A Paleolimnological Assessment of the Diatom Communities of Lake Opeongo, Ontario, Canada

by

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Abstract

Lake Opeongo is an oligotrophic lake in Algonquin Provincial Park, Ontario, Canada (45°42' N, 78° 22' W). Limnological conditions over the past 300 years were tracked using diatoms preserved in the sediments and applying diatom-based transfer functions to infer total phosphorus (TP). Before European settlement, Lake Opeongo was highly oligotrophic, with the diatom community consisting of *Cyclotella stelligera* complex with subdominants *Tabellaria flocculosa* IIIp and *Aulacoseira distans*. No changes occurred until ca. 1956 when the diatom community shifted to the current mesotrophic assemblage consisting of *Asterionella formosa* with lesser amounts of *Cyclotella bodanica* var *lemanica*, *C. stelligera* complex, *Fragilaria crotonensis* and *T. flocculosa IIIp*. This shift could have occurred due to increased direct human impacts on the watershed; increased post-war fertilizer use; global warming, including changes in the thermal conditions of the lake; and trophic level changes caused by human manipulation of the fish community of the lake.

To Allan and Anita with thanks

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Chapter 1: Introduction

Many ecological studies of lakes are hampered by a poor understanding of the lake's past environmental conditions. Paleolimnological studies offer a solution to this problem as it is possible to reconstruct the history of a lake using the fossil remains of various organisms preserved in the lake sediments. With such background information one can begin to characterize the lake's previous environmental conditions. One type of organism widely used in such studies are diatoms. Diatoms are unicellular algae belonging to the class of Bacillariophyceae that are useful environmental and paleoenvironmental indicators. The community composition reflects and responds to environmental conditions such as water depth, aquatic pH, nutrient availability, salinity and current conditions, etc. (Stoermer and Smol, 1999). They preserve well in many sediments due to the siliceous composition of their cell walls and can therefore be used to reconstruct past environmental conditions (Dixit et al., 1999). Knowledge of taxa's ecological optima (i.e., estimates of the value along an environmental gradient at which a taxon consistently achieves its highest abundance relative to other taxa) and tolerances (i.e., estimates of the range of occurrence of a taxon along an environmental gradient) allows for transfer functions to be constructed and hence, a lake's ecological history or paleohistory can be reconstructed by analysis of diatoms within a sedimentary core.

Imbrie and Kipp (1971) revolutionized Quaternary paleoecology by developing a procedure for the quantitative reconstruction of past environmental variables from fossil assemblages using transfer or calibration functions. Since this pioneering work on marine foraminifera and their relationship with ocean surface temperatures and salinity, the general approach of quantitative environmental variable reconstruction has been expanded to many different groups of organisms, including diatoms, pollen, chrysophytes, chironomids, ostracods and radiolaria (Charles and Smol, 1994). The main goal of this approach is to express the value of an environmental variable (e.g. lake-water epilimnetic total phosphorus (TP)) as a function of biological or proxy data (e.g. diatom assemblages) (Birks, 1995). The function that does this is the "transfer function". Its construction involves building a modern "training or

calibration set" consisting of quantitative counts of the modern assemblages of the study organisms from many different sites. The modern values of the environmental variables to be reconstructed are also measured at these sites. The modern relationships between the biological assemblages and the environmental variables are modeled statistically and the resulting function is used as a transfer function to transform fossil assemblage data into quantitative estimates of the past environmental variables. Many diatom-based transfer functions derived from weighted-averaging regression (WA) and calibration models exist to quantitatively infer trophic-related variables from sedimentary diatom assemblages (e.g. reviewed in Hall and Smol, 1999). The validity of using diatom-based WA inferences to reconstruct paleoenvironmental variables has been tested by comparison to actual historical records (i.e., Renberg and Hultberg, 1992; Bennion et al., 1995; Lotter, 1998) and shown to be reliable. Several calibration sets for TP exist and more are in progress. Among these are four transfer functions that infer TP from North American diatom assemblages, those of Reavie et al. (1995a), Hall and Smol (1996), Wilson et al. (1996) and Reavie and Smol (*in press*).

Eutrophication may be defined as the process of enrichment of a water body due to an increase in nutrient loading (Horne and Goldman, 1994). It is synonymous with increased growth rates of the biota of lakes and an increase in biomass at all levels of the food chain, including fish (Wetzel, 1983). The most important limiting nutrients that cause eutrophication are phosphates, nitrates and ammonium ions; although much more rarely increases in carbon dioxide, silica, iron or trace minerals may cause eutrophication. Significant changes occur at all levels in the trophic hierarchy and entire communities can be greatly altered or disappear (Hall and Smol, 1999). Changes in the N:P ratio can result in shifts in the phytoplankton from diatoms and other smaller edible algae towards the larger and less edible cyanobacteria that are better competitors for nitrogen (Tilman et al., 1986; Reynolds, 1984). Decomposition of the increased algal biomass reduces oxygen availability in the hypolimnion, causing changes in the benthic community and a reduction of fish habitat, and in extreme cases, massive fish kills (Hall and Smol, 1999). Increased phytoplankton loads causes an increase in turbidity and lessens light

levels which can reduce the submerged aquatic macrophytes which form important habitats and food for many organisms in the lake.

Eutrophication may occur naturally during the course of lake succession (Horne and Goldman, 1994) or in response to natural disturbances in watershed such as fire (Hickman et al., 1990) or forest die-off (St-Jacques et. al., 2000; Hall and Smol, 1993). Climatic changes, such as droughts, may concentrate lakewater nutrients or increase the relative contribution of a nutrient-rich groundwater (Webster et al., 1996). From historical and paleolimnological studies, a shift towards more eutrophic conditions caused by anthropogenic impacts has been identified as the most widespread type of environmental pollution affecting the world's lakes and reservoirs (Harper, 1992). The commonest direct causes of such cultural eutrophication are discharge of sewage, excess fertilizer runoff from local fields and urban storm runoff into lakes and rivers. Lake Washington, Seattle, Washington, (Griffiths and Edmondson, 1975; Wetzel, 1983) is a classic case of eutrophication due to direct human impact. The lake became enriched by increasing volumes of sewage effluent during 1941 to 1953. Algal production and phosphate concentrations increased markedly. Effluent was diverted away from the lake in 1963, and by 1969, phosphate and summer chlorophyll *a* had decreased to roughly one fourth of their 1963 values.

Other possible causes of eutrophication include increased nutrient run-off from the watershed from logging (e.g., Bormann and Likens, 1979), increased North American post-WWII inorganic fertilizer use, and anthropogenic induced climate changes. Increased inorganic fertilizer use by the industrialized countries in the post-war period has led to a severe problem with long distance airborne transport and deposition of excess fertilizer (Vitousek, 1994; Langan, 1999; Jefferies and Maron, 1997). Even remote areas such as Great Slave Lake (Stoermer et al., 1990) and alpine lakes in the Colorado Rockies (Wolfe et al., 2000) that have never had any large human settlements in their watersheds are showing eutrophying trends because of this airborne deposition. Anthropogenic climate changes includes global warming which can alter boreal forest lakes' thermal structures, and increase chemical concentrations in boreal lakes because of decreased water renewal and forest fires in catchments (Schindler et al.,1990;

King, 1997). Global warming, acting together with lake acidification from airborne pollution, can also cause declines in dissolved organic carbon (DOC), allowing deeper penetration of ultraviolet-B light which is harmful to aquatic organisms (Schindler et al, 1996).

The following three North American paleolimnological studies demonstrate the use of diatoms in examining the course of eutrophication. Bradbury (1975) examined the diatoms preserved in the sediments of seven lakes in Minnesota and South Dakota and was able to show limnological changes associated with enrichment following European-American settlement, land clearance and logging. Brugam (1988) in his diatom study of the long-term history of Meridian Lake, Washington, found two periods of eutrophication: the first in the 1880's when the watershed was deforested by logging and the second in the late 1940's when the watershed was developed for suburban housing. Reavie et al. (1995b) examined six currently eutrophic British Columbia lakes and applied a transfer function to reconstruct past trophic conditions. They found that only three of the lakes showed significant eutrophication since European settlement, and in two of the three cases this eutrophication started after WWII as land use, including agriculture, in the region intensified. The other three appeared to have always been eutrophic.

Concern about eutrophication has directed the attention of limnologists towards management of nutrient supply as a regulator of lake productivity. However, nutrient supply alone cannot explain all the variation in the primary productivity of lakes. The trophic cascade concept explains the variation in the primary productivity of aquatic systems that is not explained by variation in nutrient supply (Carpenter and Kitchell, 1993; Carpenter *et al.*, 1985). Consider a lake food web that consists of limiting nutrients and four basic trophic levels: piscivores, zooplanktivores (both vertebrate and invertebrate), zooplankton (rotifers, and small and large crustacean zooplankton) and phytoplankton. Decreases in the density of large piscivorous fishes result in increases in density and changes in the species composition of zooplanktivorous fishes; increases in the density of large piscivorous fishes result in decreases in density and changes in the species composition of zooplanktivorous fishes. High planktivory by vertebrates is associated with low planktivory by invertebrates, together with high densities of rotifers and small

crustaceans. In the absence of planktivorous fish, invertebrate planktivores and large crustacean zooplankton dominate. Differences in size among herbivorous zooplankton lead to pronounced differences in grazing and nutrient recycling rates. Herbivorous zooplankton effect changes in phytoplankton size structure and species composition directly by selective consumption and indirectly (but sometimes more importantly) by nutrient recycling (Bergquist et al., 1985; Bergquist and Carpenter, 1986). Hence, changes in the biomass at one trophic level can cause changes in the biomass in the trophic level immediately below it and thereby cascade through-out an ecosystem.

The overall objective of this study is to reconstruct the paleoenvironmental conditions for Lake Opeongo, a softwater Canadian Shield lake using diatoms preserved in the sediments. This lake is the site of the Ontario Ministry of Natural Resources Harkness limnological research station. Baseline limnological conditions of 200-300 years ago, prior to European settlement, will be established. A further objective is to identify and track at high resolution the diatom response to increased human impact over the past 150 years. This research will identify the timing and degree, if any, of eutrophication in Lake Opeongo, as well as the effect that fish introductions of smallmouth bass (*Microterus dolomieui*) and cisco *(Coregonus artedii*) might have had. Although this lake has been well-studied by fisheries scientists, no paleoecological studies have been conducted on this lake.

This thesis consists of six chapters. Chapter 1 introduces the background theoretical concepts of the thesis. The study site is discussed in Chapter 2. Coring procedures, sample preparation and statistical methods are discussed in Chapter 3. Paleolimnological reconstruction results are presented in Chapter 4 and an overview of the diatom assemblage and taxonomic details are given in Chapter 5. Chapter 6 contains a discussion and a summary of the results of the study.

Chapter 2: Site Description

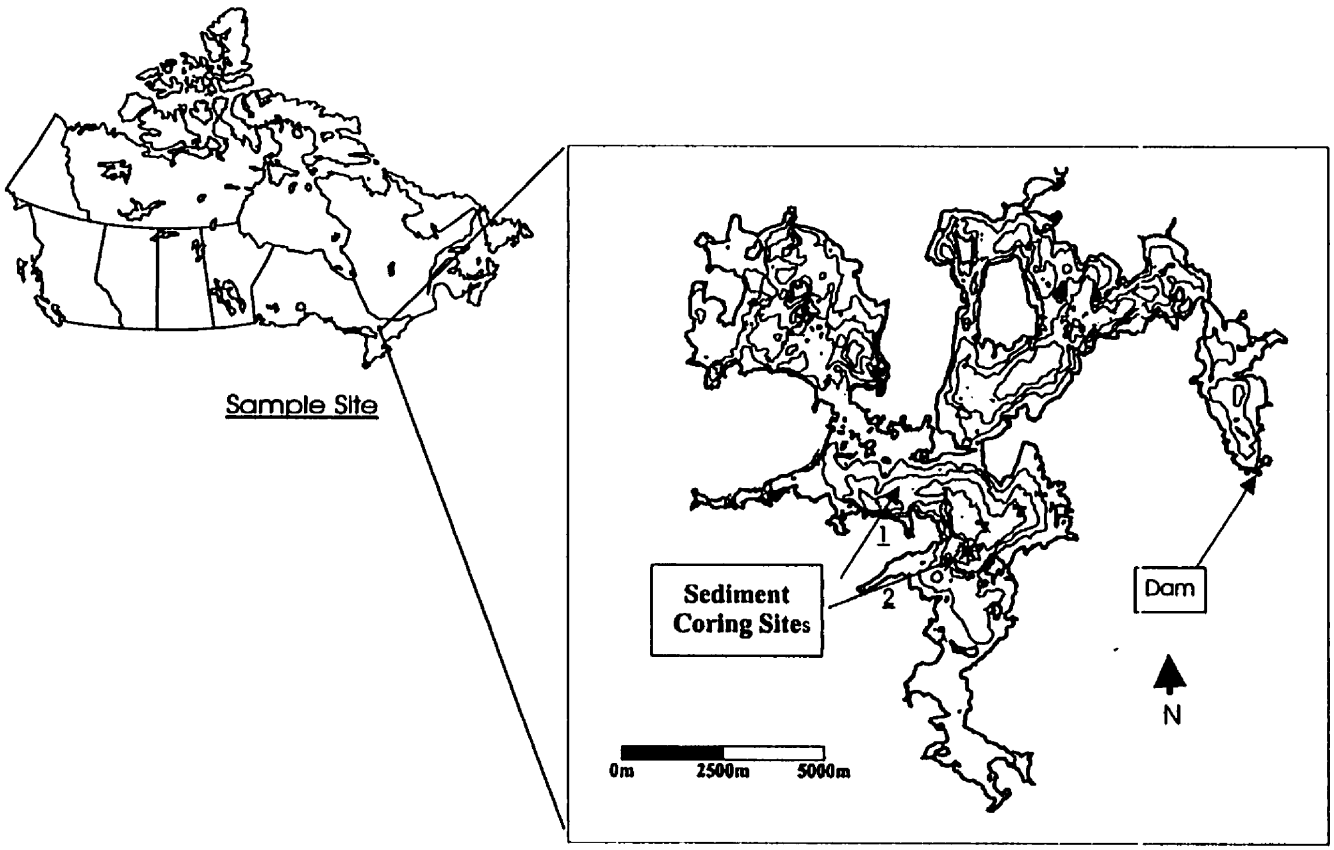
Natural conditions:

Lake Opeongo is a multibasin, oligotrophic lake in the southeastern region of Algonquin Provincial Park, Ontario (45° 42' N, 78° 22' W) (Fig. 1). Physical and chemical characteristics of Lake Opeongo are provided in Table 1. It lies on the Canadian Shield, specifically on Precambrian gneiss within the Grenville Province (Chapman and Putnam, 1984). In the Georgian Bay-Ottawa Valley region the bedrock has been warped to form broad arches, including the Algonquin Arch. This large upwarping, running northeastward from Chatham to Algonquin Provincial Park, together with another arch, the lesser Frontenac Axis, gives the region its dome-shaped build. In fact, Algonquin Park lies on the highest land in Southern Ontario. Considerable bedrock is found in the area and the hills are covered with a 1-3 m layer of stony to bouldery granitic till of sandy to silty sand texture (Martin and Fry, 1972). These soils are acidic and low in fertility. The lowest temperatures and shortest growing season for Southern Ontario are observed in the area.

Lake Opeongo is separated into three major arms (South, East and North) and two smaller bays (Sproule and Annie). Shallow sills separate the three arms and the two bays; hence, in many ways the large lake functions as five separate lakes. The South and East arms are divided by a channel that is roughly 3 m deep, the South and North arms by a channel that is roughly 6 m deep, the South arm is separated from Sproule Bay by a channel 7 m deep and Annie Bay is separated from the East arm by a channel only 2 m deep. The single outlet from Annie Bay to the Opeongo River is blocked by a dam. Water level variations are minor and are usually less than a meter (Martin and Fry, 1972). The Opeongo River drains south and west to the Madawaska River, and then, 160 km downstream, into the Ottawa River. It is one of the major headwaters of the Madawaska River.

Lake Opeongo has been well studied by fisheries scientists (Matuszek *et al.*, 1990; Martin, 1970; Fry and Kennedy, 1937). The most important fish are lake trout (*Salvelinus namaycush*), smallmouth

Figure 1. Location of Lake Opeongo, Ontario showing coring sites 1 and 2. Cores 1 and 2 from site 1, cores 3 and 4 from site 2.


Sample Site
Sediment Coring Sites
1
2
Dam
N
0m
2500m
5000m

bass (*Micropterus dolomieui*), lake whitefish (*Coregonus clupeaformis*), round whitefish (*Prosopium cylindraceum*), cisco (*Coregonus artedii*), burbot (*Lota lota*) and yellow perch (*Perca flavescens*) (Martin and Fry 1972). All these fish with the exception of the bass and the perch are pelagic fishes. The zooplanktivorous cisco is the dominant prey of the lake trout (Matuszek et al., 1990). The smallmouth bass is restricted to the warmer littoral zone and 80% of its food is crayfish (Martin and Fry, 1972). The

Table 1 Selected physical and chemical characteristics of Lake Opeongo.

