USE OF PALEOLIMNOLOGY TO DOCUMENT THE EFFECT OF LAKE SHORELAND DEVELOPMENT ON WATER QUALITY

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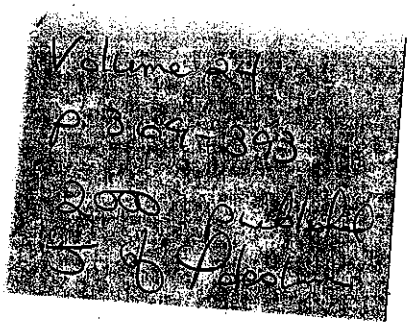

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Abstract

Four Wisconsin, USA lakes were examined with paleolimnological techniques to determine the effects of shoreland development on water quality. Geochemical parameters such as aluminum, iron, manganese, phosphorus, nitrogen, and carbon were used to document watershed inputs while redox sensitive elements provided information on changes in hypolimnetic oxygen levels. Changes in the diatom community were used to assess the impacts of development upon the lakes' trophic status as well as changes in the littoral community. We assessed the relative sensitivity of high versus low ANC lakes.

The initial shoreland development, late in the nineteenth or early in the twentieth century, involved seasonal cabins and minimal disturbance of the shoreland. This development phase had minimal impact upon the lakes compared to that during the mid- and late twentieth century. Increased levels of iron and aluminum indicated the highest input of sediment occurred during the construction phase of converting shoreland homes from seasonal to year-round usage. Phosphorus deposition increased moderately in the high alkalinity lakes but more so in the low alkalinity lakes. In the low alkalinity stratified lake, increased levels of iron and manganese in the last decade indicated more anoxia in the hypolimnion. Phosphorus levels have also increased during this time period most likely as a result of elevated phosphorus release from the sediments.

In the high alkalinity lakes, as the nutrient levels increased, diatom production initially shifted from benthic taxa of the family Fragilariaceae to metalimnetic taxa and as nutrients increased further, epilimnetic species. In the low alkalinity lakes, prior to settlement the major site of diatom prodution was the open water. Coincidental with the early shoreland development was an increase of macrophyte density as indicated by the epiphytic diatom *Achnanthidium minutissimum* ((Kützing) Czarnecki (=*Achnanthes minutissima* Kützing). The water quality in the high alkalinity lakes showed some improvement following completion of the home construction phase, especially in the lake with lower development density. The low alkalinity lakes did not show improved water quality and appear more sensitive to shoreland development.

Introduction

In the upper Midwest of the United States, lakeshore development has greatly expanded during the last 25 years as people have more leisure time and post World War II babies reach retirement age. In addition to the development on smaller lakes and shoreland sensitive to erosion around previously developed lakes, lake lots have become more numerous and existing lots have become more suburbanized. While in the upper Midwest initial cottage development on many lakes occurred in the first half of the twentieth century, most of these structures were for seasonal usage and consisted of small cabins with minimally developed lawns. These cottages were built on grade with minimal land disturbance. With improved roads and more people reaching retirement age, many of these lake lots have been redeveloped. Homes are much larger and often there are ancillary structures such as garages and storage sheds. Frequently driveways are paved and more extensive. These improvements result in larger imperious surfaces which contribute to greater runoff of pollutants. Because residents are spending more time at their lake homes, the landscape is more highly managed, usually resulting in removal of native vegetation and replacement with lawns. Often these lawns extend to the lakeshore with few buffers to intercept sediments and nutrients.

Although more nutrients are exported from a low density residential watershed compared with one that is forested (Dennis, 1986), only a few studies have examined the impact of this development upon a lake's water quality. In a comparison of data from the 1940's and the 1980's, Eilers et al. (1989) found that many northern Wisconsin lakes showed long-term increases in limnological parameters such as alkalinity and conductivity, which they attributed to, increased runoff from shoreland development. A paleolimnological study of lakes in south central Ontario by Hall & Smol (1996) found that most lakes have present day phosphorus concentrations that are similar to pre-industrial levels despite moderate levels of cottage development. These studies did not examine the effect of development during and following cottage construction.

We use paleolimnological techniques to examine in detail the history of four lakes in Wisconsin, USA, which have undergone shoreland development. These lakes differ in their development history as well as their limnological characteristics. Two lakes are high alkalinity marl lakes in the more densely populated southern part of the state, while the other two lakes possess low alkalinity and are located in the less populated northern portion of the state. This comparison allows us to contrast the sensitivity of high alkalinity and lower alkalinity lakes to watershed disturbances. These lakes are ideally suited to assess the impact of development, as the major landuse in their

watersheds the last 50 years has been shoreland development. Because of their proximity to a major metropolitan center (Milwaukee), the southern lakes were developed earlier than the northern lakes. This provides the opportunity to examine the long-term impacts of shoreland development.

The limnological history of these lakes is reconstructed through the integration of several independent paleolimnological techniques. Changes in landuse are tracked through geochemical proxies for soil erosion such as aluminum, iron, and porosity. Redox-sensitive elements provide information about changes in the oxygen regime of the hypolimnion. Nutrient concentrations and accumulation rates as well as qualitative changes in diatom species composition and diatom accumulation data are used to reconstruct past nutrient levels within the lakes.

Site Description

Moose and Silver lakes are located in southeastern Wisconsin in Waukesha County about 48 km from the city of Milwaukee (Figure 1). This region was one of the first to be settled following the Black Hawk War and the ceding of land by the Indian tribes in 1833 (Stark, 1984). At the time of settlement, the land around these lakes was largely oak savanna with prairie vegetation. With the arrival of settlers, much of the land was converted from prairie to subsistence type farming. Beginning around 1850, farming practices shifted to cultivation of wheat and later corn, hay, and oats but lack of tractors limited the amount of land that was in production. From the 1940's to the 1960's, there was a large increase in the population, especially around lakes in Waukesha County. Lake shorelines that were once farmed were sold for seasonal homes (Langill & Loerke, 1984).

Moose Lake has a mean depth of 12.2 m (Table 1), alkalinity of 167 mg L^{-1}, and total phosphorus concentration of 11 μg L^{-1} (Table 2). Although Silver Lake is also dimictic, its mean depth is less than Moose Lake at 4.8 m. Silver Lake has similar alkalinity and total phosphorus values of 163 mg L^{-1} and 11 μg L^{-1} respectively. Both of these lakes have small effective watersheds with nearly all of the land use in the contributing watershed consisting of residential lakeshore development. The shoreland homes are mostly year-round residences with private septic systems. The intensity of development around Moose Lake (33 homes/km of shoreline) is more than double that around Silver Lake (14 homes/km of shoreline) (Table 1). The slope of the shoreland is generally steeper around Moose Lake, especially along the western side.

Long and Round lakes are located in northwestern Wisconsin in Chippewa County (Figure 1). Because of their geographical location, these lakes were settled later than the southern lakes.

European settlements in the northern portion of the state did not occur until railroads penetrated the wilderness enabling extensive logging between the 1870's and 1890 (Lane, unpublished data).

Long Lake was very popular with the wealthy families of the city of Eau Claire during the logging boom when a number of large summer homes were built on the lake shore. Although the shoreland was not initially logged because of the homes, most of the shoreland and all of the homes and resorts were burned during widespread wildfires in this region in 1893-94 (Lane, unpublished data). At the present time, this area is heavily used by seasonal residents although some homes are inhabited year-round. The density of residential development around Long Lake is considerably less than the southern lakes being 6 homes/km of shoreline. Structures on both these lakes are served by private septic systems.

Long Lake is the only lake in this study that is a drainage lake, i.e. inflowing streams, and has a watershed of 1200 ha. Most of the watershed is forested with the lakeshore ringed by both seasonal and year-round homes served by private septic systems. This lake is dimictic with a mean depth of 6.1 m (Table 1), alkalinity of 53 mg L^{-1}, and total phosphorus concentration of 12 μg L^{-1} (Table 2). Round Lake is the only lake in the study that is polymictic. It has a mean depth of 1.6 m, alkalinity of 5 μg L^{-1} and a total phosphorus value of 21 μg L^{-1}. The intensity of development around these lakes is considerably less than the southern lakes with the density of homes around Long Lake being 6 homes/km of shoreline. Although at Round Lake the density of homes is similar to Long Lake and they are mostly located on the eastern shoreland. Most of the rest of the lake is surrounded by a wetland dominated by *Alnus rugosa* (DuRoi) Sprengel., *Chamaedaphne calyculata* (L.) Moench., and *Larix laricina* (DuRoi) K.Koch. The homes consist of seasonal and year-round residences and are served with private septic systems.

Methods

Sediment coring

Cores were collected from Moose and Silver lakes in July and August 1995 using a gravity corer with a 6.5-cm plastic liner. Cores were extracted from the deepest area of the lakes, in 18 m of water for Moose Lake and 12 m in Silver Lake. Cores were stored upright with a stopper in the bottom until returned to the laboratory. In the laboratory, cores were hydraulically pushed up from the bottom and sectioned into 2 cm slices. The core from Moose Lake was 44 cm long and the core

from Silver Lake was 46 cm in length. Samples were placed into clean, labeled plastic containers and stored in a refrigerated until dried.

The core from Long Lake was collected in October 1994. A 13-cm plastic tube was pushed into the sediment via SCUBA at a water depth of 20 m in the central basin. Upon retrieval, the core is stored upright with a stopper in the bottom until return to land. The 67 cm core was immediately extruded on shore by pushing the core through the top of the tube. The upper 50 cm was sliced into 1 cm sections and the remainder of the core was sectioned into 2 cm intervals. Sediment samples were placed into clean, labeled plastic freezer bags and stored on ice until return to the laboratory where they were frozen until analyzed.

The 105 cm core from Round Lake was collected in August 1995 in the deepest portion of the lake at 5.5 m. A Peterson piston corer equipped with a 13-cm plastic core tube was used to collect the core. The core was collected by pushing a plastic coring tube with aluminum push rods into the sediment until the piston inside the tube reached the top of the tube. The piston was held in a fixed position by attaching a line from it to the boat. Upon retrieval, the core is stored upright with a stopper in the bottom until return to land. The core was immediately extruded by pushing the core through the top of the tube. The upper 50 cm was sectioned into 1 cm segments and the remainder of the core was sectioned into 2 cm slices. Sediment samples were placed into clean, labeled plastic freezer bags and stored on ice until return to the laboratory. Samples were frozen until analyzed.

Sediment dating

Samples from Moose and Silver lakes were dried for 24 hours at 105°C for radiometric analyses. Samples were hand ground in a mortar and pestle. Every sample depth was analyzed in both cores and this was conducted at the University of Wisconsin-Milwaukee, Great Lakes WATER Institute. A sample of dried core section was digested in HCl and ^{210}Po, a daughter of ^{210}Pb, was plated onto a copper disc. Polonium-210 activity was determined by alpha counter and a multichannel analyzer. Polonium-208 was added as an internal tracer to determine recovery efficiency of radioisotopes through sample processing (Robbins & Edgington, 1975). Supported ^{210}Pb was estimated as the asymptote of ^{210}Pb activity at depth where a constant value was obtained. ^{137}Cs activity was determined for the upper 5 depths (0-10 cm) for the Moose Lake core to corroborate ^{210}Pb dates. This was done at the Wisconsin State Laboratory of Hygiene by direct gamma counting.

Samples from Long and Round lakes were freeze dried for one week prior to radiometric analyses. Eighteen samples from the Long Lake core and 10 samples from the Round Lake core were analyzed to determine age and sediment accumulation rates. Isotopic activities (^{210}Pb, ^{226}Ra, and ^{137}Cs) were measured by direct gamma counting (Schelske et al., 1994). The Round Lake core was analyzed at the Wisconsin State Laboratory of Hygiene, and the Long Lake core was analyzed by Dr. Jae Cable of Dr. Claire Schelske's laboratory at the University of Florida. Unsupported ^{210}Pb activity was calculated by subtracting ^{226}Ra activity from the total ^{210}Pb activity (Appleby et al., 1990). ^{137}Cs activity as measured in an effort to identify the period of maximum fallout from atmospheric nuclear weapons (Krishnaswami & Lal, 1978) testing and corroborate ^{210}Pb dates.

Sediment age for the various depths of sediment were determined by the constant rate of supply (CRS) model (Appleby & Oldfield, 1978), with dating errors calculated by first-order propagation of counting uncertainty (Binford, 1990). Bulk sediment accumulation rates (g cm^{-2} yr^{-1}) were calculated from output of the CRS model (Appleby & Oldfield, 1978). Accumulation rates of geochemical variables and diatoms were computed for each sediment depth by multiplying the bulk sediment accumulation rate (g cm^{-2} yr^{-1}) by the corresponding concentration (mg g^{-1}) of each constituent in the bulk sediment.

Physical and chemical analyses

Percentage dry weight was determined by measuring weight loss after 24 hours at 105°C. Organic matter content was measured by weight loss after ashing at 550°C for one hour (Dean, 1974). Sediment bulk density was determined by placing a known volume of sediment into a preweighed crucible, reweighed to obtain wet mass, dried at 105°C for 24 hours, and reweighed to obtain the dry mass per unit wet volume of sediment.

Porosity was calculated using the formula:

$$\text{Porosity} = \frac{(1-f)/D_w}{(1-f)/D_w + (f/D_s)} \qquad (1)$$

where: D_w = water density (1.0 g cm^{-3})

D_s = sediment density (2.45 g cm^{-3})

f = fraction dry weight (g cm^{-3})

Geochemical analyses of the cores from Moose and Silver lakes were performed at the University

of Wisconsin-Milwaukee, Great Lakes WATER Institute. All of the depths from each core were analyzed. Total and organic carbon as well as organic nitrogen were measured with a Carlo Erba Elemental Analyzer 1106. Concentration of inorganic carbon was calculated as the difference between total carbon and organic carbon. Phosphorus was analyzed spectrophotometrically following digestion with nitric and sulfuric acids. Samples for total iron and manganese were analyzed with an atomic absorption analyzer following digestion with nitric acid and hydrogen peroxide.

