to mix the tube water rapidly; the spike was poured into the propellor wash to ensure rapid mixing.

Integrated water samples were taken in linear polyethylene bottles, pumping water through a peristaltic pump. Samples were taken immediately to the field laboratory where separations were done as follows: each sample was filtered through 0.45 µm millipore membrane. The filtrate was then passed through an activated charcoal column, followed by a column of mixed cation and anion exchange resins. Non-polar organically bound isotopes were expected on the charcoal fraction, and polar organic compounds plus ionic species on the ion exchange fraction. Following this treatment, water was digested with persulphate, and any remaining radionuclides precipitated with cobalt ferricyanide and a mixed manganese, iron, and mercuric hydroxide.

All samples were radioassayed on a Princeton Gamma Tech lithium-drifted germanium detector equipped with a Canberra model 8100 pulse height analyzer (4096 channels) and interfaced to a Hewlett-Packard 9830 computer. An automatic sample changer constructed by the Freshwater Institute shop (W. Burton) allowed continuous operation. All samples were counted in 5 cm diameter plastic petri dishes with interlocking lids, to maintain constant geometry. Typical counting times were 2000 to 4000 sec, depending on sample activity.

# 3. RESULTS

## 3.1 LAKE CHEMISTRY

### 3.1.1 pH, Bicarbonate, and Sulphate

Despite the weekly sulphuric acid additions, there was no significant change in pH in 1976 (Figure 5). Reasons are discussed in detail later.

Dissolved inorganic carbon (DIC = $HCO_3^- + CO_3^= + H_2CO_3 + CO_2$) decreased greatly, as expected (Figure 6).

Sulphate concentrations in the lake doubled during acidification in 1976, from 3 to 6 $g/m^3$, as one might expect (Figure 7). Chloride remained unchanged (Figure 8).

### 3.1.2 Major Cations

There was little or no change in the concentration of dissolved calcium, magnesium, sodium, or potassium (Figures 9 and 10). Both iron and manganese (Figures 11 and 12) showed the usual increase in hypolimnetic concentration under anoxic conditions during summer stagnation, but only manganese increased significantly in the lake as a whole between July and December 1976. Due to the lack of long-term background data for this element it is not clear whether this increase is within the range of normal year to year variation, but this should become apparent from the trend in future years. As the lake becomes more acid, manganous ion released into the hypolimnion during summer stagnation would be expected to re-oxidize more slowly, due to the increase stability of the manganous ion in acidic water.

### 3.1.3. Trace Metals

No trace metal showed any significant response due to acid addition. This was expected, because solubilities of most do not increase significantly until pH values of 6 or lower are reached. Figures 13 to 19 show seasonal patterns, along with the unaltered Lake 224 as a reference.

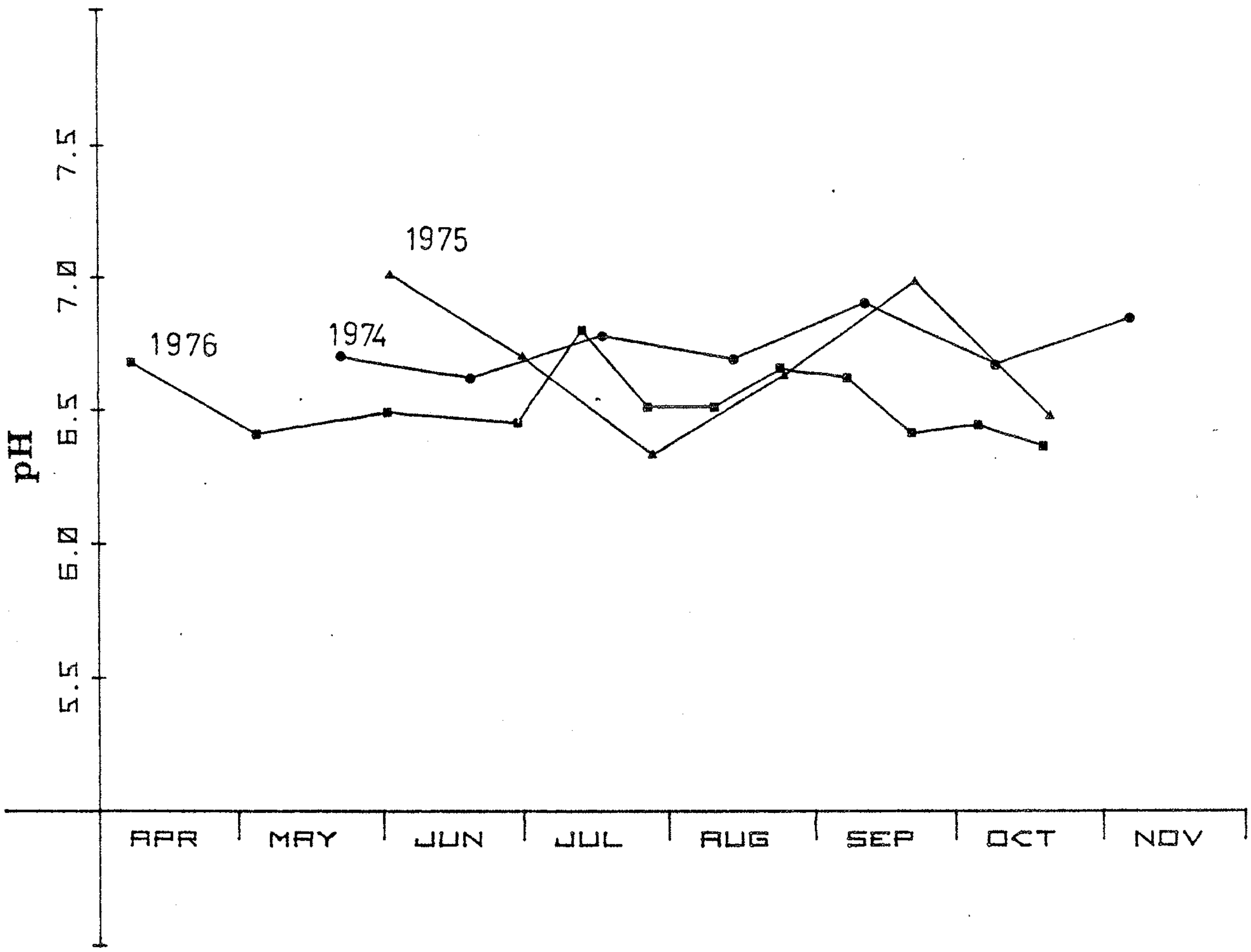


Figure 5. The pH of epilimnion water in Lake 223, 1974, 1975, and 1976.

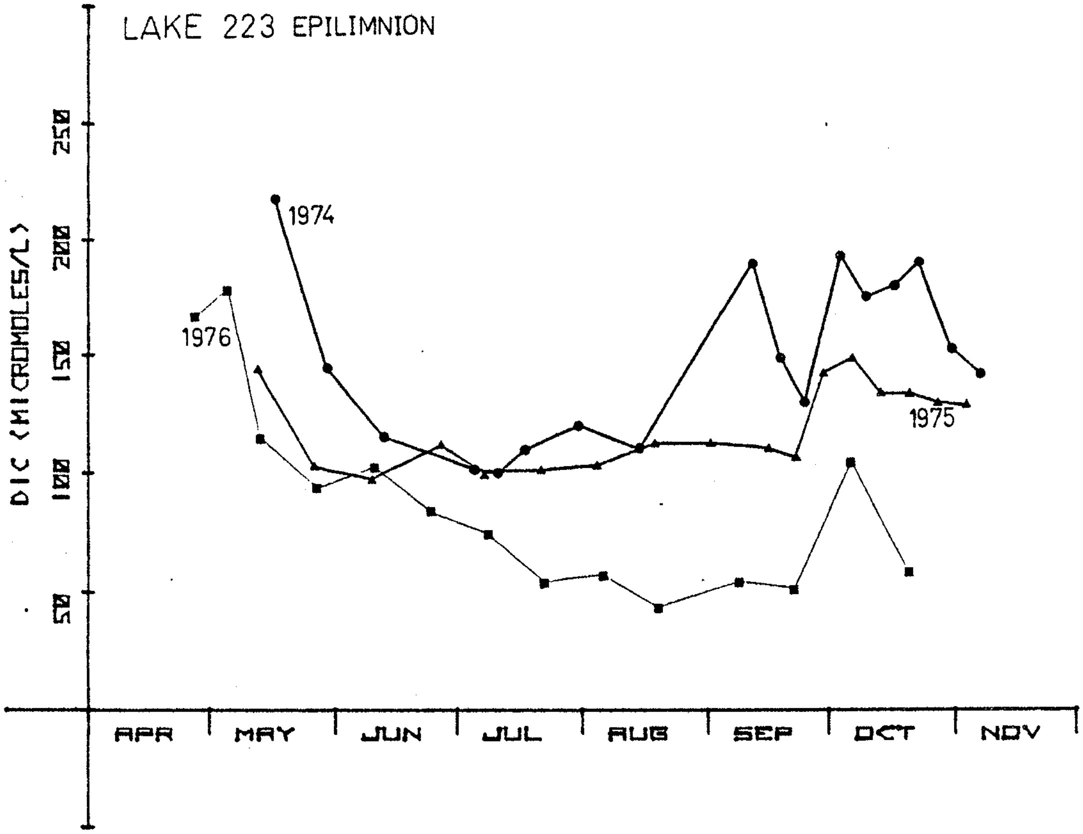


Figure 6. The average concentration of DIC in Lake 223, 1974, 1975, 1976. Note the decrease in 1976 due to acid addition.

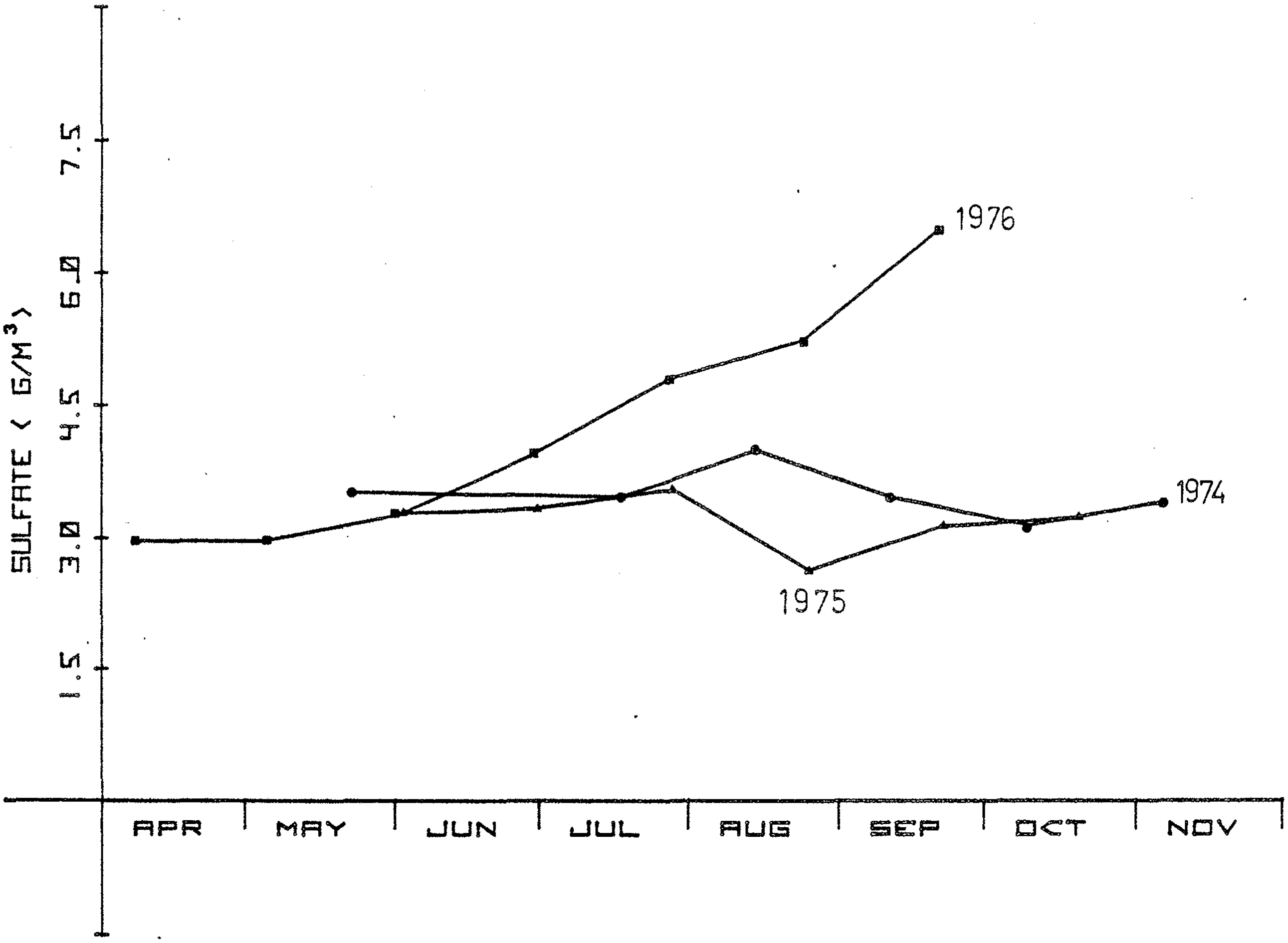


Figure 7. The average concentration of sulphate in Lake 223, 1975 and 1976.

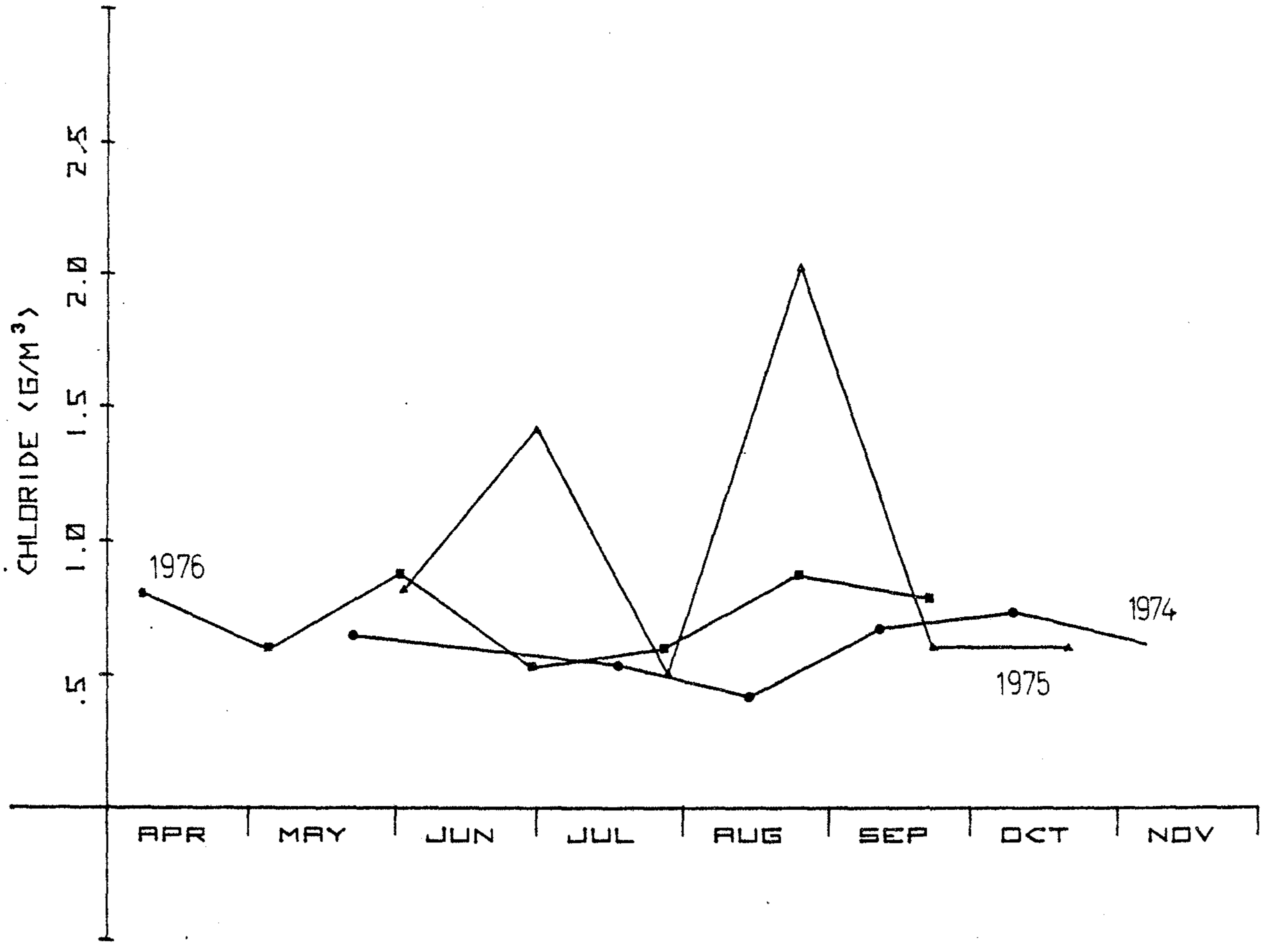


Figure 8. The average concentration of chloride in Lake 223, 1974, 1975, and 1976.

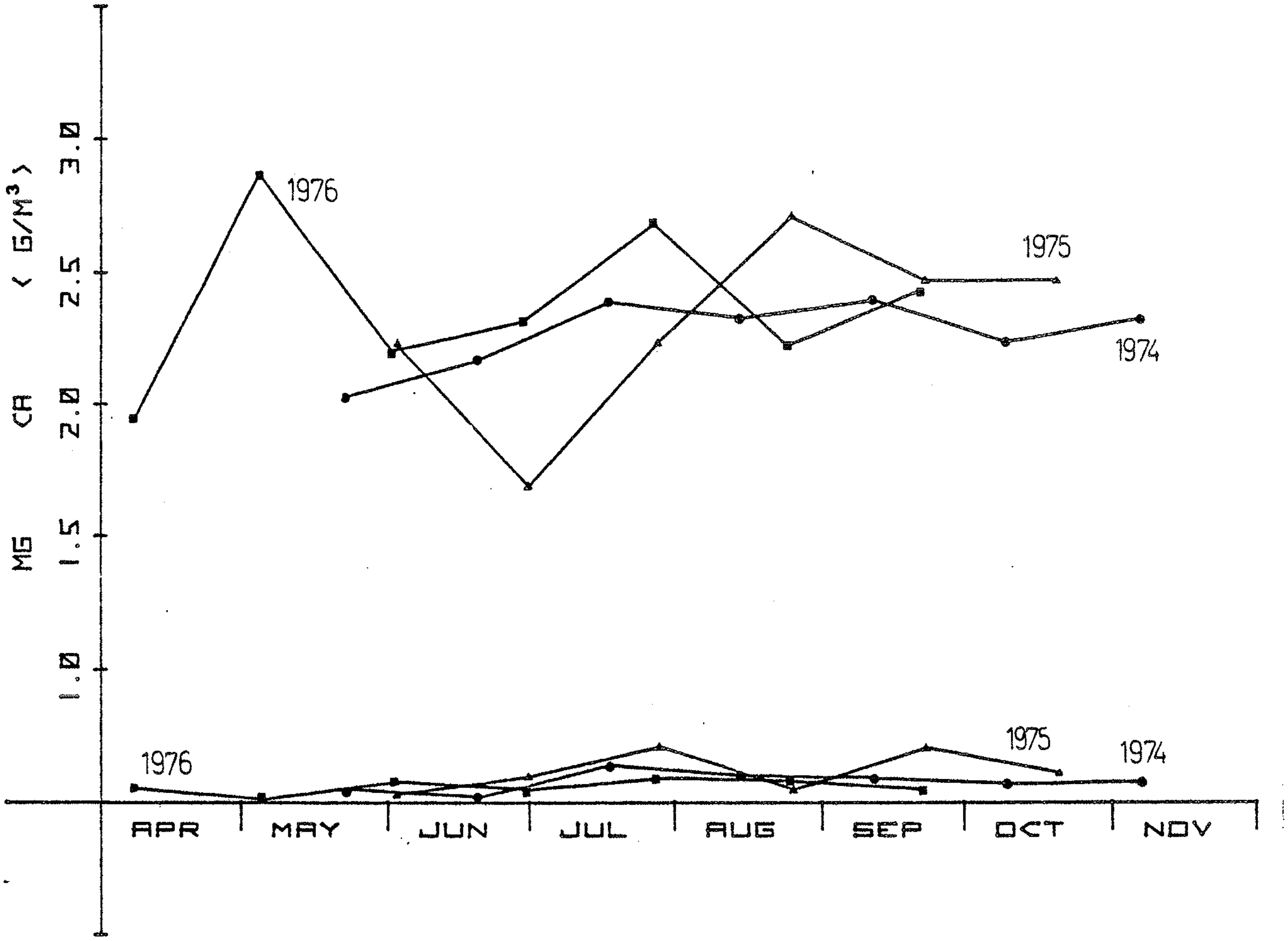


Figure 9. Average concentration of calcium and magnesium in Lake 223, 1974, 1975, and 1976

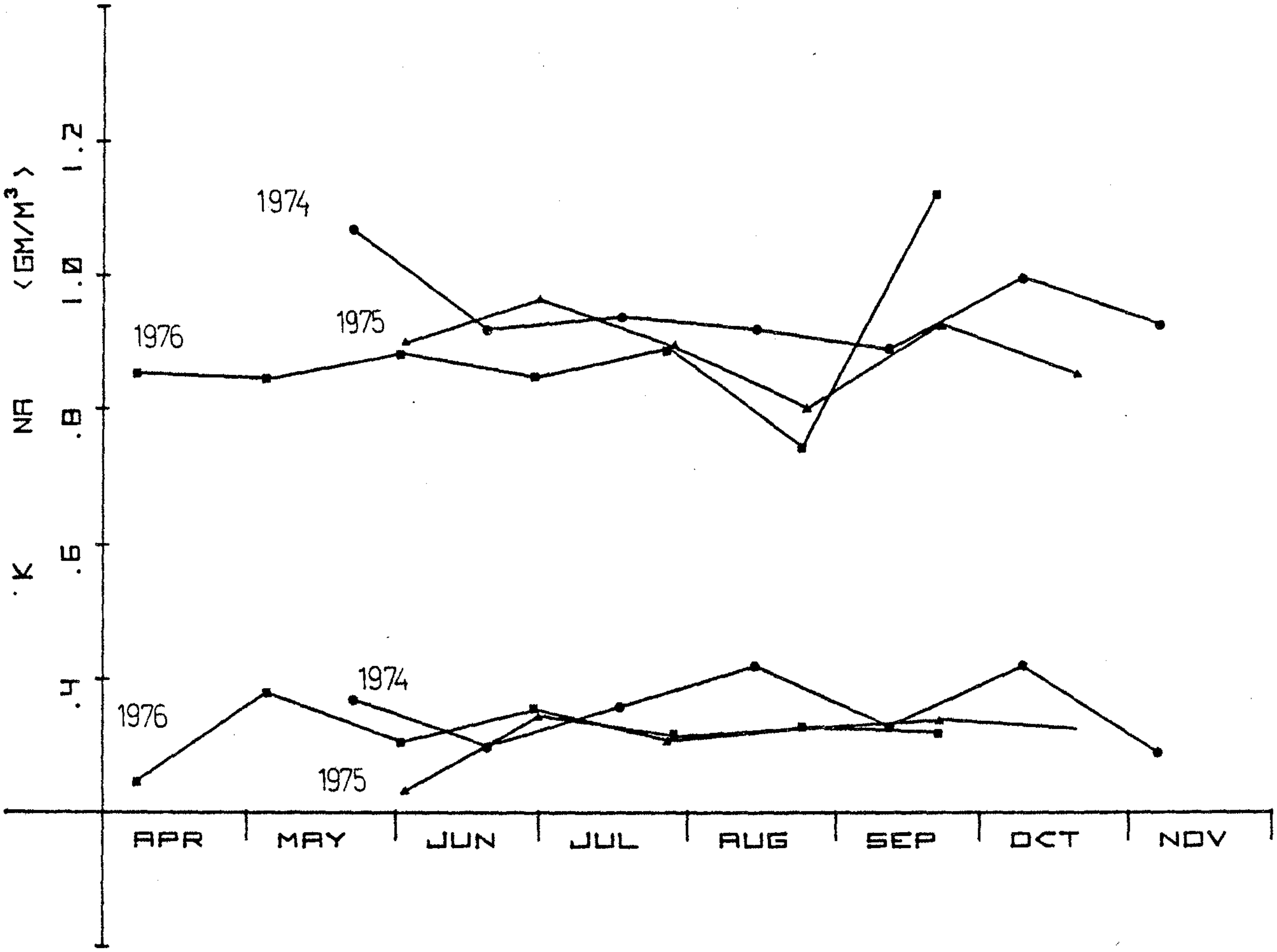


Figure 10. Average concentration of sodium and potassium in Lake 223, 1974, 1975, and 1976.

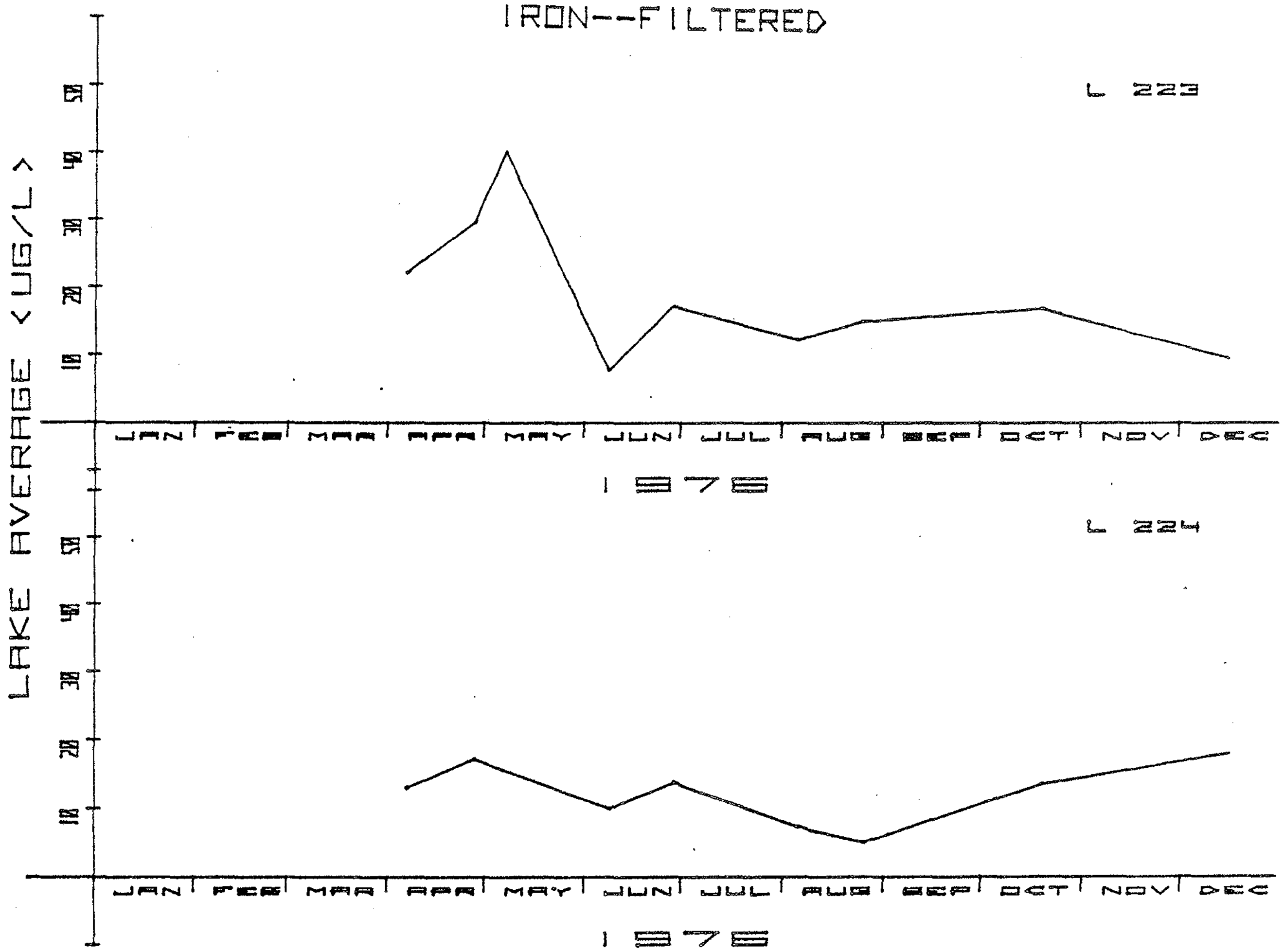


Figure 11. The volume-weighted average concentration of dissolved iron in Lakes 223 and 224, 1976. The greater seasonal trends in Lake 223 are due to the development of low $O_2$ concentrations in near-bottom waters. Samples were filtered through 0.22 μm millipore membranes.