Physical	
Drainage area (km^2)[2]	189.6
Surface area (km^2)[1]	58.6
Shoreline (km) (including islands)[1]	171
Mean depth (m)[1]	14.8
Maximum depth (m) (in South arm)[3]	51.8
Elevation (m) asl.[1]	403.4
Highest relief of watershed (m) asl.[1]	578
Secchi depth (m)[3]	5.1
Chemical	
Chlorophyll (mg/m^3)[9]	1.6
Total Kjeldahl nitrogen (mg/L)[7]	0.29
Total phosphorus (mg/L)[4]	0.008
Alkalinity as $CaCO_3$ (mg/L)[6]	4.60
Hardness (mg/L)[8]	15 - 30
pH[5]	6.8
Conductivity (µmho/cm)[7]	39
Dissolved Organic Carbon (DOC) (mg/L)[7]	4.2

[1]Shaw (1998), [2]Martin and Fry (1972), [3]King (1997), [4]Appendix IV, [5]Wong and Nriagu (1985), [6]Wong et al. (1984), [7]Ontario Ministry of the Environment, unpublished data, [8]Foerster and Schlichting (1965) and [9]Lake Opeongo Field Course, Dept. of Zoology, University of Toronto (http://www.cquest.utoronto.ca/env/opeongo/limnological.html).

perch also is a denizen of shallow water (Scott and Crossman, 1973). The whitefish eat zooplankton and the burbot are deepwater fish eaters. None of these three species has been studied intensively. It is thought that the overall biomass tied up in their populations is significant, but likely less than that in the lake trout, cisco and bass (Dr. B. Shuter, Ontario Ministry of Natural Resources, pers. com.). Their respective roles in the food web are not well defined.

Background studies:

An unusually long physical and instrumental climatic data set exists for the lake (King 1997). Summer lake temperature profiles exist from 1958-1998. Summer Secchi depth data exists for 1971-1983. Meteorological data exists for 1958-1998: iceout (for Lake Opeongo itself), mean air temperature (from Madawaska, 36 km southeast), bright sunshine hours (Petawawa, 87 km northeast) and windspeed (for Lake Opeongo itself but only for 1964-1974 and 1976). The sediment accumulation rate has been determined by ^{210}Pb geochronology and the sediment concentrations of, and deposition rates for, the pollutant metals (Pb, Cu, Zn, Ni, and Cd) and the rare earth elements have been measured (Nriagu and Wong, 1986; Wong and Nriagu, 1985; Wong *et al.* 1984). Foerster and Schlichting (1965) surveyed the littoral zone phyco-periphyton, including diatoms, of the lake. Algonquin Park's fire record begins in 1920 and Guyette and Dey (1995) have constructed a dendrochronological fire history for Opeongo Lookout at the south end of the South Arm which spans 1636-1994.

Human impacts on the watershed:

Over the past 200 years the lake has been impacted by Euro-Canadian settlement which, in fact created the lake. The lake originally existed as three lakes in the South, East and North arms. Loggers built a wooden dam at the outflow in Annie Bay ca. 1867 which caused the water level to rise approximately 5 m, thus linking the three separate lakes into one (Shaw, 1998). The dam level was raised in 1942; and the wooden dam was replaced by a concrete one in 1955 which remains today, turning Lake Opeongo into a reservoir for regional water control. The watershed was logged, often heavily and primarily for white pine (*Pinus strobus)*, beginning circa 1860; and selective logging continues

today. Portions of the watershed were also cleared for farms to produce supplies for the logging companies. The provincial park was established in 1893, prohibiting trapping. The farms seem to have gradually disappeared as access improved to the lake. Early in 1902, the St. Anthony Lumber Company built a railway spur line to the south end of Lake Opeongo in order to haul logs and supplies. This line was removed in the late 1920's and in 1933-1936 Highway 60 was built across the park, 4 km to the south of Lake Opeongo, cutting across its watershed. This road greatly increased tourist and angler use of the park. Furthermore, there have been changes in the trophic structure of the lake: the introduction of smallmouth bass (*Micropterus dolomieui*), a large piscivore in 1900 (Shaw, 1998); the beginning of fishing of the bass and lake trout (*Salvelinus namaycush*), another piscivore in 1936; and the introduction of cisco (*Correngus artedii*), a planktivorous fish which greatly increased the population of lake trout and filled an underutilized niche in 1948 (Matuszek *et al.* 1990).

Chapter 3: Materials and Methods

Coring procedures:

Locations of the coring sites are shown on the map (Fig. 1). Two initial cores were recovered on February 14, 1999, for preliminary taxonomic studies, using ice as a coring platform. Core 1 was taken in 27 m of water, using a Glew maxi corer and was 49 cm long. Core 2 (50 cm long) was taken with a Glew maxi corer in 22 m of water (Glew, 1989). Both cores were sampled at 1 cm intervals using a Glew extruder (Glew, 1988). On May 20, 1999, two additional Glew maxi cores (cores 3 and 4) were taken the deepest site, at 50 m, for stratigraphic analysis using a 6 m boat as a coring platform. They both were 43.5 cm in length and 63 mm in diameter. Core 3 was sampled at 0.25 cm intervals for the top 13 cm, at 0.5 cm intervals for the next 7 cm, and at 1 cm intervals for the remainder. Core 4 was sampled at 0.25 cm intervals for the first 10 cm, at 0.5 cm intervals for the next 10 cm, and at 1 cm intervals for the remainder. Core 1 was used for initial taxonomic work and to confirm trends seen in the main diatom analysis of core 3. Core 2 was used for pollen analysis in order to roughly find the rise of ragweed and the beginning of heavy European settlement in Southern Ontario. Core 3 was used for the main diatom analysis and ^{210}Pb and pollen dating. Core 4 was archived under refrigeration at 4^0 C in the Paleoenvironmental Assessment Laboratory (PAL), Department of Geology, at the University of Toronto.

Microfossil preparation and analysis:

Samples from cores 1 through 3 were prepared for diatom analysis as follows (Battarbee, 1986). Absence of calcium carbonate in the sediments precluded the need for a HCl wash. To remove organic matter, a 1:1 mixture of concentrated sulphuric and nitric acids was added and then samples were heated twice in a hot water bath for three hours. The samples were washed eight to ten times with distilled water and allowed to settle 24 hours between rinses. Because of the similar densities of the diatoms and the

acids, core 3 samples were allowed to stand for 48 hours in order to be certain that the diatoms had settled out. Each rinse was removed by careful aspiration. Strewn mounts were made and plated in Naphrax, a mounting medium of high refractive index (1.74). Samples and slides are stored in the Paleoenvironmental Assessment Laboratory, Department of Geology, University of Toronto.

The slides were examined under 1000x magnification with either a Nikon Optiphot X-2 (n.a. = 1.25) or a Leica DMRB (n.a. = 1.3) microscope both equipped with differential interference contrast (DIC) and phase contrast oil immersion lens. At least 500 diatom valves per slide were identified and counted. To aid in identification, photographs were taken using black and white Kodak Techpan film (50 ASA) or using a CoolSnap digital camera (RS Photometrics, Roper Scientific Inc.). Taxonomically difficult species, i.e., *Cyclotella stelligera* complex and *Aulacoseira* species, were studied using a JEOL 840 scanning-electron microscope at 15 kV.

Sediment chronology:

Dating of the core was done using ^{210}Pb analysis. Eighteen sediment samples from core 3, spanning 0 - 30 cm, were analysed to determine the age and sediment accumulation rates for the past 150-200 years. In order to obtain enough mass per depth, some sediment samples were recombined into 1 cm samples. Bulk density (wet and dry mass) of the samples was determined at PAL and the samples were ground and pre-weighed before being sent to J. Cornett, MyCore Ltd., for ^{210}Pb analysis. The Constant Rate of Supply (CRS) model (Appleby and Oldfield, 1978) was used to determine the geochronology, as according to Binford (1990) and Olsson (1986). The assumptions of this model are that the lake receives a constant ^{210}Pb flux at its surface, the supported ^{210}Pb (that produced from uranium present in the sediments) is produced in the sediments at a constant rate, the initial sediment ^{210}Pb concentration is variable, and the sediment influx rate is variable. Pollen analysis was done on cores 2 and 3 by J. McAndrews with identifications following McAndrews et al. (1973) in order to determine an approximate depth in the lake sediments for the rise in ragweed *(Ambrosia)* pollen which signals the beginning of

European forest clearance (McAndrews, 1994). Samples were prepared according to Cwynar et al. (1979).

Statistical analysis:

Diatom species that occurred in more than one sample and greater than 1% abundance in at least one sample were used in all statistical calculations. Species that did not meet this criterion were dropped because their occurrence might have been due to chance and also because species with such low presence were much more likely to be identified incorrectly. Stratigraphic zones were drawn by visual inspection.

Lake-water total phosphorus concentrations (TP) were inferred from diatom percent abundance data using a weighted averaging (WA) regression and calibration model (WACALIB v. 3.3; Line et al., 1994, Birks et al., 1990). Because of the high number of taxa included in the calibration sets, Birks (1994) and Wilson et al. (1996) suggest that WA rather than tolerance weighted averaging (tol(WA)) gives the least $RMSE_{boot}$. For the transfer function approach to have any meaning, a calibration model with close modern analogs to the fossil diatom assemblages must be used. Two approaches were used to determine how well a calibration set of modern samples provides analogs for fossil core samples as recommended by Birks et al. (1990) and Laird et al. (1998a,b). The first approach, that of minimum dissimilarity coefficients, evaluates how well the calibration set represents the fossil samples in terms of the relative abundance of all taxa found in the core sample. The second approach, based on canonical correspondence analysis (CCA) ordination, evaluates the degree to which the calibration set reliably estimates TP for each core sample, because only those taxa found in the calibration set are used in this analysis.

In the minimum dissimilarity measure approach, every fossil sample was compared to a calibration set by using a squared chi-squared distance as a dissimilarity measure using the computer program ANALOG version 1.6 (H.J. Birks and J. M. Line, unpub. program). Birks et al. (1990), Overpeck et al. (1985) and Birks (pers. com.) recommend either a squared chi-squared or squared chord distance

measure for diatom relative abundance data. The mean minimum dissimilarity coefficient (DC) with respect to the modern training set was calculated for each fossil value. Every sample in the training set was compared with all the other modern samples in that training set to determine the 2.5%, 5% and 10% percentiles of the training set dissimilarity coefficients (Anderson et. al., 1989). These gave critical values with which to compare the fossil samples (Birks, pers. com.): fossil samples with a minimum DC in the least extreme 2.5% of the modern training set were considered to have a close modern analog, and those samples with DCs between the 2.5% and 5% percentiles were considered to have good analogs. Those fossil samples with minimum DCs between the 5% and 10% percentiles were considered to have poor analogs and those with minimum DCs greater than the 10% percentile were deemed to have very poor or no modern analogs. Using Birks and Line's ANALOG program, permutation test one was run as suggested with (number of modern samples)(number of modern samples - 1)/2 iterations and permutation test two was run with 99 iterations.

In the CCA approach, CCA ordination keeps TP as the only explanatory variable to calculate the squared residual distance of each fossil (passive) diatom sample from the TP axis (CCA axis 1). The degree to which the calibration set reliably provides analogs for inferring TP is determined by comparing the squared residual distances of the fossil samples to those of the modern samples in the calibration set. Fossil samples with residual distances within the 90% confidence intervals of the modern samples are considered to have good analogs for estimating TP. Those with residual distances in the extreme 10% of the modern samples are considered to have poor analogs for estimating TP and lastly, those with residual distances in the extreme 5% are deemed to have very poor analogs (Birks et al., 1990; Laird et al., 1998b). This was done using CANOCO 4 with the following options chosen: inter-sample distances, biplot scaling, no species transformations, no forward selection, no Monte Carlo tests, and with fossil samples made supplementary (ter Braak and Smilauer, 1998; ter Braak, 1988).

In order to ensure the best nutrient reconstruction, four calibration sets were examined for close modern analogs to the Lake Opeongo assemblages: those of Reavie et al. (1995a), Reavie and Smol (*in press*), Hall and Smol (1996) and Wilson et al. (1996). These four sets were chosen because of species

in common with the Lake Opeongo fossil assemblages and because the authors had published extensive photomicrographs of their taxa which allowed taxonomic consistency. Reavie et al.'s (1995a) set was developed from diatoms in 64 non-shallow, circumneutral to alkaline British Columbia lakes with a TP gradient from 5 to 138 µg/L. It included the lakes that were used by Hall and Smol (1992). Reavie and Smol's (*in press*) set was developed from diatoms in 64 circumneutral to alkaline southeastern Ontario lakes with a TP gradient from 4 to 54 µg/L. Hall and Smol's (1996) set was developed from 54 acidic to circumneutral south-central Ontario lakes on the Canadian Shield with a TP gradient from 2.7 to 24.3 µg/L. Wilson et al.'s (1996) set was developed from 111 circumneutral to alkaline British Columbia lakes with a TP gradient from 2 to 268 µg/L. This last set included some of the lakes of Wilson et al. (1994) with the high salinity lakes removed. No level of the Lake Opeongo fossil assemblages had a close modern analog in Reavie et al.'s (1995a) calibration set as assessed by the method of minimum dissimilarity coefficients (see Fig. 8d), therefore this training set was dropped from further analysis.

Although published in their entirety, further decisions about the statistical procedures must be made when using these calibration sets. As recommended in Hall and Smol (1996), one outlier lake was dropped in both analog analysis and fossil TP inference, leaving 53 lakes in the calibration set. As the authors recommend, classical deshrinking and Hill's N_2 (Hill, 1973) were used in WACALIB; and it was not necessary to transform the TP data in order to have a normal distribution of the data. As well, species data were untransformed. A plot of observed TP values against TP concentrations calculated by WA for the 53 calibration lakes had RMSE = 3.5 µg/L and r^2 = 0.62. As discussed in Reavie and Smol (*in press*), five outlier lakes were dropped in both analog analysis and fossil TP inference, leaving 59 lakes in the calibration set. As Reavie and Smol suggested, classical deshrinking and Hill's N_2 were used in WACALIB; and it was not necessary to transform the TP data for normality. The species data were square-root transformed for WA analyses to reduce the effect of dominant taxa on model calculations as suggested. A plot of observed TP values against TP concentrations calculated by WA for the 59 calibration lakes had RMSE = 7.29 µg/L and r^2 = 0.637. It was not necessary to drop any outlier lakes in Wilson et al. (1996), and classical deshrinking, Hill's N_2, and untransformed species data were used.

However, in order to have normality, their TP data was transformed to log (TP + 1). This makes the standard error envelopes on the TP inferences more difficult to interpret as the envelopes apply only to the diatom-inferred log (TP + 1) values and cannot be transformed to give standard error envelopes for the diatom-inferred TP values. A plot of observed TP values against TP concentrations calculated by WA for the 111 calibration lakes had RMSE = 0.331 log (µg TP/L + 1) and r^2 = 0.661.

In order to assess the significance of any trend in the WA inferred TP values, some method of estimating the errors of the inferences must be used. Bootstrapping was used to estimate the root mean squared error (RMSE) of prediction of TP for the fossil samples since it can be used to assess paleolimnogical inference reliability (Line et al., 1994; Birks et al., 1990). WACALIB 3.3 was used to do 1000 bootstrap cycles for each calibration set combined with the Lake Opeongo fossil data set. In each cycle, WACALIB selects at random but with replacement from the original training set a subset of training samples of the same size as the actual training set. Any samples not selected for the training set form a bootstrap test set for that cycle. WA regression and calibration are then used with the bootstrap training set to infer the TP values for the modern samples in the bootstrap test set (i.e., those not selected) and also for the fossil samples. Let s_{i1} be the standard deviation of the inferred values for each modern and fossil sample *i*. This is the part of the prediction error due to estimation error in the optima of the taxa. Let s_{i2} be the difference between the observed TP value and the mean bootstrap estimate in all bootstrap cycles when the modern sample *i* is in the bootstrap test set (this is only defined for modern samples). This is the part of the prediction error due to variations in the abundance of taxa at a given environmental value, e.g., two lakes could have the same TP value but have different diatom assemblages. Let v_1 be the root mean square of the s_{i1} over all modern and fossil samples *i*. Let v_2 be the root mean square of the s_{i2} over all modern samples *i*. The estimated standard error of prediction (Est Se_{pred}) for any given sample *i* (both modern and fossil) is the square root of the sum of squares of s_{i1} and v_2. The Est Se_{pred} is an estimate of the standard deviation of the sample-specific WA inferred value. The RMSE of prediction for samples in the training set ($RMSE_{boot}$) is the square root of the sum of squares of v_1 and v_2. For each calibration set, the $RMSE_{boot}$ was calculated; the observed values of the training set were regressed

against the mean bootstrap values to obtain r^2_{boot}; and for each inferred TP value using this calibration set, the Est Se_{pred} was calculated to serve as an error estimate for that value.