Geochemical analyses for Long and Round lakes were performed by the Wisconsin State Laboratory of Hygiene. Thirteen depths in Long Lake and 15 depths in Round Lake were analyzed. Total aluminum, iron, and manganese were analyzed using ICP-MS procedures following digestion of the sediment with $KClO_4$ (Wisconsin State Laboratory of Hygiene, 1993). Sediment samples for phosphorus and nitrogen analyses were digested with sulfuric acid as well as $CuSO_4$ and K_2SO_4 and the digestate measured using a technicon autoanalyzer

Samples for diatom analysis were cleaned with hydrogen peroxide and potassium dichromate (van der Werff, 1956). A portion of the diatom suspension was dried on a coverslip and samples were mounted in either Hyrax® or Naphrax®. Specimens were identified and counted under oil immersion objective (1400X) until at least 100 frustules were examined except for Long Lake where at least 250 frustules were counted. A known amount of glass microspheres was added to each sample following the procedure of Battarbee & Keen (1982) in order to determine absolute concentrations. Common nationally and internationally recognized keys were used including Patrick & Reimer (1966, 1975), Camburn et al. (1984-86), Dodd (1987), and Krammer & Lange-Bertalot (1986, 1988, 1991a,b).

Results

Sediment dating

Of the four lakes, the lowest total residual unsupported ^{210}Pb activity occurred in the core from Moose Lake at 5.15 pCi cm^{-2} (Table 3). The ^{210}Pb activity in the core from Silver Lake was higher at 8.26 pCi cm^{-2}. The northern lakes were considerably higher with the highest value occurring in the Long Lake core at 73.78 pCi cm^{-2}. These inventories are equivalent to mean annual ^{210}Pb fluxes of 0.16 to 2.30 pCi cm^{-2} yr^{-1}. All but Long Lake are similar to the usually cited global range for

atmospheric flux of 0.2 to 1.1 pCi cm^{-2} yr^{-1} (Krishnaswamy & Lal, 1978). The higher flux in Long Lake indicates significant sediment focusing is occurring in the lake. Binford et al. (1993) indicated that in the PIRLA lakes, focusing was more common in drainage lakes than seepage lakes. Long Lake is the only drainage lake in this data set. The cores from the southern lakes (Moose, Silver) exhibited higher mean sedimentation rates (0.022 and 0.027 g cm^{-2} yr^{-1} respectively) even though the linear sedimentation rates were lower. This was the result of the greater density of the sediments in the southern lakes since these are high alkalinity lakes. They experience considerable precipitation of calcium carbonate ($CaCO_3$).

Figure 2 presents the stratigraphic profiles for ^{210}Pb, ^{226}Ra, and ^{137}Cs in the cores from the study lakes. The cores from all of the lakes reached supported ^{210}Pb activity before the bottom of the cores. This was reached at 28-30 cm in Moose Lake while it was deeper in the other cores. In Silver Lake supported ^{210}Pb activity was reached at 30-32 cm. Supported ^{210}Pb activity was reached at 60-62 cm in Long Lake and 45-46 cm in Round Lake.

Radium-226 activities were much lower in the core from Round Lake than Long Lake. Concentrations in the Round Lake core were similar with a mean of 0.57 pCi g^{-1}. The activity of ^{226}Ra was higher and more variable in the Long Lake core. Activities ranged from 1.51 pCi g^{-1} to 5.56 pCi g^{-1} with higher values occurring in the upper portion of the core. This necessitated subtraction of supported ^{210}Pb (i.e. ^{226}Ra) from total ^{210}Pb activity on a level-by-level basis to estimate unsupported ^{210}Pb activity.

In the cores from two of the 3 lakes (Moose, Long, Round) where ^{137}Cs activity was measured, a definitive bomb peak was present (Figure 2). In the Round Lake core, the highest activity was above 20 cm but the peak was spread over 3 depths. This suggests that there is some postdepositional mobility of this highly soluble radioisotope in this shallow lake.

Under ideal conditions with no mixing from physical or biological processes, log-linear plots of ^{210}Pb activity versus depth should yield a straight line. The profile from Moose Lake exhibits a break from this between 6-10 cm (Figure 2). The CRS model treats the lower activity in these layers as periods of rapid deposition. A similar break also occurred in Long Lake in the core at the 35-36 cm depth. Similar breaks did not occur in either the Silver or Round lake cores. Neither Moose nor Silver lakes exhibit flattening of the profile at the top of the core. Both Long and Round lakes exhibited flattening of the profile in the top section of the cores. This may indicate that

mixing processes, either from biota or physical factors, have occurred in recent years. Oldfield and Appleby (1983) indicate that the CRS model is robust enough that the mixed zone can be as much as 15% of the depth of unsupported ^{210}Pb before dates lower in the core become unreliable. Since this flattening only occurred at the very top of the cores, this should not adversely affect the dating results.

The profiles of sediment depth versus age are presented in Figure 3. All of the cores except the one from Moose Lake exhibit increases in the slope upcore indicating an increase in the sediment accumulation rate. This is especially true in the upper portion of the cores from Long and Round lakes. These profiles also exhibit large errors near the bottom of the datable portion of the cores. In this part of the cores unsupported ^{210}Pb activity is so low that it can not be measured accurately.

Cesium-137 can be used to identify the period of maximum atmospheric nuclear testing (Krishnaswami and Lal, 1978). The peak testing occurred by the USSR in 1963 and thus the ^{137}Cs peak in the sediment core should represent a date of 1963. In the Moose Lake core, the ^{137}Cs peak was one depth shallower for this date calculated with the CRS model (Figure 3). In the Long Lake core, the peak was exactly at the depth indicated as 1963 by the lead dating. In the Round Lake core, there was not a definitive peak but the ^{137}Cs peak fell close to the date of 1963 indicated by the ^{210}Pb dating. The ^{137}Cs peaks fall reasonably close to the ^{210}Pb age-depth curve thus supporting the accuracy of the CRS model results.

Geochemical profiles

In the Moose Lake core, the concentrations of the geochemical variables were relatively unchanged from the bottom of the core until 14 cm (ca. 1900) (Figure 4a). At this depth porosity, phosphorus, nitrogen, and organic carbon exhibited a small peak and then declined. Inorganic carbon began to decline around 1900. All of the variables continued to decline until about 1950 and then increased to the top of the core. Inorganic carbon was the exception as it maintained a relatively constant concentration to the top of the core. In contrast, iron, and to a lesser extent, manganese began to increase around 1900. Manganese continued to slowly increase until near the top of the core while the iron concentration peaked in the late 1930's. Iron declined after the mid-1960's to the top of the core

In the Silver Lake core, the concentrations of the geochemical variables were generally unchanged from the bottom of the core until the 1880's (Figure 4b). During the period between the 1880's and 1910's, phosphorus, nitrogen, and organic carbon concentrations peaked. The concentrations of these three parameters then declined and concentrations remained lower until circa 1970, after which they steadily increased to the top of the cores. In contrast, iron, manganese, and inorganic carbon declined during the period between the 1880's and 1910's. Both manganese and inorganic carbon concentrations were steady but higher throughout the remainder of the core while iron was present in lower concentrations. The higher concentration of inorganic carbon likely resulted from increased precipitation of $CaCO_3$ as the lakewater pH became elevated with increased primary productivity. The increased concentration of manganese may indicate decreased oxygen levels in the hypolimnion. Engstrom et al. (1985) noted that manganese is mobilized at a higher redox potential than iron. They observed that the concentration of manganese increased with anoxia while iron did not.

In the Long Lake core, the concentrations of the geochemical variables, aluminum, iron, manganese, and phosphorus were unchanged from the bottom of the core until 1920. Beginning in the 1920's, phosphorus increased with peak concentrations around 1950 (Figure 5a). Manganese steadily increased after 1980 until near the top of the core. Aluminum showed the opposite trend, steadily declining after 1960 to the top of the core. Organic matter increased slowly from the bottom of the core to 1900. From 1900 until 1940 the concentration steadily declined because of dilution by clastic material from soil erosion (Engstrom et al., 1985). After 1960 organic matter slowly increased until 1980 when the concentrations increased at a faster rate to the top of the core.

The core from Round Lake exhibited much greater changes in the concentrations of the geochemical variables than the cores from the other lakes. For example, the highest aluminum concentrations occurred near the bottom of the core (Figure 5b), well before European settlement. Organic matter concentrations also exhibited variability downcore. An extensive wetland system is hydrologically connected with Round Lake and it is possible that this lake has exhibited significant water level changes. After 1800, aluminum and organic matter exhibited more stable concentrations although aluminum peaked during the mid-1920's and declined at the surface. The concentration of organic matter was steady from the early 1800's until around 1940. Its concentration steadily increased from 1940 until the top of the core. Phosphorus was more dynamic with the lowest concentrations in the core occurring at the bottom of the core, prior to the beginning

of European settlement. Phosphorus increased some during the 1800's and peaked during the middle 1920's. A second peak occurred around 1975.

Material accumulation rates

In the core from Moose Lake, the bulk sedimentation rate exhibited two peaks. One peak occurred circa 1880 and a second peak occurred around 1950 (Figure 6a). The sedimentation rate at the top of the core at 0.010 g cm^{-2} yr^{-1}, was similar to the pre-settlement mass sediment accumulation rate. The accumulation rate of iron, manganese, phosphorus, nitrogen, and carbon also exhibited two peaks in the datable portion of the core. The bulk sedimentation rate was the primary determinant of the trends of phosphorus and inorganic carbon. While iron and manganese accumulation rates also exhibit two peaks, the 1950 peak was greater. The opposite was true for the accumulation rates of nitrogen and organic carbon with the highest peak occurring during the 1880's.

In the core from Silver Lake, the bulk sedimentation rate did not increase until the 1920's (Figure 6b). The sedimentation rate remained high until the late 1970's when it declined and reached near pre-settlement levels at the surface (0.028 g cm^{-2} yr^{-1}). The accumulation rates of iron, manganese, and inorganic carbon exhibited a similar trend as the bulk sedimentation rate. Phosphorus, nitrogen, and organic carbon first increased around 1890 and remained elevated throughout the upper portion of the core.

In the core from Long Lake, the bulk sedimentation rate initially increased in the 1890's and then slowly increased further beginning in the 1920's and peaked at the surface of the core. (Figure 7a). The initial bulk sedimentation rate increase in the 1890's was only reflected in organic matter and to lesser extent in aluminum. The accumulation rates of phosphorus, manganese, iron, and aluminum did not increase until the 1940's. Their rates remained high throughout the upper portion of the core and all of these variables except aluminum, peaked near the surface of the core. The ratio of iron to aluminum was generally unchanged from the bottom of the datable portion of the core until 1950. A short-lived peak occurred around this time. The ratio of iron to aluminum increased during the 1990's with the highest levels occurring at the top of the core. The ratio of iron to manganese was unchanged from the bottom of the core until the 1910's. It increased around 1920 and remained high until the mid-1980's when it declined.

In the datable portion of the Round Lake core, the bulk sedimentation rate increased slightly around 1880 but the rate was higher during the 1920's (Figure 7a). The sedimentation rate increased further in the late 1970's with the peak accumulation rate occurring at the surface of the core. The accumulation rates of aluminum, phosphorus, and organic matter generally reflected this same trend. Aluminum and phosphorus accumulation rates decline in the surface sediments while rates of organic matter accumulation remained elevated. The ratio of aluminum to phosphorus was constant throughout the datable portion of the core except the bottom most sample.

Diatoms

During the early 1800's the diatom community in the Moose Lake core was dominated by benthic diatoms such as *Staurosira construens* var. *venter* (Ehren.) Williams and Round (Figure 8a). The planktonic diatom *Cyclotella bodanica* var. *lemanica* (O. Müller) Bachmann was also an important component of the diatom community. During the late 1800's, these diatoms became less important and were replaced by the planktonic taxon *Cyclotella michiganiana* Skvortzow. Throughout the 1800's the diatom accumulation rate was low (Figure 8a) indicating low productivity of the diatom community. Throughout the twentieth century, the diatom community was dominated by the planktonic species *Fragilaria crotonensis* Kitton, *Asterionella formosa* Hasall, and *Stephanodiscus minutulus* (Kütz.) Cleve & Möller. The diatom accumulation rate also was higher during the twentieth century compared with the 1800's.

Prior to the 1930's the diatom community in the Silver Lake core was dominated by benthic taxa such as *Staurosirella pinnata* (Ehren.) Williams and Round, *S. construens* (Ehren.) Williams and Round, *S. construens* var. *venter*, and *Pseudostaurosira brevistrata* (Grun.) Williams and Round as well as the planktonic taxa *C. michiganiana* and *C. bodanica var. lemanica* (Figure 8b). The diatom accumulation rate was also low during the time period. By 1940 benthic taxa had become virtually absent and been replaced by the planktonic taxa *Cyclotella glomerata* Bachmann, *Aulacoseira ambigua* (Grunow) Simonsen, *A. formosa*, and *F. crotonensis* typically found in the epilimnetic waters. The diatom accumulation rate increased after 1920 indicating increased diatom productivity. Although the diatom accumulation rate remained high near the top of the core, *F. crotonensis* and *A. ambigua* declined and *C. michiganiana* increased.

In the Long Lake core, prior to 1900, the diatom assemblage was dominated by *Tabellaria flocculosa* (Roth) Kütz. strain IIIp *sensu* Koppen, *Cyclotella atomus* Hustedt, and *A. ambigua*

(Figure 9a). Around 1900, *C. atomus* and *T. flocculosa* strain IIIp declined and *C. michiganiana* and *C. glomerata* increased. The diatom accumulation rate increased during the early part of the twentieth century and peaked during the 1940's. Around 1950, diatoms such as *F. crotonensis* and *A. formosa* which are indicative of higher nutrients increased. During the 1990's these taxa declined and were replaced by the benthic diatom *S. pinnata*. The highest diatom accumulation rate occurred at the top of the core and was largely driven by the high sedimentation rate at this level.