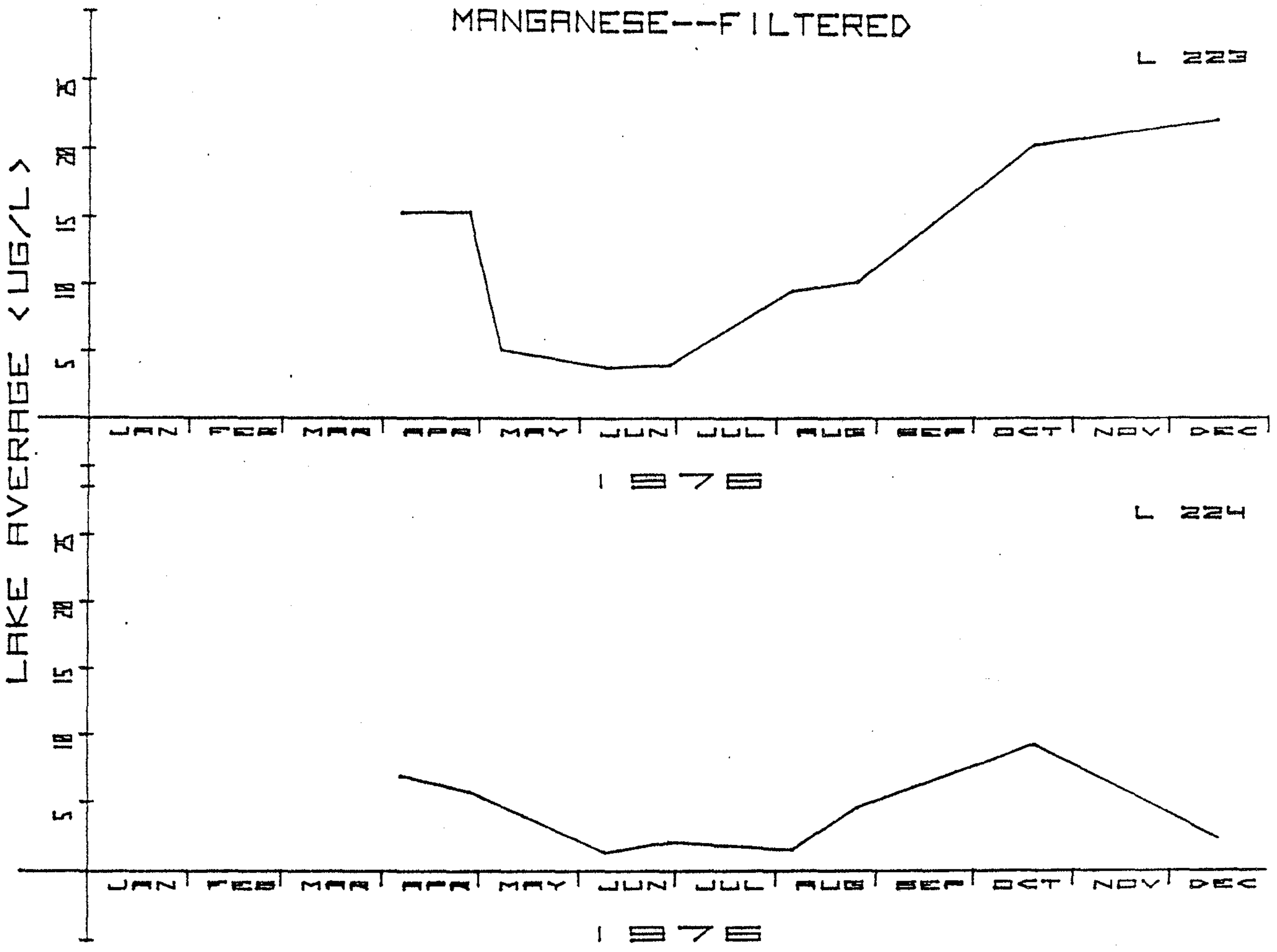


Figure 12. The volume-weighted concentrations of dissolved manganese in Lakes 223 and 224, 1976. Sample treatment and differences between the two lakes are the same as those for iron.

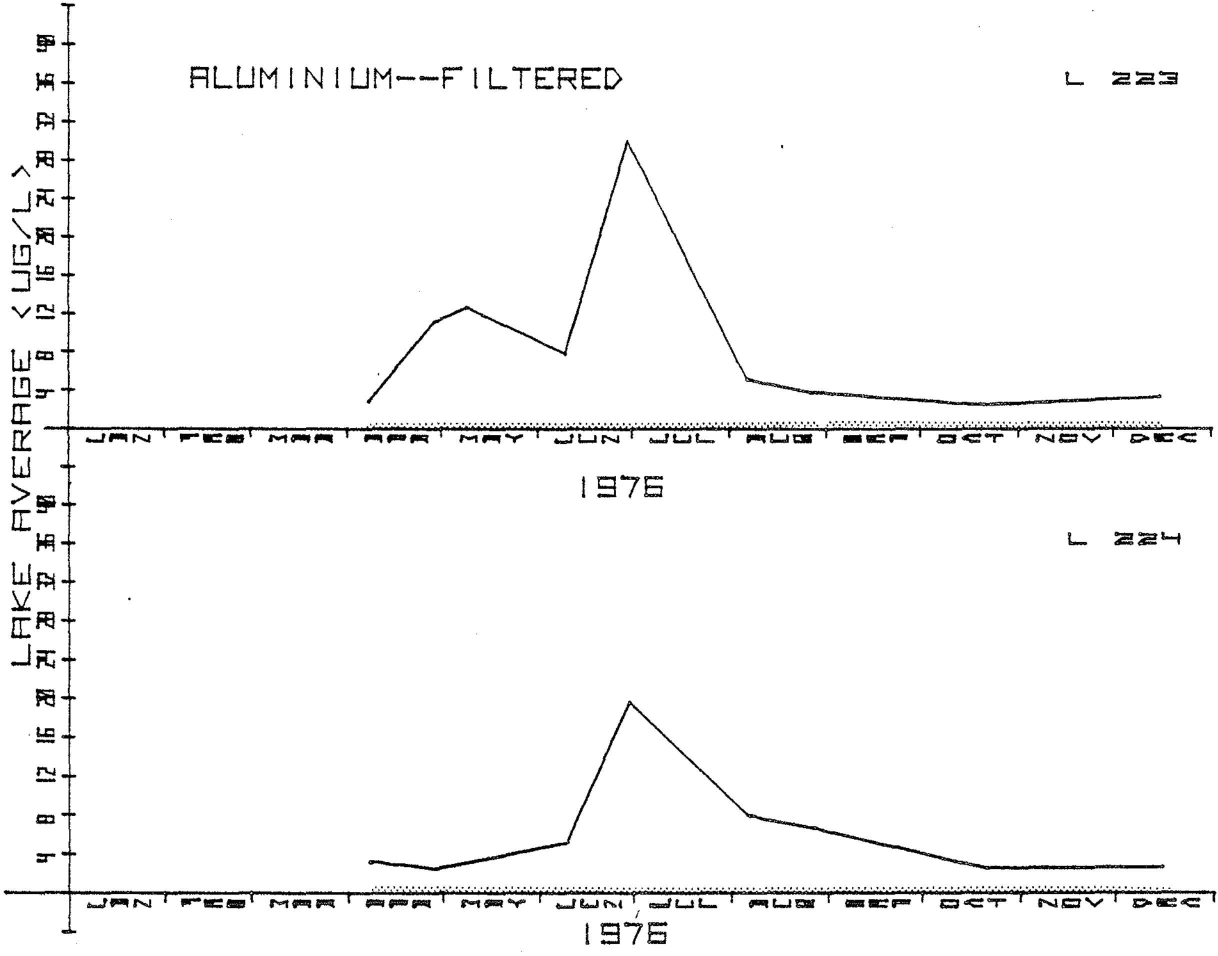


Figure 13. The volume-weighted concentrations of dissolved aluminum in Lakes 223 and 224, 1976. Sample treatment was the same as for iron and manganese. Stippled areas represent the lower analytical detection limit.

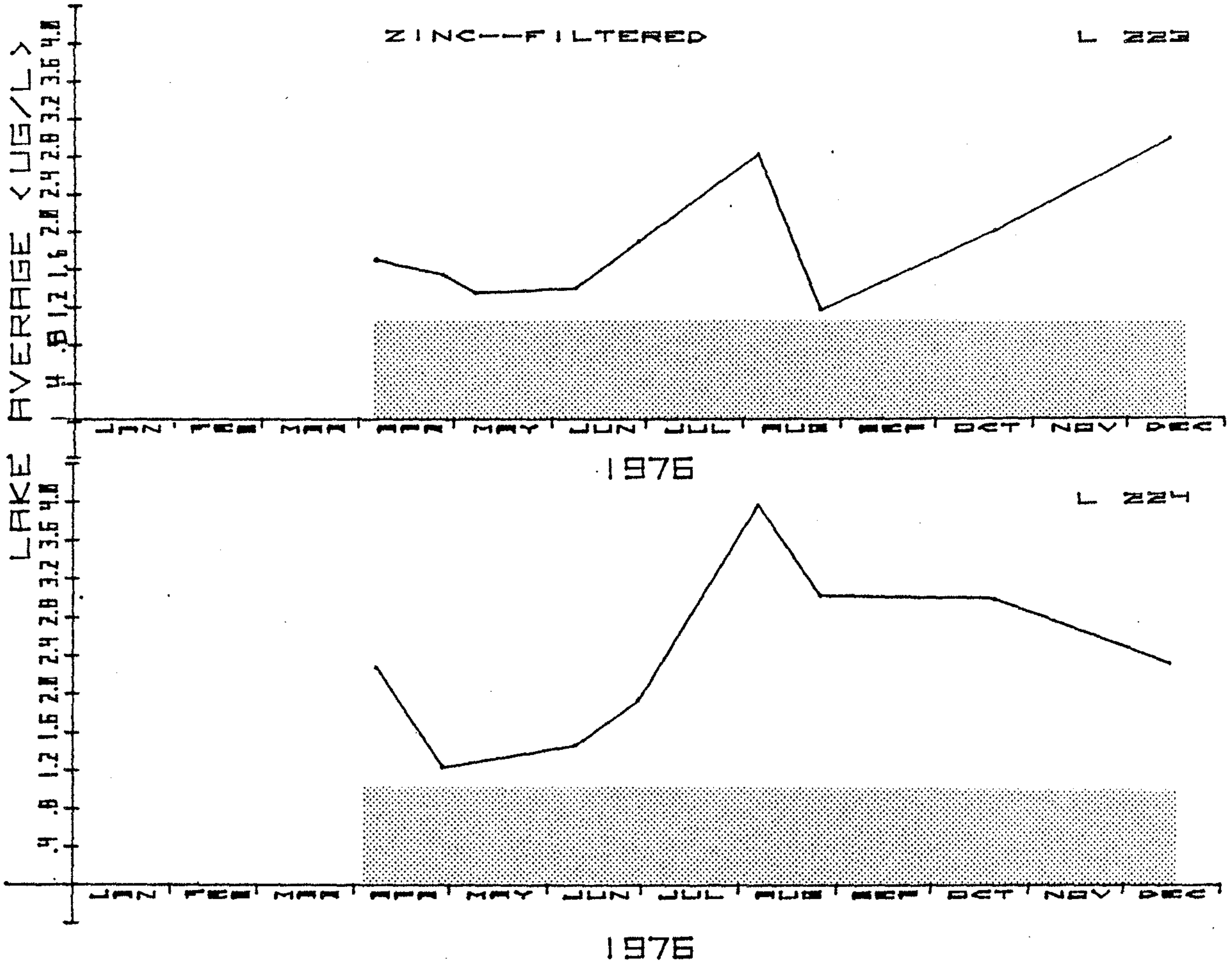


Figure 14. The volume-weighted concentrations of dissolved zinc in Lakes 223 and 224, 1976. Sample treatment was the same as for iron and manganese. Stippled areas represent the lower analytical detection limit.

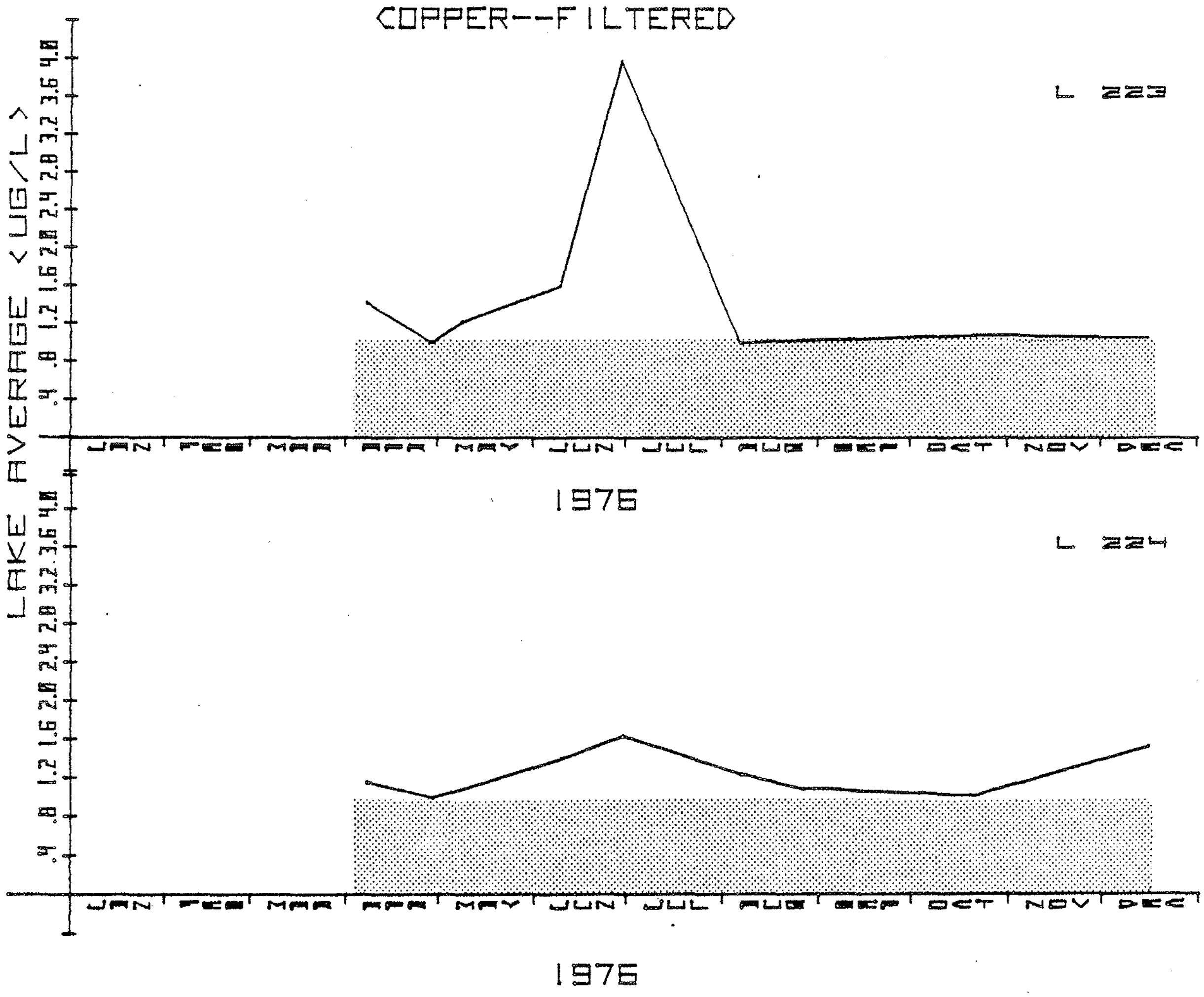


Figure 15. The volume-weighted concentrations of dissolved copper in Lakes 223 and 224, 1976. Sample treatment was the same as for iron and manganese. Stippled areas represent the lower analytical detection limit.

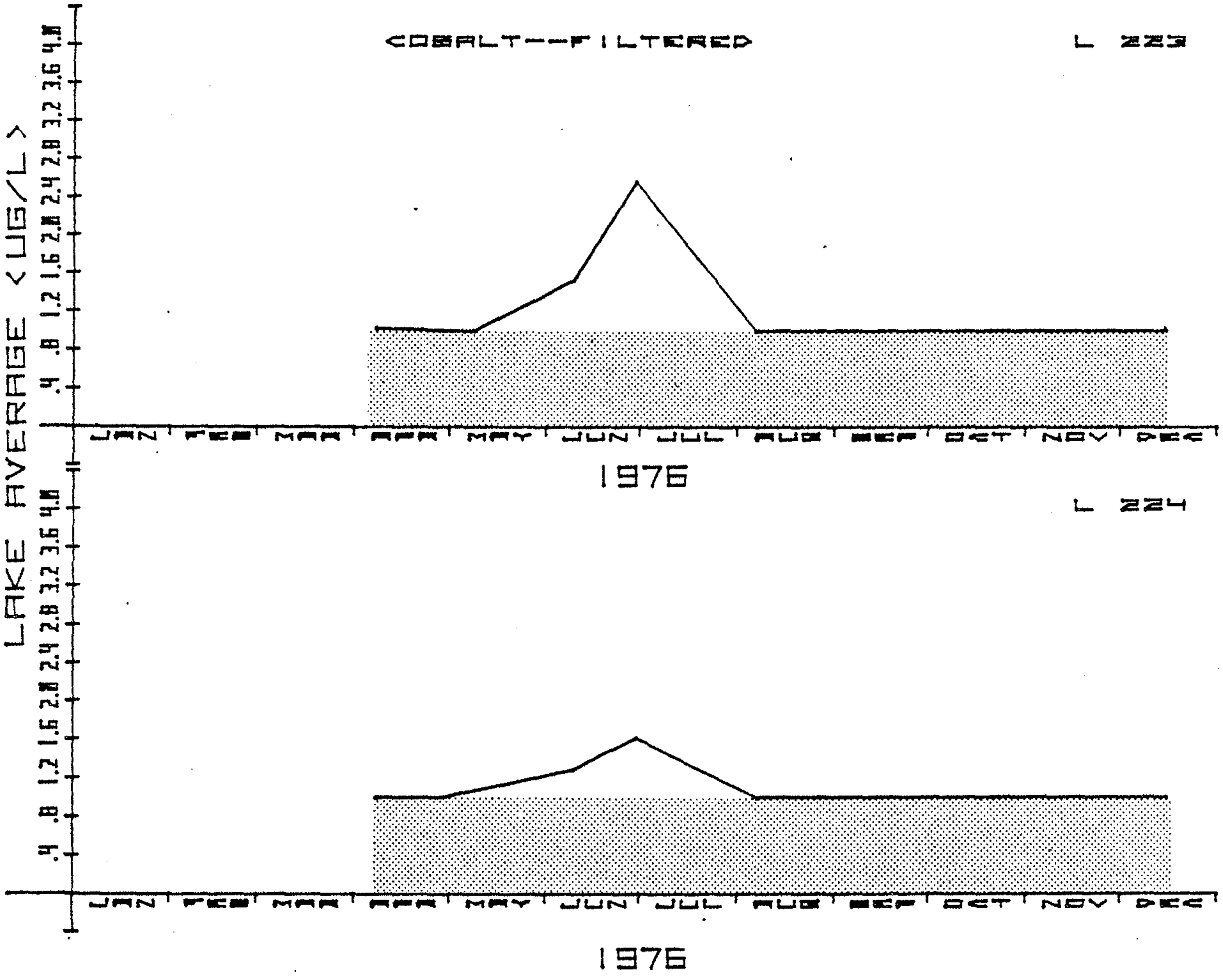


Figure 16. The volume-weighted concentrations of dissolved cobalt in Lakes 223 and 224, 1976. Sample treatment was the same as for iron and manganese. Stippled areas represent the lower analytical detection limit.

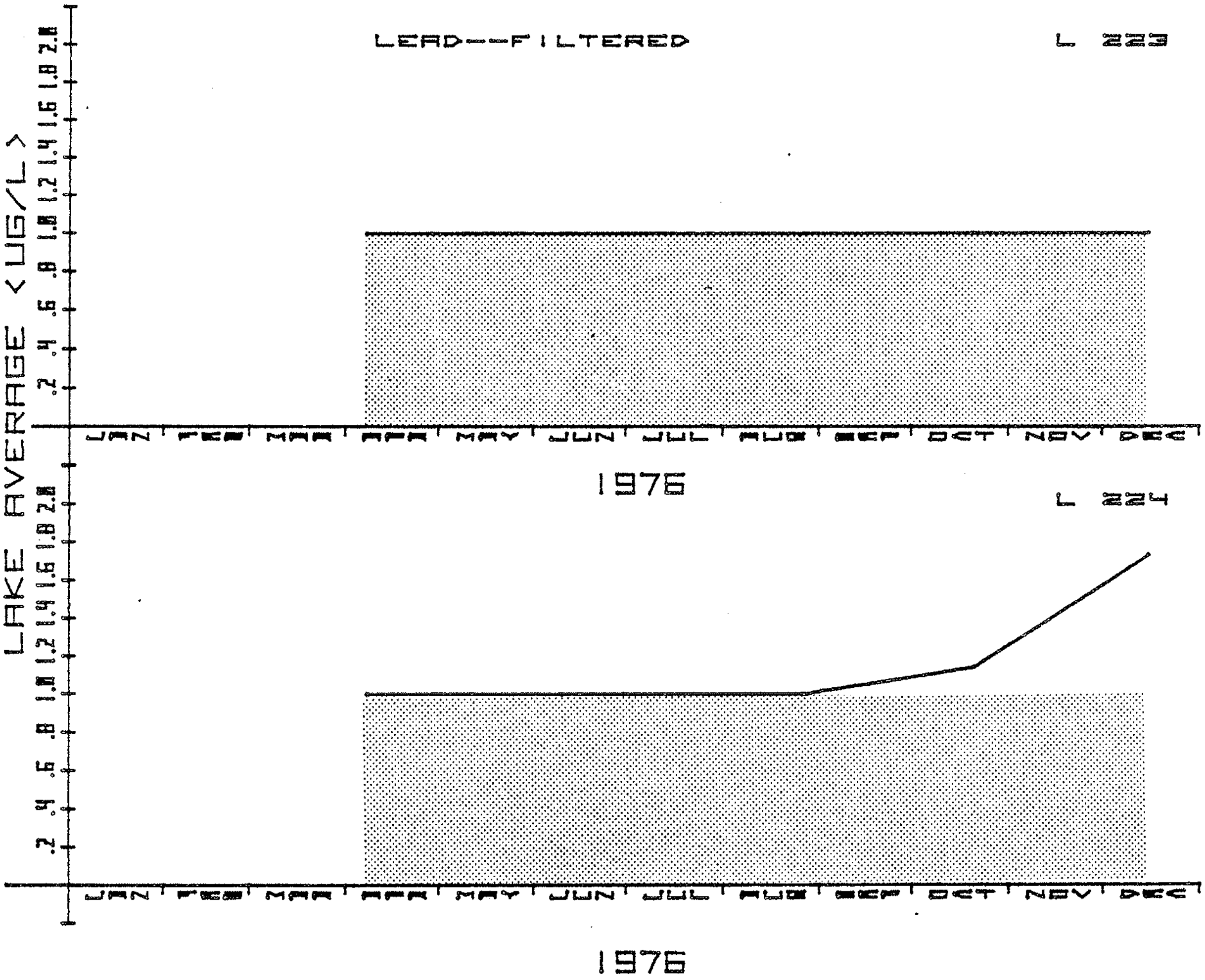


Figure 17. The volume-weighted concentrations of dissolved lead in Lakes 223 and 224, 1976. Sample treatment was the same as for iron and manganese. Stippled areas represent the lower analytical detection limit.

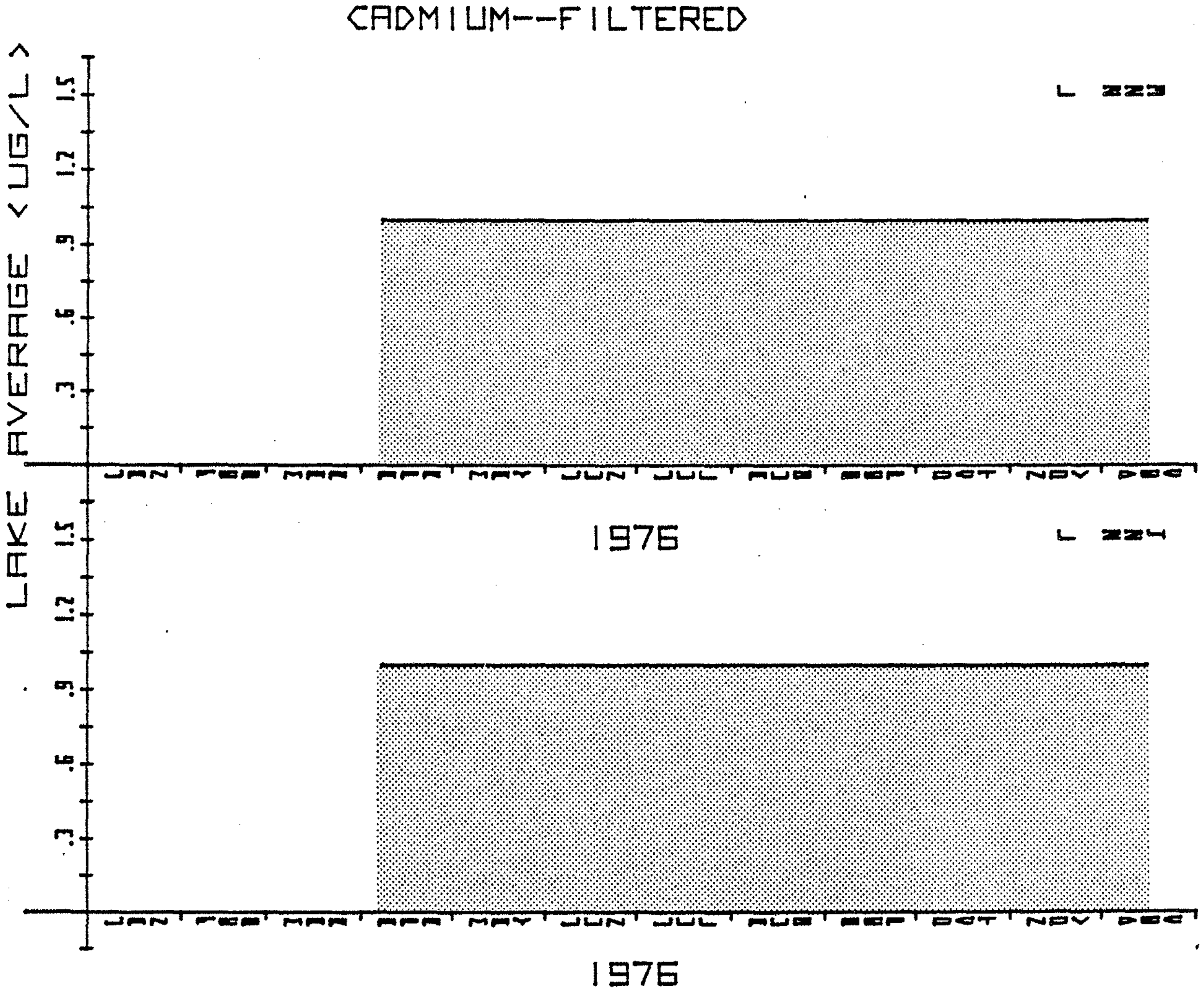


Figure 18. The volume-weighted concentrations of dissolved cadmium in Lakes 223 and 224, 1976. Sample treatment was the same as for iron and manganese. Stippled areas represent the lower analytical detection limit.