Chapter 4: Paleolimnological Reconstruction Results

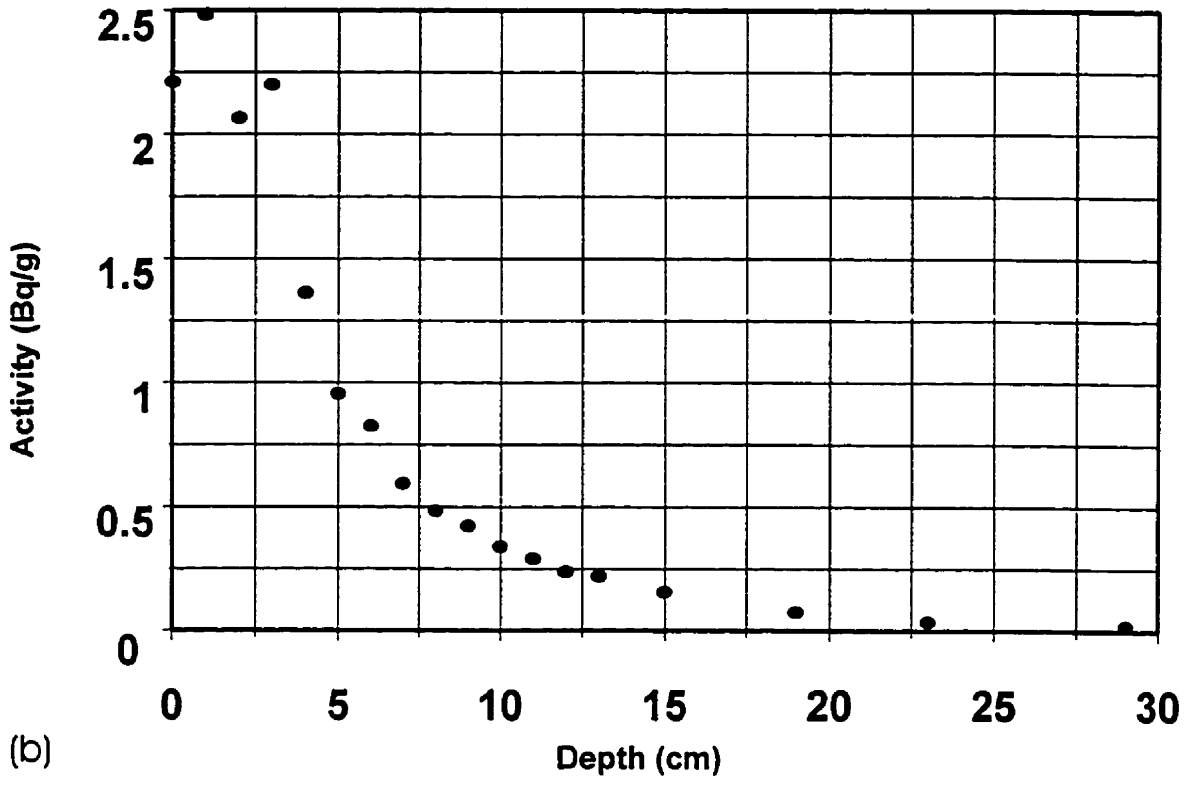
The sediment characteristics of core 3 were as follows: The sediment consisted of highly organic mud with no visible grain size changes. The top 2.75 cm consisted of red-brown flocculant material with an irregular surface. This changed to greyish-brown flocculant material at 2.75 cm which gradually became a grey-black sediment with very rare red flecks by 5 cm. From 6.25 - 43 cm, the sediments were very black.

The plot of ^{210}Pb activity (Bq/g) against core depth (cm) for Lake Opeongo core 3 (Fig. 2a) shows the classical exponential decay ^{210}Pb profile except in the top 4 cm (see Appendix I for full details). The sedimentation rates in the lake have changed over the past 200 years (Fig. 2b). Initially, prior to European impact, it was quite low, an average of 86 g/m^2/yr (dry weight). Once the dam was constructed and heavy logging and forest clearing began in 1860's, it tripled to an average of 258 g/m^2/yr from 1900-1962. Since then it has declined to an average of 207 g/m^2/yr. In core 3, the rise of the ragweed pollen occurs between 13 and 18 cm which is between 1856 and 1924 according to the ^{210}Pb dating (see Appendix II). In core 2, the rise of ragweed pollen occurs at approximately 13 cm or ca. 1850 (McAndrews, 1994). This does not correspond well with core 3, as according to the ^{210}Pb dating, 13 cm corresponds to 1924 ± 7.7. Hence, the sedimentation rate over the past 150 years is less at the site of core 2.

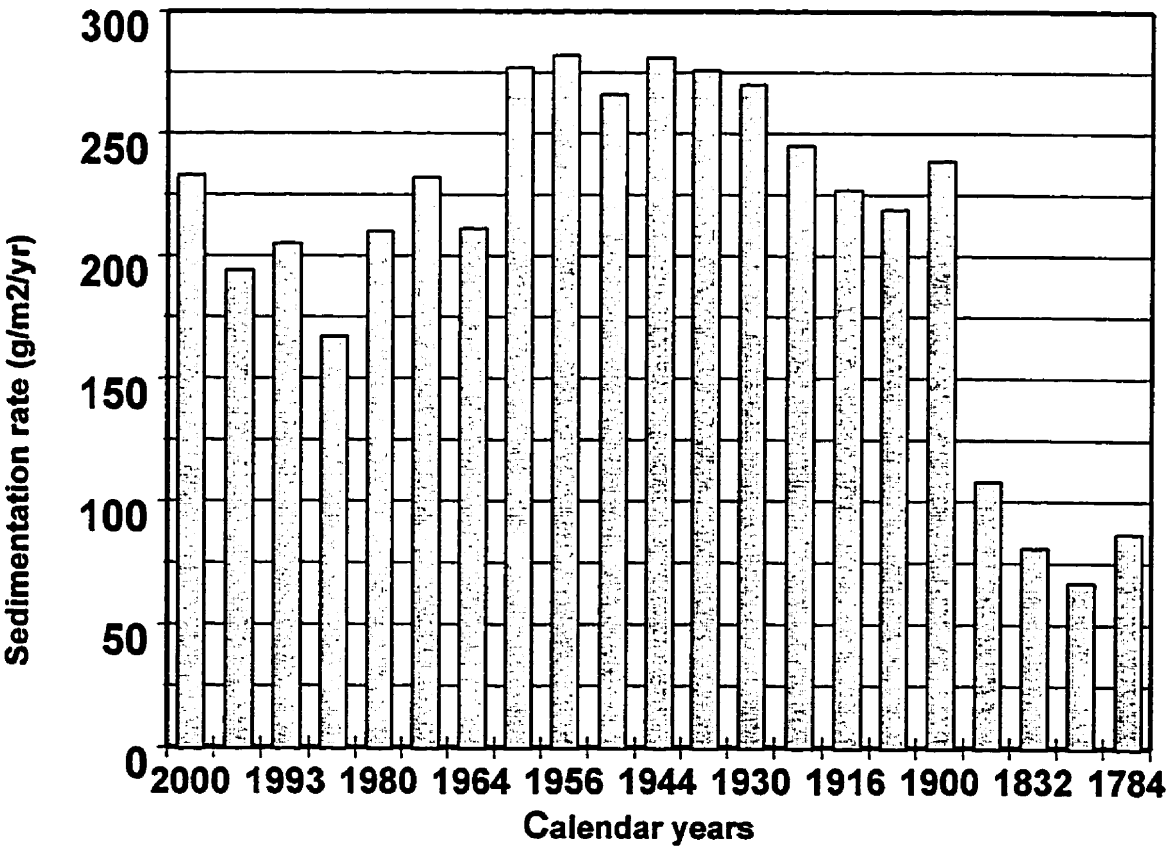
In this study, a total of 221 taxa from 32 genera were identified. Taxonomic details are provided in Chap. 5 and photographs of the taxa are included in Plates 1-3. The majority of the diatom frustules were from planktonic species with an average abundance of 64.2%. In all cores examined, the smaller diatoms were well preserved while the long slender diatoms such as *Asterionella* spp, *Tabellaria* spp, *Fragilaria* spp, *Nitzschia* spp and *Rhizosolenia* sp. were often broken, making it necessary to count tips and adjust the counts accordingly by dividing the sum of the tips by two.

Figure 2. (a) Plot of ^{210}Pb activity (Bq/g) against core depth (cm) for Lake Opeongo core 3. (b) Plot of sedimentation rates (g/m^2/yr) from ^{210}Pb analysis versus calendar year dates, assuming constant rate of supply.

(a)


2.5
2
1.5
1
0.5
0
Activity (Bq/g)
0
5
10
15
20
25
30
Depth (cm)

(b)


300
250
200
150
100
50
0
Sedimentation rate (g/m2/yr)
2000
1993
1980
1964
1956
1944
1930
1916
1900
1832
1784
Calendar years

The relative abundances of the nine common taxa (greater than or equal to 5% on at least one level) in core 3 are shown in Fig. 3. Because the diatom stratigraphy shows clear transitional states, a readily-available bottom up or agglomerative constrained cluster analysis algorithm such as CONISS (Grimm, 1987) is not appropriate (Birks, 1986; Grimm, 1987). Therefore, the diatom zone boundary was drawn visually using the common species. The boundary was drawn at 8.1 cm because above this point *Asterionella formosa, Cyclotella bodanica* var *lemanica, Fragilaria crotonensis* and *Rhizosolenia eriensis* all increase in relative abundance to values greater than their maxima lower down in the core, and the dominant *Cyclotella stelligera* decreases from a local maximum in the top of the core. The relative abundances of the 38 rare taxa (greater than or equal to 1% on at least one level) in core 3 are shown in Fig. 4. The combined common and rare taxa contributed 84.6-94.1% (average 89.3%) of the total frustules counted in the samples from core 3. See Appendix III for the relative abundances of the common and rare taxa. For most statistical purposes and in Figures 3 and 4, *Aulacoseira distans, Fragilaria capucina* and *Tabellaria flocculosa* varieties were combined, although they were counted separately. Figure 5 shows the varietial breakdowns of these three species.

Zone 1 (42 - 8.25 cm)

The diatom assemblage in this zone characterizes oligotrophic conditions in Lake Opeongo. The assemblage is dominated by the highly oligotrophic, planktonic *Cyclotella stelligera* complex which had an average abundance of 39.1% in this zone. *Tabelleria flocculosa IIIp* and *Aulacoseira distans*, both planktonic, were the co-dominants, with averages of 13.7% and 6.6%, respectively. Benthic *Achnanthes minutissima* and the small, benthic, alkaliphilous *Fragilaria pinnata* were also present. The more eutrophic *Asterionella formosa, Cyclotella bodanica* var *lemanica*, and *Fragilaria crotonensis* (all planktonic) were present in very low abundances (2.6%, 1.6%, 0.9%, respectively) as was the planktonic *Rhizosolenia eriensis*. The diatom community appears relatively stable. The average percent abundance of the planktonic species was 59.5%. Note that there do not appear to be significant changes in the diatom community in response to when the dam was constructed and logging became heavy circa

1867, or at the introduction of smallmouth bass at 1900, or when the road across the Park and the Harkness Fisheries Laboratory were built in 1936 and fishing and access greatly increased.

Zone 2 (8 - 0 cm)

The diatom assemblage in this zone indicates a transition to more meso-eutrophic conditions in Lake Opeongo starting at 1956. From the data, it is not clear if a stable community has been reached. The lake could easily be in a dynamic state. The oligotrophic *Cyclotella stelligera* complex declines in abundance, reaching a minimum of 12.1% in this zone. The more eutrophic *Asterionella formosa*, *Cyclotella bodanica* var *lemanica*, and *Fragilaria crotonensis* all greatly increase in abundance, reaching respective maxima of 26.2%, 9.2% and 11.8%. The planktonic *Rhizosolenia eriensis* also increases in abundance. However, this latter increase may well be an artifact of preservation in the sediments as the diatom is extremely fragile and it is extremely rare that intact frustules are preserved. (Only one whole frustule was seen; counts were based on its spine tips and dividing by two.) *Tabelleria flocculosa* IIIp first decreased in percentage, then appeared to regain its former numbers. *Aulacoseira distans* and *Achnanthes minutissima* declined somewhat in abundance. The planktonic *Cyclotella michiganiana* and *Fragilaria nanana* both appear to increase slightly in percentage, while planktonic *Aulacoseira alpigena* and benthic, alkaliphilous *Fragilaria brevistriata* appear to decline somewhat. The average percentage of planktonic diatoms increases in this zone to 68.1%. A two-tailed t-test assuming unequal variances shows that this increase is significant at the 0.05 level (41 df, t-statistic = 9.2853).

In order to address concerns relating to the above analysis being based on a single core, core 1 was also examined at a higher resolution to confirm the trends seen in core 3. Eight levels at 2 cm intervals were examined (see Fig. 6). At least 300 diatoms where counted per level and only the common taxa of core 3 which showed significant change (*A. distans*, *A. formosa*, *C. bodanica* var *lemanica*, *C. stelligera* complex, *F. crotonensis*, *T. flocculosa* and *R. eriensis*) were enumerated. (All other taxa were combined into a category of *other*). The results are similar to those of core 3: The initial assemblage at 14 cm

Figure 3. The relative abundance of common diatoms of Lake Opeongo core 3 plotted against core depth. Shown are the taxa with a relative abundance of at least 5% in at least one stratigraphic level. Included is the percent planktonic species. Also included are calendar year dates derived from ^{210}Pb analysis together with their standard deviation.

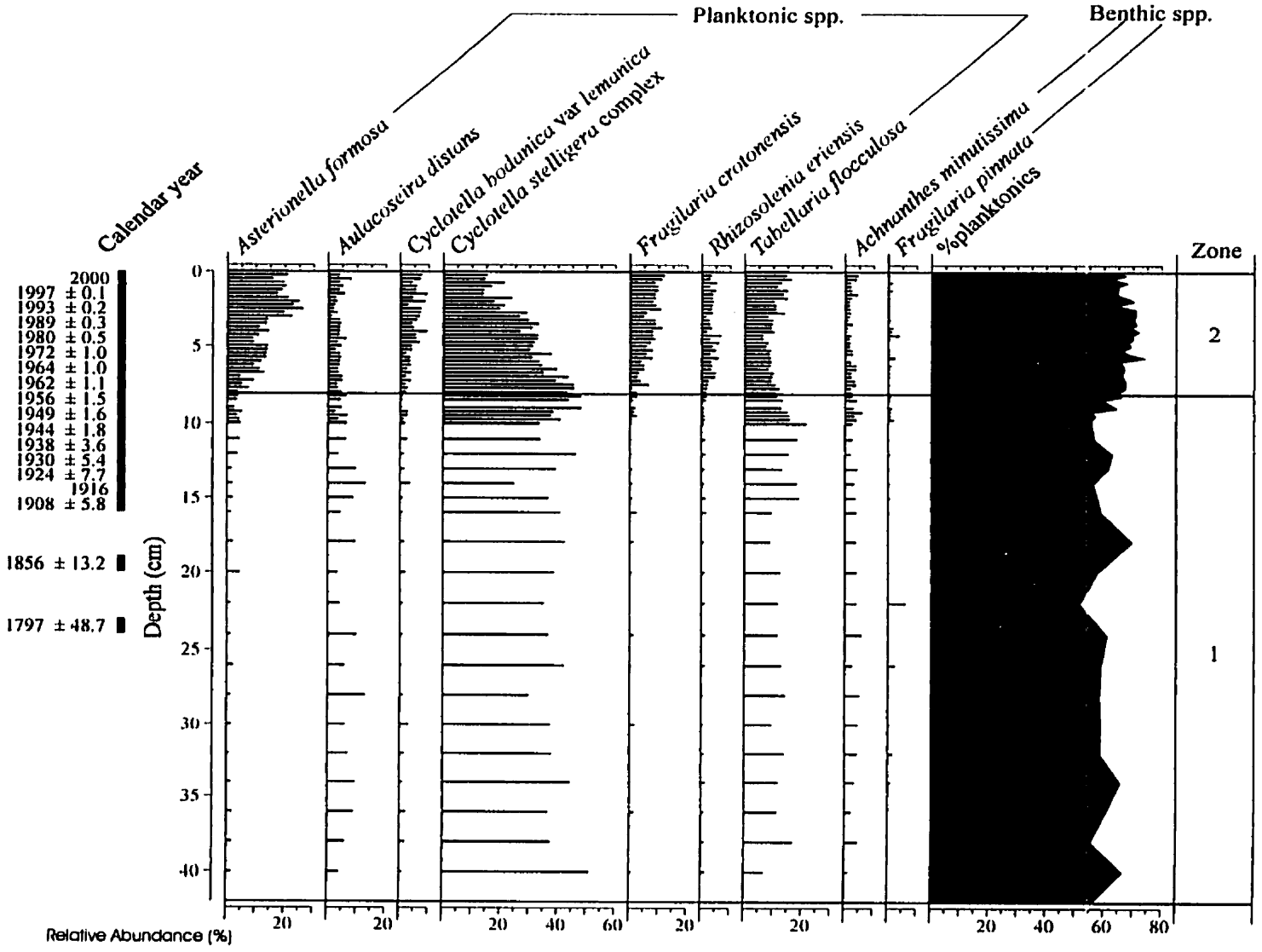

Planktonic spp.
Benthic spp.
Calendar year
Asterionella formosa
Aulacoseira distans
Cyclotella bodanica var lemanica
Cyclotella stelligera complex
Fragilaria crotonensis
Rhizosolenia eriensis
Tabellaria flocculosa
Achnanthes minutissimu
Fragilaria pinnata
%planktonics
Zone
2000
1997 ± 0.1
1993 ± 0.2
1989 ± 0.3
1980 ± 0.5
1972 ± 1.0
1964 ± 1.0
1962 ± 1.1
1956 ± 1.5
1949 ± 1.6
1944 ± 1.8
1938 ± 3.6
1930 ± 5.4
1924 ± 7.7
1916
1908 ± 5.8
1856 ± 13.2
1797 ± 48.7
Depth (cm)
0
5
10
15
20
25
30
35
40
2
1
Relative Abundance (%)
20
20
20
40
60
20
20
20
40
60
80

Figure 4. Rare diatoms (percentage) of Lake Opeongo core 3 plotted against core depth. Shown are the taxa of at least 1% relative abundance in at least one stratigraphic level but always less than 5% relative abundance. The x-axis scales show a maximum relative abundance of 10%. Also included are calendar year dates derived from ^{210}Pb analysis together with their standard deviation.