In Round Lake the diatom community prior to 1870 was dominated by the benthic diatoms *S. pinnata* and *Eunotia* spp. (Figure 9b). Although a number of *Eunotia* taxa were encountered, one of the most common was *Eunotia incisa* Gregory. During the 1870's, the planktonic diatom *T. flocculosa* str. IIIp became dominant and remained an important component of the diatom community throughout the rest of the core. Beginning in the 1940's, *Asterionella ralfsii* var. *americana* Körner increased as well as *T. flocculosa* strain IIIp. The percentage of planktonic diatoms also increased around this time and remained high throughout the upper portion of the core.

Discussion

Moose Lake

The first settlement on the lakeshore was by Hans Gasmann in 1843 on the southeastern shore of the lake (Stark, 1984). Early landuse consisted of subsistence farming, which resulted in little change in the lake's sedimentation rate (Figure 6a). Shoreland development began in the 1920-30's, and by 1940 much of the shoreland had been developed (SEWRPC, 1997). Early cottage construction most likely disturbed less soil compared with later construction because the use of heavy equipment was not common. Resident usage was seasonal and structures were smaller resulting in fewer impervious surfaces. This would reduce water runoff and cause less sediment and nutrient delivery to the lake. The improvement of roads following World War II and proximity to the large city of Milwaukee resulted in cottages being converted to year-round homes as well as construction of new homes. Larger homes and increased use of heavy equipment caused more land disturbance and likely resulted in a greater amount of soil erosion being delivered to the lake. The highest sedimentation rate in the core occurred around 1950 (Figure 6a), and much of the increased sedimentation was likely due to soil erosion. Porosity reached its lowest point in the core at the sedimentation peak in 1950 suggesting a greater portion of the deposited material was from coarser soil particles that entered the lake during home construction. Both iron concentration and accumulation increased at this time period. It is likely that at least some of the iron accumulated as

a result of input from the watershed as iron may be associated with clastics derived from erosion (Engstrom & Wright, 1984). Another hypothesis for the increase in iron concentration is increased mobilization of sediment bound iron resulting from increased anoxia. However, manganese, which would be expected to be more mobile than iron under anoxic conditions (Jones & Bowser, 1978; Engstrom et al., 1985), does not exhibit a peak. It is our contention that the elevated iron is a result of soil erosion from home construction. Although phosphorus accumulation increased during construction, it was driven the sedimentation rate. Sediment phosphorus concentration actually declined. Apparently only a small amount of phosphorus was deposited at the site of the sediment core during construction. With the completion of the major construction phase by 1960, the sedimentation rate declined, porosity increased, and the accumulation of nutrients and carbon also declined. The sedimentation rate continued to decline and by 1990 was nearly the same as pre-settlement rates. Although the carbon accumulation rate also returned to pre-settlement levels, both phosphorus and nitrogen accumulation remain greater than rates observed prior to European settlement.

The diatom community in Moose Lake during the early 1800's was dominated by benthic diatoms such as *S. construens* var. *venter* (Figure 8a). This taxon as well as other benthic species of the family Fragilariaceae have been found in shallow, alkaline lakes (Hickman, 1974) as well as the littoral regions of deeper lakes (Hickman & White, 1989). They are able to tolerate a wide variety of environmental conditions and have been found in lakes of variable trophic status. Wilson et al. (1997) reported that these species were found in British Columbia in lakes with total phosphorus (TP) values ranging from 5-100 $\mu g\ L^{-1}$. It appears that nutrients may not be an important determinant in their growth. Other factors such as sufficient light and suitable substrate may be more important. We suggest that benthic taxa are an indication of water clarity. These diatoms are more abundant under conditions of high water clarity as a larger lake bottom area is available for colonization. We suggest that these benthic Fragilariaceae indirectly indicate nutrient status as high water clarity occurs under low nutrient levels. Following this reasoning water clarity in Moose Lake was high prior to European settlement allowing these taxa to colonize a large portion of the lake bottom. The other important diatom taxa at the bottom of the core, *C. michiganiana*, and *C. bodanica* var. *lemanica*, have been found in the metalimnion of other high alkalinity Wisconsin lakes that possess low TP values and Secchi depths exceeding 3 meters (Garrison, unpublished data). These taxa also indicate high water clarity, as sufficient light must reach the metalimnion for their growth.

With the transition from subsistence to wheat farming in the mid-1800's (Western Historical Company, 1880; Langill & Loerke, 1984), farming intensified around the lake. Apparently soil delivery did not increase as porosity values and iron concentrations remained unchanged (Figure 6a). Phosphorus accumulation did increase along with the bulk sedimentation rate. Much of this increase likely was the result of increased precipitation of calcium carbonate ($CaCO_3$) as both the concentration and accumulation of inorganic carbon increased. In these high alkalinity marl lakes, increased $CaCO_3$ precipitation occurs as the lake water pH is elevated during increased primary productivity.

Around 1880, the dominant diatom taxa shifted from benthic to planktonic diatoms such as *C. michiganiana* (Figure 8a) indicating a modest increased in nutrient levels. Around 1900, the diatom community suggests the lake's productivity began to substantially increase, even though the geochemistry did not reflect increased deposition of phosphorus or nitrogen. Benthic taxa of the family Fragilariaceae nearly disappeared and metalimnetic taxa e.g. *C. michiganiana* and *C. bodanica* var. *lemanica* were replaced by *F. crotonensis*, *A. formosa*, and *S. minutulus*. *F. crotonensis* is often considered a reliable indicator of cultural disturbance associated with nutrient loading (Ennis et al., 1983, Engstrom et al., 1985, Christie & Smol, 1993, Stager et al., 1997). *A. formosa* and *S. minutulus* are also common in nutrient enriched waters (Bradbury, 1975; Carney, 1982; Fritz et al., 1993). The diatom accumulation rate increased around the early 1900's, further indicating increased nutrients in the lake.

Around 1910 the abundance of the diatom *Achnanthidium minutissima* Kützing increased. Reavie & Smol (1997) have found this taxa to be associated with rocks and macrophytes. Since few rocks are present in this lake, it is likely that the increased presence of *A. minutissima* suggests an increase in the macrophyte community. It appears that with the increased nutrients, the macrophyte community expanded or became denser. Reavie et al. (1998) found in a lake in the St. Lawrence River that the major site of diatom growth switched from rocks to macrophytes as nutrient levels increased. *A. minutissima* has been found in abundance in macrophyte beds in other Wisconsin lakes (Garrison, unpublished data).

Around 1980, there was a decline of eutrophic diatom *S. minutulus* and an increase in the metalimnetic diatoms *C. michiganiana* and *C. bodanica* var. *lemanica*. However, the diatom accumulation rate increased during the 1990's. The change in taxonomic composition suggests that nutrients have declined but remain higher than levels experienced prior to shoreland development.

Further, even though nutrients have declined, diatom productivity remains high. In three oligotrophic marl lakes in Michigan, Fritz et al. (1993) also found that diatom taxa indicated declining nutrient levels, but their productivity remained elevated.

Silver Lake

While the general developmental history of Silver Lake was similar to Moose Lake, the most intensive shoreland development period occurred later. The later development likely was due to the lake being located farther from Milwaukee. Although development had begun by 1920, the shoreland was not completely developed until 1970 with greatest development occurring between 1950 and 1963 (SEWRPC, 1997). Land clearing and early farming in the 1800's did not have as great an impact on the lake as occurred in Moose Lake. Although it is likely the intensity of farming was similar around both lakes, the topography around Silver Lake is flatter leading to less sediment entering the lake.

Around 1900, there was a large increase in nutrients (Figure 6b), especially phosphorus, although the sedimentation rate did not increase and porosity remained unchanged. This suggests that the nutrient source was not from soil erosion. Inorganic carbon (C) also did not change indicating these nutrients were not associated with $CaCO_3$ precipitation. Beginning around 1920, soil erosion did appear to increase as porosity values declined. The increased sedimentation rate may have been theh result of increased soil erosion as well as increased precipitation of $CaCO_3$. The peak sedimentation rate occurred between 1940 and 1965 which was the period of greatest shoreland development. Porosity values remained depressed during this time period and nutrient accumulation rates remained elevated. Inorganic C accumulation and concentration continued to be elevated, most likely from $CaCO_3$ precipitation in this marl lake. Since 1980 the sedimentation rate and accumulation of inorganic C has declined and porosity values have increased, suggesting less soil erosion after the construction of shoreland homes was complete.

The pre-settlement water quality in Silver Lake was similar to that in Moose Lake. Benthic diatoms of the family Fragilariaceae such as *S. pinnata*, *S. construens*, *S. construens* var. *venter*, and *P. brevistrata* dominated the diatom assemblage (Figure 8b) indicating high water clarity. The metalimnetic diatom, *C. michiganiana,* increased beginning around 1890 suggesting a small increase in lake nutrients but water clarity remained high, as benthic taxa were still an important component of the diatom assemblage. Beginning around 1920, the diatom accumulation rate

increased as did the accumulation of nutrients and inorganic carbon. By 1940 benthic taxa had nearly disappeared and metalimnetic taxa such as *C. michiganiana* and *C. bodanica var. lemanica* declined. *C. glomerata* which is considerated a planktonic diatom of mesotrophic to eutrophic lakes (Sreenivasa & Duthie 1973) began to increase. *A. ambigua*, *A. italica* (Ehren.) Simonsen, *A. formosa*, and *F. crotonensis*, indicative of higher nutrient levels (Bradbury, 1975; Reavie et al., 1995), became the dominant taxa in the diatom assemblage. The diatom accumulation rate also generally increased during this period and reached its peak around 1965, suggesting that increased soil erosion during home construction led to more nutrient runoff and stimulated algal growth.

The increased nutrients during home construction also had a significant influence upon the lake's littoral community. *A. minutissima* increased beginning in 1940 indicating an expansion in the macrophyte community. This coincides with the disappearance of the benthic Fragilariaceae taxa which largely grow on the sediment surface.

Around 1980, diatoms indicating elevated nutrient levels, e.g. *F. crotonensis* and *A. ambigua*, declined, while *C. michiganiana*, which indicates lower nutrient levels and greater water clarity, increased. It appears the lake's water quality has improved in the last 15 years despite the intensive shoreland development. Although the diatom accumulation rate remains elevated, unlike Moose Lake, productivity has generally declined since 1970.

Long Lake

The logging activity near Long Lake in the 1870's had only a small effect upon the lake's sedimentation rate (Figure 7a). This lack of significant effect of logging upon a lake's water quality was also reported for a lake in upper New York (Stager et al., 1997). A greater effect occurred during the extensive wildfires in the mid-1890's. The destruction of vegetation during these fires resulted in increased soil erosion as suggested by the increases in aluminum accumulation and sedimentation rate. Aluminum is associated with clastic minerals and is a good indicator of past soil erosion rates (Mackereth, 1966; Sasseville & Norton, 1975; Huttunen & Tolonen, 1977; Engstrom et al., 1985). There was little phosphorus associated with these soil particles as the phosphorus concentration and accumulation rate did not increase. Soil erosional rate as well as the sedimentation rate declined after the fires. Around 1940 both the sedimentation rate and the aluminum accumulation rate increased. This change likely was the result of shoreland development but agricultural activity may also have contributed. With the increased development on the lake

shore, sediment phosphorus concentration and accumulation greatly increased, peaking in 1950 (Figure 6). Aluminum accumulation continued to increase until the late 1970's although phosphorus declined. Aluminum accumulation has declined in the last 15 years indicating a reduction in soil erosion but phosphorus levels have increased.

In Long Lake, iron and to a lesser degree manganese accumulation rates increased around 1940, similar to the profiles of Al and P. Since the Fe:Mn increased during this time period it is likely that at least some of the iron accumulated as a result of input from the watershed as iron may be associated with clastics derived from erosion (Engstrom & Wright, 1984). During this time, the ratio of Fe to Al remained constant adding further evidence that the watershed was the source of the increased accumulation of Fe and not changing redox conditions. Beginning in the late 1980's, both Fe and Mn concentrations and accumulation increased significantly and the Fe:Mn declined. Since Fe and Mn are strongly affected by redox cycling, this increase suggests increased anoxia in the hypolimnion. As these elements are readily mobilized from sediments during anoxic conditions (Mortimer, 1941, 1942) they may be mobilized from surficial sediments on the slope of the lake basin and transported into profundal regions. Manganese can be selectively enriched because it dissolves at higher redox potential and tends to remain in solution longer than Fe (Jones & Bowser, 1978). A similar response was noted by Engstrom et al. (1985). The increased levels of Fe and Mn in the core are supported by increased concentrations of Fe and Mn measured in the hypolimnion of the lake since 1986 (Sorge, unpublished data).

The elevated accumulation rates of iron and manganese near the top of the core are largely driven by the sedimentation rate. The elevated sedimentation rate maybe due in part to an artifact of the CRS modeling. The ^{210}Pb stratigraphic profile (Figure 2) indicates a decline in activity at the top of the core. This may indicate sediment mixing by biota or physical processes. The CRS model interprets this as an increase in sedimentation rate. While it is likely that the elevated accumulation rates of iron and manganese in the upper sediments is the result of mixing processes, the decline of the Fe:Mn ratio (Figure 7a) as well as the general increase in concentrations of iron and manganese (Figure 5a) support our contention of increased anoxia in the hypolimnion.