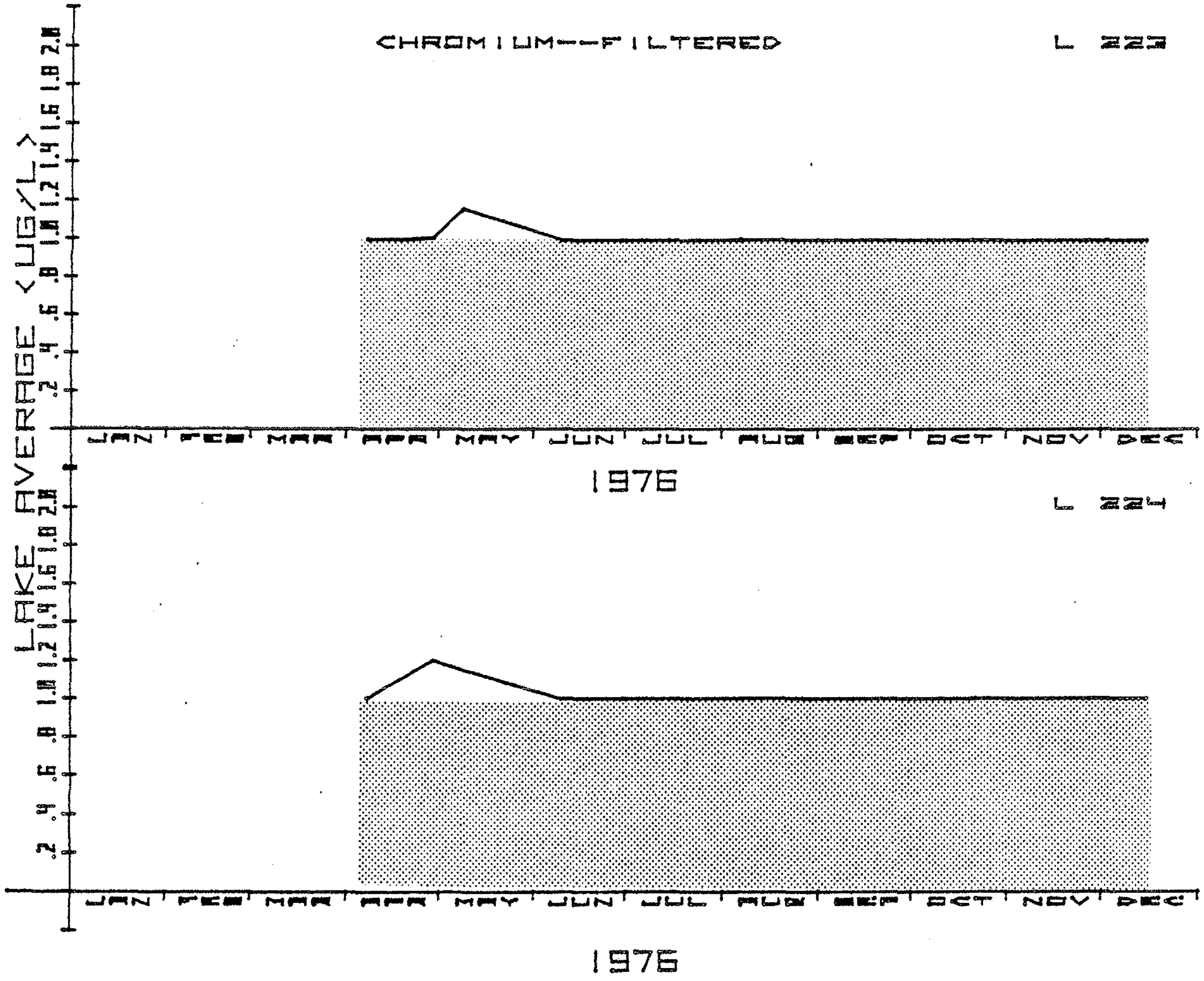


Figure 19. The volume-weighted concentrations of dissolved chromium in Lakes 223 and 224, 1976. Sample treatment was the same as for iron and manganese. Stippled areas represent the lower analytical detection limit.

Aluminum had an early July peak in both lakes (Figure 13), followed by a decrease in late summer and fall. The maximum average concentration in Lake 223 was 32 μg/L, while the minimum was 4 μg/L. The Lake 224 maximum was only 20 μg/L, although the minimum was very similar to that in Lake 223.

Zinc also had a midsummer peak in both lakes, reaching 3 μg/L in Lake 223 and 4 μg/L in Lake 224 in late July (Figure 14). Minimum values of 1.2 to 1.5 μg/L were reached during spring overturn. Fall and early winter values remained high, and there is some evidence for a winter increase beginning in Lake 223.

Copper and cobalt also peaked in summer, at the same time as aluminum (Figures 15 and 16). For most of the rest of the year, both elements were undetectable.

Lead, cadmium, and chromium were undetectable or very near the limit of detection in Lake 223 for the entire year (Figures 17 to 19). The pattern in Lake 224 was similar, except for lead, which increased late in the fall, for reasons which are not clear.

### 3.1.4 Nitrogen, Phosphorus, Carbon, and Silica

No significant differences between years were observed for any nitrogen or phosphorus fractions, either dissolved or particulate (Figures 20 to 25). For carbon, on the other hand, a considerable decline in dissolved organic carbon (DOC) was noted in 1975, but not in 1976 (Figure 26). Particulate carbons were similar in the two years (Figure 27). Reactive silicate concentrations showed neither seasonal nor year-to-year changes (Figure 28).

Conductivity in the summer of 1976 was higher than in the other two years (Figure 29). This increase in electrolyte is due to the large additions of acid to the lake.

## 3.2 LIGHT EXTINCTION AND SECCHI DEPTH

The transparency data for 1976 show increased light penetration over 1975. However, 1975 transparency was also higher than 1974 (Figure 30), so that it is difficult to attribute the change to acidification.

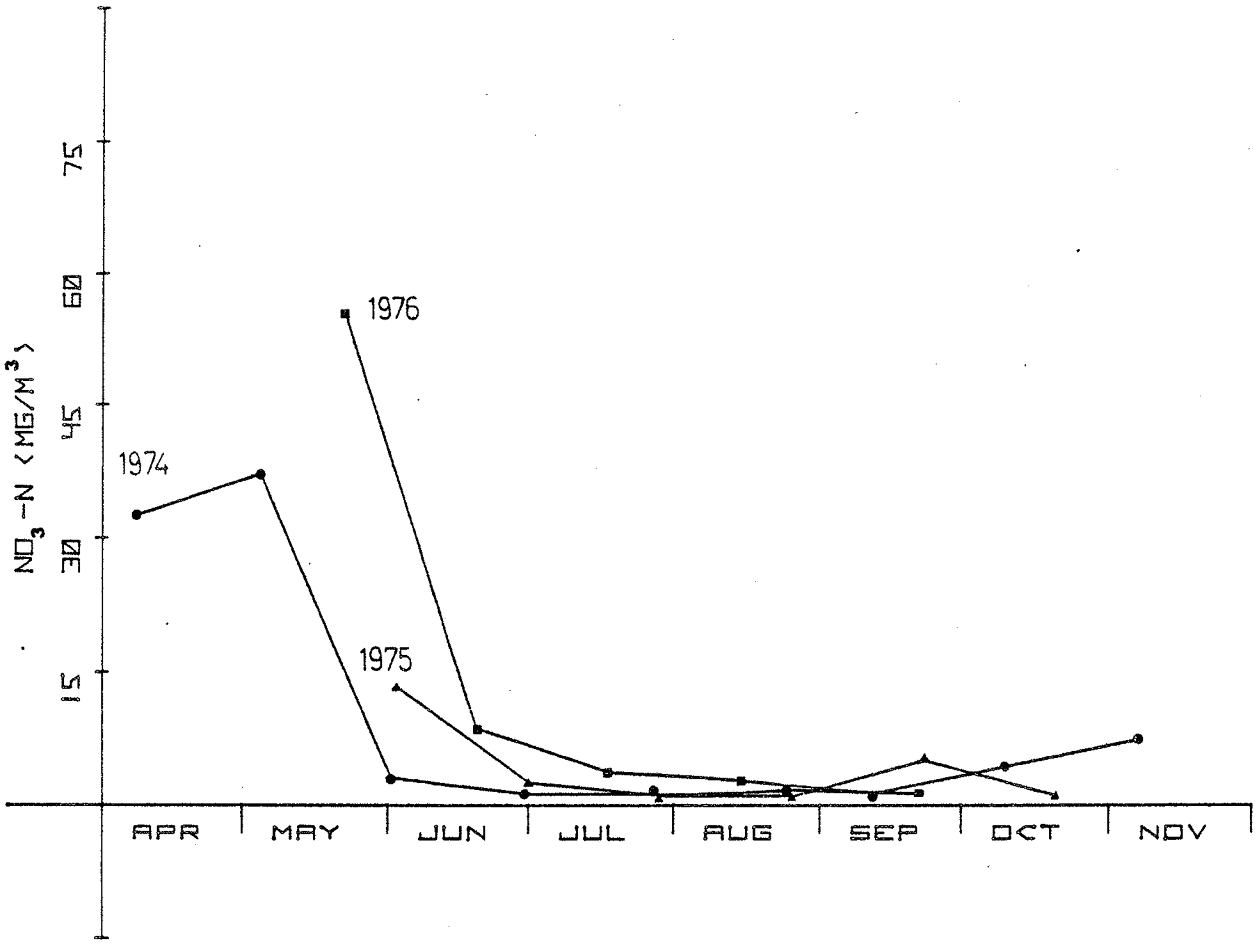


Figure 20. Average nitrate concentration in Lake 223, 1974-1976.

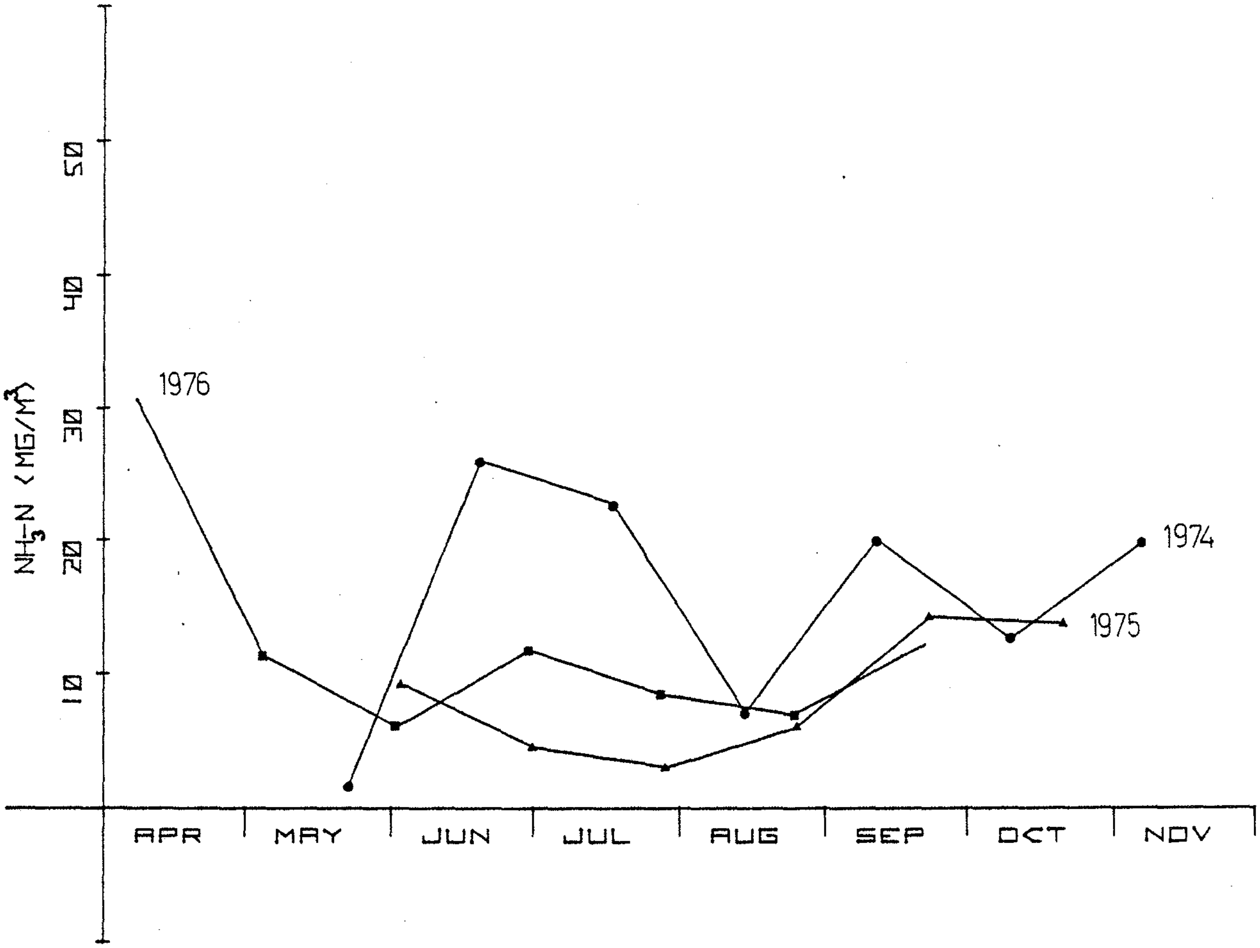


Figure 21. Average ammonium concentration in Lake 223, 1974-1976.

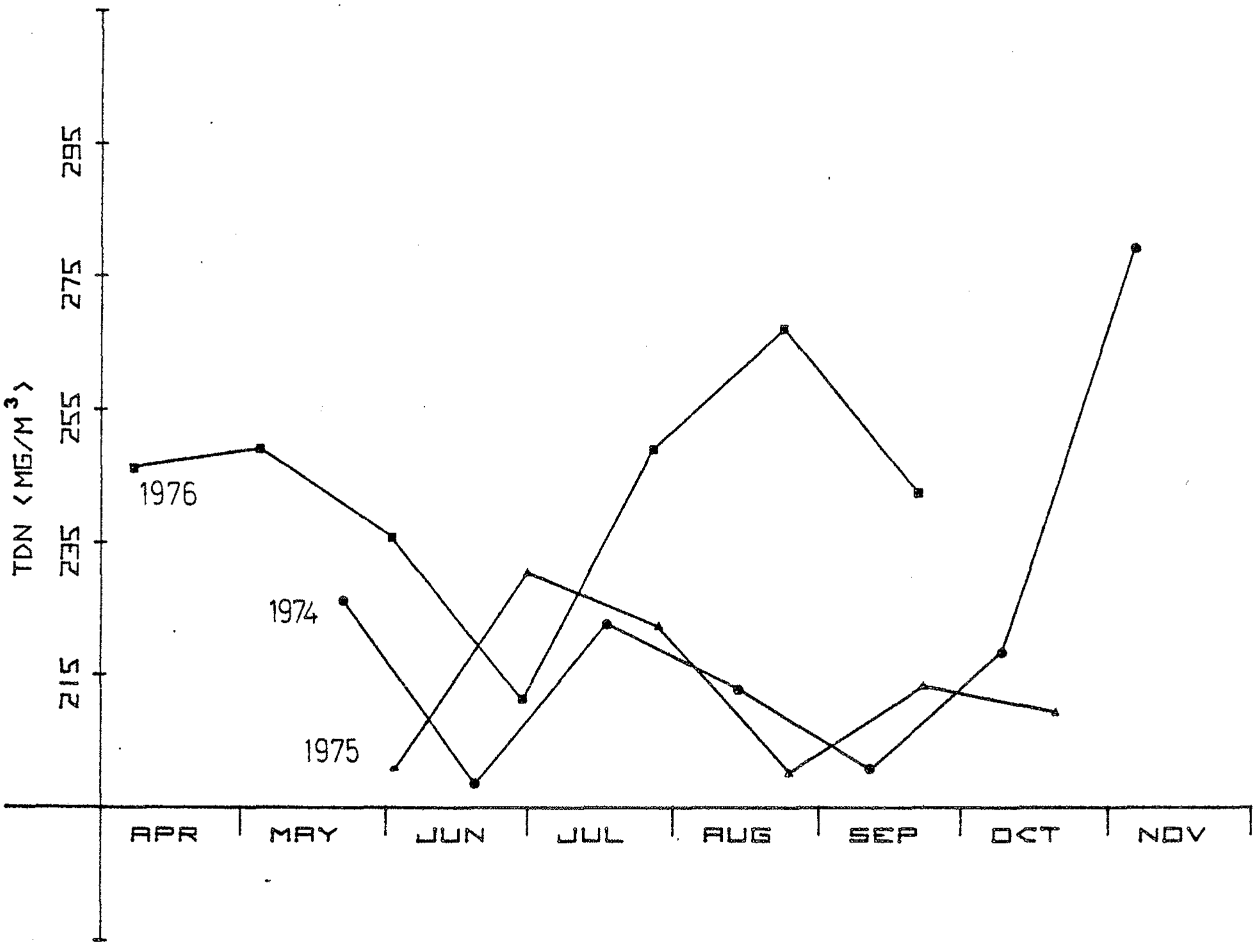


Figure 22. The average concentration of TDN in Lake 223, 1974-1976.

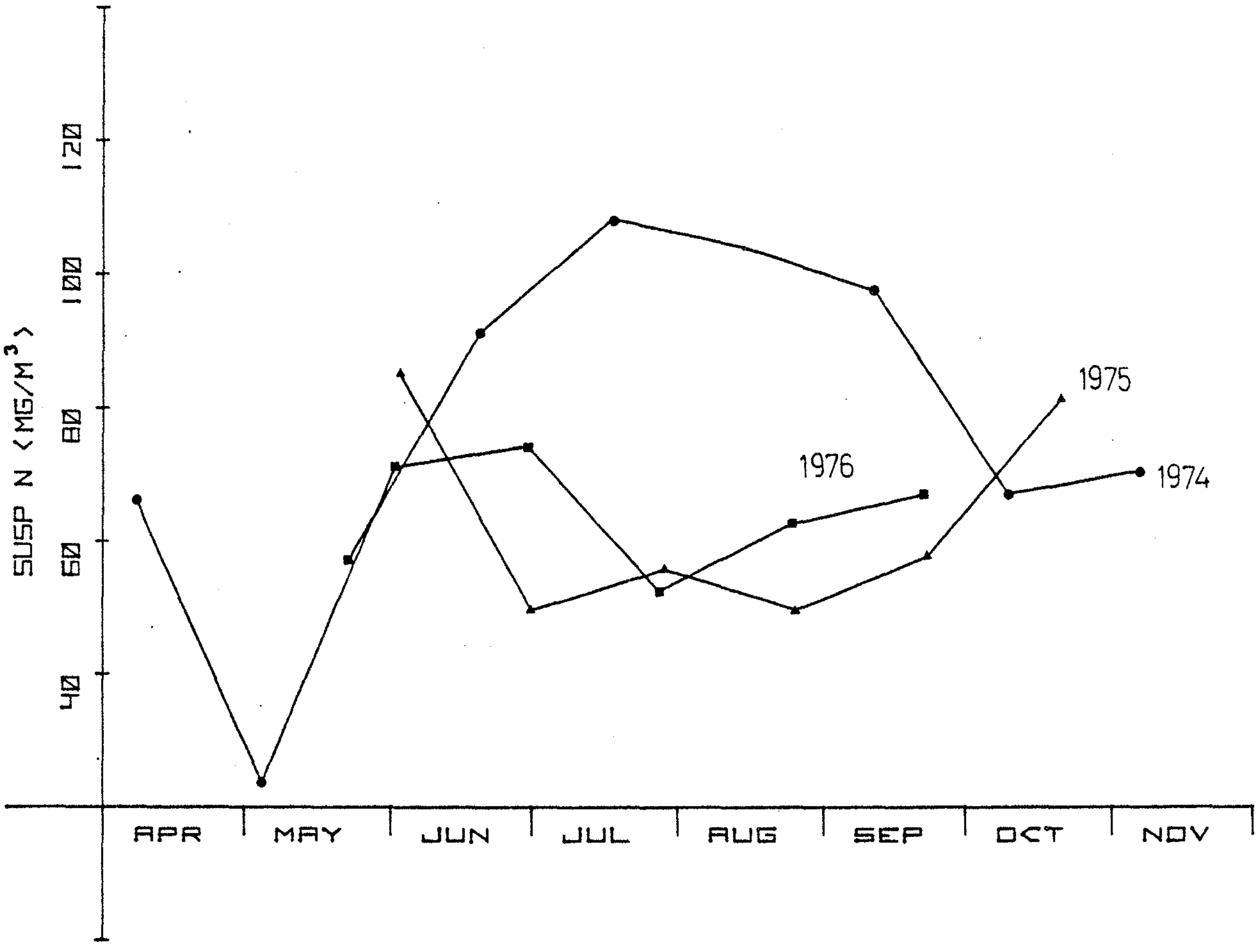


Figure 23. The average concentration of particulate nitrogen in Lake 223, 1974-1976.

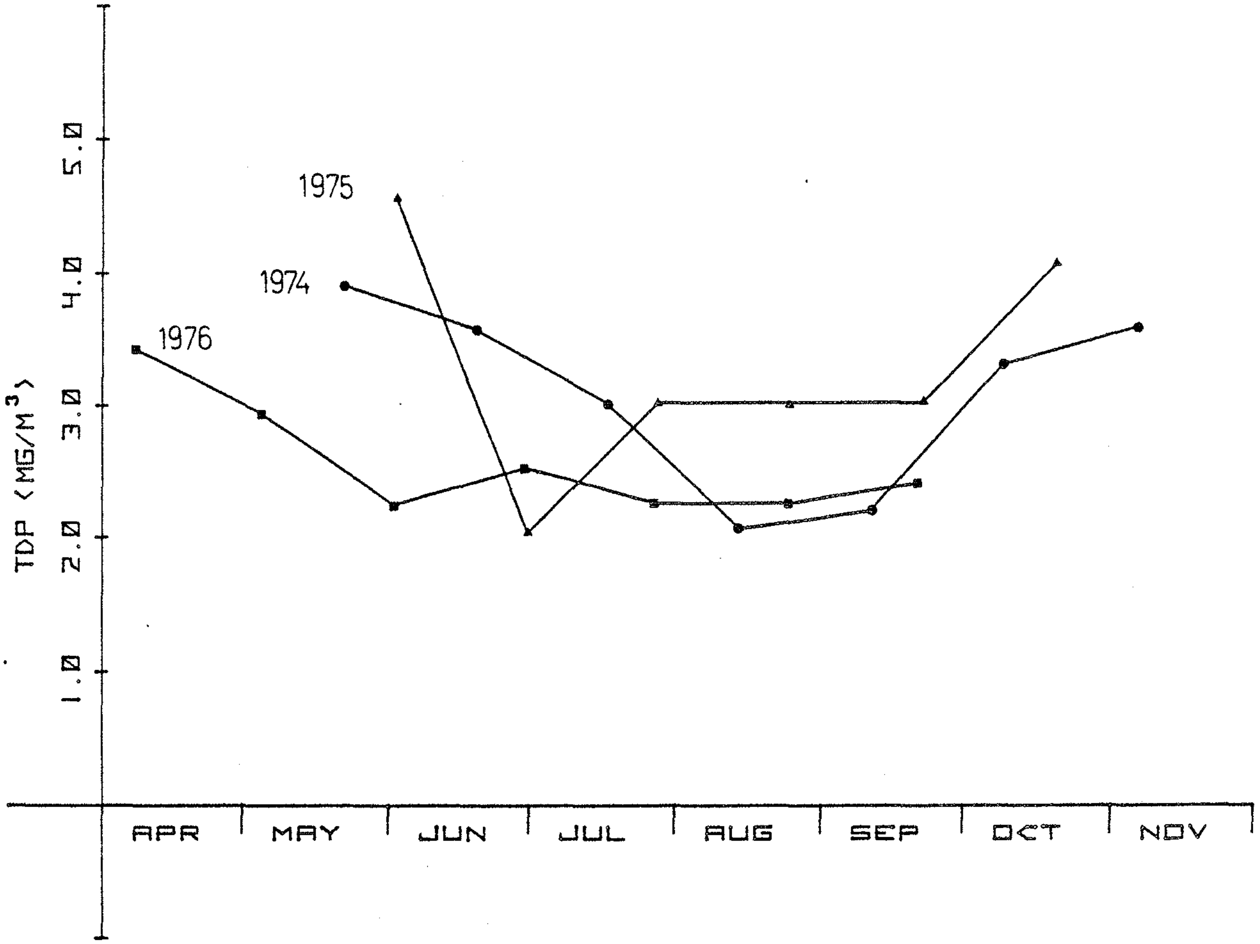


Figure 24. The average concentration of TDP in Lake 223, 1974-1976.

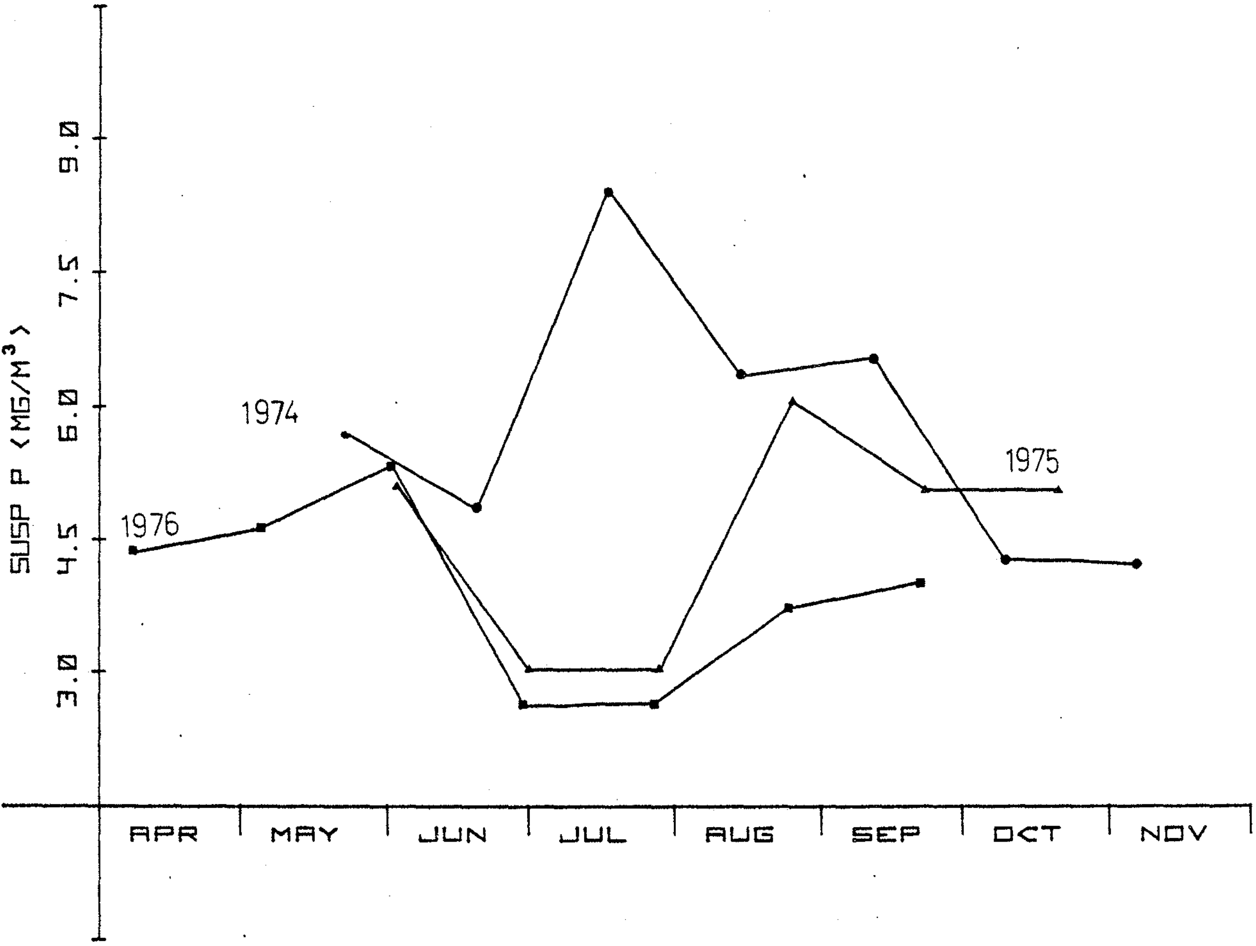


Figure 25. The average concentration of particulate phosphorus in Lake 223, 1974-1976.