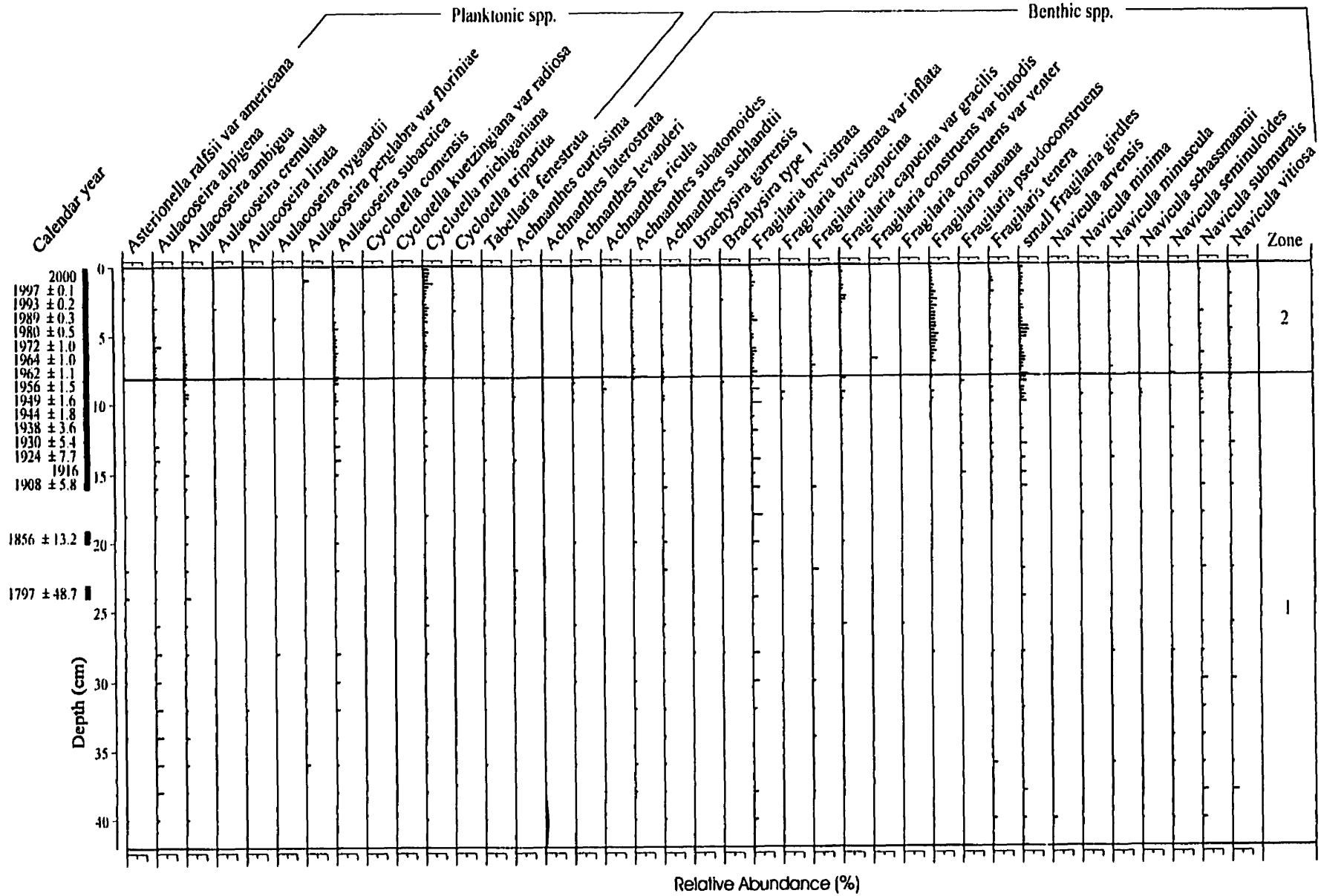

Planktonic spp.
Benthic spp.
Calendar year
Asterionella ralfsii var americana
Aulacoseira alpigena
Aulacoseira ambigua
Aulacoseira crenulata
Aulacoseira lirata
Aulacoseira nygaardii
Aulacoseira perglabra var floriniae
Aulacoseira subarctica
Cyclotella comensis
Cyclotella kuetzingiana var radiosa
Cyclotella michiganiana
Cyclotella tripartita
Tabellaria fenestrata
Achnanthes curtissima
Achnanthes laterostrata
Achnanthes levanderi
Achnanthes ricula
Achnanthes subatomoides
Brachysira suchlandtii
Brachysira garrensis
Fragilaria type 1
Fragilaria brevistrata
Fragilaria brevistrata var inflata
Fragilaria capucina
Fragilaria capucina var gracilis
Fragilaria construens var binodis
Fragilaria construens var venter
Fragilaria nanana
Fragilaria pseudoconstruens
Fragilaria tenera
small Fragilaria girdles
Navicula arvensis
Navicula minima
Navicula minuscula
Navicula schassmannii
Navicula seminuloides
Navicula submuralis
Navicula vitiosa
Zone
2000
1997 ±0.1
1993 ±0.2
1989 ±0.3
1980 ±0.5
1972 ±1.0
1964 ±1.0
1962 ±1.1
1956 ±1.5
1949 ±1.6
1944 ±1.8
1938 ±3.6
1930 ±5.4
1924 ±7.7
1916
1908 ±5.8
1856 ±13.2
1797 ±48.7
0
5
10
15
20
25
30
35
40
Depth (cm)
2
1
Relative Abundance (%)

Figure 5. Breakdown of *Aulacoseira distans*, *Fragilaria capucina* and *Tabellaria flocculosa* varieties in Lake Opeongo core 3 plotted against depth in core.

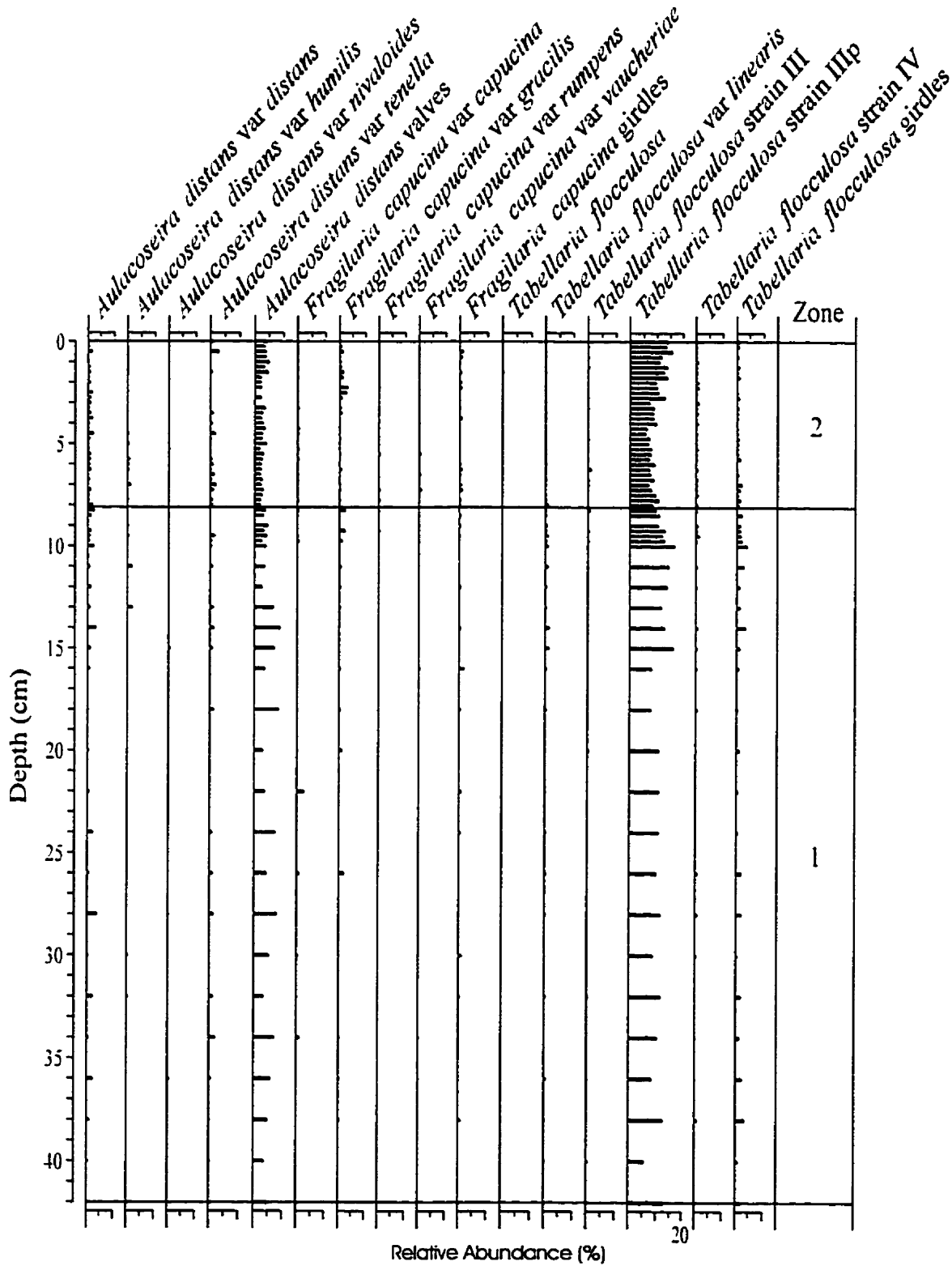
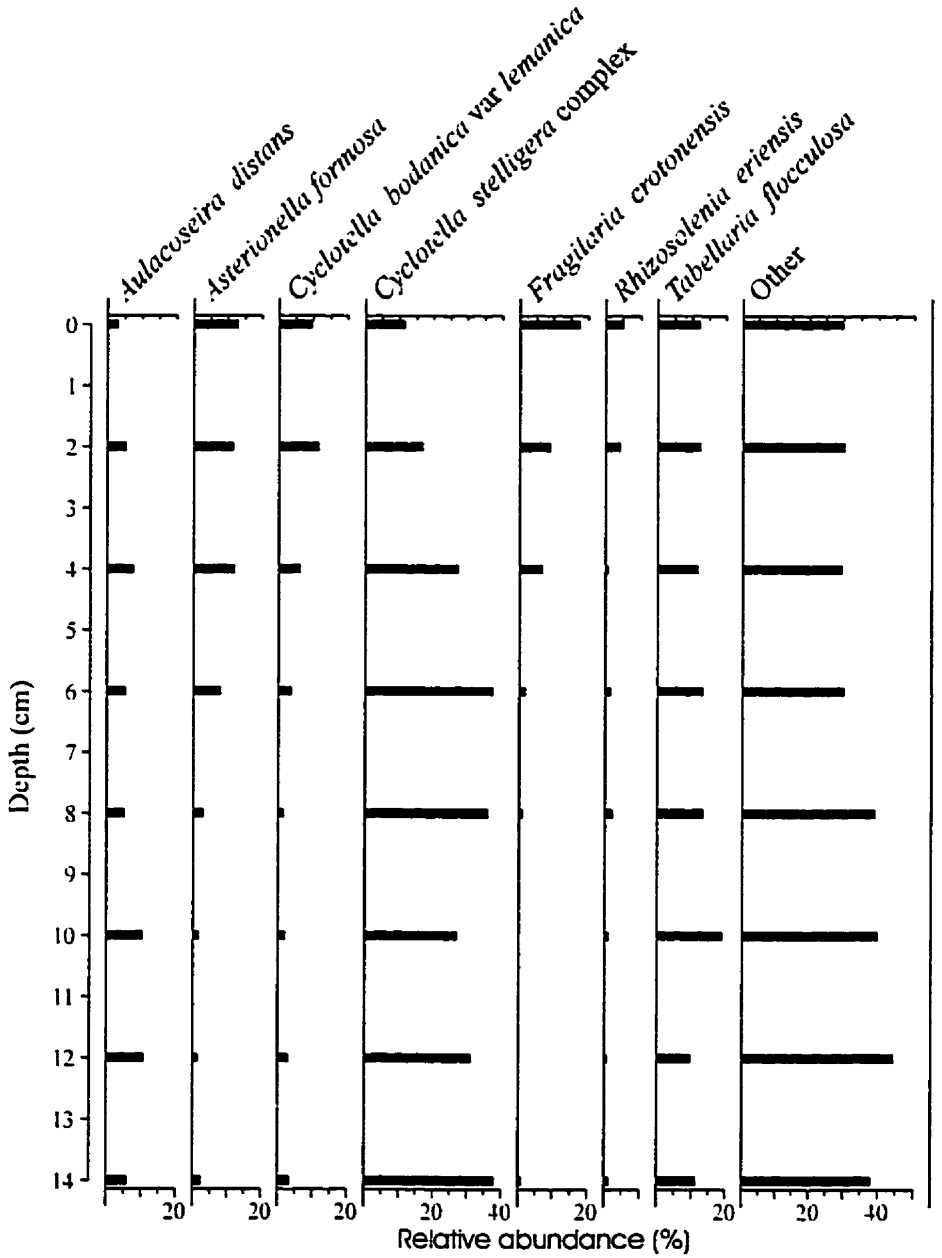

Aulacoseira distans var distans
Aulacoseira distans var humilis
Aulacoseira distans var nivaloides
Aulacoseira distans var tenella
Aulacoseira distans valves
Fragilaria capucina var capucina
Fragilaria capucina var gracilis
Fragilaria capucina var rumpens
Fragilaria capucina var vaucheriae
Tabellaria flocculosa
Tabellaria flocculosa girdles
Tabellaria flocculosa var linearis
Tabellaria flocculosa strain III
Tabellaria flocculosa strain IIIp
Tabellaria flocculosa strain IV
Tabellaria flocculosa girdles
Zone
2
1
Depth (cm)
0
5
10
15
20
25
30
35
40
20
Relative Abundance (%)

Figure 6. Plot of major diatoms in Lake Opeongo core 1 plotted against core depth. The major species are the common species from core 3 which showed a significant change between diatom zones 1 and 2.


Aulacoseira distans
Asterionella formosa
Cyclotella bodanica var lemanica
Cyclotella stelligera complex
Fragilaria crotonensis
Rhizosolenia eriensis
Tabellaria flocculosa
Other
Depth (cm)
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
20
20
20
20
40
20
20
20
40
Relative abundance (%)

consists of mainly *C. stelligera* complex and *T. flocculosa*. As the surface is approached, the diatom community changes to an assemblage of *A. formosa, C. bodanica var lemanica, F. crotonensis*, and *R. eriensis* with comparatively lesser amounts *C. stelligera* complex and *T. flocculosa*.

The diatom-inferred TP from Hall and Smol (1996)'s calibration set shows that Lake Opeongo has become slightly more eutrophic (Fig. 7a). The signal is somewhat noisy. The 5-point running mean shows that from 38 cm to 10 cm, the diatom-inferred TP was relatively constant at roughly 5.5 µg/L. From 10 cm to 8 cm, TP falls from 6 µg/L to 5 µg/L. At 7.5 cm, TP begins rising, reaching a current peak of 8 µg/L. These changes are within the envelope of the estimated sample-specific standard errors of prediction. For this calibration set, $RMSE_{boot}$ = 4.159 µg TP/L and r^2_{boot} = 0.411 (for comparison, see Table 2). The diatom-inferred TP from Reavie and Smol (*in press*)'s calibration set shows that Lake Opeongo has actually become less eutrophic (Fig. 7b). Once again, the signal is somewhat noisy. The 5-point running mean shows a maximum of 12 µg/L at 30 cm. For the remaining 30 cm of the core, the diatom-inferred TP declines to a current low of -1 µg/L. This oligotrophic trend is outside the envelope of the estimated sample-specific standard errors of prediction. For this calibration set, $RMSE_{boot}$ = 9 µgTP/L and r^2_{boot} = 0.414 (for comparison, see Table 2). On the other hand, the diatom-inferred TP from Wilson et al. (1996) shows that the lake has become more eutrophic (Fig. 7c). The reconstructed TP values show that from 10 - 42 cm highly oligotrophic and fairly constant conditions of 2 µg/L TP prevailed in the lake. There is a slight dip from 8 - 10 cm similar to that seen from Hall and Smol's calibration set. Then at approximately 8 cm, the diatom-inferred TP begins fairly steadily rising to a current peak of 4.5 µg/L, showing a trend towards more mesotrophic conditions. This mesotrophic trend is nearly outside the envelope of the estimated sample-specific standard errors of prediction (see Fig. 7d). For this calibration set, $RMSE_{boot}$ = 0.390 log(µg TP/L + 1) and r^2_{boot} = 0.467 (see Table 2).