Studies have documented that with anoxic conditions in the deep water sediments, phosphorus is released into the overlying waters (Mortimer, 1941,1942; Kamp-Nielson, 1974). Although the accumulation of Al has declined in recent years, P accumulation has increased. It is likely that this was largely the result of the mobilization of P from surficial sediments during anoxic conditions and

translocation to the profundal region. Thus it appears that even though soil erosion has declined during the last few years as shoreland home construction has slowed, anoxia has increased in the hypolimnion in the lake contributing to sharp increases in P, Fe, and Mn.

Prior to 1900, the diatom assemblage was dominated by *T. flocculosa* strain IIIp, *C. atomus*, and *A. ambigua* (Figure 9a). While *A. ambigua* was indicative of elevated P levels in Silver Lake, the same relationship apparently is not true in a low alkalinity lake like Long Lake. This taxon is has been found to be common in many low alkalinity oligotrophic lakes in the upper Midwest (Camburn & Kingston, 1986; Kingston et al., 1990) as well as a dominant taxon in other lakes that historically had low nutrient levels (Dixit et al., 1996). Apparently *A. ambigua* has differenet nutrient requirements in high versus low alkalinity environments.

Along with nutrient profiles, the pre-settlement diatom assemblage indicates that the lake's nutrient levels did not increase as a result of the logging practices and fires in the late 1800's. Likewise the early development of summer homes did not adversely affect the lake. The diatom assemblage was unchanged until about 1900. At this time *C. atomus* and *T. flocculosa* strain IIIp declined and *C. michiganiana* and *C. glomerata* increased. Dixit et al. (1996,1998) reported that *T. flocculosa* strain IIIp declined in response to increased nutrients although Stager et al. (1997) noted an increase coincidental with *F. crotonensis* and *A. formosa*. With the increase of metalimnetic taxa, e.g. *C. michiganiana*, it appears that some of the increase in diatom productivity occurred in the metalimnion in response to a subtle increase in nutrient levels. At about 1950 *F. crotonensis* and *A. formosa* became the dominant taxa in the diatom assemblage indicating increased nutrient levels. This change in the diatom community occurred at the same time that the sediment phosphorus concentration accumulation increased, suggesting a cause and effect relationship. The latter two taxa remained dominant until about 1990 when they declined.

Around 1920 there was an increase in the macrophyte density as indicated by the increase in *A. minutissima*. Similar to Moose Lake, the increased macrophytes occurred earlier than the increase in nutrients during the major construction phase. *A. minutissima* abundance began to decline in the 1970's indicating a reduction in the macrophyte community. Rusty crayfish, *Orconectes rusticus*, invaded the lake in the 1960's and by the mid-1970's they had reduced the extent of the aquatic plant community (Borman, 1993). With the reduction of the plant community, the epipelic diatom *S. pinnata* became more common. The reduction since 1990 of the taxa *F. crotonensis* and *A.*

formosa, which are common under higher nutrient levels, indicates that water clarity has improved even though anoxia in the hypolimnion has become more extensive.

Round Lake

The initial shoreland development during the 1920's resulted in a short term increase in the sedimentation rate (Figure 7b). There was a significant increase in soil erosion during the construction of the cottages as indicated by the increase in aluminum concentration and accumulation. This increased soil erosion resulted in an increase in phosphorus accumulation as well. Most of the increased phosphorus was associated with soil particles as the Al:P was constant throughout the core (Figure 7b). After the initial construction was finished, the sedimentation rate declined to near pre-settlement levels. Accumulation of Al and P also declined although not to pre-settlement levels. A second construction phase occurred during the 1970's and into the early 1980's. During this construction phase, many older cottages were converted to year-round homes which resulted in an increase in impervious surface area as well as a decline in native vegetation and an expansion of mowed lawns. This resulted in increased soil erosion as suggested the increased concentration and accumulation of aluminum. Phosphorus accumulation also increased during construction. During the 1990's the accumulation of Al and P declined indicating a reduction in soil erosion rates, although today they remain higher than pre-settlement levels. Even though soil erosion has declined, the highest sedimentation rates occur at the top of the core.

Prior to logging in the late 1800's, benthic and attached algae such as *S. pinnata* and *Eunotia* spp. dominated the diatom assemblage (Figure 9b). Following the logging, *S. pinnata* became less important and was replaced by the planktonic diatom *T. flocculosa* strain IIIp. Because the geochemistry does not indicate any significant increases in soil erosion or phosphorus at this time it appears that the decline in the benthic diatoms was not a result of decreased water clarity but more likely a result of changes in the lake's water level. Since diatom taxa indicative of increased dissolved organic carbon did not change, it appears that increased water color did not drive the change in the diatom community.

The initial cottage development in the 1920's had little effect upon the lake's nutrient status. The macrophyte community did appear to increase in the early 1900's as indicated by the increase in the *A. minutissima.* The diatom assemblage indicates that the major construction period during the 1960-70's had a detrimental effect on the nutrient status of the lake. *A. ralfsii* var. *americana*

increased around 1960 as well as *T. flocculosa* strain IIIp. The increase in *A. ralfsii* var. *americana* likely indicates a nutrient increase (Kingston et al., 1990) in the lake. The percentage of planktonic diatoms also began to increase at about this time. Other studies have found that planktonic diatoms increase as nutrient levels rise (Bradbury & Winter, 1976; Battarbee, 1978). Since 1985 the abundance of the benthic diatom *S. pinnata* has declined and the planktonic diatom *T. flocculosa* strain IIIp has increased, indicating a further decline in water clarity. This has occurred even though soil erosion has declined. Asplund (1996) found that sediments from this lake when suspended in the water column could stimulate algal growth. This lake has experienced increased motor boat usage in the last few years (Sorge, personal communication) in part because of the construction of an improved boat launch. It is likely that the increased motorboat activity (both numbers and boat size) has increased sediment resuspension in this shallow lake and may be contributing to a decline in water quality.

All of these lakes suffered degradation as a result of shoreland development, although the degree of the degradation varied among the lakes. The initial development, whether following the arrival of Europeans in the 1800's or during the 1920's, resulted in small but significant impacts on the lake's nutrient status. The construction period, whether during the early development phase or during later home improvement, caused the greatest increase in sediment and nutrient inputs. Between the two high alkalinity southern lakes, increased sediment and nutrients was most evident in Moose Lake (Figure 2), likely a result of the steeper topography as well as the greater density of shoreland development

The reconstruction phase in the last half of the twentieth century has had the largest impact on all of the lakes. The sediment delivery was higher and the nutrient input was also elevated compared with the initial construction period. As cottages were converted to year-round homes or upgraded for seasonal use, more land was disturbed in order to enlarge the habitations as well as add additional structures such as garages. In many cases, lots were suburbanized which removed much of the native vegetation and replaced it with manicured lawns.

In the southern high alkalinity lakes, the input of sediments and nutrients declined following completion of construction in the redevelopment phase. This decline was especially evident in Moose Lake presumably because of its steep topography. In Silver Lake, the sedimentation rate declined following construction although nutrient accumulation did not. Apparently there is a nutrient source other than soil erosion to this lake. During the last decade, there has been a smaller

but noticeable decline in soil erosion in the northern lakes as indicated by the decline in aluminum accumulation (Figure 7) most likely as a result of reduced construction activity.

In the high alkalinity lakes, nutrient accumulation rates currently are higher than pre-settlement levels although the sedimentation rate is similar to pre-settlement rates,. Even though there is reduced input of sediment there are other significant sources of nutrients. Grass clippings from lawn mowing and other debris could be washed into the lake following storm events and during spring runoff. It is also likely that at least some of these lawns are periodically fertilized. Historically, native shoreline vegetation acted as a buffer against the input of nutrients from overland runoff. In many cases these buffers have been removed as lawns are mowed to the water's edge.

This study documents the limnological changes in these lakes as nutrient inputs gradually increased during the period of European settlement. A summary of the diatom community demonstrates the changes that have occurred in the three stratified lakes. In the high alkalinity lakes (Moose and Silver lakes), planktonic diatom production was low historically and the major site of diatom growth was the sediments (Figure 10). The planktonic diatom production that did occur was primarily found in the metalimnion. As nutrient levels increased with early development, more of the diatom production shifted to the metalimnion. However, water clarity remained high as indicated by the presence of benthic taxa. As nutrient levels further increased in the lakes, the major site of diatom production shifted from the metalimnion to the epilimnion. These changes coincide with the major shoreland development period. As nutrient input increased, the diatom assemblage shifted to taxa indicative of eutrophic conditions such as *F. crotonensis, A. formosa* , *S. minutulus*, and *A. ambigua*. In Long Lake, changes in the the diatom groups was not great as the two southern lakes. This was largely because during the pre-settlement period the majority of the diatoms inhabited the epilimnion (Figure 10). However the same trend is still observed. Even though the geochemical variables indicated little change in nutrient deposition in the early part of the twentieth century, there was an increase in metalimnetic diatoms. With the increase in shoreland development beginning around 1940, epilimnetic diatom abundance increased. This coincided with increases of phosphorus concentration and accumulation in the sediments (Figures 5a, 7a). This same analysis was not done for Round Lake. Since this lake does not stratify all diatom production occurs in the epilimnion and the lake bottom. However the increase in the percentage of planktonic diatoms indicates an increase in the nutrient status of the lake during the period of greatest shoreland construction activity (1970-1990) (Figure 9b).

Along with the general shift from metalimnetic to epilimnetic taxa with increased nutrient and sediment input, the character of the littoral zone also changed. Macrophytes became much more important as suggested by the increase of *A. minutissima* which often grows as an epiphyte on macrophytes. With the expansion of the macrophytes, the epipelic diatoms became comparatively rare (Figure 10). In both lakes the macrophyte community does not appear to have declined with the reduction in water column P levels since the epiphytic diatom *A. minutissima* is still common and there are almost no epipelic taxa (Figure 8).

With completion of the major construction phase, both southern lakes have shown some improvement in water quality. Sedimentation rates have declined to near pre-settlement levels although nutrient accumulation rates have remained elevated. Greater improvement has occurred in Silver Lake compared to Moose Lake most likely because the latter lake has a higher density of development. In Silver Lake the diatom assemblage has reverted to dominance by metalimnetic taxa (Figure 10). The diatom accumulation rate is less than that during the construction period but is still considerably above background levels (Figure 8b). Silver Lake currently has an extensive algal community in the metalimnion. The highest summer chlorophyll *a* levels occur, not in the surface waters, but instead in the metalimnion (Asplund, unpublished data). In Moose Lake the diatom assemblage is still dominated by epilimnetic taxa but there has been an increase in metalimnetic taxa in recent years (Figure 10).

Despite improvement of water quality in recent years in the pelagic zone in the southern lakes, studies of other lakes have clearly documented the adverse effects of development upon the nearshore area. Jennings et al. (1996) found that fish assemblages are less diverse along developed shorelines when compared to undeveloped sites. This decrease in diversity is the result of a degradation of the littoral zone because of changes in the composition and density of the macrophyte community (Bryan & Scarnecchia, 1992), quantity and composition of shoreline habitat such as woody debris (Christiansen et al., 1996) and size and uniformity of substrate particles (Jennings et al., 1996).

The marl lakes have shown some improvement in recent years despite moderate to high levels of shoreland development, while the low alkalinity lakes have shown minimal improvement if any. In the deeper Long Lake, the diatom assemblage indicates similar phosphorus levels for the last 50 years but the geochemistry indicates more extensive hypolimnetic anoxia. This increased anoxia

could eventually result in additional release of P from the sediments thus enhancing internal P loading, and stimulating higher algal production. In Round Lake which is shallower, P levels have not improved even though soil erosion has declined with the completion of home construction. In fact the diatom assemblage indicates that the lake's P level has increased during the last decade.

Based upon results from these four lakes, it appears the low alkalinity lakes are more sensitive to shoreland development than marl lakes. This is not surprising because marl lakes are able to buffer the adverse effects of phosphorus inputs (Wetzel, 1970) because of high levels of calcium carbonate which compete with algae for phosphorus. Low alkalinity lakes do not have this innate buffering ability and therefore are more sensitive to nutrient inputs. This is especially significant as many areas that are undergoing intensified development, such as the upper Midwest, northeastern USA, and southern Ontario, possess a large number of low alkalinity lakes. Shoreland development, even at low densities, increases nutrient export to the lake. While the greatest export often occurs during construction, nutrients continue to enter the lake from runoff from impervious surfaces and lawns. This nutrient input can be minimized by the wise use of riparian buffer strips, building homes set back from the lakeshore, and maintaining development at a moderate density.

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Table 1. Geographic and morphometric characteristics of study lakes.

	Latitude North	Longitude West	Area (ha)	Max Depth (m)	Mean Depth (m)	Hydrologic Type	Homes/km of Shoreline
Moose	43°08'06"	88°24'16"	34	18.6	12.2	Seepage	33
Silver	43°04'09"	88°29'31"	90	13.4	4.8	Seepage	14
Long	45°15'19"	91°26'55"	426	30.8	6.1	Drainage	6
Round	45°14'52"	91°24'04"	87	5.5	3.0	Seepage	5

Table 2. Limnological characteristics of study lakes.