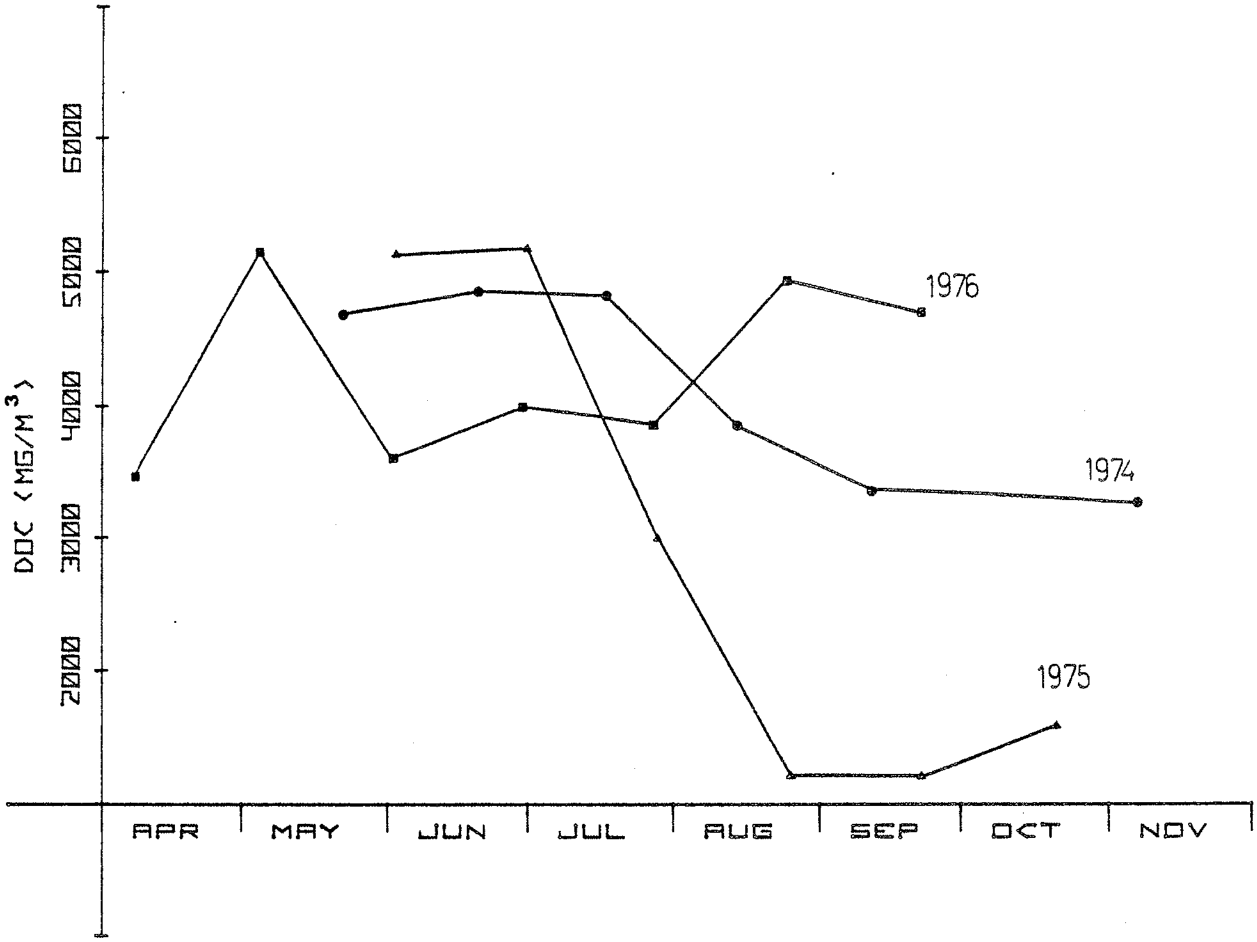


Figure 26. The average concentration of DOC in Lake 223, 1974-1976.

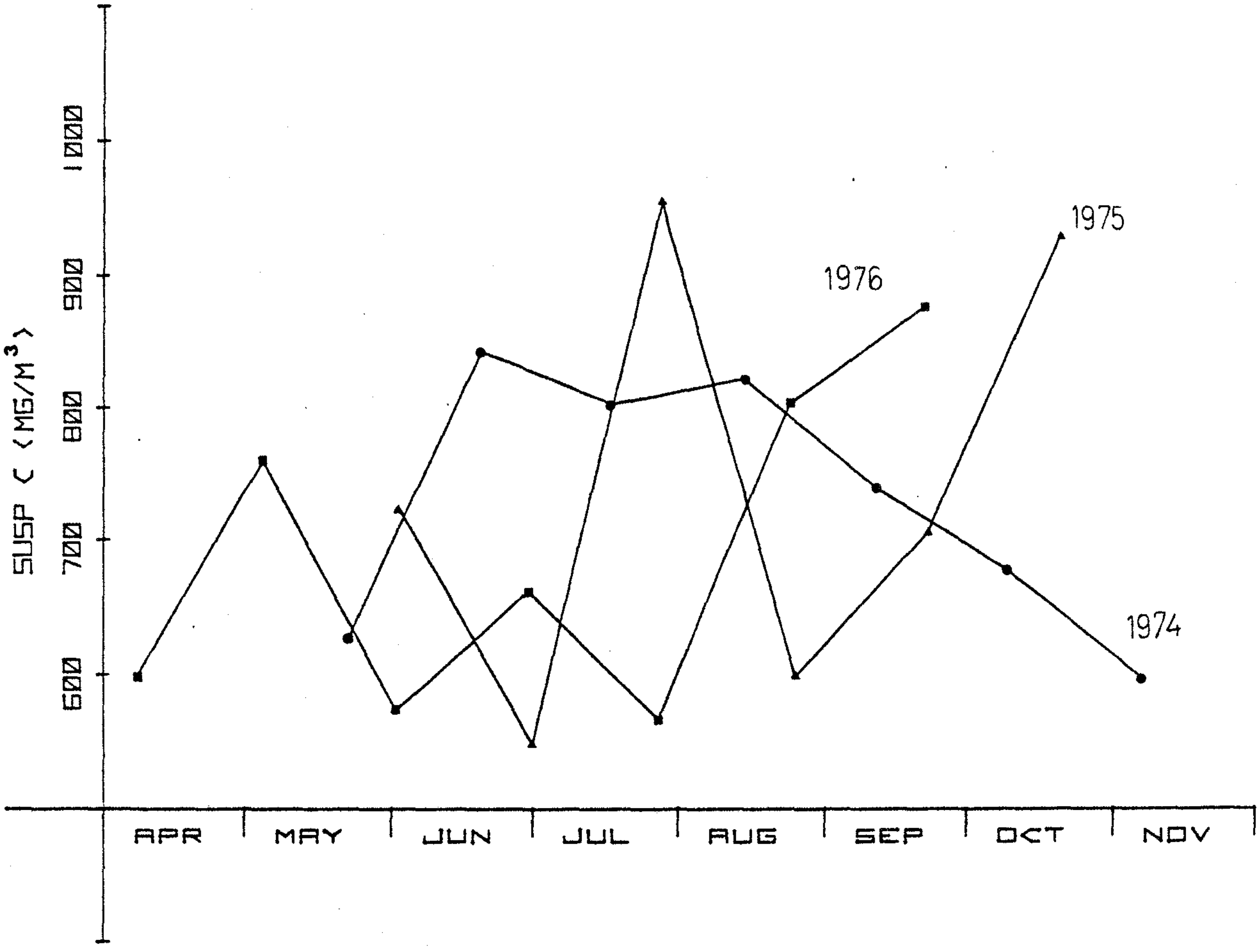


Figure 27. The average concentration of particulate carbon in Lake 223, 1974-1976.

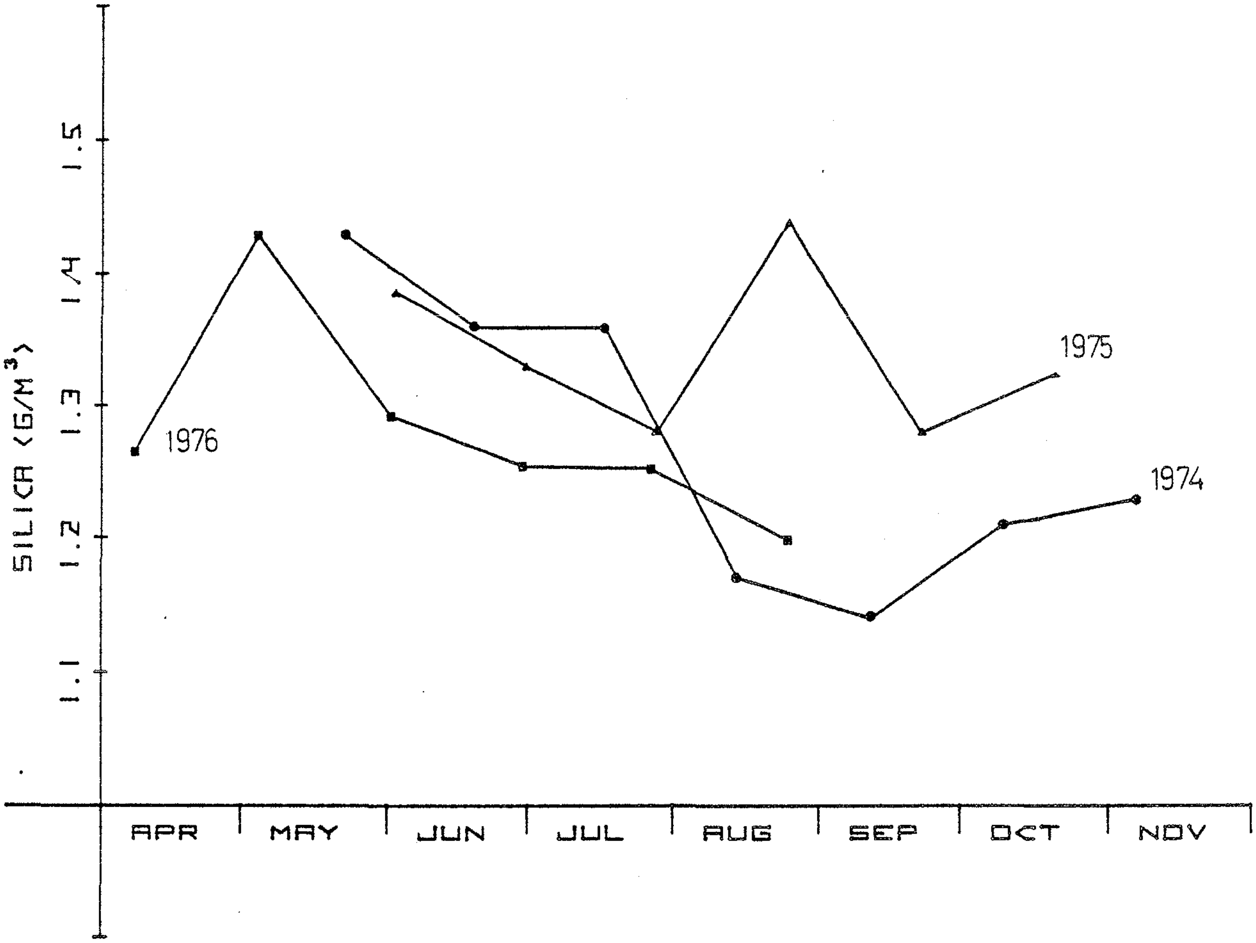


Figure 28. The average concentration of reactive silicate in Lake 223, 1974-1976.

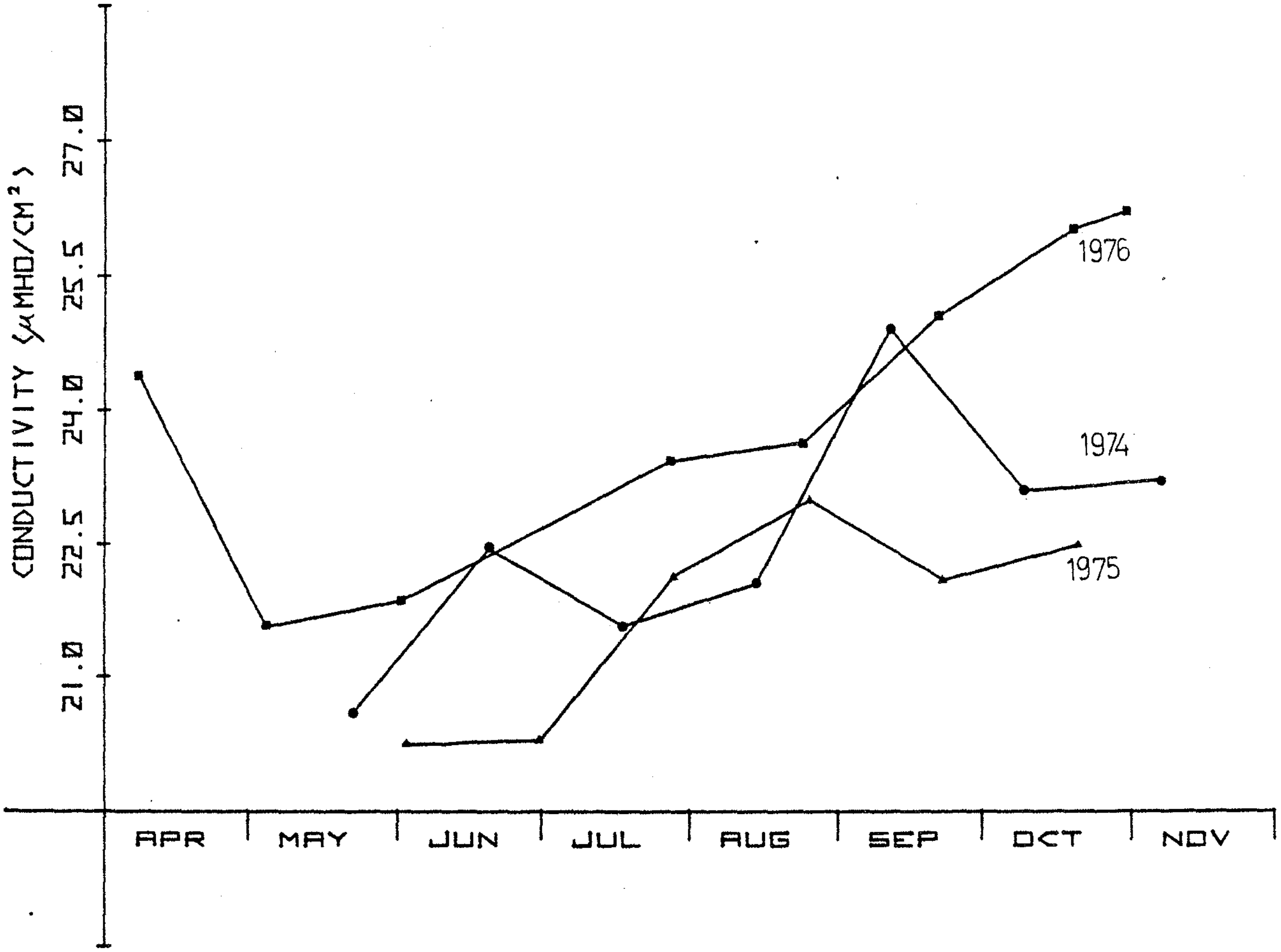


Figure 29. Volume-weighted average conductivities for Lake 223, 1974-1976. Values are μmho/cm$^2$ at 25°C.

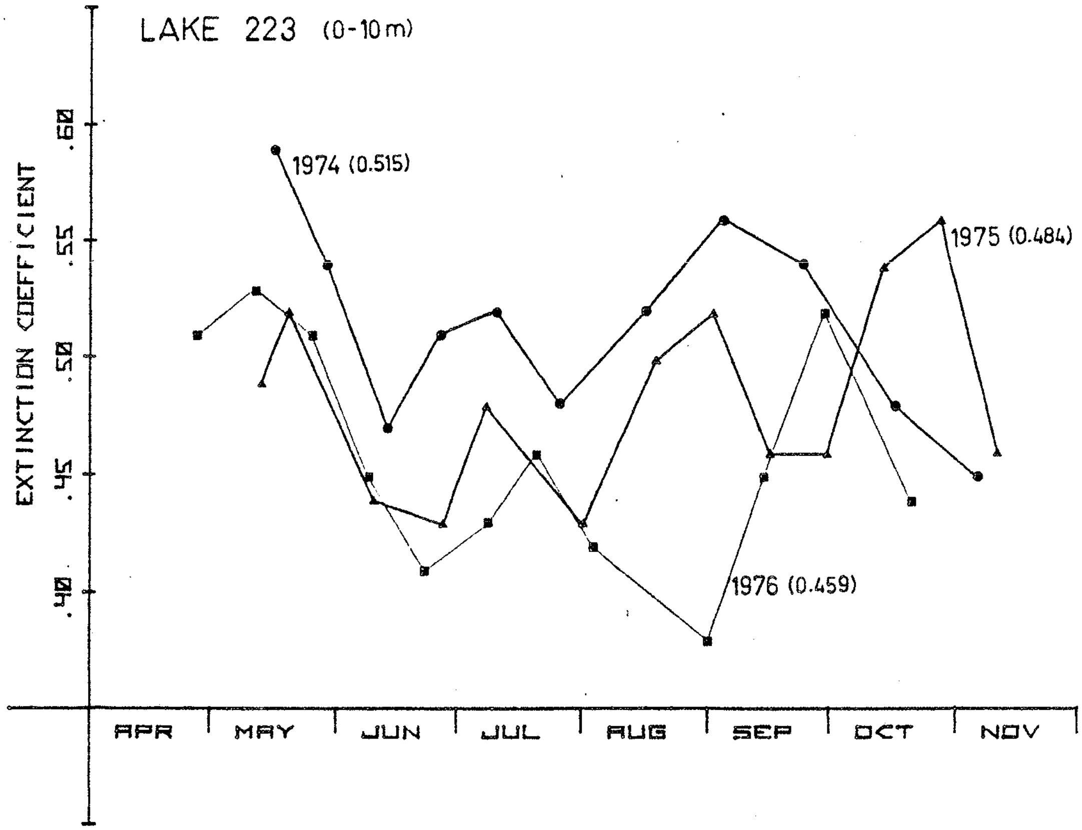


Figure 30. The vertical extinction coefficient for light in Lake 223, 1974-1976.

Secchi depths gave information similar to light extinction, with highest transparencies in early summer of 1976 (Figure 31).

## 3.3 TEMPERATURE

The thermocline of Lake 223 was at 5 to 6 m in all three years (Figure 32). Epilimnion maxima were 21 to 24°C; 1976 was slightly warmer than other years.

## 3.4 PHYTOPLANKTON PRODUCTION, RESPIRATION, AND CHLOROPHYLL CONTENT

Integrated annual phytoplankton production calculations for 1976 are still not complete, so this description will be limited to other properties of the photosynthetic system. The rate of production under optimum light conditions (P opt) in the incubator in 1976 was intermediate between 1974 and 1975 (Figure 33).

Respiration rates were measured for the first time in 1976, so year-to-year comparisons are not yet possible. The decline in respiration in later summer (Figure 34) is at the time when both epilimnion temperature and chlorophyll were highest (Figures 32 and 35), which seems unusual. Chlorophyll concentrations in 1976 were similar to other years.

Chlorophyll-depth distributions in 1976 were also similar to those of 1975 (Figures 36 to 39). The pattern is typical for ELA lakes.

## 3.5 PHYTOPLANKTON STANDING CROP

There was not major change in phytoplankton biomass or species composition in 1976, when compared to 1974 and 1975. Chrysophyceans dominated in all three years (Figure 40). The phytoplankton volume averaged about 1000 $g/m^3$ in all three years, and chlorophyll concentrations during the ice-free season ranged from 1.5 to 6 $mg/m^3$ (Figure 35).

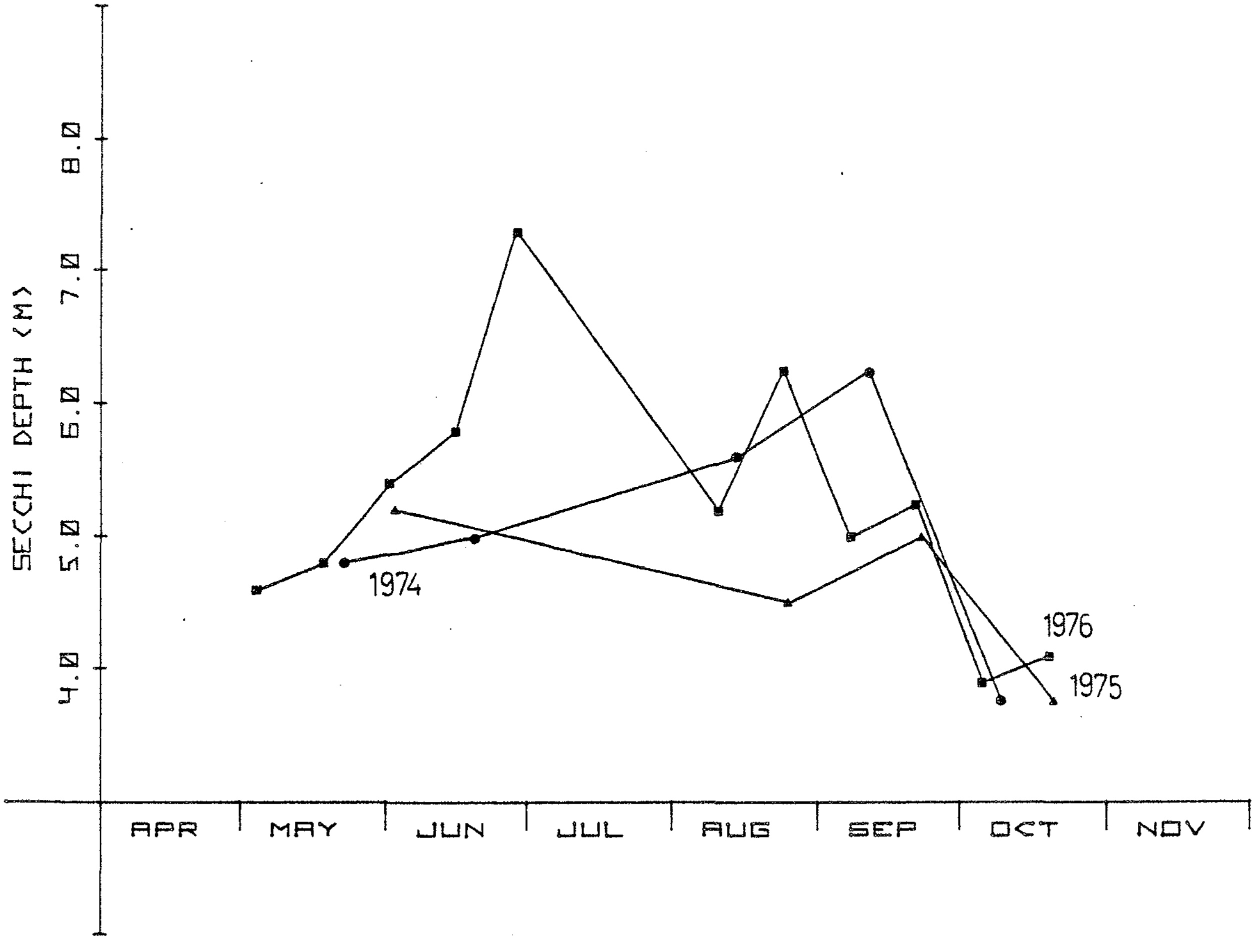


Figure 31. Secchi disc visibility (m) in Lake 223, 1974-1976.

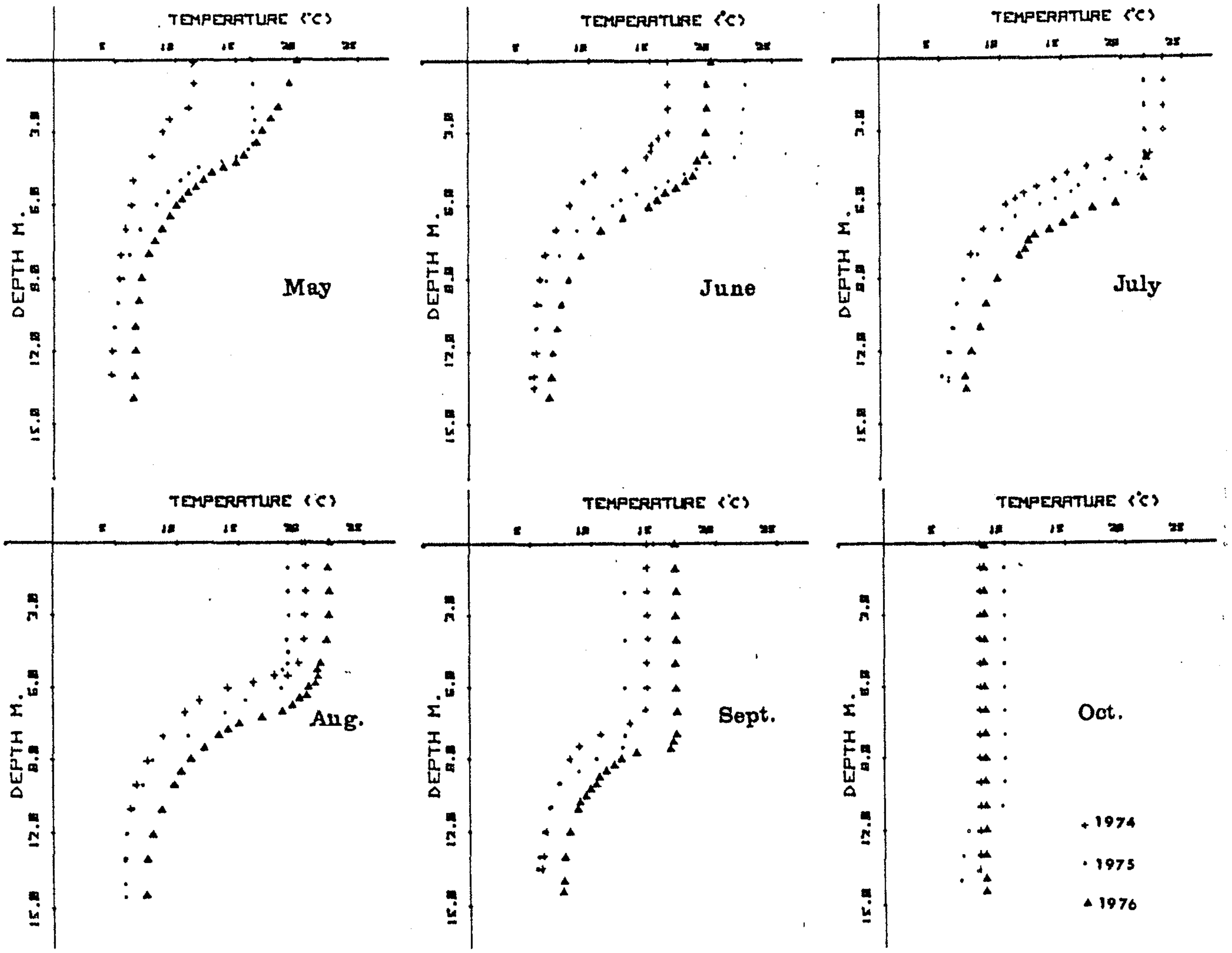


Figure 32. Temperature profiles (°C) of Lake 223, 1974-1976.

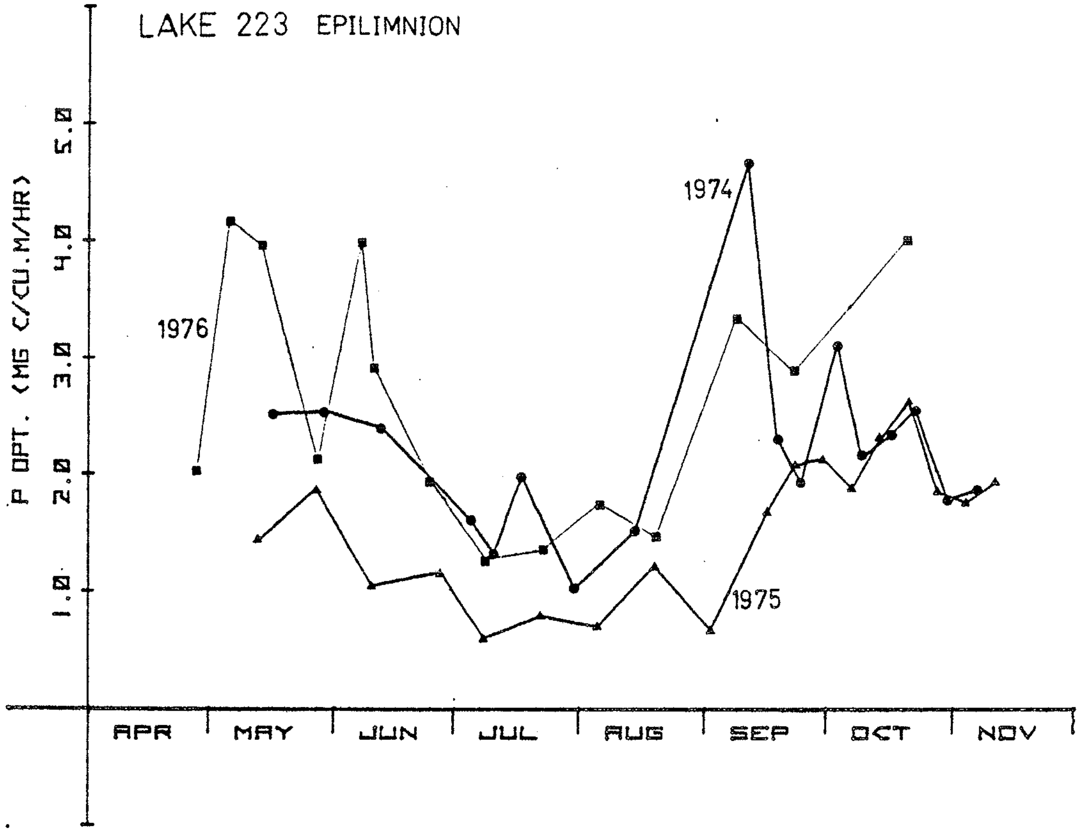


Figure 33. The rate of phytoplankton production under optimum light conditions (P opt) in the light incubator, 1974-1976 ice-free seasons.