Figure 7. Diatom-inferred Total Phosphorus (µg/L) based upon the transfer functions of: (a) Hall and Smol (1996), (b) Reavie and Smol (*in press*), and (c) Wilson et al. (1996). Also shown for each graph is a 5-point running mean (heavy line). Diatom-inferred TP in µg/L. Also shown in 7a and 7b are the estimated standard errors of prediction (Est Se_{pred}) for the WA inferences. The Est Se_{pred} were derived from bootstrapping (see methods for details). Shown in 7d are the estimated standard errors of prediction for the log (TP+1) inferences derived from Wilson et al. (1996)'s calibration set, also derived from bootstrapping.

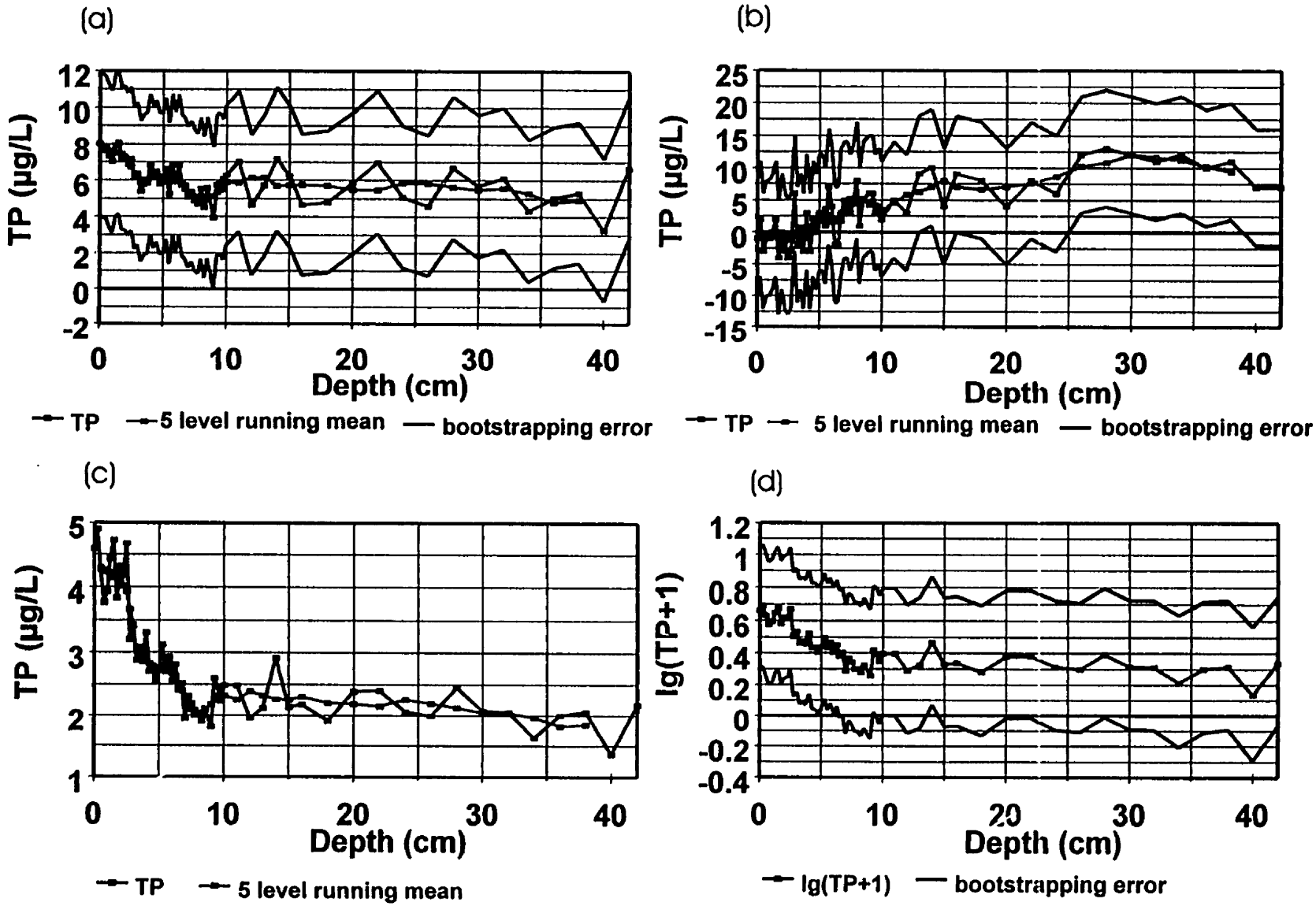

(a)
TP (μg/L)
12
10
8
6
4
2
0
-2
0
10
20
30
40
Depth (cm)
TP
5 level running mean
bootstrapping error
(b)
TP (μg/L)
25
20
15
10
5
0
-5
-10
-15
0
10
20
30
40
Depth (cm)
TP
5 level running mean
bootstrapping error
(c)
TP (μg/L)
5
4
3
2
1
0
10
20
30
40
Depth (cm)
TP
5 level running mean
(d)
lg(TP+1)
1.2
1
0.8
0.6
0.4
0.2
0
-0.2
-0.4
0
10
20
30
40
Depth (cm)
lg(TP+1)
bootstrapping error

Table 2. Summary of calibration set statistics: r^2 and RMSE calculated directly from the calibration set and r^2_{boot} and $RMSE_{boot}$ derived from bootstrapping.

Calibration set	r^2	RMSE	r^2_{boot}	$RMSE_{boot}$
Hall and Smol (1996)	0.620	3.5 µg/L	0.411	4.159 µg TP/L
Reavie and Smol (*in press*)	0.637	7.29 µg/L	0.414	9 µg TP/L
Wilson et al. (1996)	0.661	0.331 log(µg TP/L +1)	0.467	0.390 log(µg TP/L +1)

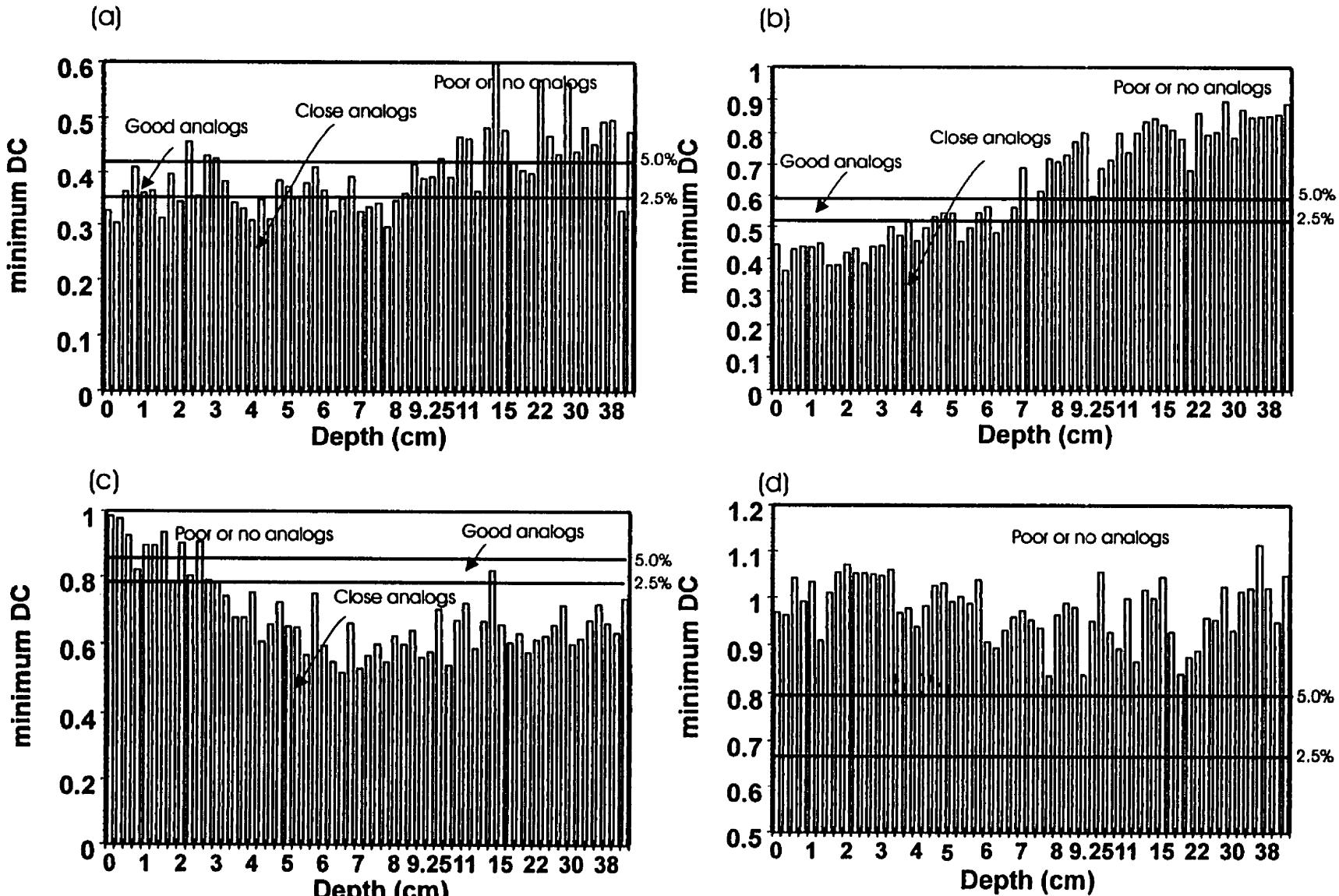
For Hall and Smol's (1996) calibration set, any fossil samples with minimum DCs > 0.4085 (i.e., greater than 5% of the dissimilarity coefficients of the calibration sample) were considered to have no or poor modern analogs and those samples with 0.3408 < minimum DC < 0.4085 (in between the 2.5 % and 5% of the calibration samples) were considered to have good modern analogs and any fossil samples with minimum DCs < 0.3408 were considered to have close modern analogs (Fig. 8a). Approximately 36% of samples have poor or no modern analogs, and 41% have good modern analogs, with the remaining 24% of the samples having close modern analogs. Fossil samples with high minimum DCs have taxa with higher percentages than the maximum relative abundance occurrence in the calibration samples or contain taxa that are absent in these samples. Those samples with minimum DCs > 0.4085 have either one or both of the following characteristics: higher percentages of *A. distans* than are in the calibration set (maximum abundance in the calibration set of 3.1% but average fossil abundance of 5.1%) or many rarer taxa (maximum abundance in the core < 5% on all levels) which are absent in this training set. Among these taxa are *Aulacoseira alpigena, Fragilaria pseudoconstruens, F. capucina, F. capucina* var *gracilis* and some of the small *Achnanthes* and *Navicula* species: *Achnanthes curtissima, A. suchlandtii* (present in abundance greater than in the training set), *Navicula minima, N. minuscula* and *N. seminuloides.*

For Reavie and Smol's (*in press*) calibration set, any fossil samples with minimum DCs > 0.5885 have no or poor modern analogs and those samples with 0.5168 < minimum DC < 0.5885 have good modern analogs and any fossil samples with minimum DCs < 0.5168 have close modern analogs (Fig. 8b). Approximately 51% of samples have poor or no modern analogs, and 10% have good modern analogs, with the remaining 39% of the samples having close modern analogs. The main problem with this calibration set is that from 7 cm down in the core there is a complete lack of modern analogs. This arises because one of the major diatoms in this area of the core, *C. stelligera* complex, is present in a higher abundance in the Lake Opeongo core than in the calibration set, and another major diatom, *A. distans*, is absent in the calibration set. From 7 - 42 cm, *C. stelligera* complex and *A. distans* have average fossil abundances of 39.7% and 6.2%, but *C. stelligera* complex has a maximum abundance of only 28.4% in the calibration set. In addition, there were further species not present in this training set: *Achnanthes levanderi, A. curtissima, A. ricula, N. seminuloides, N. minuscula, N. vitiosa* and *F. pseudoconstruens.*

For Wilson et al.'s (1996) calibration set, any fossil samples with minimum DCs > 0.9464 have no or poor modern analogs and those samples with 0.7710 < minimum DC < 0.9464 have good modern analogs and any fossil samples with minimum DCs < 0.7710 have close modern analogs (Fig. 8c). Approximately 14% of the samples have poor or no modern analogs, and 9% have good modern analogs, with the remaining 78% of the samples having close modern analogs. Most of these samples with poor or no fit lie in the upper 2.5 cm of the core. The problem with these levels is that *T. flocculosa* IIIp in the Lake Opeongo samples exceeds samples in the calibration set (a maximum calibration abundance of 8.0% but with an average abundance of 13.1% in the top 2.5 cm) and that *Rhizosolenia eriensis* does not occur in the calibration set at all but has an average abundance in the top 2.5 cm of 3.6%.

For Reavie et al.'s (1995a) calibration set, any fossil samples with minimum DCs > 0.7943 have no or poor modern analogs and those samples with 0.6665 < minimum DC < 0.7943 have good modern analogs and any fossil samples with minimum DCs < 0.6665 have close modern analogs (Fig. 8d). All

Figure 8. Analog analysis measured by a dissimilarity coefficient (DC) based on a chi-squared distance. The minimum distance of each fossil sample to a modern calibration sample is shown. (a) shows the results for Hall and Smol's (1996) training set, (b) shows the results for Reavie and Smol's (*in press*) set, (c) shows the results for Wilson et al.'s (1996) set and (d) shows the results for Reavie et al.'s (1995a) set. The 2.5% and 5% percentile lines divide the fossil samples into those with close, good, and poor or no modern analogs.


(a)
minimum DC
0.6
0.5
0.4
0.3
0.2
0.1
0
Good analogs
Close analogs
Poor or no analogs
5.0%
2.5%
0 1 2 3 4 5 6 7 8 9.25 11 15 22 30 38
Depth (cm)
(b)
minimum DC
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Good analogs
Close analogs
Poor or no analogs
5.0%
2.5%
0 1 2 3 4 5 6 7 8 9.25 11 15 22 30 38
Depth (cm)
(c)
minimum DC
1
0.8
0.6
0.4
0.2
0
Poor or no analogs
Good analogs
Close analogs
5.0%
2.5%
0 1 2 3 4 5 6 7 8 9.25 11 15 22 30 38
Depth (cm)
(d)
minimum DC
1.2
1.1
1
0.9
0.8
0.7
0.6
0.5
Poor or no analogs
5.0%
2.5%
0 1 2 3 4 5 6 7 8 9.25 11 15 22 30 38
Depth (cm)

samples had poor or no modern analogs. Hence, this calibration set was not used for further analysis. The main problem with this calibration set is that many common diatoms (those with an abundance of at least 5% on a level) are present in a higher abundance in the Lake Opeongo core than in the calibration set. Among these are *T. flocculosa IIIp, C. bodanica* var *lemanica, C. stelligera complex, A. formosa* and *A. distans.* They had respectively maximum calibration abundances: 3.9%, 6.6%, 30.5%, 20.8%, and 6.8%, but maximum fossil abundances: 21.4%, 9.2%, 51.0%, 26.3% and 13.0%. In addition, there were many rarer taxa which were not present in this training set: *A. alpigena, Fragilaria nanana, F. capucina,* and some of the small *Achnanthes* and *Navicula* species: *A. curtissima, A. ricula, Navicula submuralis, N. minuscula* and *N. vitiosa.*

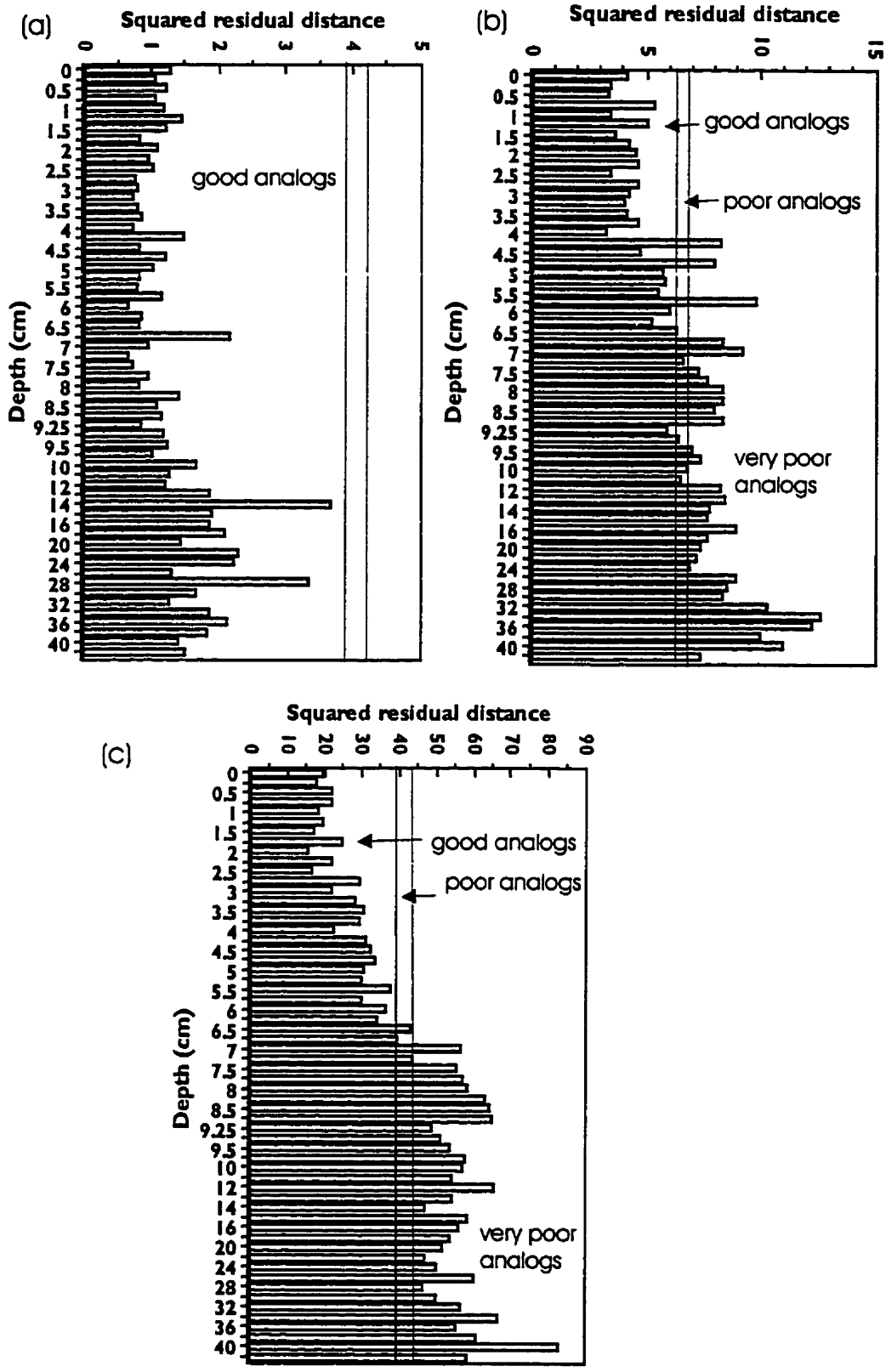
The second method of testing for modern analogs, that from CCA ordination gave the following results (Fig. 9): For Hall and Smol's (1996) calibration set, fossil samples with squared residual distances greater than 4.18 had very poor modern analogs in the calibration set for estimating TP and fossil samples with squared residual distances greater than 3.87 but less than 4.18 had poor analogs (Fig. 9a). All fossil samples had good analogs in this calibration set.