	Total P (ug L^{-1})	Total N (u g L^{-1})	Chlorophyll (ug L^{-1})	Secchi (m)	Alkalinity (mg L^{-1})	Color (PCU)
Moose[1]	11	0.7	2.2	4.4	167	7.5
Silver[1]	11	1.0	5.6	2.4	163	N/A
Long[2]	12	0.6	4.8	3.1	53	17
Round[2]	21	0.7	10.8	1.6	5	26

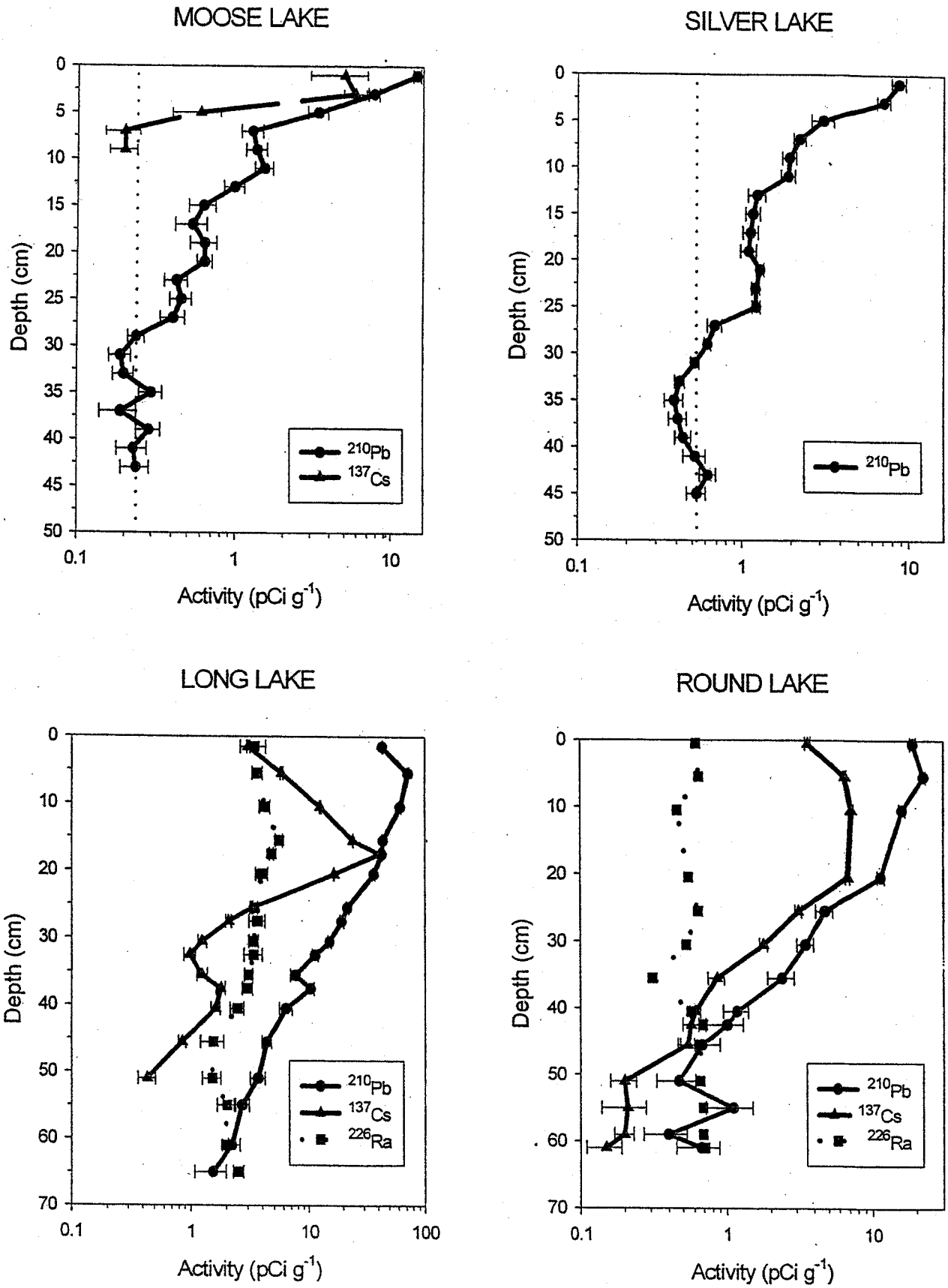
[1]Garrison, unpublished data
[2]Wisconsin Dept. Natural Resources, Long Term Trends Data

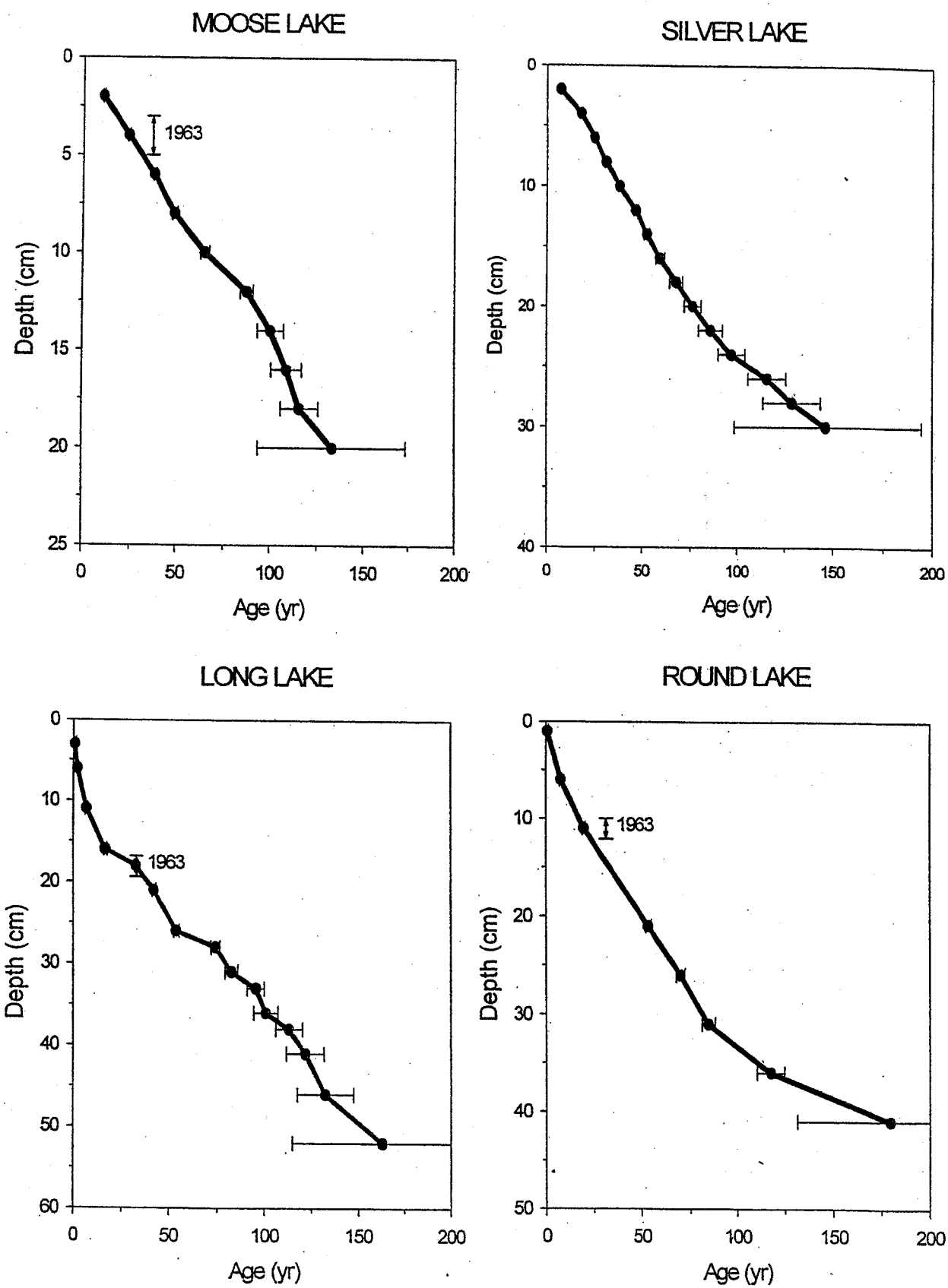
Table 3. ^{210}Pb parameters and sediment accumulation rates for the study lakes.

	Cumulative Unsupported ^{210}Pb (pCi cm^{-2})	Unsupported ^{210}Pb at surface (pCi g^{-1})	Mean Sedimentation rate (g cm^{-2} yr^{-1})	Mean ^{210}Pb flux (pCi cm^{-2} yr^{-1})
Moose	5.15	14.19	0.022	0.16
Silver	8.26	8.24	0.027	0.26
Long	73.78	40.22	0.020	2.30
Round	23.20	18.22	0.015	0.72

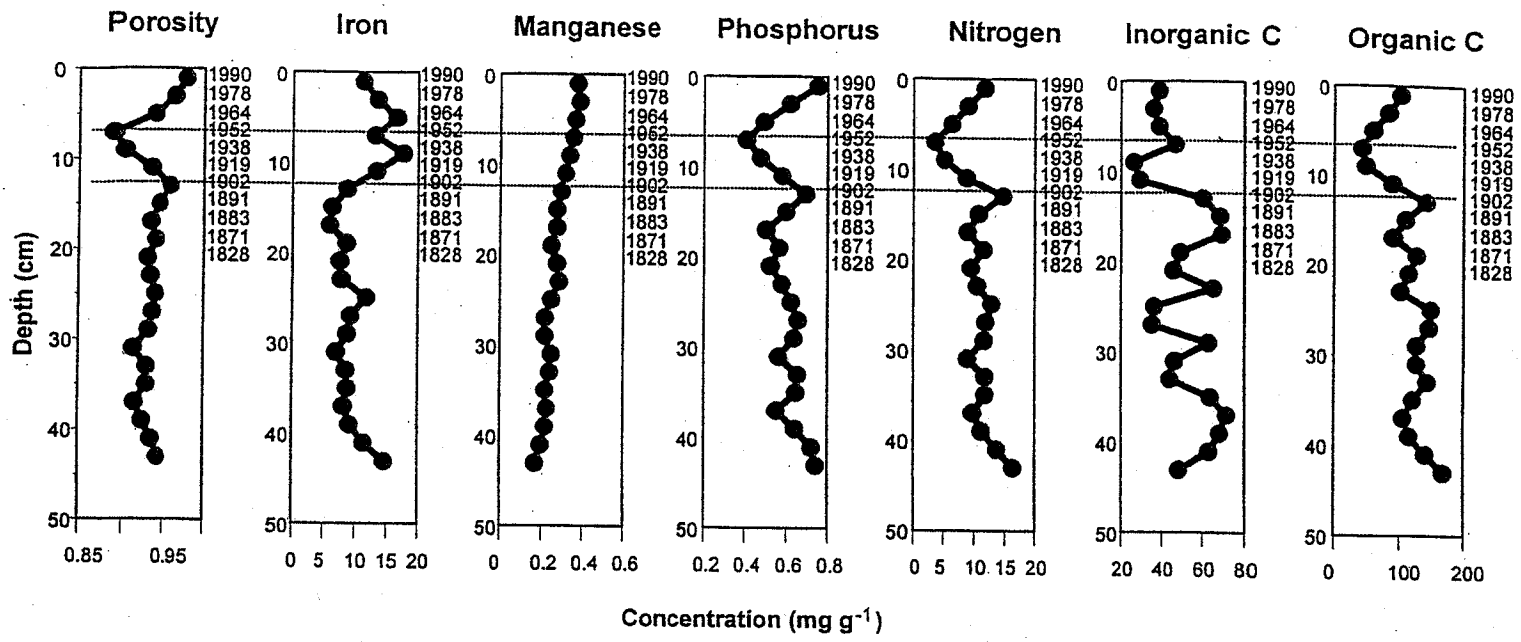
List of Figures


Round
Long
Moose
Silver


MOOSE LAKE
Depth (cm)
Activity (pCi g-1)
210Pb
137Cs
SILVER LAKE
Depth (cm)
Activity (pCi g-1)
210Pb
LONG LAKE
Depth (cm)
Activity (pCi g-1)
210Pb
137Cs
226Ra
ROUND LAKE
Depth (cm)
Activity (pCi g-1)
210Pb
137Cs
226Ra


MOOSE LAKE
1963
Depth (cm)
Age (yr)
SILVER LAKE
Depth (cm)
Age (yr)
LONG LAKE
1963
Depth (cm)
Age (yr)
ROUND LAKE
1963
Depth (cm)
Age (yr)

MOOSE LAKE



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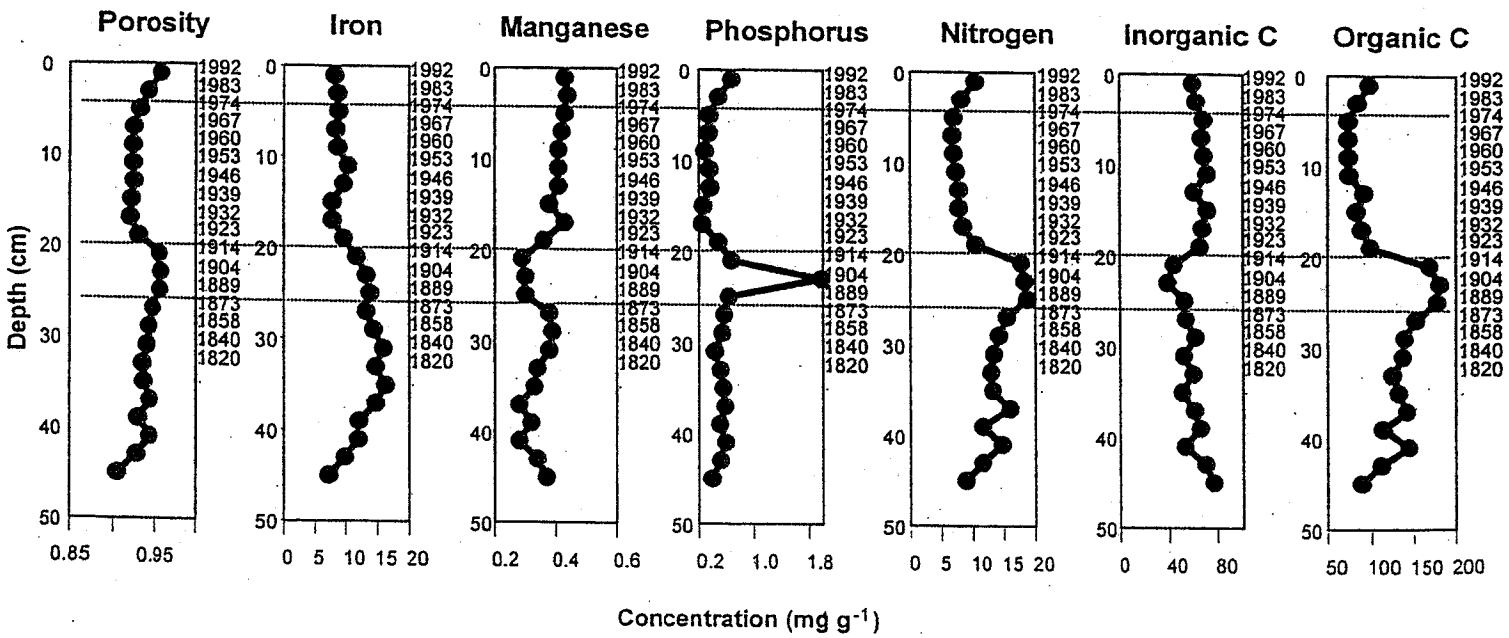