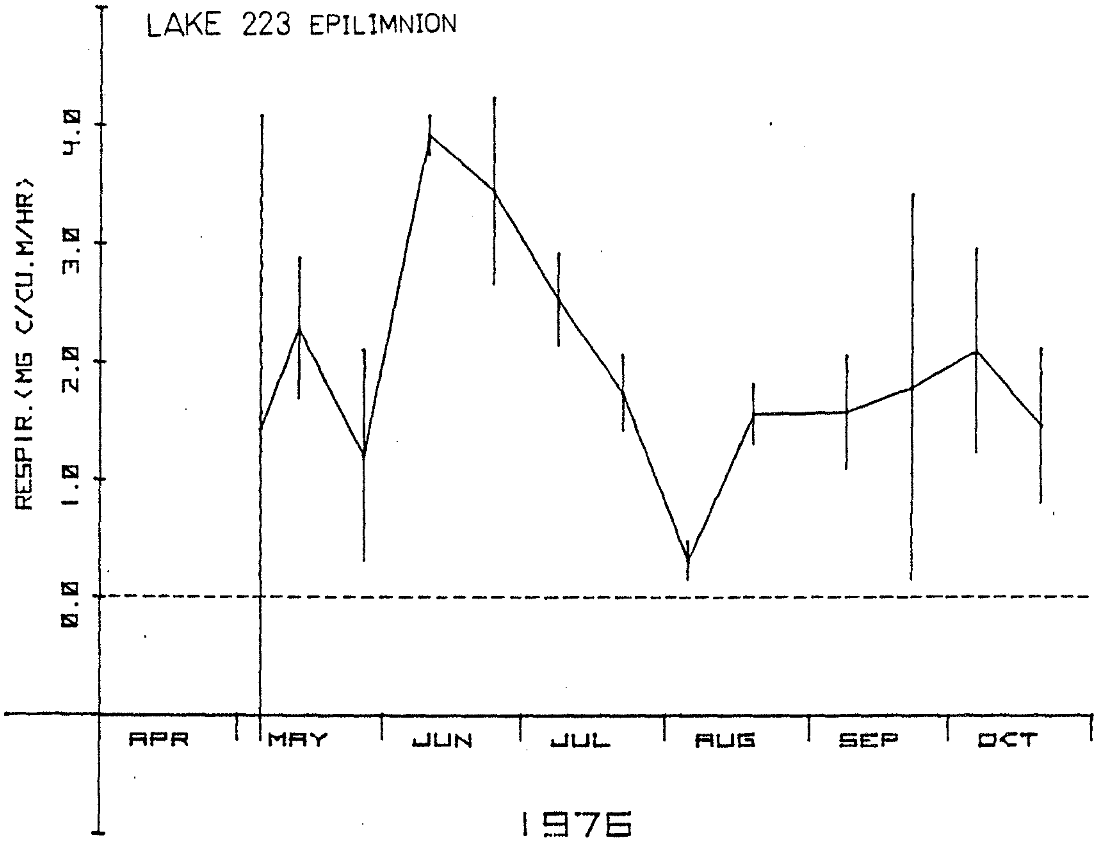


Figure 34. Phytoplankton respiration rates in 1976. The vertical bars represent measured extremes calculated from replicate samples. The line connects the mean values of these extremes.

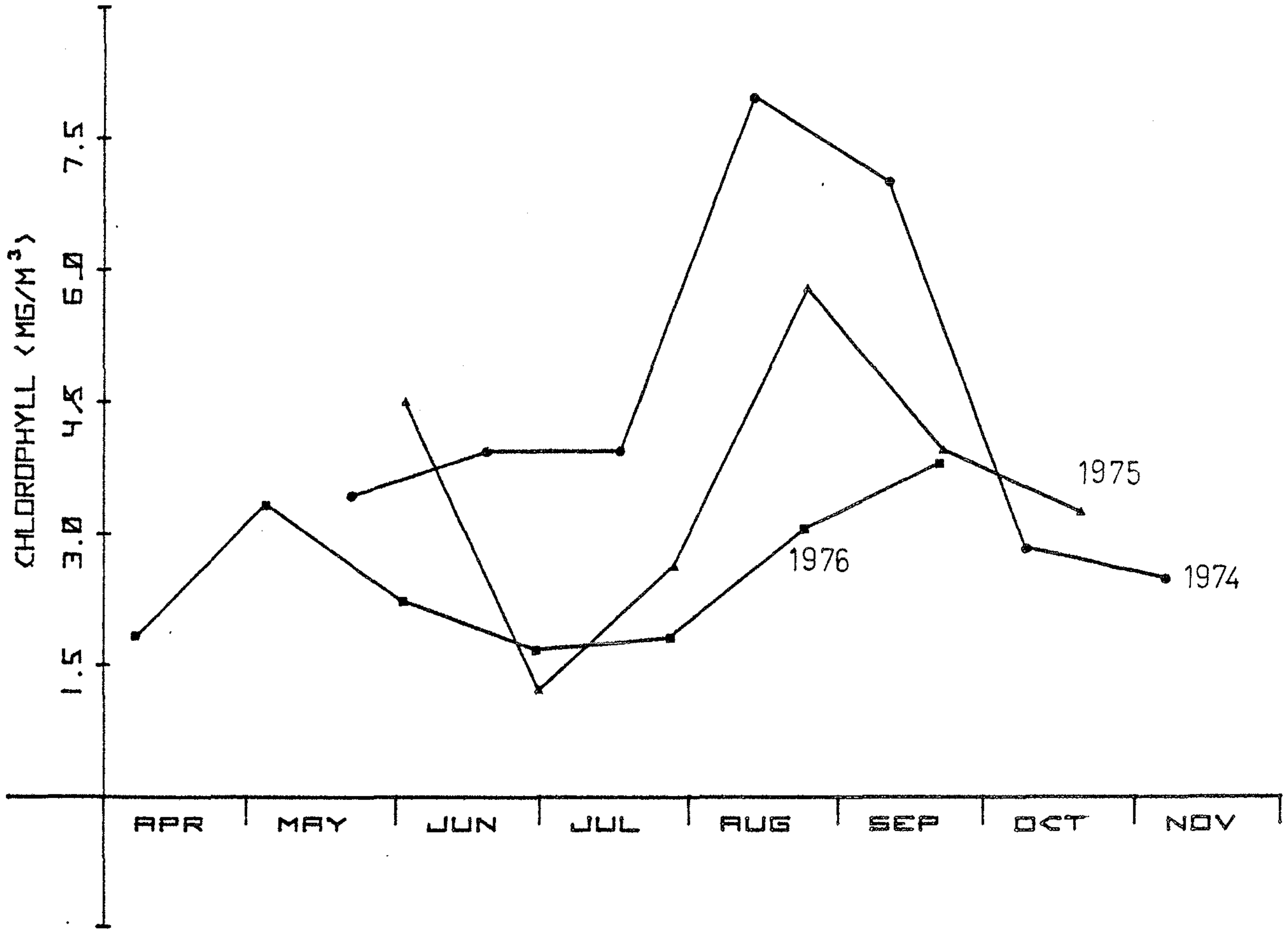
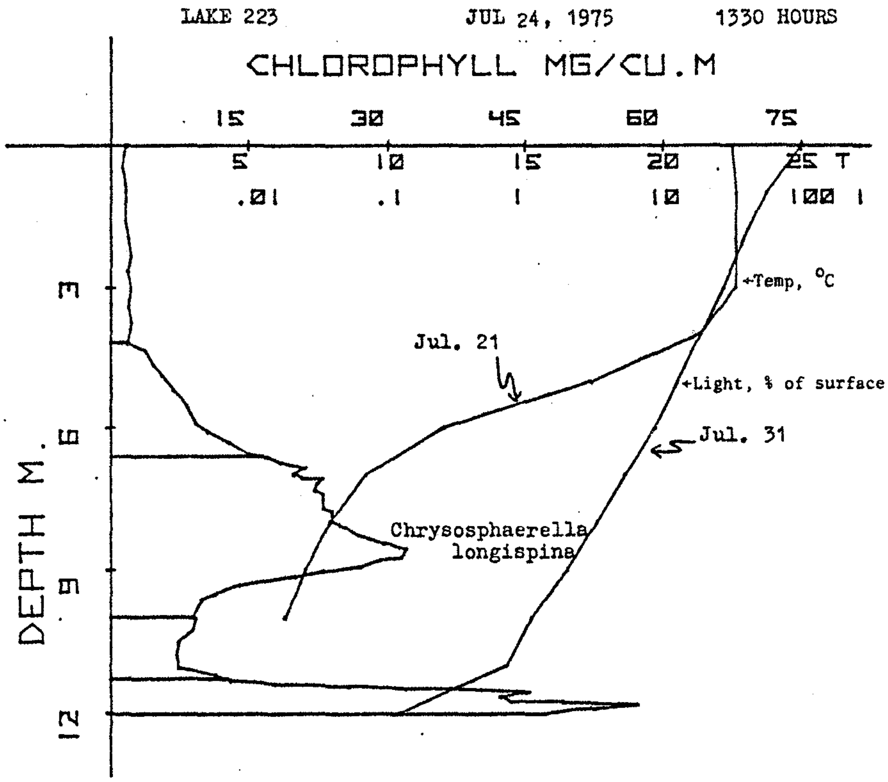


Figure 35. Average chlorophyll concentrations in Lake 223, 1974-1976.

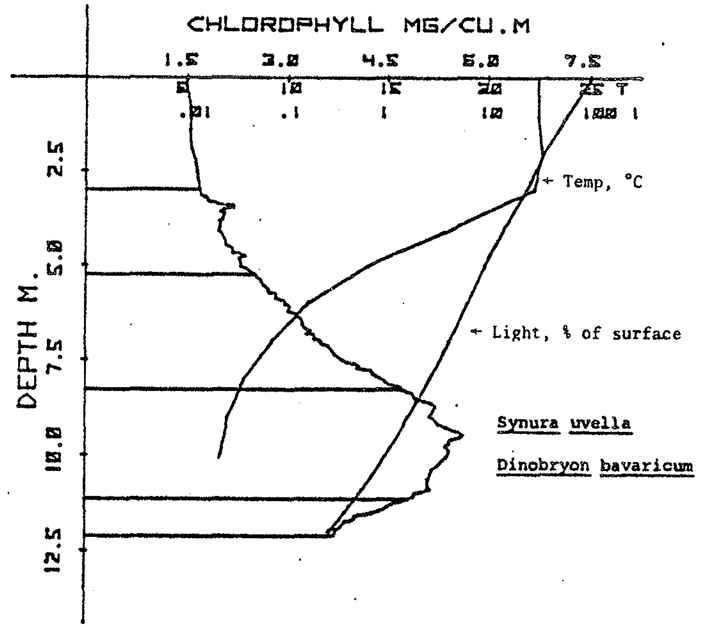


| Depth Range (m) | Volume ($10^5 m^3$) | Chlor. (kg) | Chlor. Conc. (mg/m$^3$) | % of Column Total |
|---|---|---|---|---|
| 0.00 TO 4.18 | 8.79 | 1.55 | 1.77 | 11.2 |
| 4.18 TO 6.60 | 3.87 | 3.00 | 7.74 | 21.6 |
| 6.60 TO 9.99 | 3.66 | 7.91 | 21.63 | 57.0 |
| 9.99 TO 11.28 | .78 | .66 | 8.38 | 4.7 |
| 11.28 TO 12.01 | .21 | .77 | 36.29 | 5.5 |
| TOTALS | 17.31 | 13.88 | AVG. 8.02 | |

SCALE 3X SLOPE .72 INTERCEPT -13.03 R .944 N 8

Figure 36. Chlorophyll-depth distribution in Lake 223, 24 July 1975. Temperature (T) and light (I) are also shown.

LAKE 223 JUN 8, 1976 1345 HOURS

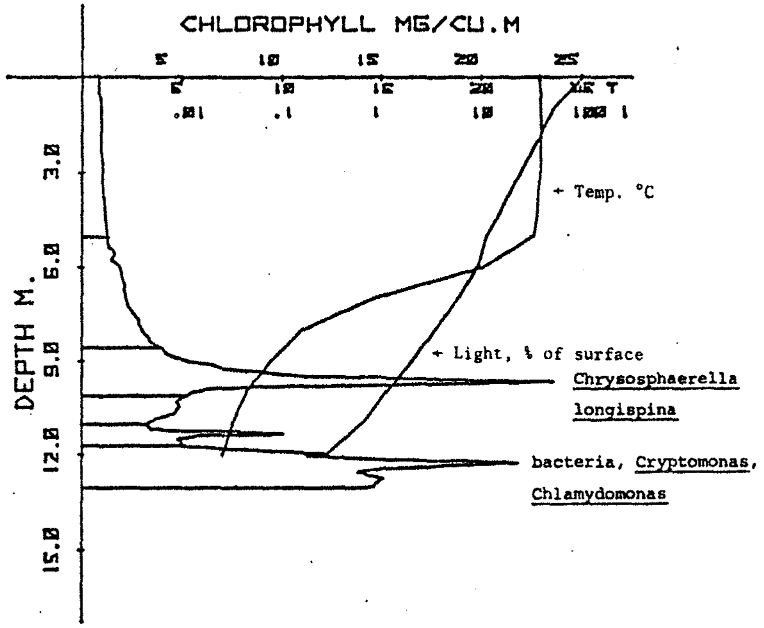


| DEPTH RANGE<br>M | VOLUME<br>10^5CU.M | CHLOR.<br>KG | CHLOR.<br>CONC.<br>MG/CU.M | %OF<br>COLUMN<br>TOTAL |
|---|---|---|---|---|
| 0.00 TO 2.99 | 6.57 | 1.03 | 1.57 | 22.5 |
| 2.99 TO 5.24 | 4.04 | .84 | 2.07 | 18.3 |
| 5.24 TO 8.28 | 4.09 | 1.37 | 3.36 | 30.0 |
| 8.28 TO 11.15 | 2.35 | 1.22 | 5.20 | 26.7 |
| 11.15 TO 12.11 | .29 | .12 | 4.20 | 2.6 |
| TOTALS | 17.33 | 4.58 | AVG. 2.64 | |

SCALE:316.0X SLOPE: .05 INTERCEPT: -.17 R: .948 N: 5

Figure 37. Chlorophyll-depth distribution in Lake 223, 8 June 1976. Temperature (T) and light (I) are also shown.

LAKE 223 JUL 27, 1976 1525 HOURS

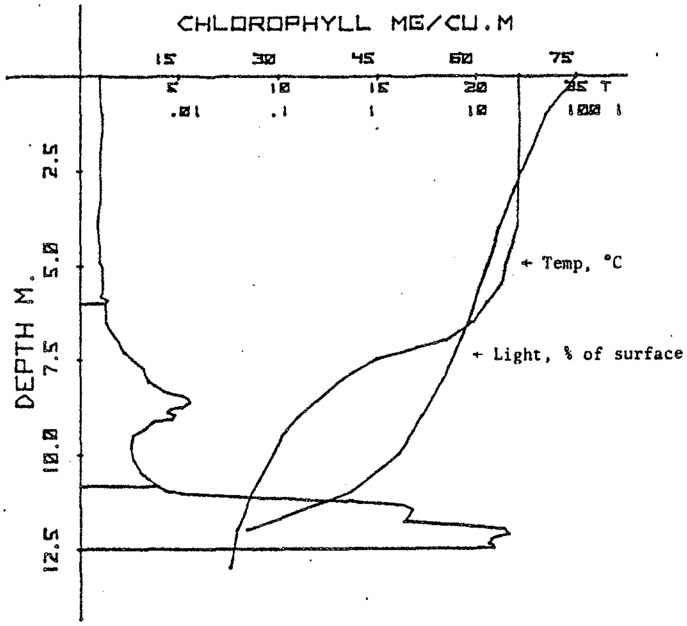


| DEPTH RANGE M | VOLUME 10^5CU.M | CHLOR. KG | CHLOR. CONC. MG/CU.M | %OF COLUMN TOTAL |
|---|---|---|---|---|
| 0.00 TO 5.04 | 10.28 | 1.09 | 1.06 | 27.3 |
| 5.04 TO 8.57 | 4.72 | 1.05 | 2.22 | 26.4 |
| 8.57 TO 10.13 | 1.42 | 1.20 | 8.45 | 30.2 |
| 10.13 TO 11.03 | .57 | .26 | 4.53 | 6.5 |
| 11.03 TO 11.74 | .26 | .14 | 5.14 | 3.4 |
| 11.74 TO 13.05 | .19 | .25 | 13.33 | 6.2 |
| TOTALS | 17.44 | 3.97 | AVG. 2.28 | |

SCALE:100.0X SLOPE: .19 INTERCEPT: -.33 R: .991 N: 6

Figure 38. Chlorophyll-depth distribution in Lake 223, 27 July 1976. Temperature (T) and light (I) are also shown.

LAKE 223 AUG 20, 1976 1020 HOURS



| DEPTH RANGE M | VOLUME 10^5CU.M | CHLOR. KG | CHLOR. CONC. MG/CU.M | %OF COLUMN TOTAL |
|---|---|---|---|---|
| 0.00 TO 6.01 | 11.80 | 3.77 | 3.19 | 37.2 |
| 6.01 TO 10.84 | 5.08 | 4.37 | 8.60 | 43.1 |
| 10.84 TO 12.50 | .50 | 1.99 | 40.04 | 19.7 |
| TOTALS | 17.38 | 10.13 | AVG. 5.83 | |

SCALE: 31.6X SLOPE: 2.32 INTERCEPT: -8.91 R: .987 N: 6

Table 39. Chlorophyll-depth distribution in Lake 223, 20 August 1976. Temperature (T) and light (I) are also shown.

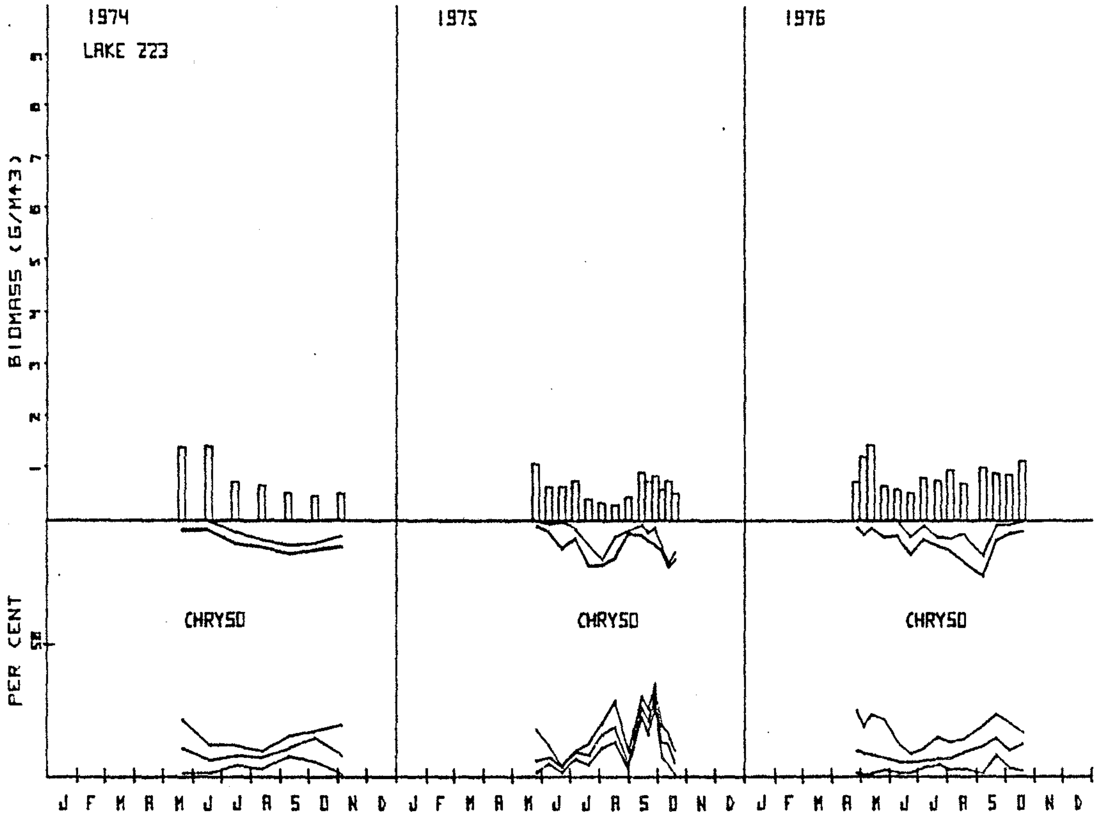


Figure 40. Phytoplankton volumes and composition by major groups, 1974-1976. Groups in the lower (percent) panel are (from top to bottom): Cyanophyta, Chlorophyta, Chrysophyta, Cryptophyta, and Peridineae.

There was, however, one slight change in the dominant species of Chrysophyceae observed during the ice-free season of 1976, when compared to the previous two years. In 1974 and 1975, the epilimnion of Lake 223 was dominated by *Botryococcus braunii, Chrysosphaerella longispina, Dinobryon bavaricum, D. sertularia, Chromulina* sp., and *Chrysococcus* sp.

In 1976, *Mallomonas elongata, Erkenia* sp., *Ochroomonas* sp., *Pseudokephyrion entezii,* and *Chromulina* sp. were commonly observed.

## 3.6 ZOOPLANKTON

Table 1 gives the species of zooplankton identified from Lake 223. The Crustacea were identical to those found in 1967 (Patalas 1971), and it seems likely that the species composition is quite stable. Rotifers were not examined in detail, and the list of species may increase if someone with considerable taxonomic experience reviews the material. Unfortunately, there is a dearth of both experts on rotifers and up to date keys for the group in North America.

Zooplankton counting is still in progress.

## 3.7 ZOOBENTHOS

Emergence of chironomids in 1976 was 4159 individuals/m$^2$, compared to 2358/m$^2$ in 1975. These figures are usual for year to year fluctuations in ELA lakes (I. Davies unpublished data).

Estimates of crayfish populations from baited minnow traps were much lower than those from direct counts or mark and recaptures done by SCUBA divers (Figure 41). The SCUBA estimates appeared to be the correct ones, because it was found that the minnow traps fished selectively for large males. Estimates for stratified areal estimates, employing direct counts on transects by SCUBA divers, and diver observations of animals caught and marked using minnow traps gave estimates of 138 561 and 162 522, respectively. Further information on crayfish trapping methods is presented elsewhere (Davies et al. 1977).

Table 1. Zooplankton species in Lake 223.

- Calanoida
  - *Epischura lacustris* Forbes
  - *Diaptomus minutus* Lilljeborg[a]
  - *Diaptomus oregonensis* Lilljeborg
- Cyclopoida
  - *Cyclops bicuspidatus thomasi* Forbes[a]
  - *Mesocyclops edax* Forbes
  - *Tropocyclops prasinus mexicanus* Kiefer
- Cladocera
  - *Daphnia galeata mendotae* Birge[a,b]
  - *Bosmina longirostris* (O.F. Muller)[a]
  - *Holopedium gibberum* Zaddach
  - *Diaphanosoma leuchtenbergianum* Fischer
- Rotifera
  - *Keratella cochlearis* (Gosse)[a]
  - *Keratella taurocephala* Myers
  - *Kellicottia longispina* (Kellicott)[a]
  - *Kellicottia bostoniensis* (Rousselet)
  - *Polyarthra vulgaris* Carlin
  - *Conochilus unicornis* Rousselet
  - *Asplanchna* cf. *A. priodonta* Gosse
  - *Synchaeta* spp.
  - *Trichocerca* spp.
  - *Polyarthra* cf. *P. remata* Skorikov
  - *Keratella* sp.

[a] Most common species.

[b] (x *thorata* Forbes; according to Patalas 1971).

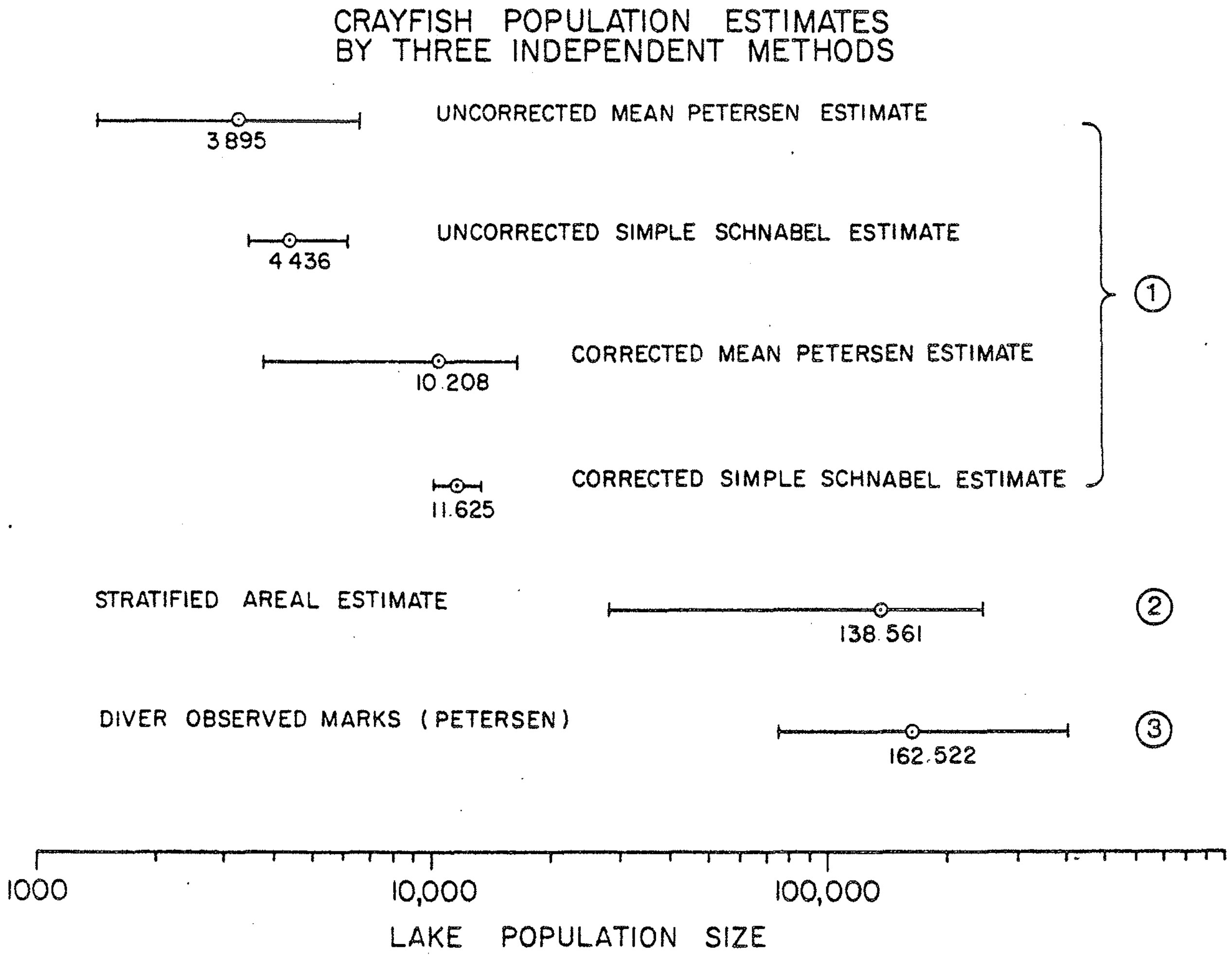


Figure 41. Means and standard deviations for various crayfish population estimates, as explained in the text.