For Reavie and Smol's (*in press*) calibration set, fossil samples with squared residual distances greater than 6.73 had very poor modern analogs in the calibration set for estimating TP and fossil samples with squared residual distances greater than 6.09 but less than 6.73 had poor analogs (Fig. 9b). All fossil samples above 6.5 cm had good analogs in this calibration set. However, basically all fossil samples deeper than 6.5 cm had poor or very poor analogs in this calibration set. This arises because one of the major diatoms in this area of the core, *C. stelligera* complex, is present in a higher abundance in the Lake Opeongo core than in the calibration set. From 7 - 42 cm, *C. stelligera* complex has an average fossil abundances of 39.7%, but *C. stelligera* complex has a maximum abundance of only 28.4% in the calibration set.

For Wilson et al.'s (1996) calibration set, fossil samples with squared residual distances greater than 42.80 had very poor modern analogs in the calibration set for estimating TP and fossil samples with

squared residual distances greater than 38.99 but less than 42.80 had poor analogs (Fig. 9c). All fossil samples above 6.5 cm had good analogs in this calibration set. Below this, the fossil samples had very poor analogs. This arises because two of the major diatoms in this area of the core, *C. stelligera* complex and *T. flocculosa IIIp*, are present in higher abundances in the Lake Opeongo core than in the calibration set. From 6.5 - 42 cm, *C. stelligera* complex has a maximum fossil abundance of 51.0%, but it has a maximum abundance of only 37.8% in the calibration set. Likewise, from 6.5 - 42 cm, *T. flocculosa IIIp* has a maximum fossil abundance of 21.6%, but it has a maximum abundance of only 8.0% in the calibration set.

Figure 9. Analog analysis measured by the squared residual distance of each passive fossil sample from the TP axis (CCA axis 1). (a) shows the results for Hall and Smol's (1996) training set, (b) shows the results for Reavie and Smol's (*in press*) set, (c) shows the results for Wilson et al.'s (1996) set. The 5% and 10% percentile lines divide the fossil samples into those with good, poor or very poor modern analogs.


(a)
Squared residual distance
0
1
2
3
4
5
Depth (cm)
0
0.5
1
1.5
2
2.5
3
3.5
4
4.5
5
5.5
6
6.5
7
7.5
8
8.5
9.25
9.5
10
12
14
16
20
24
28
32
36
40
good analogs
(b)
Squared residual distance
0
5
10
15
Depth (cm)
good analogs
poor analogs
very poor analogs
(c)
Squared residual distance
0
10
20
30
40
50
60
70
80
90
Depth (cm)
good analogs
poor analogs
very poor analogs

Chapter 5: Taxonomy

Diatom taxonomy is dynamic and in a current state of flux reflecting, for instance, new taxonomic revisions based upon the increased resolution arising from scanning electron microscopy. A comparison of the diatom genera and species used by Krammer and Lange-Bertalot (1986-1991) to those used by Round et al. (1990) demonstrates the discrepancy in two different schools of thought. Round et al. totally reorganize the genera, splitting many of them and creating many new genera. Krammer and Lange-Bertalot keep the older, classical generic names but split many established species, creating many new ones. As well, diatom taxonomy is based upon morphological characteristics and it is thus open to wide interpretation. For this reason, in order to ensure taxonomic consistency, a name and an authority are not sufficient, and at a minimum, light microscope photomicrographs are needed. In cases where there is much taxonomic controversy and confusion, a detailed discussion of the taxonomic reasoning and references used must be given.

In this study, a total of 221 taxa from 32 genera were identified. A list of all species found in the lake, together with the authorities and the species' number of occurrences and individual maximum abundances in core 3 is included in Table 3. In order to minimize taxonomic confusion, Plates 1-3 show photomicrographs of all diatom species which had an abundance of at least 1% on at least one level and hence appeared in the statistical data analysis.

The diatom species were not completely characteristic of the geographical location of Lake Opeongo. In fact, there was much similarity between the Lake Opeongo diatoms and those found by Fallu (1998) in the James Bay and Hudson Bay regions. Presumably the higher elevation and depth of Lake Opeongo allows a flora more typical of northern regions to flourish. Out of the 221 taxa found: nine were common (i.e., had a relative abundance equal to or greater than 5% on at least one level), 38 were rare (i.e., had a relative abundance equal to or greater than 1% on at least one level but less than 5% on all levels) and the remaining 174 were extremely rare (i.e., never reached a relative abundance of 1% on

any level). Among the extremely rare taxa, there are two undescribed *Achnanthes* taxa (P. Hamilton, Canadian Museum of Nature, pers. com.). There could be additional undescribed extremely rare *Achnanthes* and *Navicula* taxa but their rarity, small size (around 5 μ) and occasional preservation problems make it difficult to be certain. Practical taxonomic problems arose with the small *Achnanthes* and *Navicula* taxa which were pushing the resolution of light microscopy, small *Navicula* girdle views, broken *Nitzschia* frustules which had lost the taxonomically important middle region, and with matching girdle and valve views for the *Aulacoseira* species.

Taxonomic identifications most closely follow Cumming et al. (1995), Reavie and Smol (1998), Krammer and Lange-Bertalot (1986-1991), Lange-Bertalot and Metzeltin (1996), Camburn et al. (1984-1986) and Fallu (pers. com.), but also used were Hustedt (1930), Simonsen (1987a,b,c), Germain (1981), Moser et al.(*in prep.*) and Patrick and Reimer (1966, 1975). Camburn and Kingston (1986) and Siver and Kling (1997) were used as the main references for the genus *Aulacoseira*. Koppen (1975) was used as the reference for the genus *Tabellaria*. Used were his three morphological strains of *T. flocculosa: T. flocculosa III, T. flocculosa IV* and *T. flocculosa* var *linearis*. The identification of the long members of *Fragilaria* (formerly *Synedra*) follows Krammer and Lange-Bertalot (1986-1991).

There has been much confusion about the identity of the five small *Cyclotella* species: *Cyclotella stelligeroides* Hustedt (1945), *Cyclotella stelligera* Cleve & Grunow (1881), *Cyclotella woltereckii* Hustedt (1942), *Cyclotella glomerata* Bachmann (1911) and *Cyclotella pseudostelligera* Hustedt (1939) (see Krammer and Lange-Bertalot (1991a) for sample photomicrographs). From the dates at which the authorities erected these names, it is clear that this problem has been a long-standing one. In an elegant study using live cultures which is unfortunately all too rare, Belcher et al. (1966) showed that *C. pseudostelligera* and *C. woltereckii* are ecotypes of the same species. Lowe (1975) used electron microscopy to examine the ultrastructure of *C. glomerata*, *C. stelligera* and *C. pseudostelligera*. He concluded that all three species formed a single morphological group with no spines or central strutted processes, a single labiate process marginal on the valve mantle and regularly spaced marginal strutted processes. Haworth and Hurley (1984) and Haworth (1984), using light and electron microscope studies

and statistical analysis, have shown that the five species cannot be reliably separated and that *C. glomerata*, *C. pseudostelligera* forma *woltereckii*, *C. pseudostelligera* and *C. stelligeroides* should be varieties of *C. stelligera*. Chang (1991) examined the type-material of *Cyclotella stelligera* Cleve & Grunow and documented its wide degree of variability.

Following Reavie and Smol (1998), the Arctic-Antarctic Diatom Workshops (informal notes) and Krammer and Lange-Bertalot (1986-1991), the small *Cyclotella* species in Plate 1, Fig. 23-26 would be called *C. pseudostelligera*. However, examination of the photomicrographs of Hustedt's type specimens (Simonsen, 1987a,b,c) and Gasse (1980) shows that *C. pseudostelligera* has long marginal processes on the mantle which are clearly visible as marginal dashes in the light microscope. As the plates show, the *Cyclotella* species in Plate 1, Fig. 23-26 has no mantle processes; and therefore, it is not *C. pseudostelligera*. The larger *Cyclotella* species in Plate 1, Fig. 21-22 would be called *C. stelligera* following Reavie and Smol (1998), Krammer and Lange-Bertalot (1986-1991), and the Arctic-Antarctic Diatom Workshops. Following Haworth and Hurley (1984) and examination of the photomicrographs of Hustedt's type specimens (Simonsen, 1987a,b,c), I called the small form *C. stelligera* var *stelligeroides* and the larger, more silicified form *C. stelligera*. An attempt was made to count the two forms separately; however, they intergrade easily as other researchers have found (see below), making separate counts unreliable. Hence, the two forms were combined into a *C. stelligera* complex.

Hall and Smol (1996) and Wilson et al. (1996) considered both these forms to be instances of the *C. stelligera* complex and they are combined in their respective training sets. Reavie and Hall (*in press*), considered these two forms to be *C. pseudostelligera* and *C. stelligera*. In this thesis, Reavie and Smol's *C. pseudostelligera* and *C. stelligera* were combined into *C. stelligera* complex which was used to infer fossil TP for Lake Opeongo and compared to the Lake Opeongo counts of *C. stelligera* complex in analog analysis.

Plate 1

Fig. 1: *Aulacoseira alpigena* (Grunow) Krammer

Fig. 2: *Aulacoseira ambigua* (Grunow in Van Heurck) Simonsen

Fig. 3: *Aulacoseira crenulata* (Ehrenberg) Thwaites

Fig. 4-5: *Aulacoseira distans* (Ehrenberg) Simonsen

Fig. 6: *Aulacoseira distans* var *humilis* (Cleve-Euler) Simonsen

Fig. 7: *Aulacoseira distans* var *nivaloides* (Camburn) Simonsen

Fig. 8: *Aulacoseira distans* var *tenella* (Nygaard)(Florin) Simonsen

Fig. 9: *Aulacoseira lirata* (Ehrenberg) Ross

Fig. 10: *Aulacoseira nygaardii* (Camburn) Simonsen

Fig. 11: *Aulacoseira perglabra* var *floriniae* (Camburn) Simonsen

Fig. 12-15: *Aulacoseira subarctica* (O. Mull.) Haworth

Fig. 16: *Cyclotella bodanica* var *lemanica* (O. Mull. ex Schrot.) Bachmann

Fig. 17: *Cyclotella comensis* Grunow in Van Heurck

Fig. 18-19: *Cyclotella kuetzingiana* var *radiosa* Fricke

Fig. 20: *Cyclotella michiganiana* Skvort.

Fig. 21-26: *Cyclotella stelligera* Cleve and Grunow

Fig. 27-28: *Cyclotella tripartita* Hakansson

Fig. 29: *Rhizosolenia eriensis* H.L. Smith

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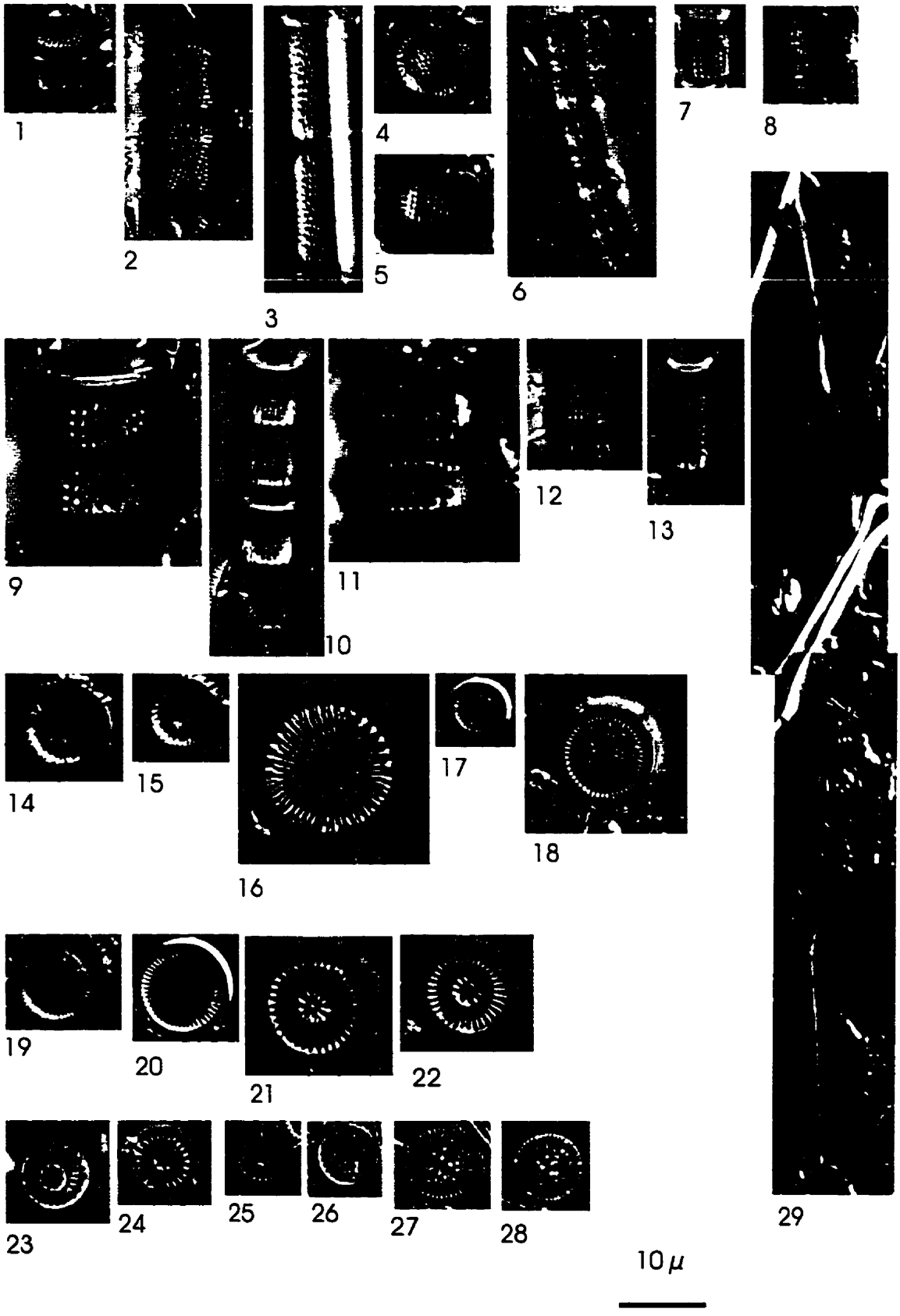
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Plate 2

Fig. 1: *Asterionella formosa* Hassall

Fig. 2: *Asterionella ralfsii* var *americana* Korner

Fig. 3-4: *Fragilaria brevistrata* Grunow

Fig. 5: *Fragilaria brevistrata* var *inflata* (Pant.) Hustedt

Fig. 6-7: *Fragilaria capucina* Desm.