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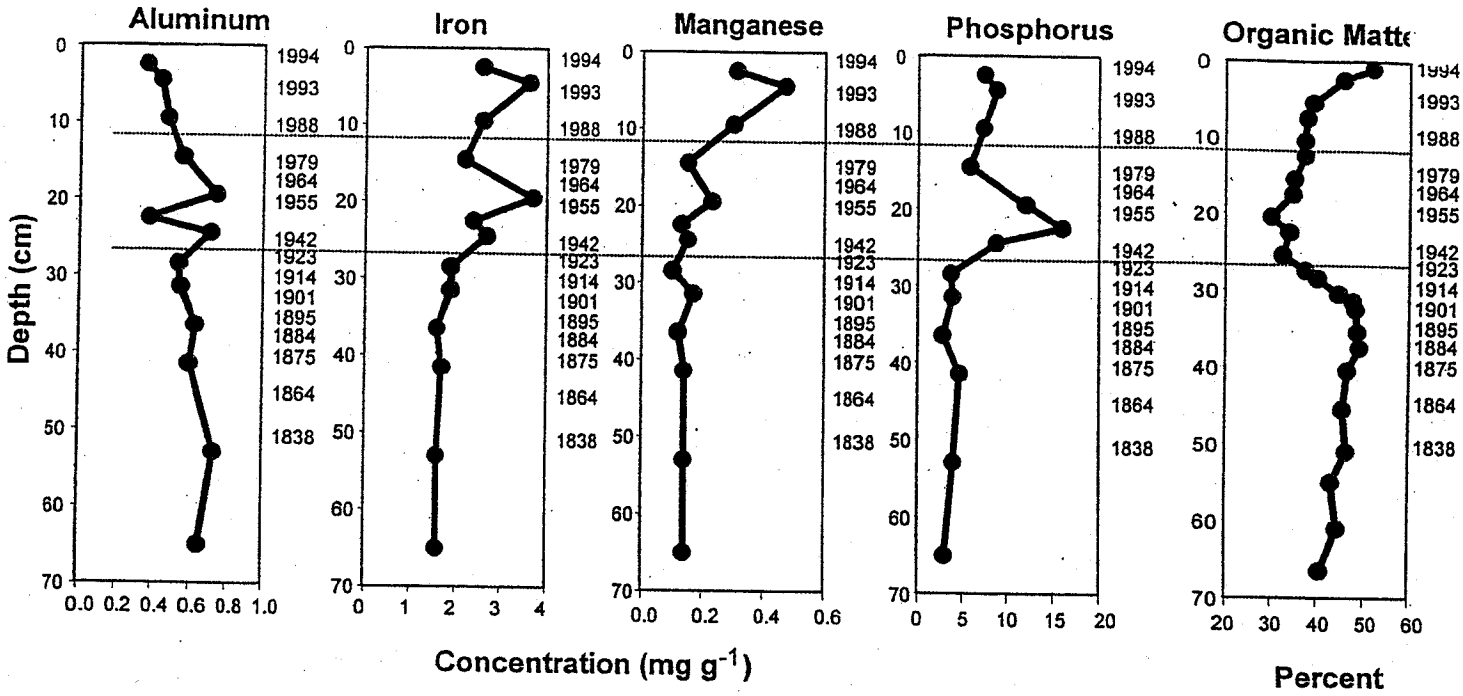

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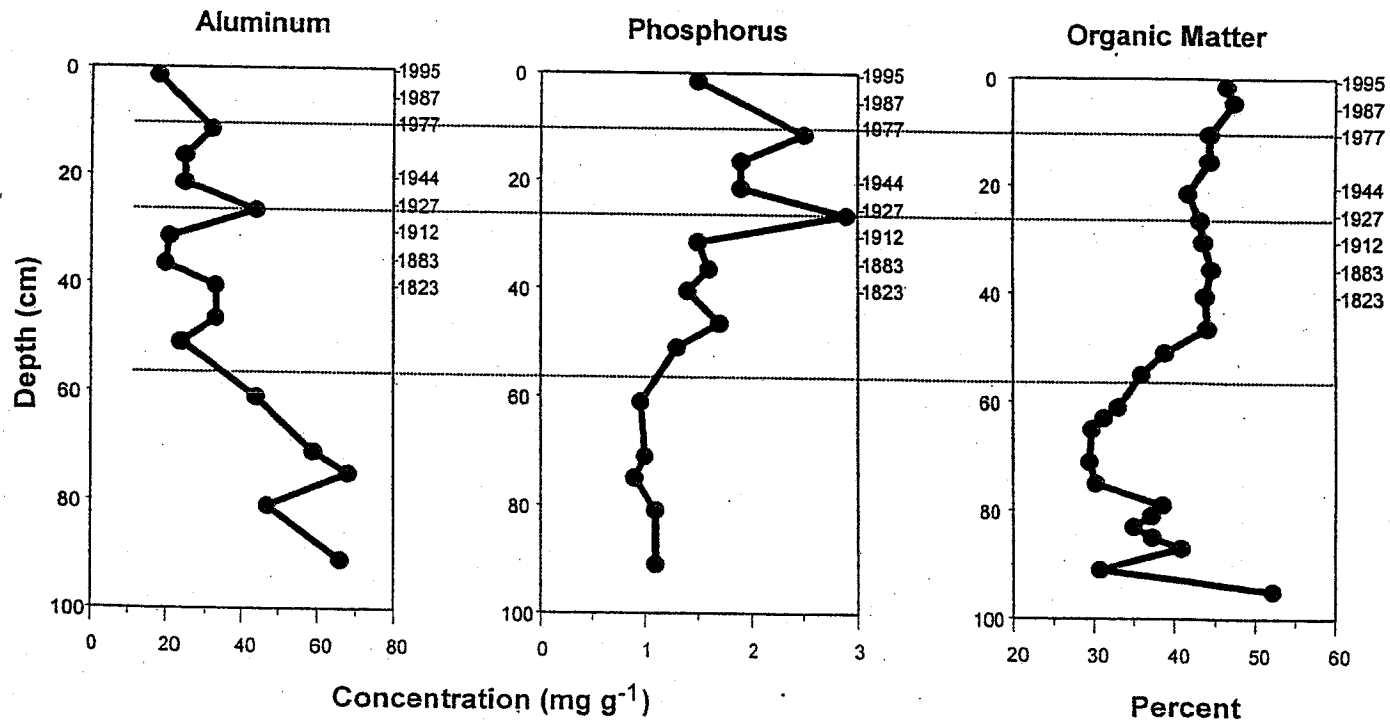

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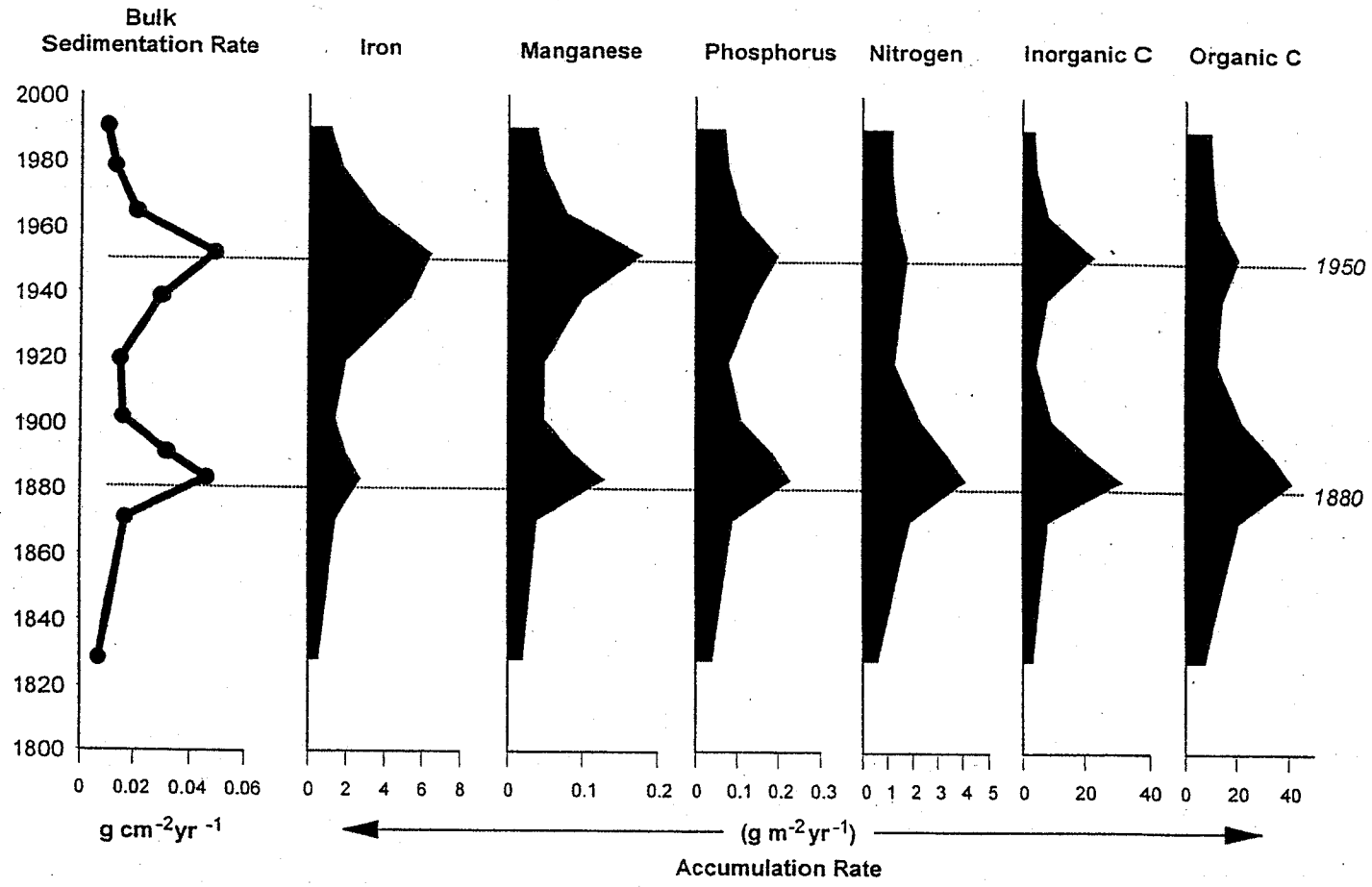

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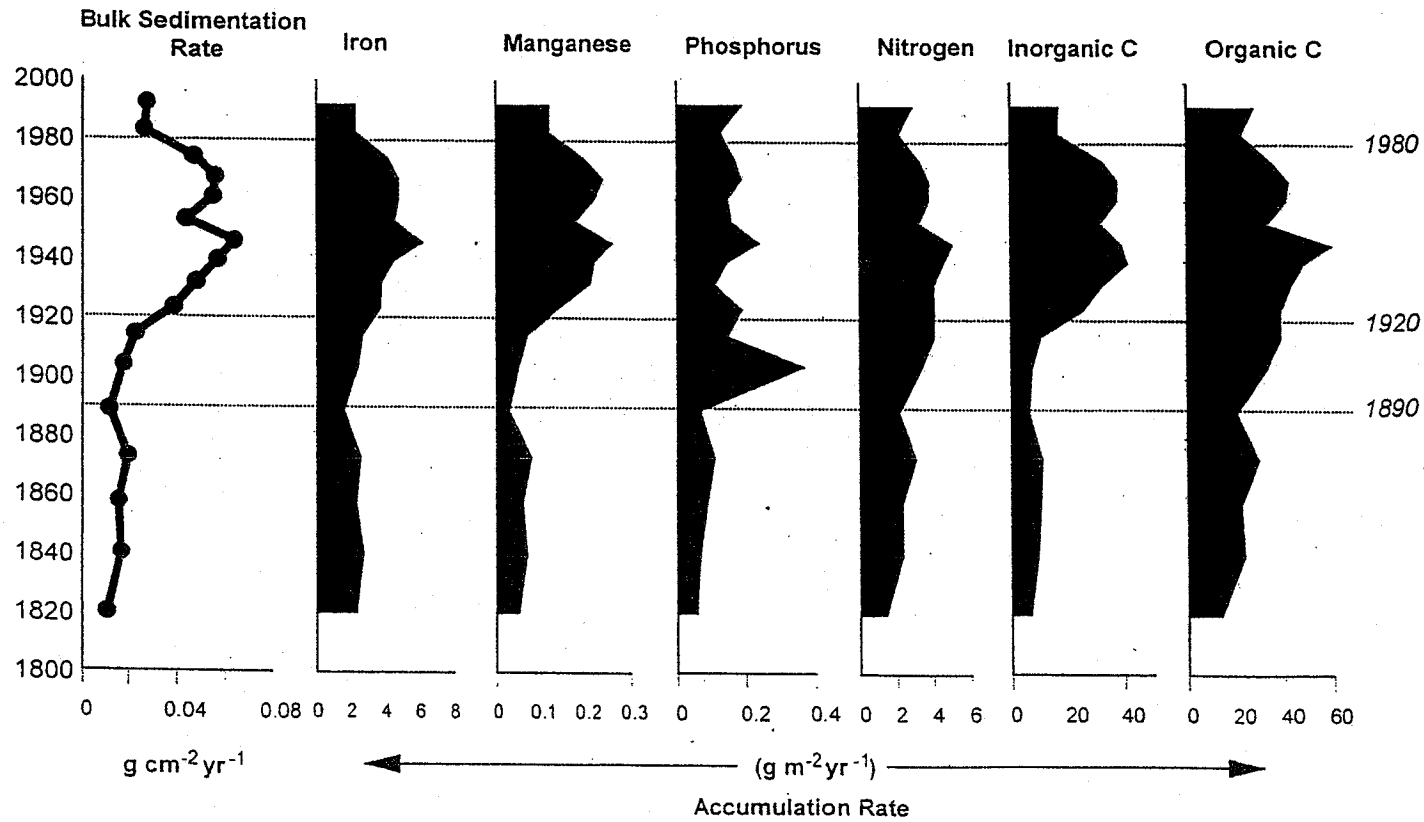