## 3.8 FISH

The fathead minnow *Pimephales promelas* Rafinesque had a unimodel size distribution, with a mean fork length near 40 mm (Figure 42). A total of 181 lake trout [*Salvelinus namaycush* (Walbaum)] and 663 suckers [*Catostomus commersoni* (Lacépède)] were captured, measured, weighed, marked, and released. Plots of fin age in years versus mean fork length are shown in Figures 43 and 44.

Various population estimates for lake trout ranged from 113 to 525 and for white sucker from 119 to 2581. Mean estimates and standard deviations were 312 ± 130 and 898 ± 114 for trout and sucker, respectively. These estimates will improve as the number of fish marked and total number of recaptures increase.

## 3.9 TUBE RADIOISOTOPE EXPERIMENTS

Because of the wealth of information on these experiments results will be described only briefly here.

### 3.9.1 Arsenic

Seston showed little affinity for As-74 as arsenate. Amounts on seston were usually undetectable, and never over 25% of the total As-74 in any tube. In all tubes, 20 to 50% of the amount in the water column was found on the activated charcoal fraction and 50 to 80% on ion exchange resins. After 20 days, 30% of the As-74 had passed to the sediments. Very little was found in sedimentation traps, so it is presumed that the transfer to sediments occurred by direct absorption. The half-time for removal from the water column was 11 to 20 days.

### 3.9.2 Barium

Seston also showed little affinity for Ba-133 as barium chloride. The seston never contained over 13% of the amount in the water column. The charcoal fraction accounted for 90% of the total initially, decreasing to 30 to 50% after 15 days. The ion exchange

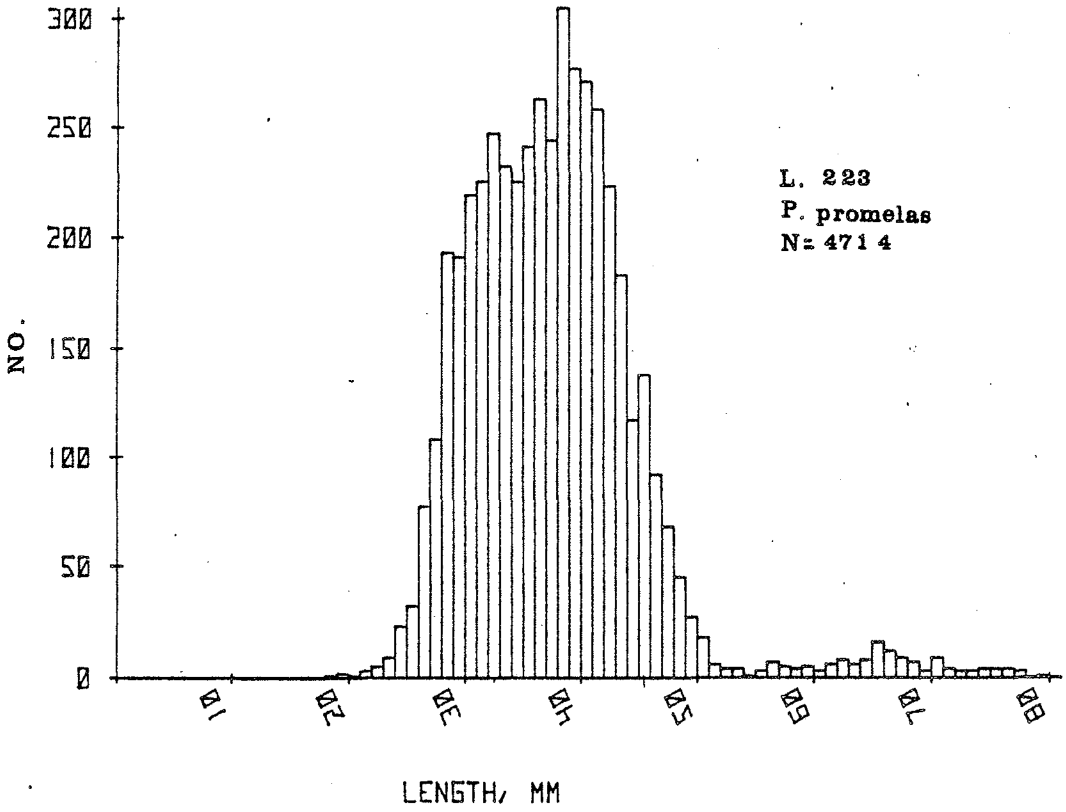


Figure 42. The size distribution of *Pimephales promelas*, the fathead minnow, in Lake 223, summer 1976.

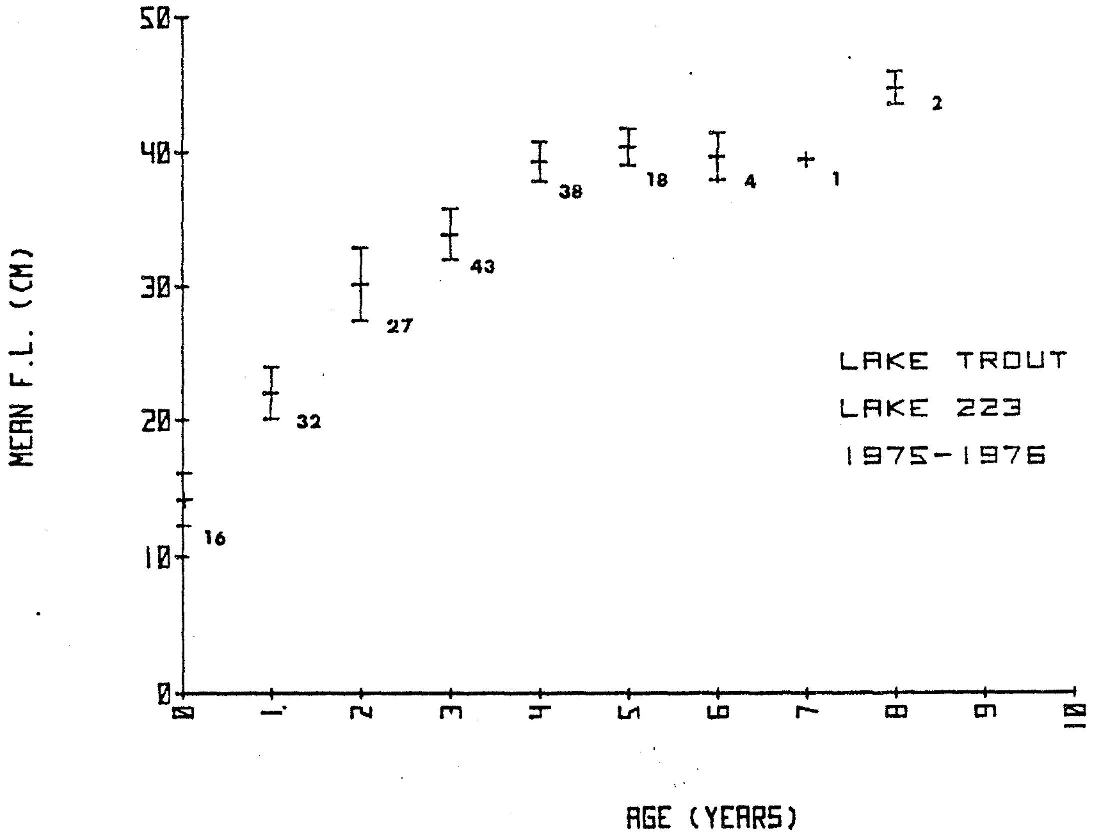


Figure 43. A fork length-age diagram for the lake trout, *Salvelinus namaycush*, in Lake 223, 1975-1976. Ages were determined from fin sections. Numbers are the number of individuals in each age class, and vertical bars are the standard deviations.

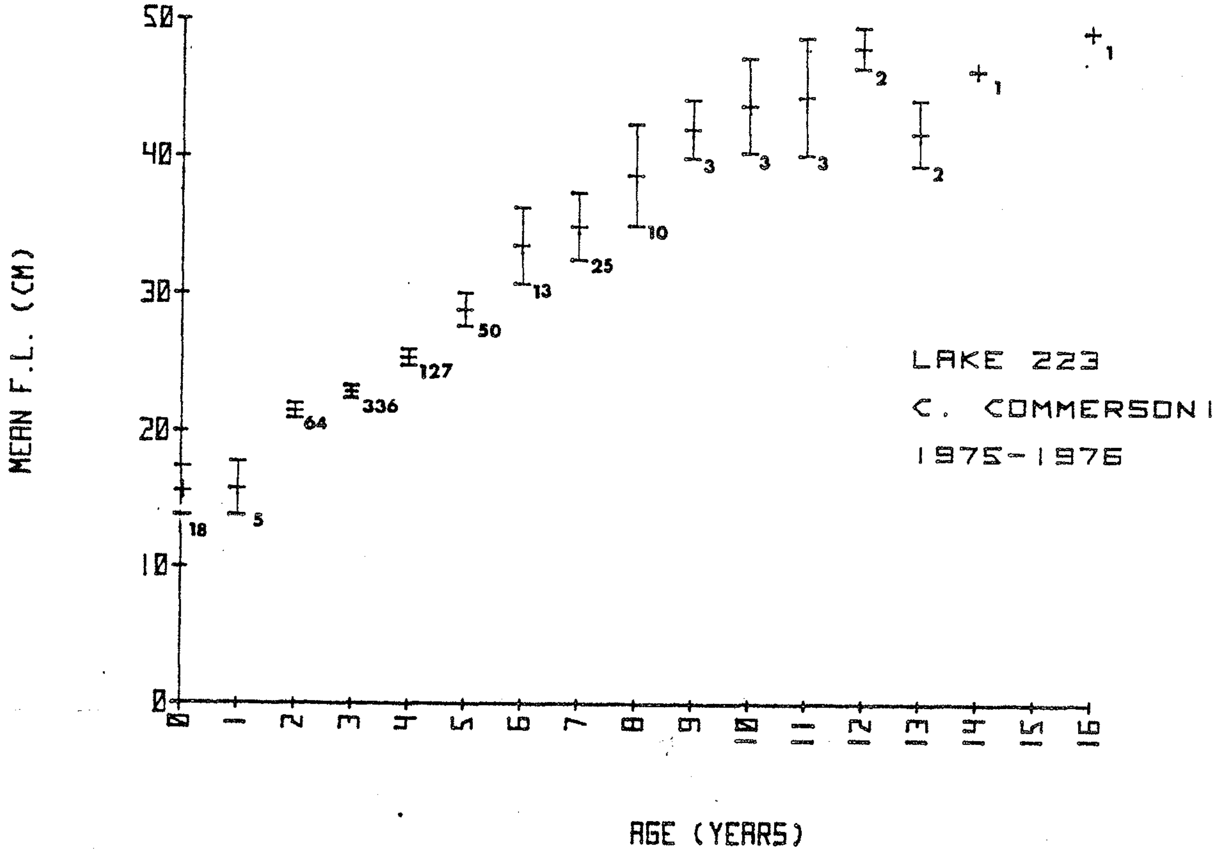


Figure 44. A fork length-age diagram for the white sucker, *Catostomus commersoni*, in Lake 223, 1975-1976. Ages were determined from fin sections. Numbers are the number of individuals in each age class, and vertical bars are the standard deviations.

fraction was quite constant at 10 to 20% throughout the first 20 days. As for As-74, 30 to 50% was in sediments, but little in sediment traps after 20 days, implying that adsorption to sediments was the primary mode of transfer. Half-time for removal was 11 to 18 days.

### 3.9.3 Cobalt

The behaviour of Co-60 cobaltous choride varied considerably with the pH of the tubes. One day after addition, nearly all of the Co-60 was in seston at lake pH (tubes D, E), about 40% at pH 6 (tube C) and only 10% at pH 5 (tube B). In all tubes, Co-60 in filtrate was largely taken out by charcoal. About 1% remained to be removed by ion resins at pH 7, ranging up to 5% at pH 5.

More Co-60 appeared in sedimentation traps at higher pH, as might be expected from the above results. At 20 days, almost all of the Co-60 was in the sediments, probably due largely to adsorption at low pH but settling of seston at higher pH. Half-time for removal was 6 to 8 days.

### 3.9.4. Chromium

Cr-51 as chromic chloride was unaffected by pH. Initially, 70% was found in seston and 30% on activated charcoal, with no significant fraction left for ion exchange. Removal was, however, faster at lower pH. The amounts remaining the water column after 20 days were 17% at pH 5, 32% at pH 6, and 37 and 58% at natural pH. Half-time for removal was 6 to 15 days.

### 3.9.5 Cesium

Cs-134 as cesium chloride showed no pH-dependent behaviour. Early in the experiment, 20 to 30% appeared on charcoal and 50 to 60% on ion exchange. Never more than 25% and usually only 3 to 5% was found on seston. Little Cs-134 was found in sediment traps as a result, but after 20 days 50 to 65% was in the sediments of all tubes. Direct adsorption therefore appeared to be the major means of transfer. Half-time for removal was 11 to 15 days.

### 3.9.6 Iron

At pH 6 and 7, 90% of the Fe-59 added as ferric chloride was in the seston after one day, with the remaining percentage on charcoal. After that, all iron in the water column was found in seston. Both sedimentation traps and cores picked up iron rapidly as a result.

At pH 5, however, only 30 to 50% of the Fe-59 was affiliated with seston, with the remainder on charcoal. Half-time for removal was 5 to 11 days.

### 3.9.7 Mercury

Hg-203, added as mercuric chloride too, was also unaffected by pH. Typically, 50 to 75% of that in the water column was in seston, 20 to 40% on charcoal, and 5 to 10% on ion exchange resin. After 20 days only 24 to 37% remained in the water column. Sediment traps were about half as active as cores, so that direct adsorption is implicated. Half-time for removal was 10 to 16 days.

### 3.9.8 Iodine

I-131 as iodine appeared to be converted rapidly to free $I_2$. This reaction is favoured at lower pH. On the few occasions when iodine was detectable it was on the charcoal or ion-exchange fractions. None was ever found in cores or sediment traps. The half-time in the water column appeared to be >20 days, but data are uncertain due to the above problem and the short (8 day) half-life.

### 3.9.9 Manganese

The behaviour of Mn-54 as manganous chloride was highly pH dependent, but the unacidified tubes also were considerably different. At pH 5, almost all of the Mn-54 stayed in filtrate. Never more than 4% was on seston. Initially, 80% was taken out by charcoal, with 16 to 20% by ion exchange resins. At pH 6, the distribution was 20% in seston, 60% on charcoal, and 20% on resins. In one unacidifed tube, 99% of the Mn-54 was retained on seston. In the other, seston and charcoal retained equal amounts, with 10% on the ion exchange fraction.

In all cases, Mn-54 appeared in both sediment traps and cores, so that direct adsorption by sediments must have taken place. Half-time for disappearance from the water was 6 to 12 days. Disappearance was slower at low pH.

### 3.9.10 Selenium

Se-75 as sodium selenate showed no pH dependent behaviour. Initially it was equally distributed between seston and charcoal, with only 5 to 15% on ion exchange. Over 20 days, the percent in seston decreased to about 15%. Little Se-75 appeared in sediment traps. After 20 days, 30 to 60% was in cores, with a higher percentage at pH 5 and 6 than at natural pH. The half-time for selenium in the water column was 15 to 25 days.

### 3.9.11 Thorium

Th-228 as thorium nitrate shifted from seston to charcoal as the pH decreased. At natural pH, the distribution was about equal; at pH 5 it had decreased to 2:1 (charcoal/seston). Little or no Th-228 was found in the ion exchange fraction. In contrast to Se-75, the percentage in particulate material decreased more slowly with time than did the charcoal. Th-228 appeared both in sediment traps and cores. Its half-time in the water was 10 to 12 days.

### 3.9.12 Vanadium

The behaviour of V-48 as vanadyl chloride showed no dependence on pH. Fractionation was erratic, with 27 to 82% in seston, 22 to 47% on charcoal, and 18 to 24% on ion exchange. V-48 was found in both sediment traps and on cores. The amount in cores was always greater than in traps, indicating adsorption. The half-time for removal from the water was 3 to 13 days.

### 3.9.13 Zinc

Zn-65 as zinc chloride was highly pH dependent. None was ever found in the ion exchange fraction, but a shift from seston to charcoal as pH decreased was apparent. The amount in seston decreased from 50% at natural pH to 10% at pH 5.

Less zinc was found in either sediment traps or cores at pH 5 than at higher pH. The half-time for disappearance from the water was 8 to 13 days, with much slower disappearance at low pH.

Table 2 summarizes half-times for removal of various isotopes from the water column of the lake.

## 3.10 STABLE METALS IN TUBES

### 3.10.1 Zinc

Zinc increased moderately at pH 6 and greatly at pH 5 (Figure 45). A maximum of 350 μg/L was recorded in mid-autumn in the pH 5 tube. The only possible source of the zinc or other metals was from lake sediments.

### 3.10.2 Manganese

Manganese increases were similar to those observed for zinc (Figure 46). A maximum of approximately 200 μg/L was observed.

### 3.10.3 Iron

Iron concentrations at pH 5 and 6 were higher than at natural pH (Figure 47). Maximum concentrations were low, never over 32 μg/L.

### 3.10.4 Aluminum

Aluminum increased greatly at pH 5, but not at all at pH 6 (Figure 48). This is to be expected from the solubility of aluminum at different pH's.

Table 2. Half-times (days) for disappearance of isotopic tracers from waters of 10 m diameter tubes at different pH values, summer 1976. All data are corrected for radiodecay.

| Isotope | pH 5 | pH 6 | pH 7-7.5 | pH 7-7.5 |
|---|---|---|---|---|
| Se-75 | 21.7 | 15.4 | 25.0 | 25.1 |
| Hg-203 | 16.3 | 11.1 | 10.1 | 12.8 |
| Cr-51 | 6.4 | 14.7 | 11.2 | 13.5 |
| Cs-134 | 13.5 | 10.6 | 14.7 | 14.2 |
| Fe-59 | 8.6 | 5.5 | 10.9 | 7.1 |
| Zn-65 | 12.7 | 9.6 | 8.5 | 8.6 |
| Co-60 | 8.0 | 6.2 | 6.8 | 6.6 |
| Ba-133 | 14.6 | 10.9 | 18.5 | 15.0 |
| Th-228 | 11.1 | 10.5 | 9.6 | 11.5 |
| As-74 | 11.3 | 13.0 | 19.9 | 12.4 |
| Mn-54 | 11.6 | 9.6 | 6.6 | 7.0 |
| V-48 | 5.4 | 5.8 | 12.7 | 7.9 |

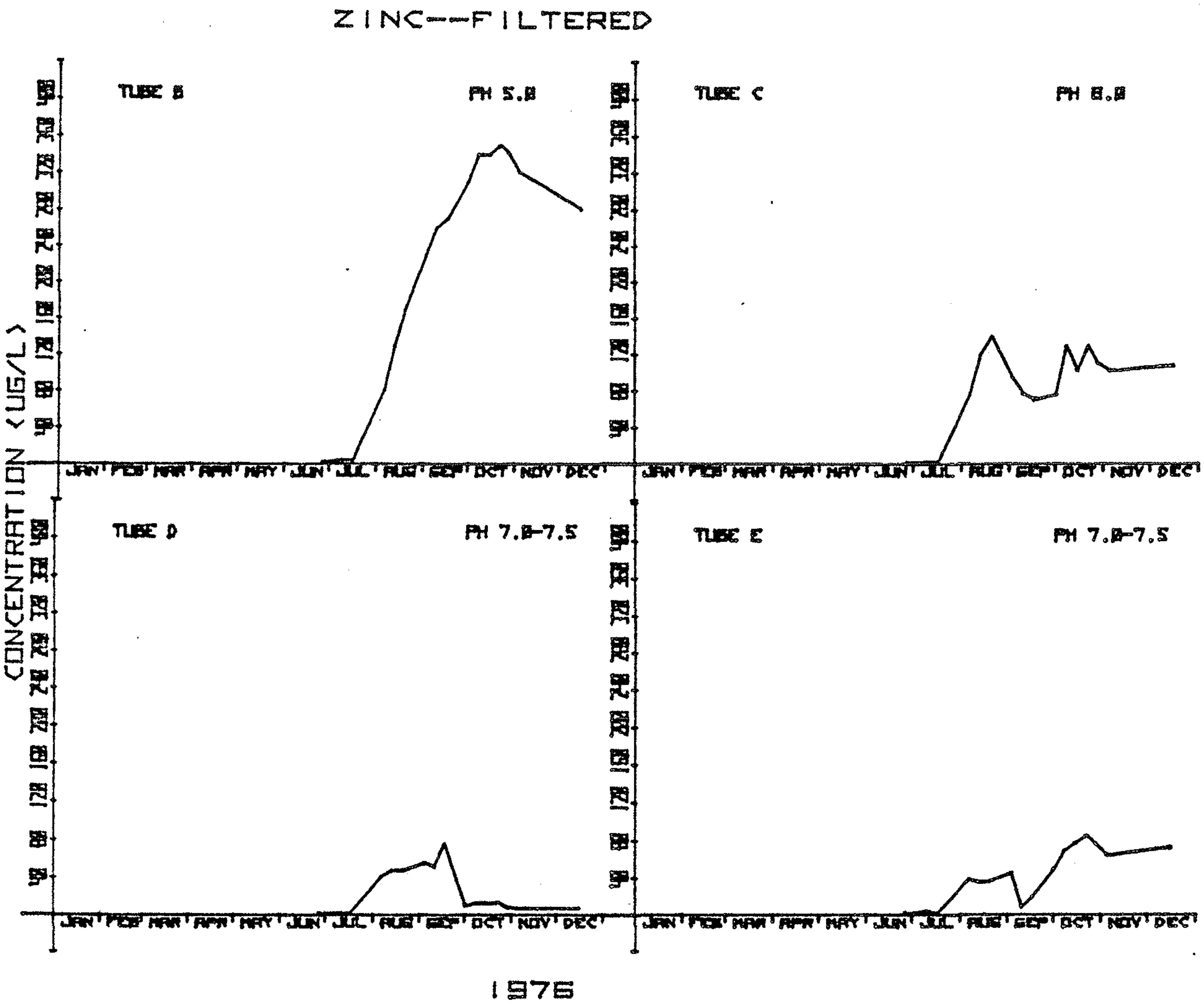


Figure 45. Concentrations of dissolved zinc in 10 m diameter tubes at different pH, summer 1976.

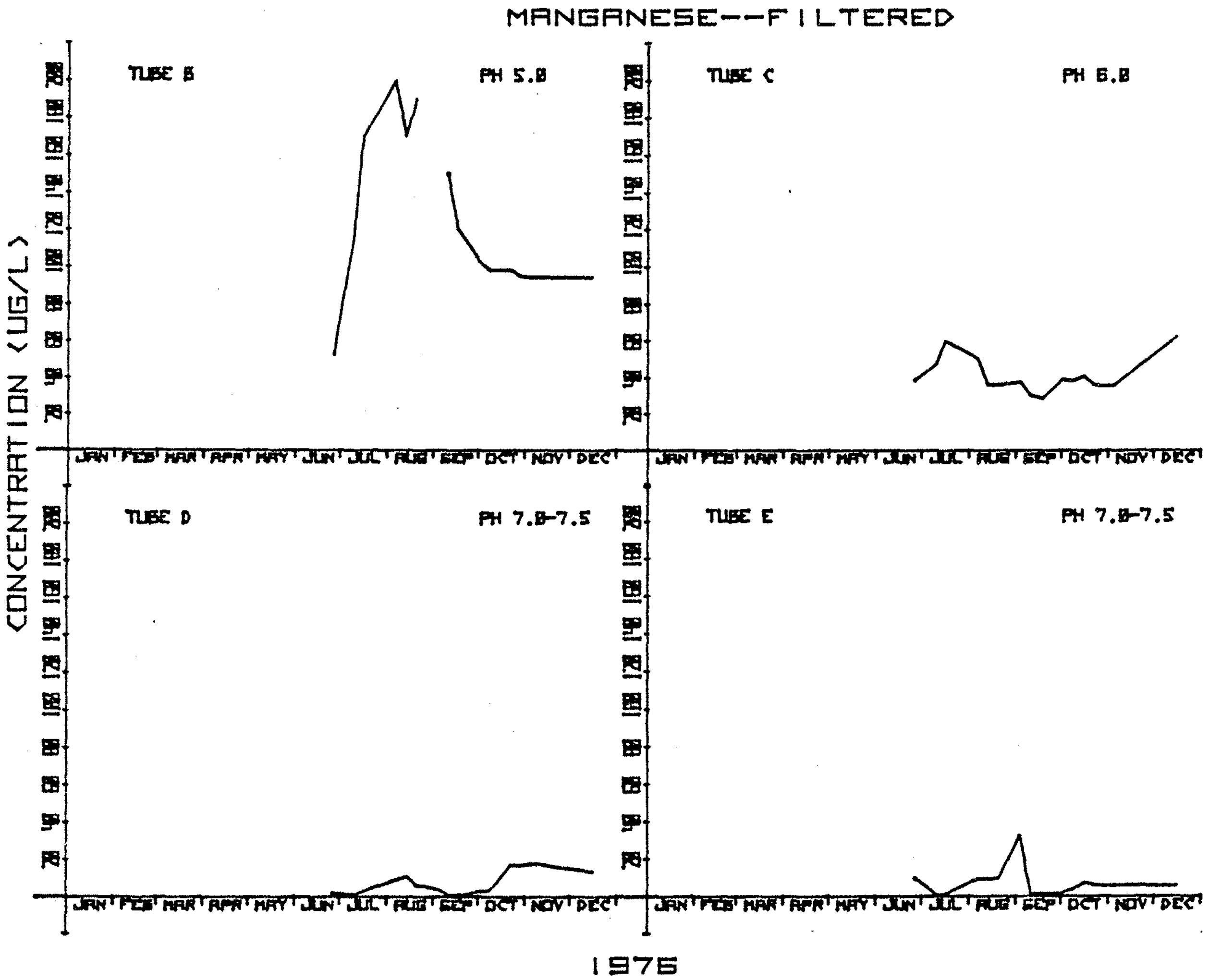


Figure 46. Concentrations of dissolved manganese in 10 m diameter tubes at different pH, summer 1976.

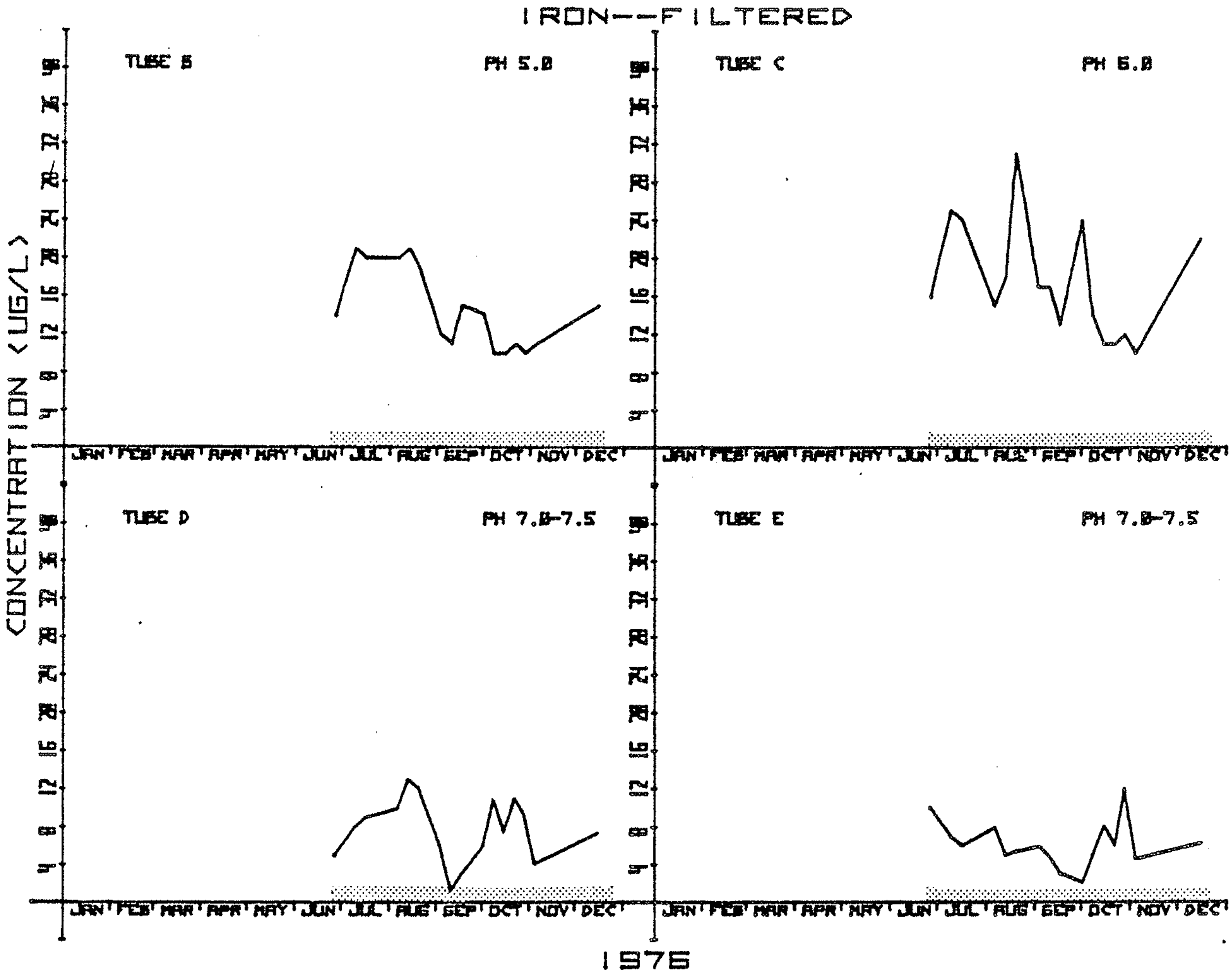


Figure 47. Concentrations of dissolved iron in 10 m diameter tubes at different pH, summer 1976.