Fig. 8-10: *Fragilaria capucina* var *gracilis* (Oestrup) Hustedt

Fig. 11: *Fragilaria construens* var *binodis* (Ehrenberg) Grunow

Fig. 12: *Fragilaria construens* var *venter* (Ehrenberg) Grunow

Fig. 13-14: *Fragilaria crotonensis* Kitton

Fig. 15: *Fragilaria nanana* Lange-Bertalot

Fig. 16: *Fragilaria pinnata* Ehrenberg

Fig. 17: *Fragilaria pseudoconstruens* Marciniak

Fig. 18: *Fragilaria tenera* (W. Smith) Lange-Bertalot

Fig. 19: *Tabellaria fenestrata* (Lyngb.) Kutzing

Fig. 20-21: *Tabellaria flocculosa* (Roth) Kutzing

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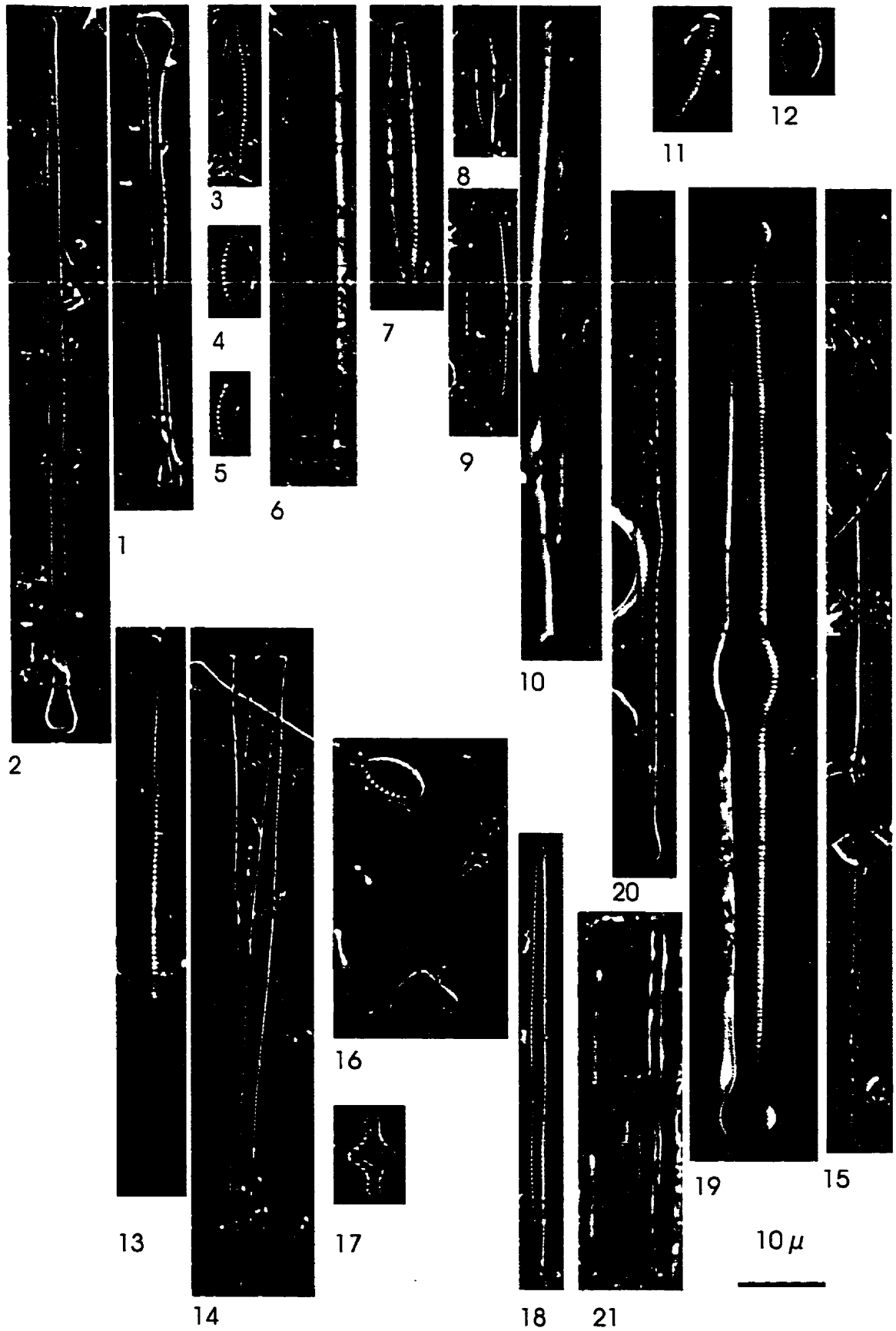
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Plate 3

Fig. 1: *Achnanthes curtissima* Carter

Fig. 2: *Achnanthes laterostrata* Hustedt

Fig. 3: *Achnanthes levanderi* Hustedt

Fig. 4: *Achnanthes minutissima* Kutzing

Fig. 5: *Achnanthes ricula* Hohn and Hellermann

Fig. 6: *Achnanthes subatomoides* (Hustedt) Lange-Bertalot and Archibald

Fig. 7: *Achnanthes suchlandtii* Hustedt

Fig. 8: *Brachysira garrensis* (Lange-Bertalot and Krammer) Lange-Bertalot

Fig. 9: *Brachysira* type 1

Fig. 10-11: *Navicula arvensis* Hustedt

Fig. 12-15: *Navicula minima* Grunow

Fig. 16: *Navicula minuscula* Grunow

Fig. 17-18: *Navicula schassmannii* Hustedt

Fig. 19: *Navicula seminuloides* Hustedt

Fig. 20: *Navicula submuralis* Hustedt

Fig. 21-22: *Navicula vitiosa* Schimanski

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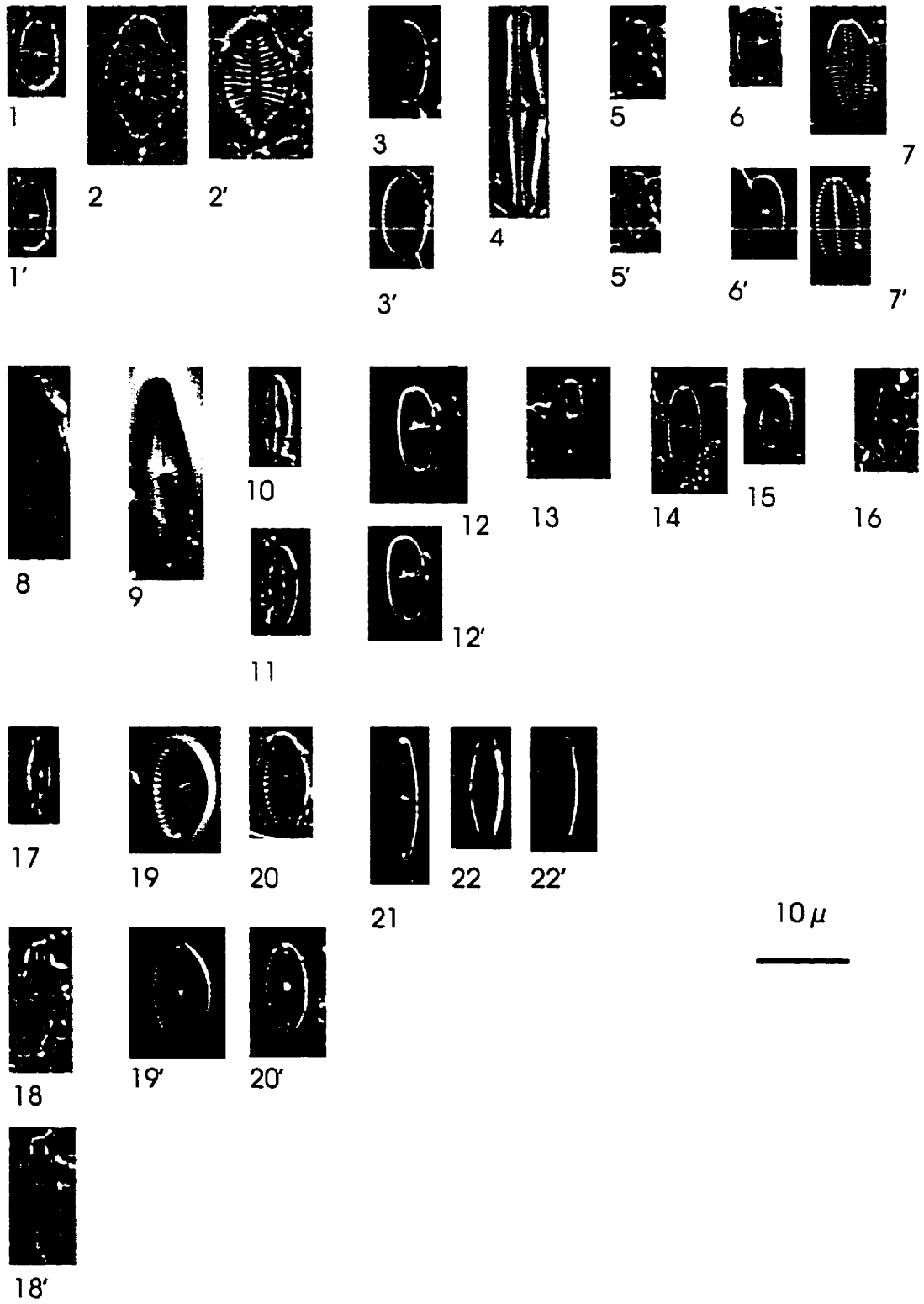
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10 μ

Table 3. Lake Opeongo diatom species list. Includes all taxa found in all core samples, their maximum relative abundance, and number of occurrences. Species with maximum abundance or number of occurrences of zero were found during initial taxonomic surveys of the slides but did not appear in the actual counts.

Diatom species names and authorites	*Maximum Abundance*	*Number of Occurrences*
Achnanthes cf *acares* Hohn & Hellermann	*0.4*	*1*
Achnanthes alpestris (Brun) Lange-Bertalot & Metzeltin	*0.6*	*6*
Achnanthes carissima Lange-Bertalot	*0.6*	*4*
Achnanthes chlidanos Hohn & Hellermann	*0.6*	*2*
Achnanthes conspicua Mayer	*0.4*	*1*
Achnanthes curtissima Carter	*1.8*	*25*
Achnanthes daonensis Lange-Bertalot	*0.6*	*6*
Achnanthes depressa (Cleve) Hustedt	*0.2*	*1*
Achnanthes didyma Hustedt	*0.8*	*29*
Achnanthes engelbrechtii Cholnoky	*0.4*	*1*
Achnanthes exigua Grun. in Cleve & Grunow	*0.4*	*2*
Achnanthes flexella (Kutzing) Brun	*0.8*	*11*
Achnanthes helvetica (Hustedt) Lange-Bertalot	*0.4*	*6*
Achnanthes hungarica (Grunow) Grunow	*0.2*	*1*
Achnanthes imperfecta Schimanski	*0.2*	*3*
Achnanthes impexa Lange-Bertalot	*0.6*	*20*
Achnanthes lacus-vulcani Lange-Bertalot & Krammer	*0.8*	*19*
Achnanthes laevis Oestrup	*0.4*	*2*
Achnanthes lanceolata (Brebisson) Grunow	*0.4*	*3*
Achnanthes lanceolata ssp.*robusta* var *abbreviata* Reimer	*0.4*	*2*
Achnanthes laterostrata Hustedt	*1.6*	*24*
Achnanthes levanderi Hustedt	*1.2*	*32*
Achnanthes linearis (W. Smith) Grunow	*0.7*	*21*
Achnanthes marginulata Grunow	*0.8*	*19*

Diatom species names and authorites	*Maximum Abundance*	*Number of Occurrences*
Achnanthes minutissima Kutzing	*5.8*	*59*
Achnanthes nitidiformis Lange-Bertalot	*0.6*	*15*
Achnanthes oestrupii (Cleve-Euler) Hustedt	*0.2*	*1*
Achnanthes peragalli Brun & Heribaud	*0.4*	*5*
Achnanthes petersenii Hustedt	*0.6*	*4*
Achnanthes ricula Hohn & Hellermann	*2.0*	*22*
Achnanthes rosenstockii Lange-Bertalot	*0.4*	*2*
Achnanthes rossii Hustedt	*0.2*	*1*
Achnanthes stewartii Patrick	*0.4*	*1*
Achnanthes subatomoides (Hustedt) Lange-Bertalot & Archibald in Krammer & Lange-Bertalot	*1.6*	*52*
Achnanthes suchlandtii Hustedt	*2.1*	*44*
Achnanthes ventralis (Krasske) Lange-Bertalot	*0.4*	*9*
Amphipleura pellucida (Kutzing) Kutzing	*0.1*	*7*
Amphora inariensis Krammer	*0.2*	*1*
Amphora libyca Ehrenberg	*0.3*	*4*
Amphora ovalis (Kutzing) Kutzing	*0.4*	*6*
Amphora pediculus (Kutzing) Grunow ex A. Schmidt	*0.4*	*12*
Amphora veneta var *capitata* Haworth	*0.2*	*1*
Asterionella formosa Hassall	*26.2*	*59*
Asterionella ralfsii var *americana* Korner	*1.8*	*25*
Asterionella ralfsii W. Smith	*0.4*	*9*
Aulacoseira alpigena (Grunow) Krammer	*3.7*	*51*
Aulacoseira ambigua (Grunow in Van Heurck) Simonsen	*2.6*	*54*
Aulacoseira crenulata (Ehrenberg) Thwaites	*1.6*	*3*
Aulacoseira distans (Ehrenberg) Simonsen	*12.3*	*59*
Aulacoseira distans var *humilis* (Cleve-Euler) Simonsen	*1.5*	*14*
Aulacoseira distans var *nivaloides* (Camburn) Simonsen	*0.4*	*4*
Aulacoseira distans var *tenella* (Nygaard)(Florin) Simonsen	*2.6*	*33*
Aulacoseira cf. *herzogii* Lemmermann	*0*	*0*
Aulacoseira italica (Ehrenberg) Simonsen	*0.5*	*10*
Aulacoseira lacustris (Grunow) Krammer	*1.0*	*9*

Diatom species names and authorites	*Maximum Abundance*	*Number of Occurrences*
Aulacoseira lirata (Ehrenberg) Ross	*1.2*	*14*
Aulacoseira lirata var *biserata* (Grunow) Haworth	*0.4*	*1*
Aulacoseira nygaardii (Camburn) Simonsen	*1.8*	*11*
Aulacoseira perglabra var *floriniae* Camburn	*0.8*	*6*
Aulacoseira subarctica (O. Mull.) Haworth	*2.8*	*56*
Brachysira brebissonii Ross	*0.4*	*2*
Brachysira garrensis (Lange-Bertalot & Krammer) Lange-Bertalot	*1.1*	*17*
Brachysira intermedia (Oestrup) Lange-Bertalot	*0.4*	*9*
Brachysira microclava Lange-Bertalot & Moser	*0.2*	*5*
Brachysira neoexilis Lange-Bertalot	*1.0*	*17*
Brachysira styriaca (Grunow) Ross	*0.2*	*1*
Brachysira zellensis (Grunow) Round & Mann	*0.2*	*1*
Caloneis silicula (Ehrenberg) Cleve	*0*	*0*
Caloneis undulata (Gregory) Krammer	*0.4*	*2*
Cocconeis diminuta Pant.	*0.4*	*7*
Cyclostephanos invisitatus (Hohn & Hellermann) Theriot, Stroermer & Hakansson	*0.2*	*1*
Cyclotella bodanica var *lemanica* (O. Mull. Ex Schrot.) Bachm.	*9.2*	*59*
Cyclotella comensis Grunow in Van Heurck	*1.0*	*7*
Cyclotella kuetzingiana Thwaites	*0.8*	*9*
Cyclotella kuetzingiana var *planetophora* Fricke	*0.8*	*22*
Cyclotella kuetzingiana var *radiosa* Fricke	*1.8*	*18*
Cyclotella michiganiana Skvort.	*4.8*	*58*
Cyclotella ocellata Pant.	*0.8*	*10*
Cyclotella stelligera Cleve & Grunow	*51.0*	*59*
Cyclotella tripartita Hakansson	*1.4*	*16*
Cymbella cesatii (Rabenhorst) Grunow in A. Schmidt	*0.4*	*5*
Cymbella cymbiformis Agardh	*0*	*0*
Cymbella delicatula Kutzing	*0.2*	*1*
Cymbella gaeumannii Meister	*0.8*	*16*

Diatom species names and authorites	Maximum Abundance	Number of Occurrences
Cymbella gracilis (Ehrenberg) Kutzing	0.5	9
Cymbella hybrida Grunow	0.4	1
Cymbella incerta (Grunow) Cleve	0.8	6
Cymbella microcephala Grunow	0.8	26
Cymbella minuta Hilse	0.4	8
Cymbella silesiaca Bleisch	0.9	12
Cymbella subcuspidata Krammer	0	0
Diatoma tenuis Agardh	0.1	1
Diploneis boldtiana Cleve	0.2	1
Diploneis marginestriata Hustedt	0.6	9
Diploneis parma Cleve	0.4	20
Entomoneis ornata (Bailey) Reimer	0.1	1
Eunotia cf *arculus* (Grunow) Lange-Bertalot and Norpel	0.2	5
Eunotia bilunaris (Ehrenberg) Mills	0.2	11
Eunotia bilunaris var *mucophila* Lange-Bertalot and Norpel	0.1	1
Eunotia elegans Oestrup	0.2	5
Eunotia exigua (Brebisson) Rabenhorst	0	0
Eunotia flexuosa (Brebisson) Kutzing	0.4	4
Eunotia formica Ehrenberg	0	0
Eunotia hexaglyphis Ehrenberg	0.2	1
Eunotia iatriaensis Foged	0.2	1
Eunotia implicata Norpel & Lange-Bertalot	0.1	1
Eunotia incisa Gregory	0.3	7
Eunotia intermedia (Krasske) Norpel & Lange-Bertalot	0.2	1
Eunotia cf *minor* (Kutzing) Grunow	0.2	1
Eunotia nymanniana Grunow	0.2	5
Eunotia pectinalis var *undulata* (Ralfs) Rabenhorst	0.4	6
Eunotia praerupta Ehrenberg	0.2	1
Eunotia rhynchocephala Hustedt	0.2	2
Eunotia cf *septentrionalis* Oestrup	0.2	4
Eunotia serra Ehrenberg	0	0
Fragilaria brevistrata Grunow	4.9	47