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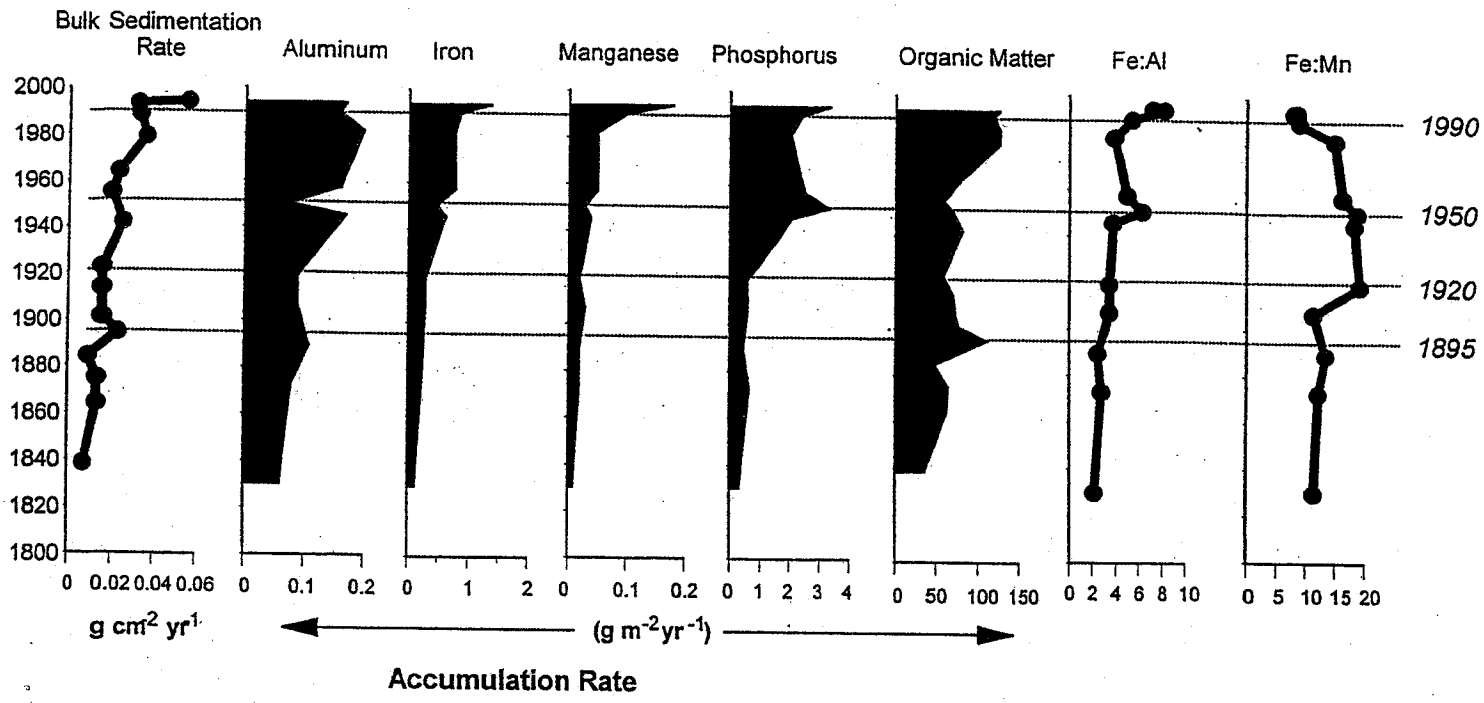

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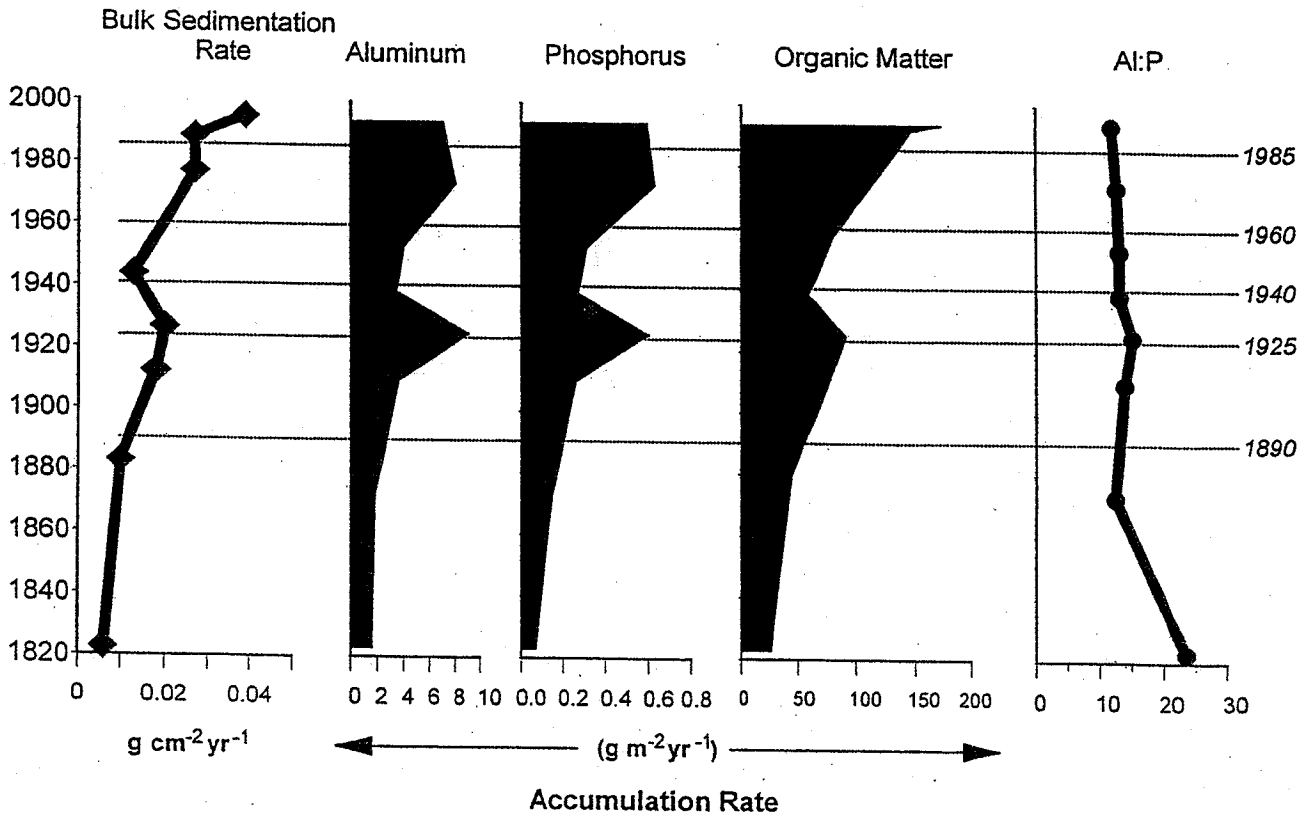

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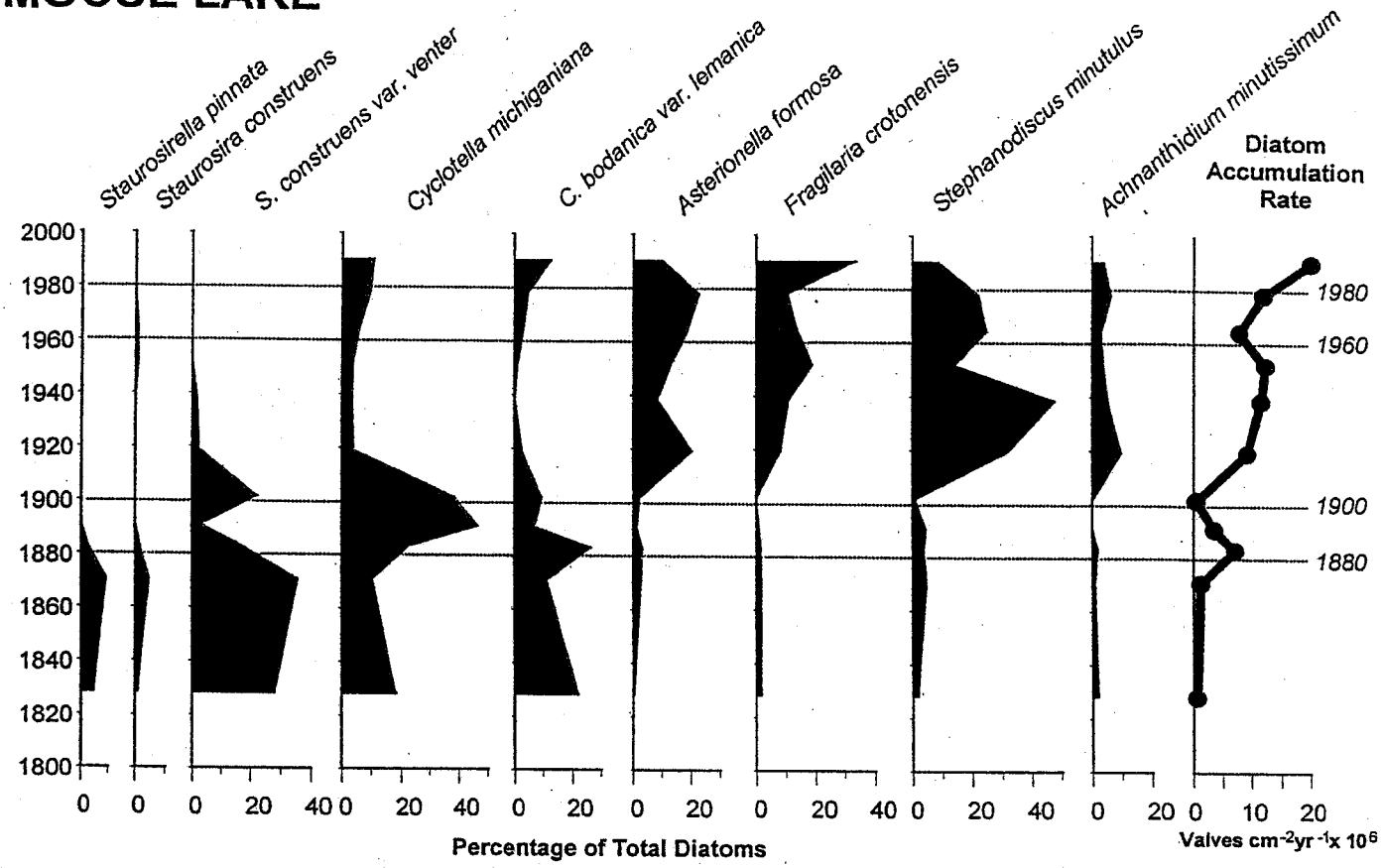