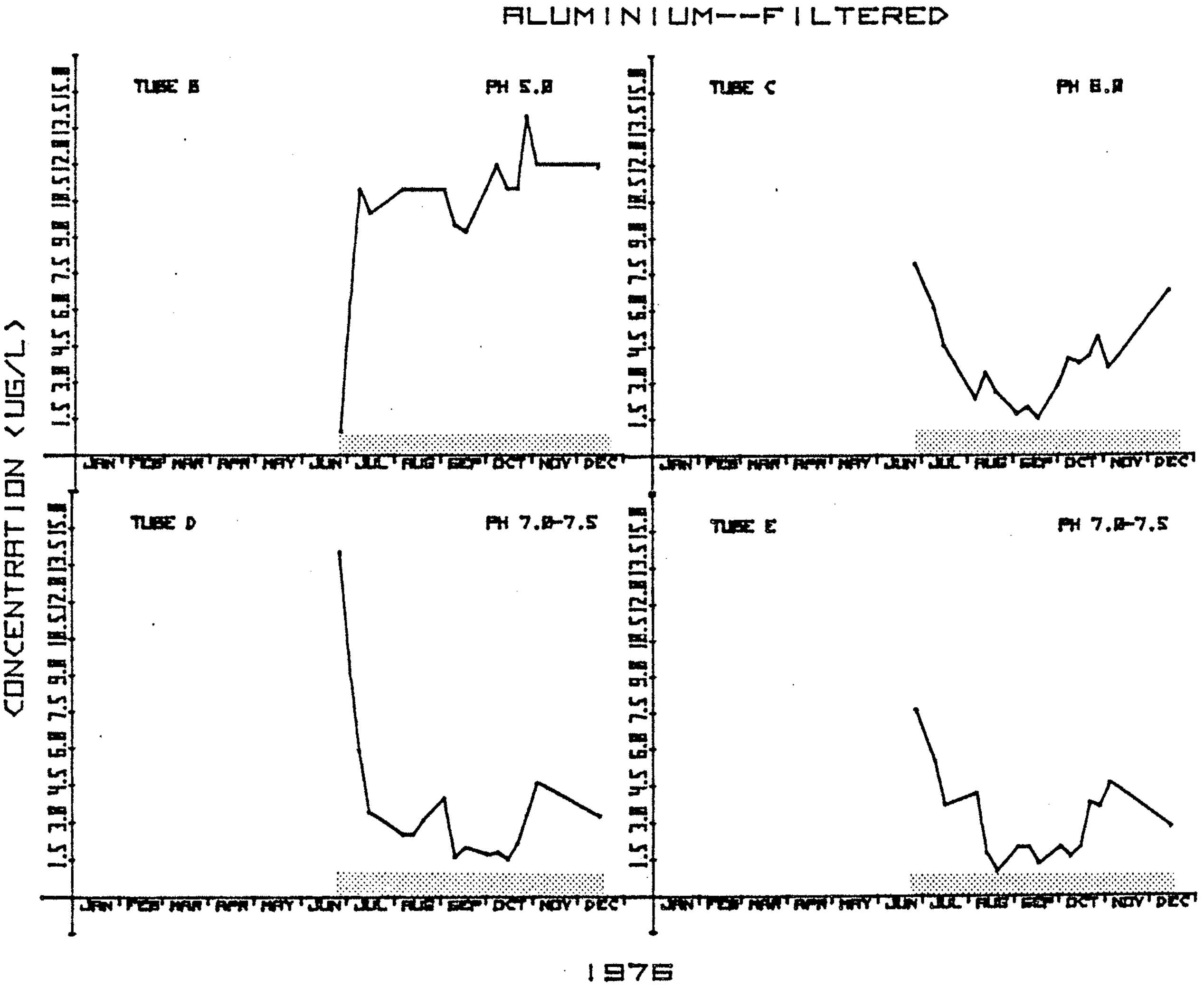


Figure 48. Concentrations of dissolved aluminum in 10 m diameter tubes at different pH, summer 1976.

### 3.10.5 Copper

Copper concentration at pH 5 showed a few interesting spikes, but results are inconclusive (Figure 49). Concentrations were generally below the limits of detection at other pH's.

### 3.10.6 Other Trace Metals

Concentrations of chromium, cadmium, cobalt, and lead were unaffected by pH (Figures 50 to 53). Once again, this was the expected result based on solubility products. The spike of cobalt in all tubes early in the experiment is puzzling, but may be due to contamination of some sort during the setup of the experiment.

Results of stable metal and gamma radionuclide experiments provided very similar answers about the mobility of metals at low pH, where it was possible to apply both techniques to the same metal (Table 3). Due to the better sensitivity and shorter analytical time, increased reliance will be put on the latter in future years.

## 3.11 MAJOR CATIONS IN TUBES

There was no significant increase in concentrations of calcium, magnesium, sodium, or potassium as pH was lowered (Figures 54 to 57). Slight increases in calcium, magnesium, and sodium were observed in the course of the experiment, probably reflecting the greater sediment surface per water volume in the tubes than in the lake.

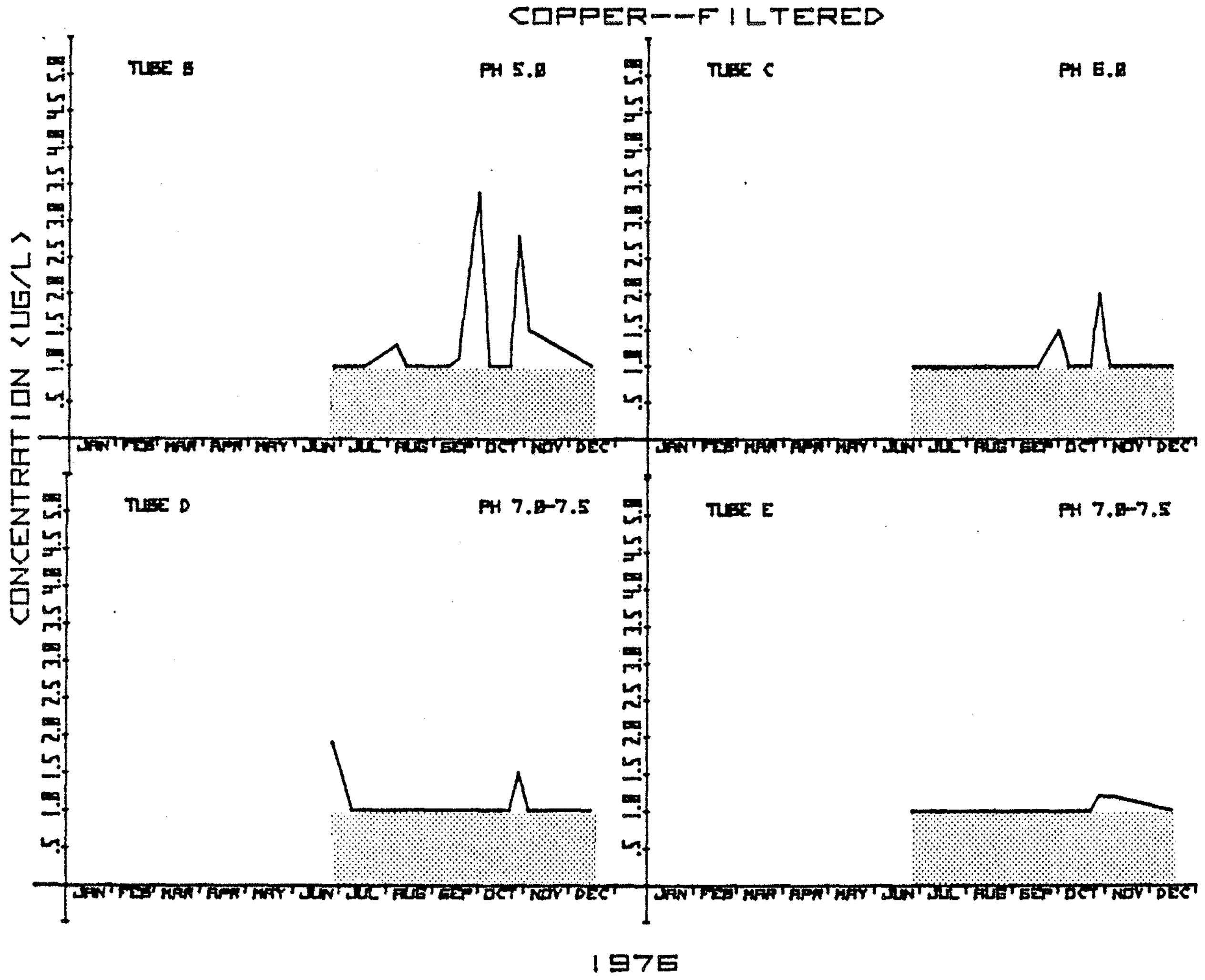


Figure 49. Concentrations of dissolved copper in 10 m diameter tubes at different pH, summer 1976.

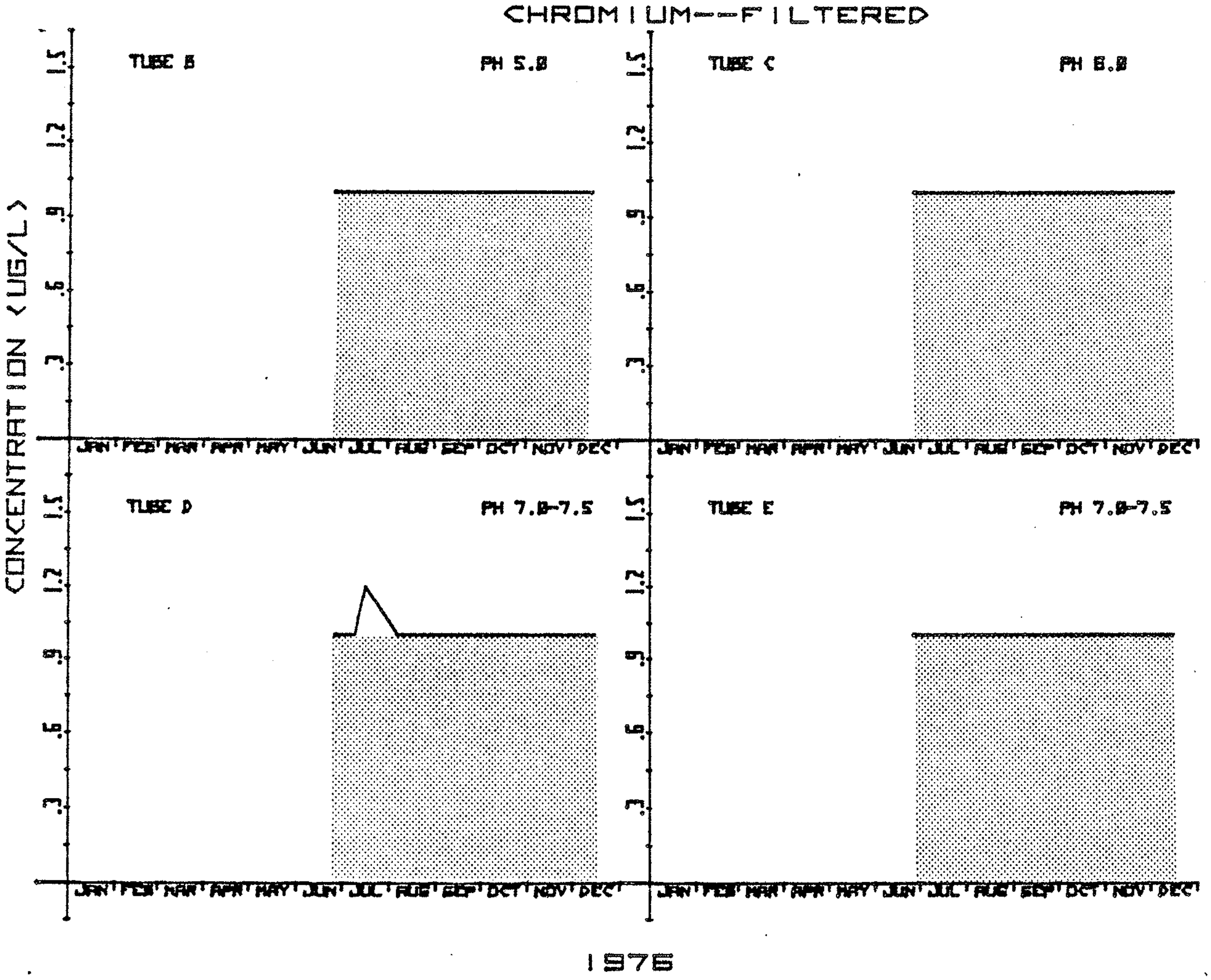


Figure 50. Concentrations of dissolved chromium in 10 m diameter tubes at different pH, summer 1976.

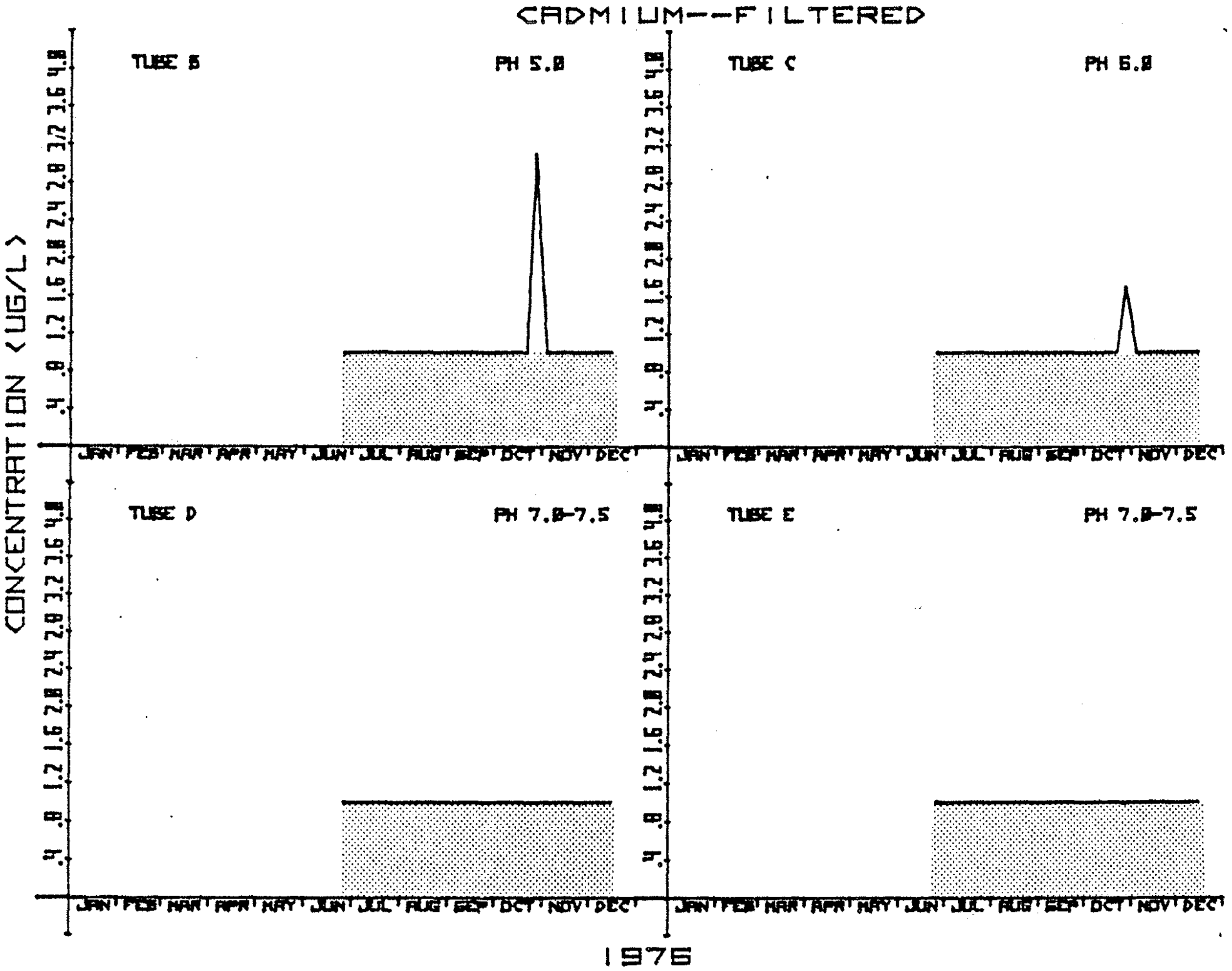


Figure 51. Concentrations of dissolved cadmium in 10 m diameter tubes at different pH, summer 1976.

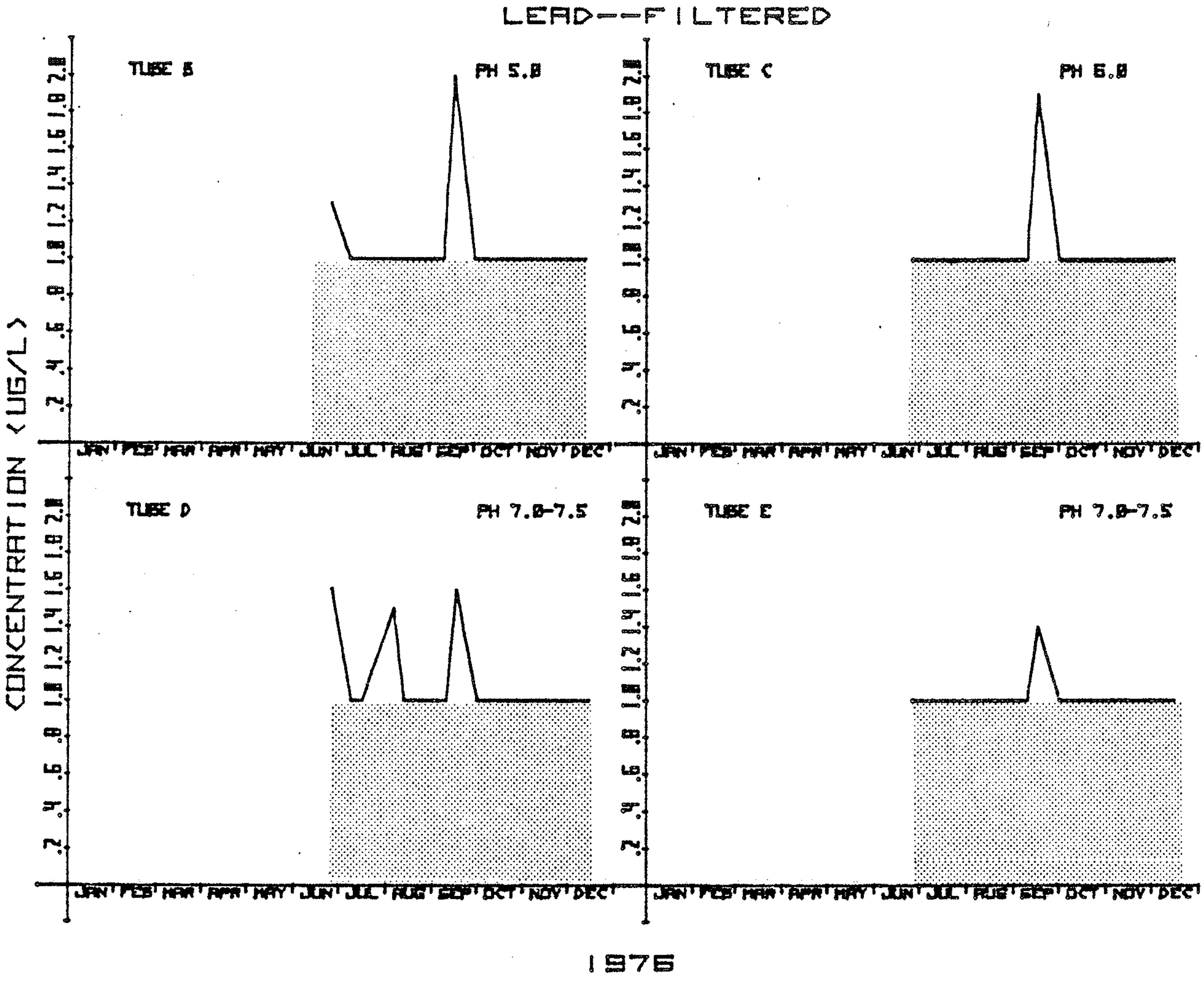


Figure 52. Concentrations of dissolved lead in 10 m diameter tubes at different pH, summer 1976.

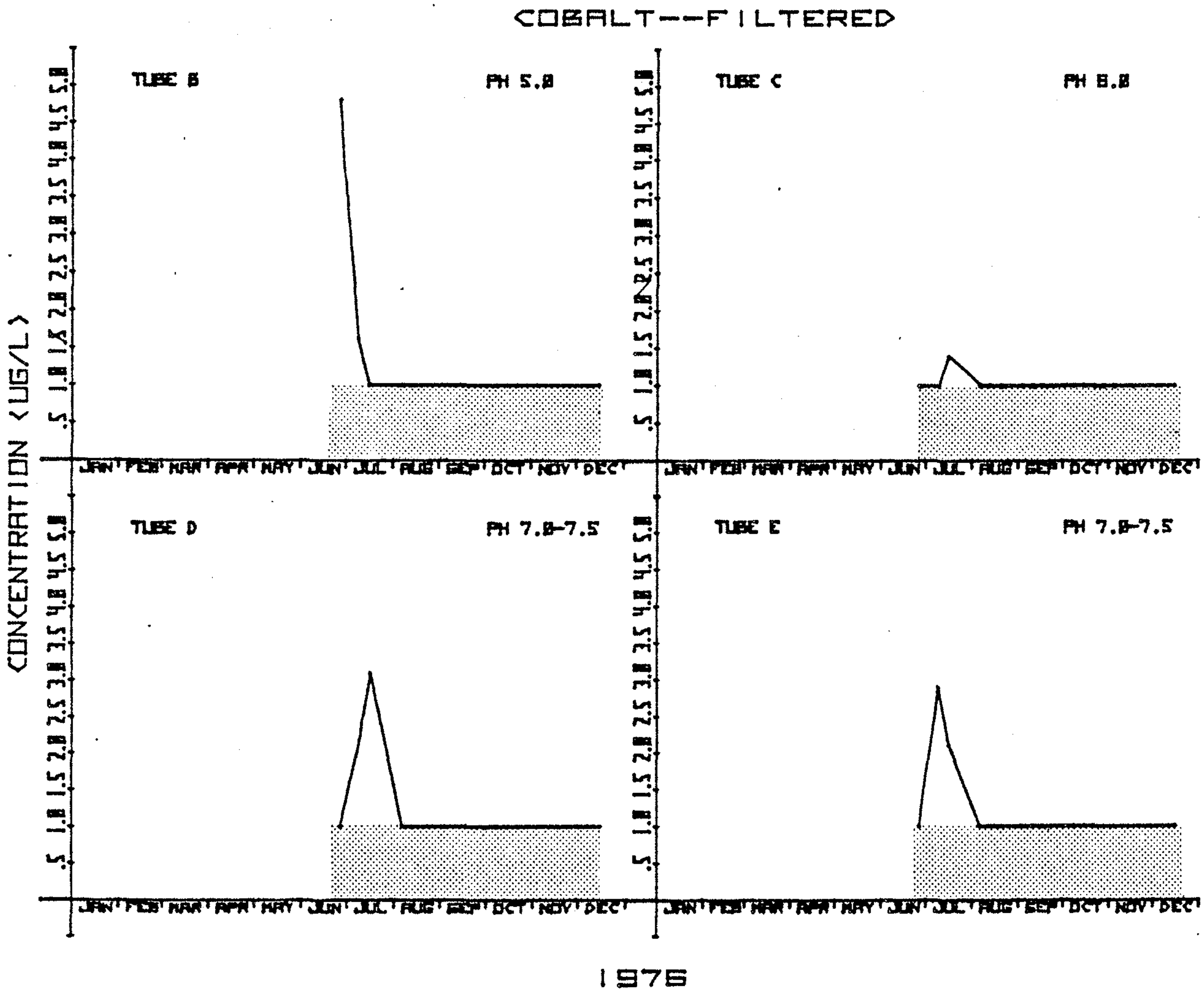


Figure 53. Concentrations of dissolved cobalt in 10 m diameter tubes at different pH, summer 1976.

Table 3. A summary of the effect of increasing acidity of the solubility of metals and other chemicals, based on radioisotope and stable metal analyses during tube experiments. The pH range tested was 5 to 7.5. A dash (-) means that analysis was not done by the particular technique.

| Element | Radioisotope Results | Stable Metal Analysis |
|---|---|---|
| Aluminum | - | great increase |
| Arsenic | none | - |
| Barium | none | - |
| Cadmium | - | possible slight increase |
| Calcium | - | none |
| Cesium | none | - |
| Chromium | none | none |
| Cobalt | increased slightly | none |
| Copper | - | none |
| Iodine | inconclusive | - |
| Iron | increased moderately | increased moderately |
| Lead | still being analyzed | none |
| Magnesium | - | none |
| Manganese | increased greatly | increased greatly |
| Mercury | none | - |
| Potassium | - | none |
| Selenium | none | - |
| Sodium | - | none |
| Thorium | increased slightly | - |
| Vanadium | none | - |
| Zinc | increased greatly | increased greatly |
| Plutonium | still being analyzed | - |
| Americium | still being analyzed | - |
| Neptunium | still being analyzed | - |
| Curium | still being analyzed | - |

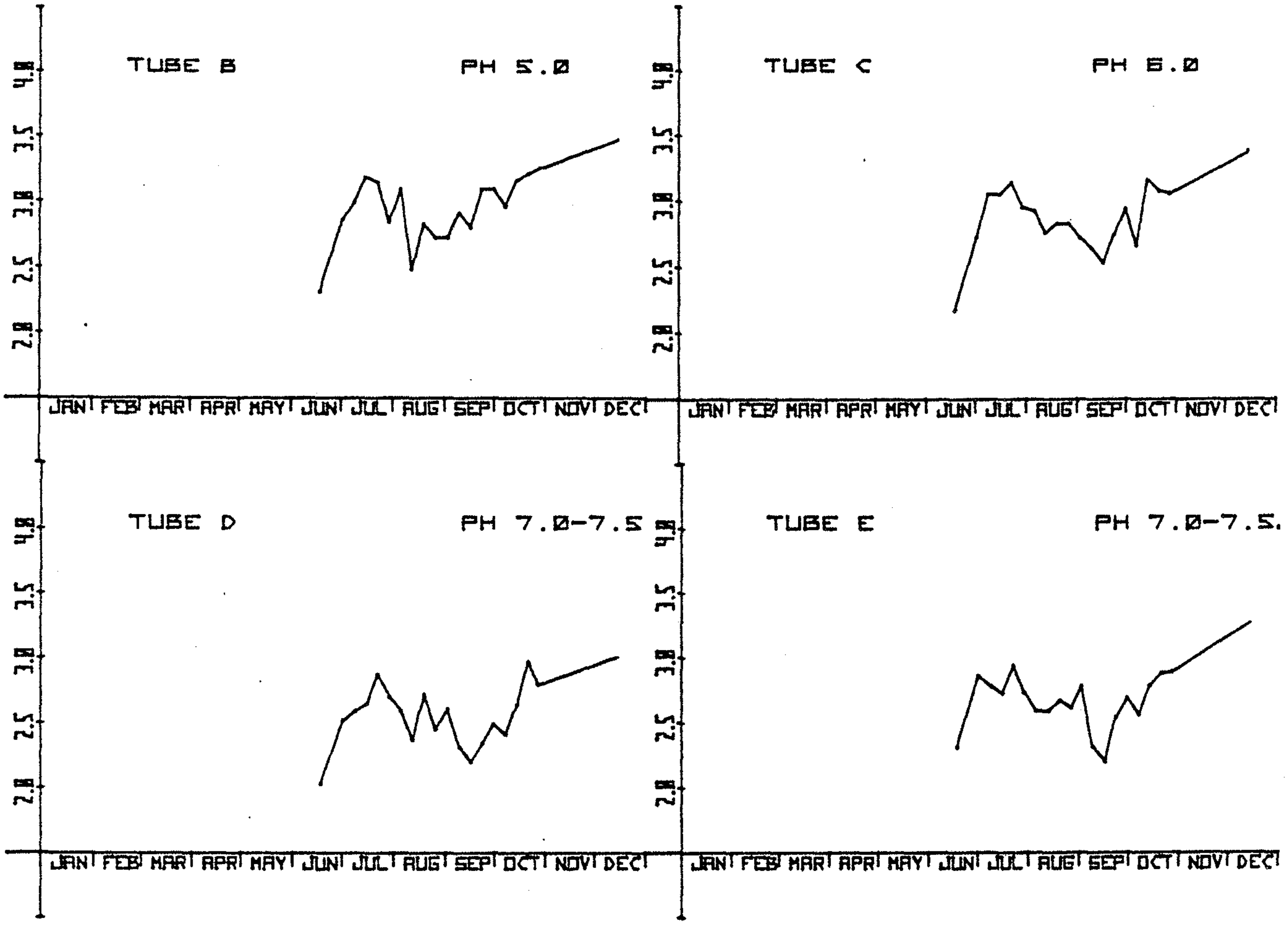


Figure 54. Concentrations of dissolved calcium in 10 m diameter tubes at different pH, summer 1976. Data are in mg/L.