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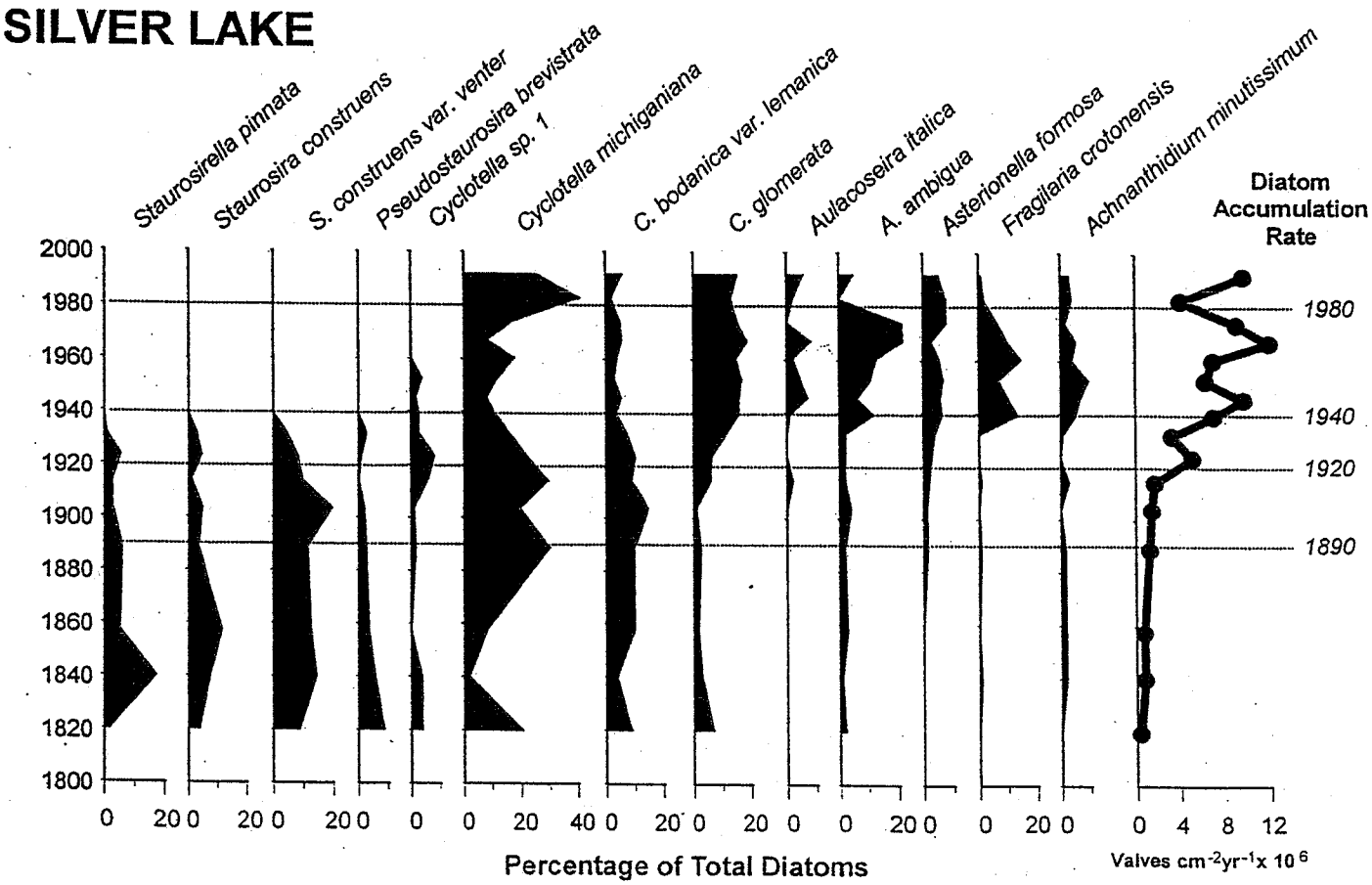

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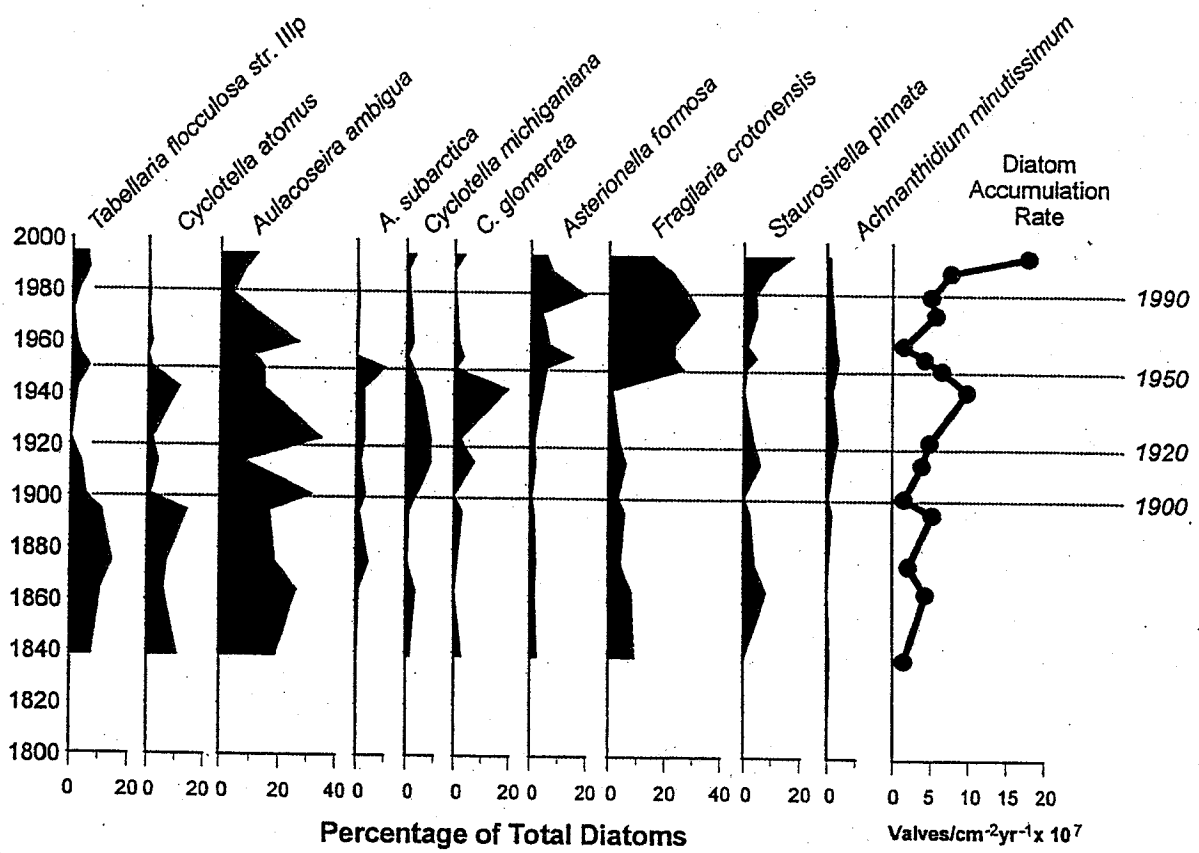

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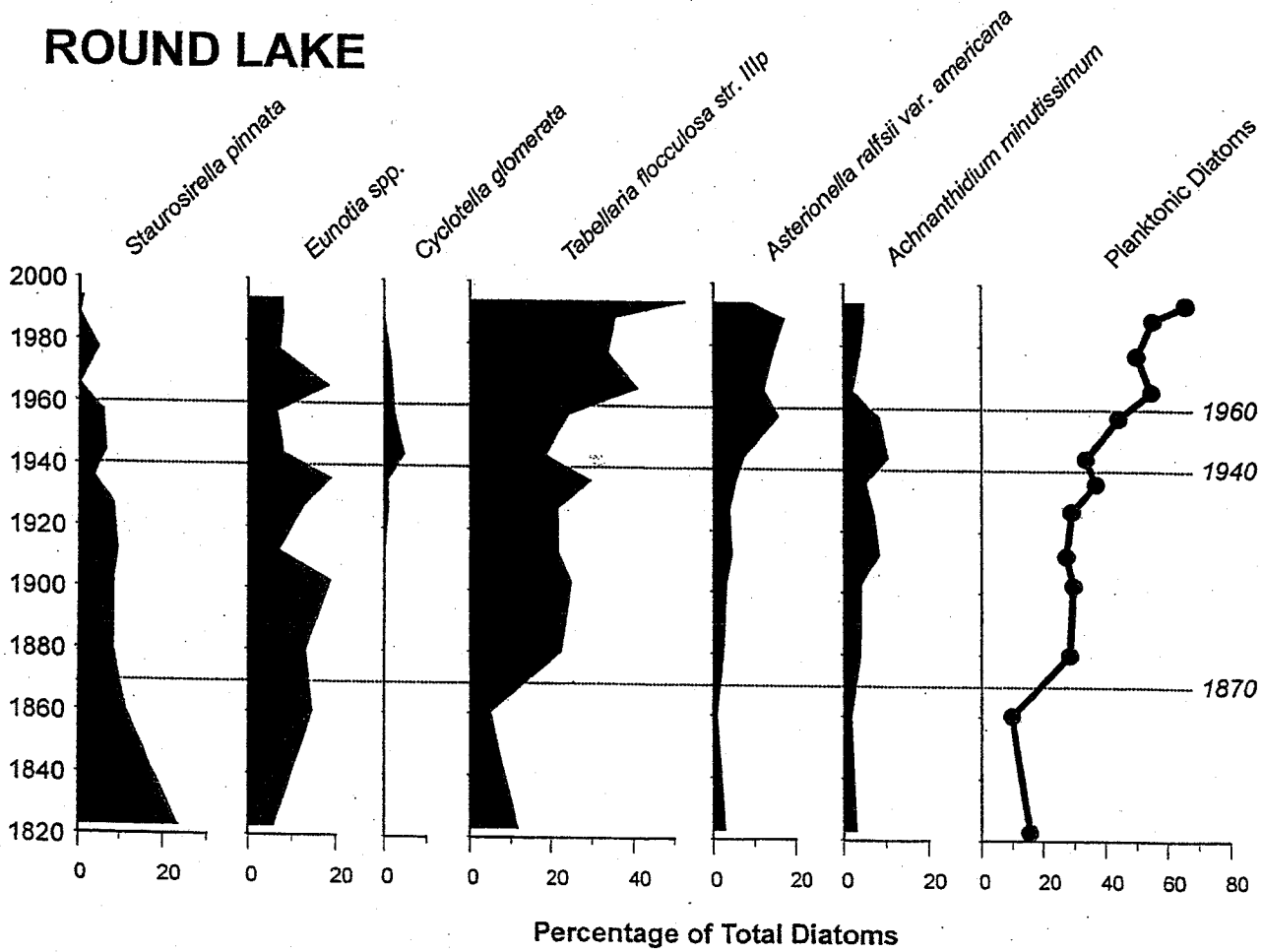

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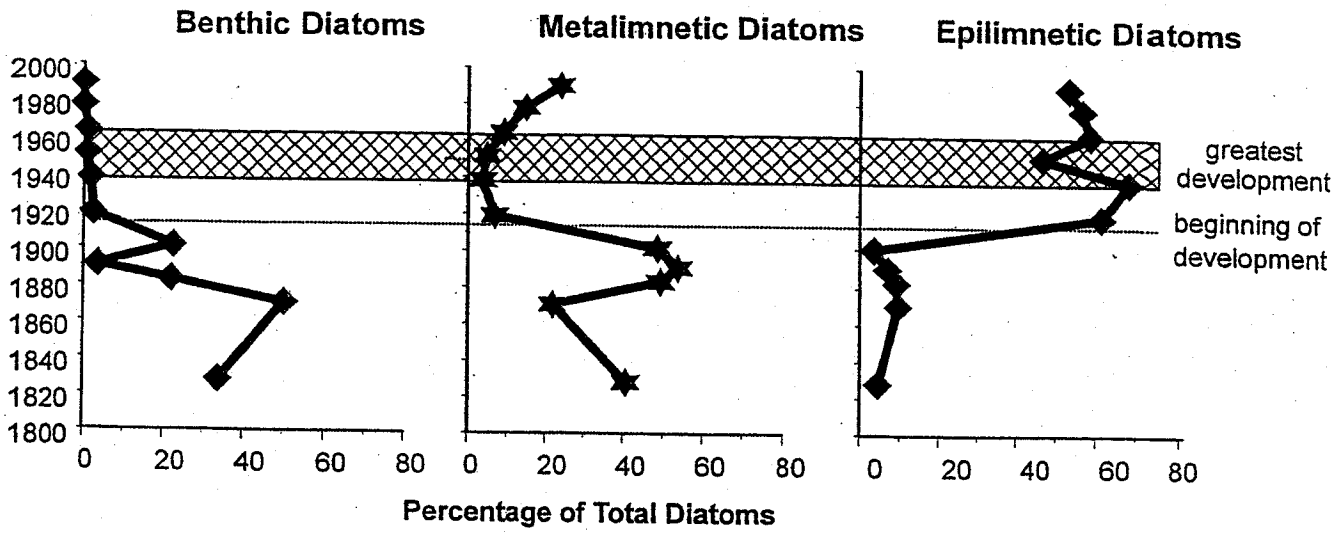

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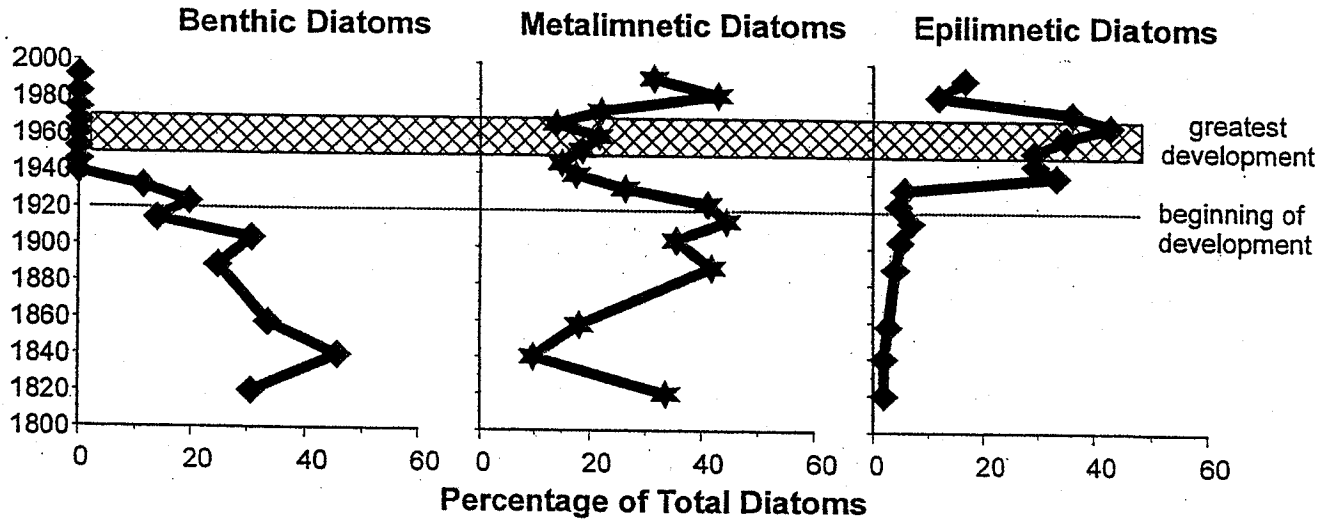

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