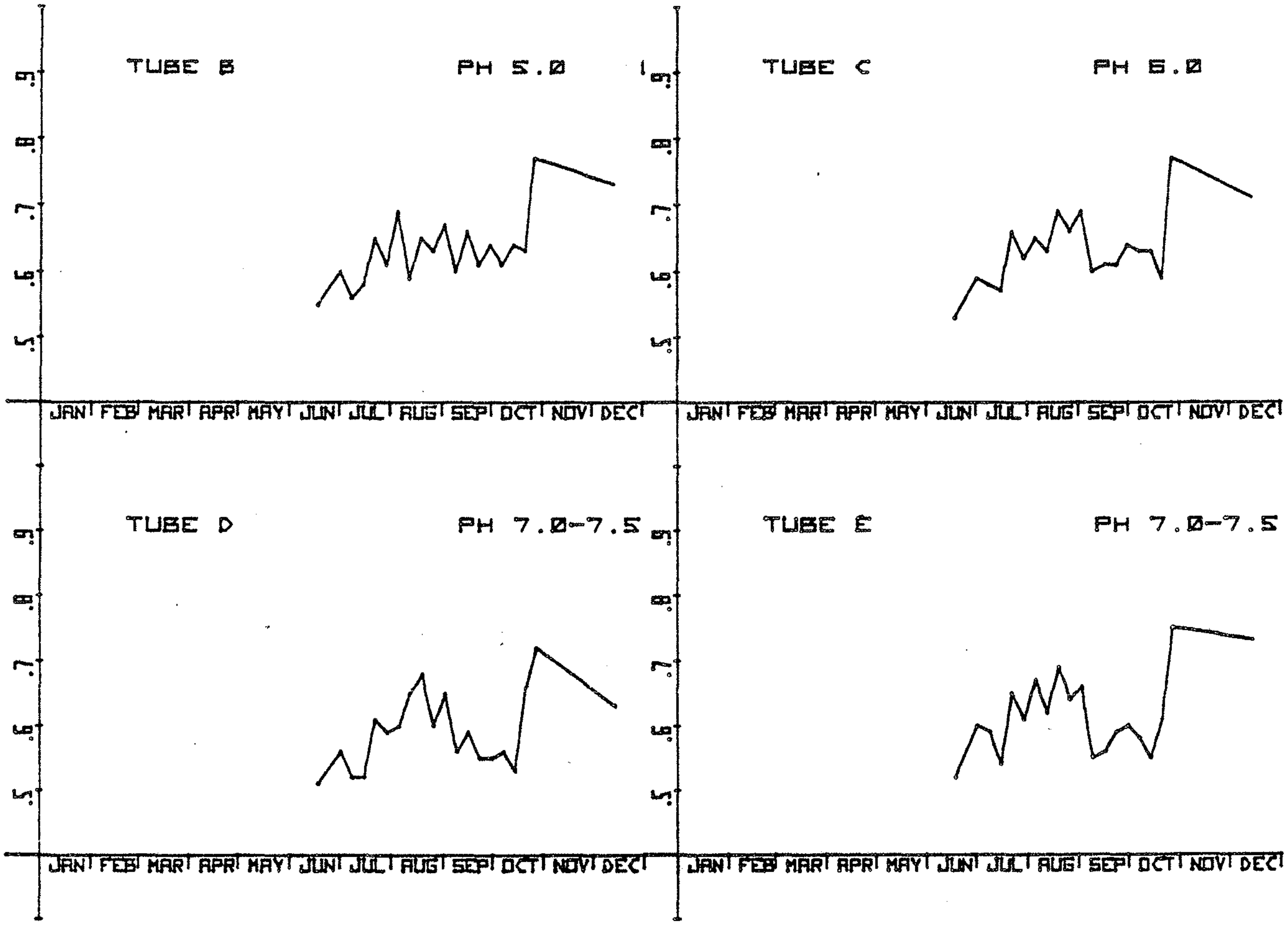


Figure 55. Concentrations of dissolved magnesium in 10 m diameter tubes at different pH, summer 1976. Data are in mg/L.

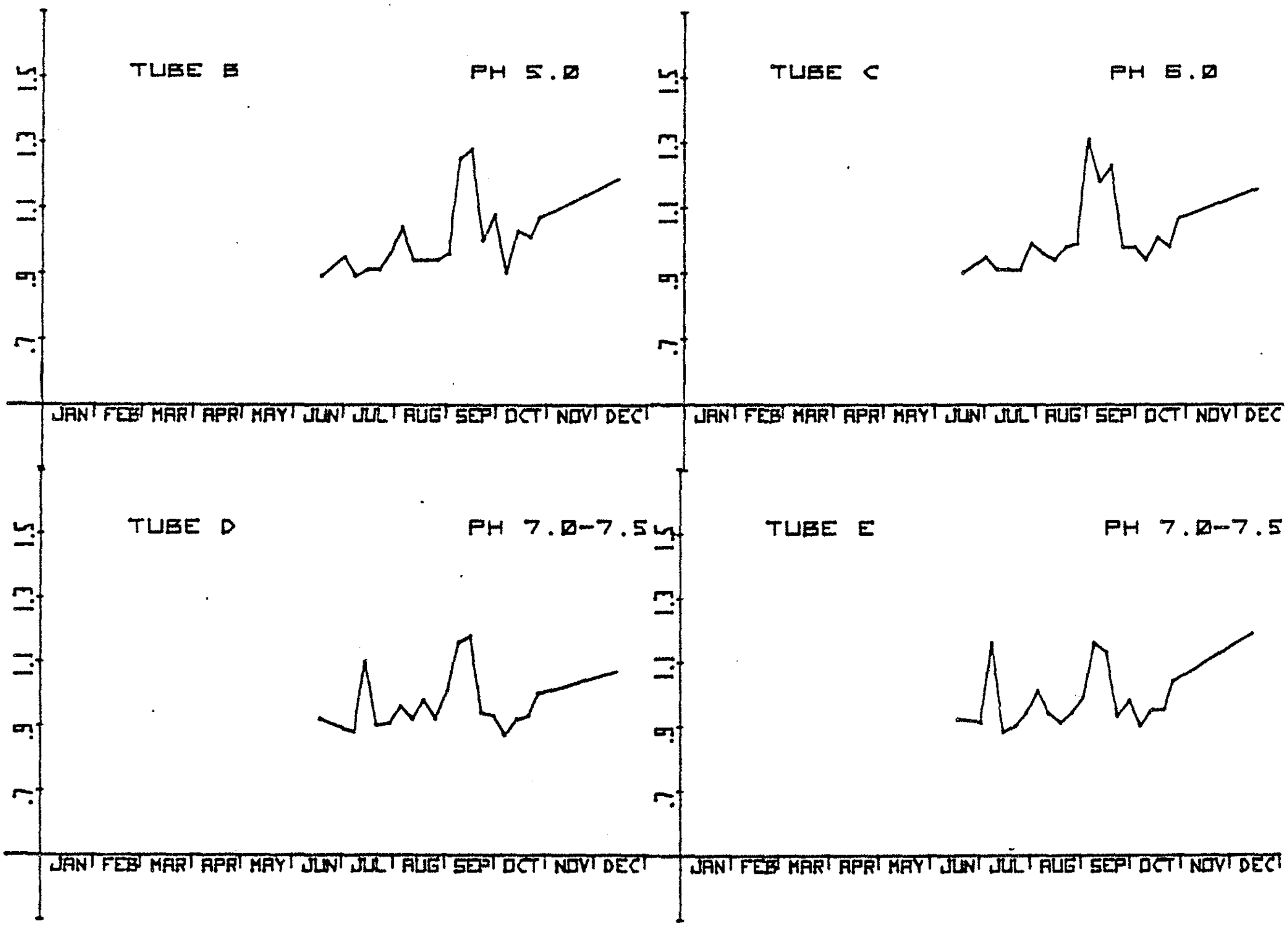


Figure 56. Concentrations of dissolved sodium in 10 m diameter tubes at different pH, summer 1976. Data are in mg/L.

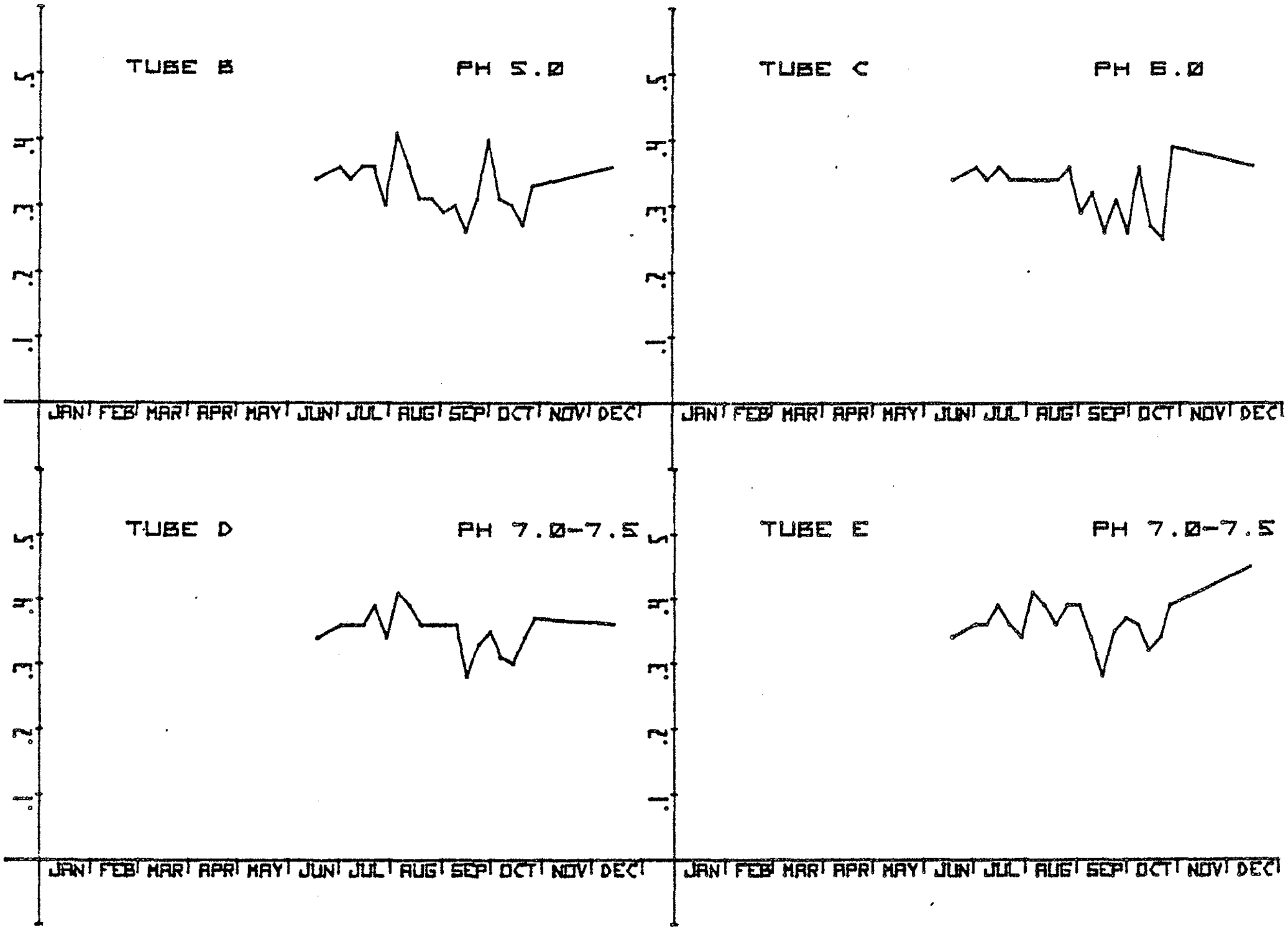


Figure 57. Concentrations of dissolved potassium in 10 m diameter tubes at different pH, summer 1976. Data are in mg/L.

## 4. DISCUSSION

The year 1976 was the driest and sunniest we have experienced at ELA, so that it is difficult to compare with the previous two years. Nevertheless, most chemical and biological parameters were remarkedly similar in the different years. It appears that acidification of the magnitude applied in 1976 had little detectable effect on ecosystem function.

The mobilization of some heavy metals, for example zinc and manganese, from lake sediments at lower pH, as demonstrated in tube experiments, indicates that more problems may develop as acidification continues. Preliminary experiments by P. Wong and Y.K. Chau of the Canada Centre for Inland Waters indicate that not only the concentration of metal increases but also the proportion of metal in the more toxic ionic form. More work on this aspect will be done in 1977. Other investigators have not found it possible to assess the magnitude of heavy metal release from sediments, because they have worked in areas of acid precipitation where metals may be leached from terrestrial soils into the lake as well (Malmer 1975).

In addition to mobilization from lake sediments and leaching from terrestrial soils, heavy metals also results from stack fallout in many areas (Kramer 1976). An important part of future experiments will therefore be to test acid-heavy metal synergisms at realistic concentrations.

Also previously unrecognized are the roles of sediments and colloidal compounds in buffering against acidification. The former would be a more important proportion of the total annual buffering in situations where acidification occurs at a slower rate, since the process is likely to be limited by the rate at which diffusion in pore waters allows the combination of acid and bicarbonate. The overall, long-term effect will be proportional to the bicarbonate content of lake sediments, while the rate of buffering will be determined by the porosity of sediments, i.e., if all else is equal, more compact sediments will buffer acid additions at a slower rate. Sediments in different areas of a lake may also vary in chemical and physical characteristics. Investigations in future years will attempt to quantify some of these relationships. The depletion of bicarbonate from lake sediments may also hinder the recovery of the lake system. The extent to which benthic organisms depend on the buffering of sediments by bicarbonate is not known.

On the other hand, colloidal compounds, including humic materials, will be more important where there is a high proportion of bog inflow. The ability of such compounds to absorb hydrogen ion is well known (D. Povoledo personal communication).

The significance of the above components in affecting the expected relationship between acid added and bicarbonate lost is illustrated in Figures 58 to 60.

Eighty-nine percent of the sulphate added as $H_2SO_4$ can be accounted for by observed increases in the water column (Figure 58). An additional 2.6% is known to have been lost through outflow. A small proportion (<0.1%) might have been taken up by biota. The remaining 7.5% has probably penetrated sediments, although this small amount is very close to our analytical confidence limits.

The ratio of $HCO_3^-$ disappearing from the water per unit observed increase in $SO_4^=$ is only 0.40, instead of the expected 0.50 (Figure 59). This means that 20% of the buffering of the system is accounted for by factors other than bicarbonates dissolved in the water column.

The ratio of $HCO_3^-$ disappearing to $SO_4^=$ added is still lower (Figure 60). Each unit of sulphate added was only 72% efficient at decreasing bicarbonate buffering. Because of the drought in 1976, this should be a near maximum efficiency, which will be slightly lower when outflow from the lake is higher.

Several problems must still be overcome with biological parts of the study. *Mysis* proved impossible to quantify properly. They remain on the bottom during daylight, but are too numerous and fast to estimate by quadrat sampling. Although animals move off the bottom at night, the maximum amplitude of migration appears to be less than 1 m, so that they cannot easily be assessed by zooplankton netting.

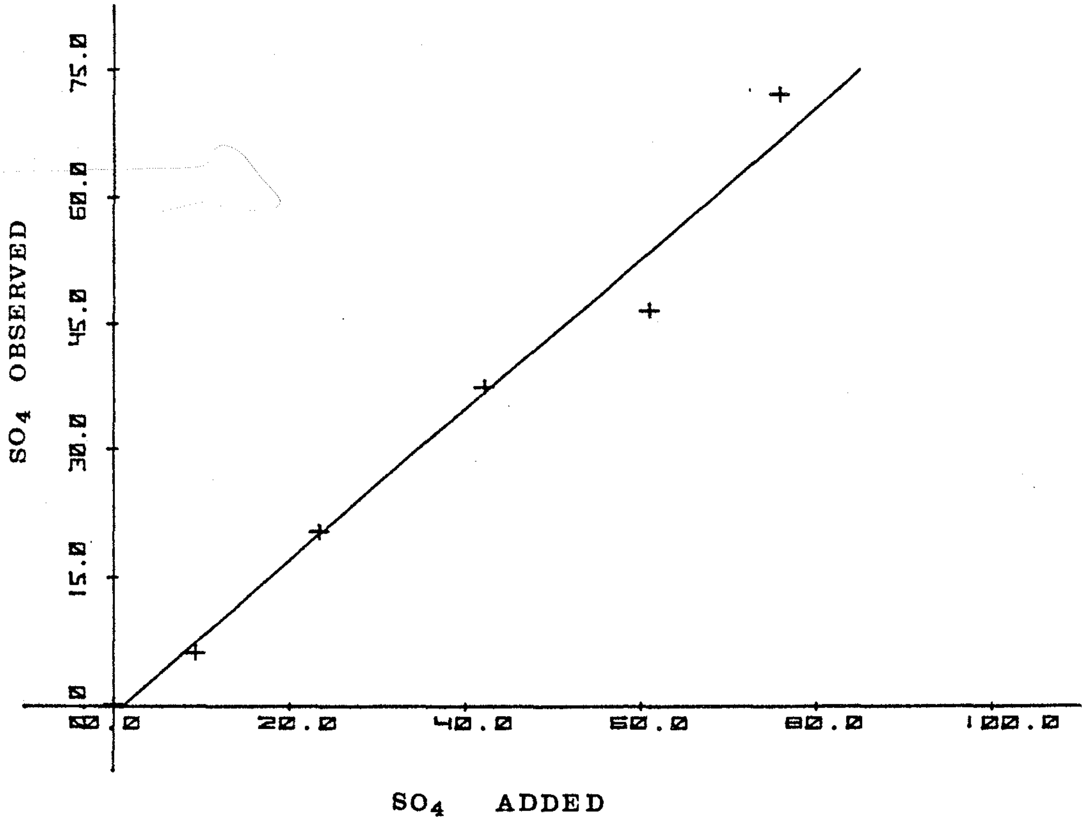


Figure 58. The relationship between added sulphate and the sulphate increase (both in meq/L) of Lake 223, 1976. The slope, 0.99, indicates the percentage of sulphate added which was still in the water column at the end of the summer. r = 0.99.

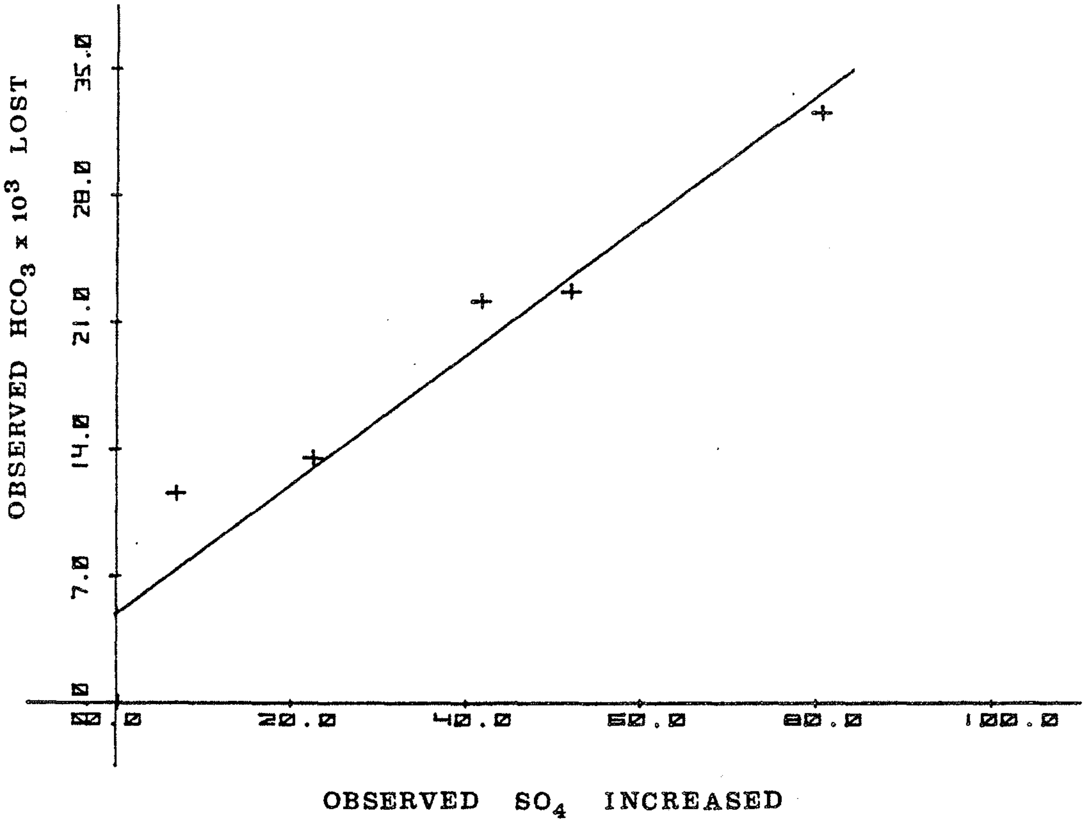


Figure 59. The relationship between the observed increase in sulphate and the observed bicarbonate decrease, in meq/L. The slope, 0.40, is less than the expected 0.50, indicating the proportion of buffering accounted for by factors other than DOC in the water column. r = 0.96.

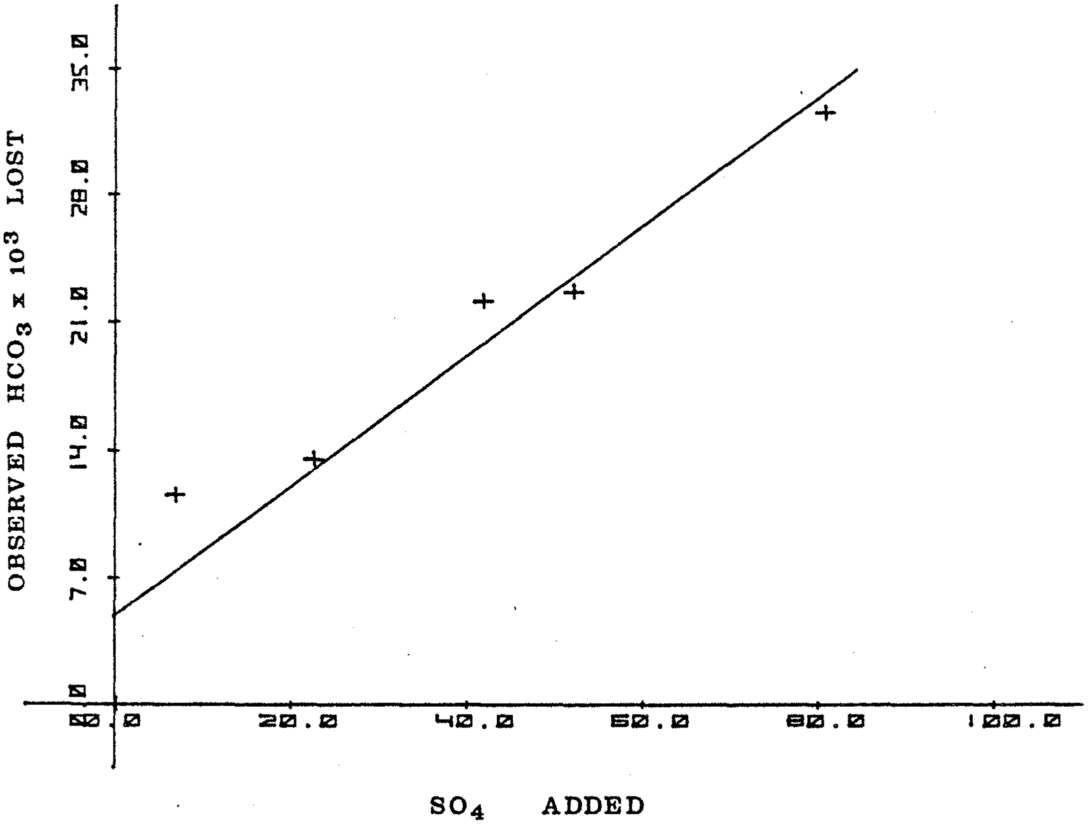


Figure 60. The relationship between added sulphate and lost bicarbonate. The slope, 0.36, indicates the added acid in 72% efficient at removing buffering capacity. This efficiency would be expected to decrease as water flow through the system increases. r = 0.96.

Crayfish population estimates by SCUBA divers appeared realistic, but were unpleasant and difficult to make, because diving was done at night, often in cold weather, and notes were kept on a slate which could often not be seen. The purchase of a diver-surface intercommunication system in late 1976 should ease this problem and allow good population estimates. The device will also allow detailed bottom mapping to assist in assessing sediment-water interaction, and it may be possible to develop a way to quantify *Mysis* populations by using it.

It has proved impossible so far to capture young-of-the-year trout. This problem has been faced, and never adequately solved, by previous investigators. Two approaches are planned for 1977. First, tows will be made with a high-speed Miller plankton trawl, which we hope will prove fast enough to quantitatively catch larval fishes. Second, emergence traps will be lowered over spawning beds prior to hatching of eggs in the spring of 1977, to catch fry as they hatch.

Fish egg/hatching experiments in the fall of 1976 were hampered by variable weather, with a warm spell causing trout to stop spawning mid-day, continuing some days later. Some variability in the maturity of eggs used for experiments was therefore experienced. Fungus growth on the eggs in spawning chambers also proved a problem, and the extent to which this occurs in nature is still unassessed.

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## 6. AOSERP RESEARCH REPORTS

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| 54. | WS 2.3 | A Preliminary Study of Chemical and Microbial Characteristics of the Athabasca River in the Athabasca Oil Sands Area of Northeastern Alberta |
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| 59. | TF 3.1 | Semi-Aquatic Mammals: Annotated Bibliography |
| 60. | WS 1.1.1 | Synthesis of Surface Water Hydrology |
| 61. | AF 4.5.2 | An Intensive Study of the Fish Fauna of the Steepbank River Watershed of Northeastern Alberta |
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| 64. | LS 21.6.1 | A Review of the Baseline Data Relevant to the Impacts of Oil Sands Development on Large Mammals in the AOSERP Study Area |
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| - | -- | Interim Report to 1978 |
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| 76. | AF 4.5.1 | An Intensive Study of the Fish Fauna of the Muskeg River Watershed of Northeastern Alberta |
| 77. | HS 20.1 | Overview of Local Economic Development in the Athabasca Oil Sands Region Since 1961. |
| 78. | LS 22.1.1 | Habitat Relationships and Management of Terrestrial Birds in Northeastern Alberta. |

These reports are not available upon request. For further information about availability and location of depositories, please contact:

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