Diatom species names and authorites	*Maximum Abundance*	*Number of Occurrences*
Fragilaria brevistrata var *inflata* (Pant.) Hustedt	*1.9*	*15*
Fragilaria capucina Desm.	*3.0*	*46*
Fragilaria capucina var *gracilis* (Oestrup) Hustedt	*3.0*	*44*
Fragilaria capucina var *rumpens* (Kutzing) Lange-Bertalot	*0*	*0*
Fragilaria capucina var *vaucheriae* (Kutzing) Lange-Bertalot	*0*	*0*
Fragilaria constricta Ehrenberg	*0.4*	*5*
Fragilaria construens fo. *binodis* (Ehrenberg) Grunow	*3.5*	*4*
Fragilaria construens var *pumila* Grunow	*0*	*0*
Fragilaria construens var *venter* (Ehrenberg) Grunow	*1.2*	*6*
Fragilaria crotonensis Kitton	*11.8*	*54*
Fragilaria exigua Grunow	*0.5*	*15*
Fragilaria nanana Lange-Bertalot	*4.2*	*46*
Fragilaria oldenburgiana Hustedt	*1.0*	*27*
Fragilaria parasitica (W. Smith) Grunow	*0.4*	*2*
Fragilaria pinnata Ehrenberg	*6.0*	*42*
Fragilaria pinnata var *lancettula* (Schumann) Hustedt	*0.9*	*17*
Fragilaria pseudoconstruens Marciniak	*2.3*	*11*
Fragilaria robusta Hustedt	*0.2*	*1*
Fragilaria tenera (W. Smith) Lange-Bertalot	*2.1*	*46*
Fragilaria ulna (Nitzsch) Lange-Bertalot	*0.8*	*7*
Frustulia rhomboides (Ehrenberg) De Toni	*0.4*	*12*
Gomphonema acuminatum Ehrenberg	*0.4*	*3*
Gomphonema exilissimum (Grunow) Lange-Bertalot & Reichardt	*0.4*	*1*
Gomphonema gracile Ehrenberg	*0.4*	*6*
Gomphonema parvulum (Kutzing) Kutzing	*0.7*	*7*
Gomphonema parvulum var *lagenula* (Kutzing) Frenguelli	*0.4*	*1*
Gomphonema subtile Ehrenberg	*0.4*	*2*
Gyrosigma acuminatum (Kutzing) Rabenhorst	*0.2*	*3*
Hantzschia amphioxys (Ehrenberg) Grunow	*0.1*	*1*
Mastogloia smithii Thwaites ex W. Smith	*0.4*	*1*
Meridion circulare var *constrictum* (Ralfs) Van Heurck	*0*	*0*

Diatom species names and authorites	Maximum Abundance	Number of Occurrences
Navicula arvensis Hustedt	2.0	3
Navicula begerii Krasske	0.4	1
Navicula bremensis Hustedt	0.4	1
Navicula bryophila Petersen	0.8	7
Navicula cocconeiformis Gregory	0.2	10
Navicula cryptocephala Kutzing	0.4	11
Navicula cryptotenella Lange-Bertalot	0.6	32
Navicula detenta Hustedt	0.4	2
Navicula difficillima Hustedt	0.4	5
Navicula cf. *digitulus* Hustedt	0.2	2
Navicula disjuncta Hustedt	0.4	2
Navicula explanata Hustedt	0.4	4
Navicula farta Hustedt	0.6	5
Navicula gerloffii Schimanski	0.2	3
Navicula halophila (Grunow) Cleve	0.2	1
Navicula jaagii Meister	0.4	1
Navicula jaernefeltii Hustedt	0.4	3
Navicula cf. *kuelbsii* Lange-Bertalot	0.4	2
Navicula laevissima Kutzing	0.4	3
Navicula mediocris Krasske	0.8	8
Navicula minima Grunow	1.3	25
Navicula minima var *pseudofossilis* (Krasske) Reimer	0.7	13
Navicula minuscula Grunow	1.6	24
Navicula mutica Kutzing	0.2	2
Navicula oblonga Kutzing	0.5	2
Navicula cf. *perminuta* Grunow	0.4	1
Navicula cf. *porifera* Hustedt	0.4	1
Navicula pseudolanceolata Lange-Bertalot	0.4	4
Navicula pseudoscutiformis Hustedt	0.7	16
Navicula pseudoventralis Hustedt	0.6	10
Navicula pupula Kutzing	0.6	12
Navicula schadei Krasske	0.4	4

Diatom species names and authorites	Maximum Abundance	Number of Occurrences
Navicula schassmannii Hustedt	1.1	21
Navicula schoenfeldii Hustedt	0.4	3
Navicula seminuloides Hustedt	2.5	44
Navicula seminulum Grunow	0.9	27
Navicula similis Krasske	0.4	2
Navicula submuralis Hustedt	2.9	45
Navicula subtilissima Cleve	0.2	1
Navicula trivialis Lange-Bertalot	0.4	1
Navicula umbra Hohn & Hellermann	0.4	5
Navicula ventralis Krasske	0.4	1
Navicula vitabunda Hustedt	0.6	6
Navicula vitiosa Schimanski	3.2	43
Neidium affine (Ehrenberg) Pfitzer	0	0
Neidium bisulcatum (Lagerstedt) Cleve	0.4	2
Neidium cf. *productum* (W. Smith) Cleve	0	0
Nitzschia acicularis (Kutzing) W. Smith	0.4	8
Nitzschia angustata (W. Smith) Grunow in Cleve & Grunow	0.7	10
Nitzschia cf. *fonticola* Grunow	0.6	6
Nitzschia microcephala Grunow	0.4	6
Nitzschia palea (Kutzing) W. Smith	0.9	11
Nitzschia paleaeformis Hustedt	0.4	6
Nitzschia cf *recta* Hantzsch in Rabenhorst	0.5	9
Nitzschia tubicola Grunow	0.4	4
Peronia fibula (Brebisson) Ross	0.2	1
Pinnularia braunii (Grunow) Cleve	0.4	1
Pinnularia interrupta W. Smith	0.2	1
Pinnularia microstauron (Ehrenberg) Cleve	0.4	1
Pinnularia nodosa Ehrenberg	0.4	2
Pinnularia viridis (Nitzsch) Ehrenberg	0.6	2
Rhizoselenia eriensis H.L. Smith	6.7	59
Rhopalodia parallela var *parallela* (Grunow) O. Mull.	0	0
Stauroneis anceps Ehrenberg	0	0

Diatom species names and authorites	*Maximum Abundance*	*Number of Occurrences*
Stauroneis anceps fo *gracilis* Rabenhorst	*0.4*	*4*
Stauroneis smithii Grunow	*0.2*	*4*
Stenopterobia delicatissima (Lewis) Brebisson Ex Van Heurck	*0.6*	*2*
Stenopterobia densestriata (Hustedt) Krammer	*0.4*	*13*
Surirella angusta Kutzing	*0.4*	*8*
Tabellaria fenestrata (Lyngb.) Kutzing	*1.2*	*49*
Tabellaria flocculosa (Roth) Kutzing	*0.2*	*1*
Tabellaria flocculosa strain III sensu Koppen	*1.0*	*22*
Tabellaria flocculosa strain IIIp sensu Koppen	*21.6*	*59*
Tabellaria flocculosa strain IV sensu Koppen	*1.1*	*48*
Tabellaria flocculosa var *linearis* sensu Koppen	*1.4*	*43*

Chapter 6: Discussion

This study attempts to answer whether Lake Opeongo has become more eutrophic over the past 200 years as a response to the heavy European settlement that occurred in Southern Ontario. The answer is a qualified yes. From the two cores analyzed, there has been a definite shift in the diatom communities that began at approximately 1956 as shown by the ^{210}Pb analysis of core 3. Prior to this time, the lake's diatom community was dominated by the *Cyclotella stelligera* complex with subdominants *Tabellaria flocculosa* IIIp and *Aulacoseira distans*. At this time, the community began to shift to one consisting of increasing amounts of *Asterionella formosa*, *C. bodanica* var *lemanica*, *Rhizosolenia eriensis* and *Fragilaria crotonensis* and decreasing percentages of *C. stelligera* complex and *A. distans*. *Tabellaria flocculosa IIIp* remained a relative subdominant in this different community. An examination of the TP preferences of the major diatoms as published in the literature shows that this shift was towards a more mesotrophic or eutrophic state. Hall and Smol (1992, 1996), Dixit et al. (1999), Reavie et al. (1995a), Reavie and Smol (*in press*), Charles (1985), Fritz et al. (1993), Cumming et al. (1995) and Wilson et al. (1996) all found that *A. formosa*, *C. bodanica* var *lemanica*, *F. crotonensis* and *R. eriensis* have higher TP optima than *C. stelligera* complex and *A. distans* do (Table 4). The *C. stelligera* complex and *A. distans* are considered to be indicators of oligotrophy and typical of pre-European settlement lakes (Brugam, 1988).

The diatom-inferred TP reconstructions from Hall and Smol (1996) and Wilson et al. (1996) both show a pronounced trend beginning from extremely oligotrophic conditions towards more mesotrophic ones. Contrastingly, Reavie and Smol (*in press*)'s calibration set gave a reconstruction showing a shift towards more oligotrophic conditions. From analog analysis using minimum dissimilarity coefficients and CCA, the calibration set of Hall and Smol (1996) provided the closest modern analogs to the fossil diatom assemblages. The calibration set of Wilson et al. (1996) provided the next closest analog and that of Reavie and Smol (*in press*) provided the worst analog.

Table 4 Total phosphorus preferences of the common diatoms from the literature. All values in µg TP/L. (na) denotes not available, (?) denotes taxonomic uncertainty.

Study	*Cyclotella stelligera complex*	*Aulacoseira distans*	*Tabellaria flocculosa*	*Asterionella formosa*	*Cyclotella bodanica var lemanica*	*Fragilaria crotonensis*	*Rhizosolenia eriensis*
Hall & Smol (1996)	6.49	6.61	7.99	7.95	7.83	7.64	8.14
Dixit et al. (1999)	7	8	8	10	na	14	na
Reavie et al. (1995)	13.5	15.1	17.9	14.6	21.0	17.6	11.8
Reavie & Smol (in press)	12*	na	13	14	11	14	13
Charles (1985)	7.2	5.5	7.6	10.4	9.6?	7.4	na
Fritz et al. (1993)	6.3	na	9.2	13.3	13.8?	13.9	na
Cumming et al. (1995)	9.7	7.7	15.0	11.8	11.7	13.8	na
Hall and Smol (1992)	9.7	7.8	15.1	11.8	11.7	13.9	na
Wilson et al. (1996)	4.3	4.3	11.3	12.3	9.1	13.8	18.3 (cysts)

Wilson et al.'s (1996) training set gives the best fit of the three as assessed by minimum DCs, by and large providing close modern analogs to the fossil samples below 3 cm. Reavie and Smol (*in press*)'s calibration set had no or very poor modern DC analogs to the lower portions of the core below 7 cm although it had mainly close modern DC analogs above that. Hall and Smol's (1996) training set also had a better fit to the upper portion of the core. It had no or very poor close modern DC analogs below 9.5 cm (ca. 1944). Between Hall and Smol's set and Reavie and Smol's set, the former provides the closer fit. Hall and Smol's training set had an excellent fit with the L. Opeongo core as assessed by squared residual distances to the CCA axis 1. Wilson et al.'s and Reavie and Smol's sets both had

problems providing modern analogs below 6.5 cm. Because Hall and Smol's and Wilson et al.'s sets had closer modern DC analogs than Reavie and Smol's and because of the excellent fit of Hall and Smol's set with respect to the CCA, the reconstructions based on the former two training sets showing a shift towards more mesotrophic conditions appear more trustworthy than the reconstruction based on the later implying a trend towards more oligotrophic conditions. Selected ranges of physical and chemical data of the three calibration sets suggests that L. Opeongo is best modeled by Hall and Smol's set (Table 5) as it falls within the measured environmental gradients. In particular, L. Opeongo lies outside the alkalinity and pH ranges of Reavie and Smol's set. According to Dixit et al. (1992), pH is the dominant factor affecting the distribution of diatoms. The lack of fit problems to modern analogs are in part due to the fact that the diatom species were not completely characteristic of the geographical location of Lake Opeongo, having strong similarities to the flora of James and Hudson Bays (Fallu, 1998). The significant increase in the relative abundance of the planktonic species in Zone 2 also suggests that the lake became more nutrient-rich. Planktonic diatoms are more likely to limited by phosphorus than periphytic diatoms which have access to further sources of phosphorus. Hence, an increase in the relative abundance of planktonic species versus periphytic ones suggests that the nutrient conditions have improved in the epilimnion.

Laird et al. (1998b) discuss the difference between the two forms of modern analog assessment: that based on dissimilarity measures and that based on CCA. They state that if more than one environmental variable is being reconstructed at the time or if many fossil samples contain taxa not in the training set or in higher abundances than found in the calibration set, then the minimum dissimilarity coefficients may be the more appropriate approach. If one wishes an assessment of the reliability of a single environmental variable reconstruction, then the CCA approach is the more appropriate. The dissimilarity approach uses the information on all the taxa, not just those in the calibration set, and thus gives evidence that the fossil samples are unusual in comparison to the fossil data set. The CCA approach only works on the taxa in the calibration set, no matter how well or how poorly they are represented in the calibration data set. Thus, a finding of very poor or poor modern analogs for a fossil

sample identifies that there is something unusual for the reconstructed value based on its residual distance in comparison to the distances of the values from the calibration set. Both methods give useful information, somewhat different in each case.

Table 5. Ranges of selected physical and chemical characteristics of the lakes included in the calibration sets. (na) denotes not available.

Calibration set characteristics	Lake Opeongo values	Hall and Smol (1996)	Reavie and Smol (*in press*)	Wilson et al. (1996)
TP (µg/L)	8	2.7 - 24.3	4 - 54	2 - 268
pH	6.8	5.61 - 7.30	7.0 - 8.6	7.3 - 9.9
alkalinity (mg/L)	4.60	0.40 - 22.90	12.4 - 154	na
hardness (mg/L)	15 - 30	na	na	10.4 - 1424
elevation (m)	403	20 - 100	80 - 320	na
maximum depth (m)	51.8	4.5 - 82.4	5.2 - 95	1.5 - 44
bedrock geology	Precambrian Shield	Precambrian Shield	limestone and/or Precambrian Shield	na

The three calibration sets had roughly equivalent r^2 values with Wilson et al.'s set being the highest (see Table 2 for a summary). Following Birks (1994), the root mean square of the error (RMSE) is usually used as a measure of the predictive capabilities of modern calibration sets. Hall and Smol's set's RMSE was roughly half that of Reavie and Smol's. Unfortunately, the RMSE of Wilson et al.'s set cannot be directly compared to that of the other two because the TP values were log-transformed in order to attain normality. Ter Braak and van Dam (1989), Birks (1995) and Birks et al. (1990) have shown that the RMSE is underestimated and the r^2 is overestimated when based only upon the calibration set. Hence, Birks et al. (1990) used bootstrapping to derive an unbiased overall root mean square error of prediction ($RMSE_{boot}$) and r^2_{boot} for the calibration set. The $RMSE_{boot}$ of Hall and Smol's calibration set is slightly less than half of that of Reavie and Smol's calibration set. This suggests that Hall and Smol's calibration set has better predictive ability than that of Reavie and Smol. Once again, the $RMSE_{boot}$ of Wilson et al.'s set

cannot be directly compared to that of the other two. The r^2_{boot} are all roughly comparable, with Wilson et al.'s training set being slightly the best. They are all somewhat low, suggesting that reconstructing TP values by mean bootstrapped values is not as reliable as straight WA. The gaps between the WA-inferred r^2 and RMSE and the bootstrapped r^2_{boot} and $RMSE_{boot}$ indicate that the optima of diatoms for TP are not as well defined as possible, i.e., as with pH (Hall and Smol, 1996). The relationships between diatoms and TP may not be as strong because other factors, such as intracellular storage, physical turbulence, dissolved organic molecules and light penetration all influence diatoms (Reynolds, 1984).

Another important statistical consideration of the predictive abilities of the calibration sets is their size. Wilson et al. (1994; 1996) found that for their salinity reconstructions, the greatest improvement in the $RMSE_{boot}$ was attained when the data set was expanded from 20 to 60 lakes, and that after roughly 100 lakes the $RMSE_{boot}$ remained relatively stable. By this criterion, the calibration set of Wilson et al. is large enough (111 lakes); but those of Hall and Smol (53 lakes) and Reavie and Smol (59 lakes) are somewhat small.

The WA model provides a single estimate of a variable that has high natural within-year variability. Therefore, the errors of the model should be considered with this in mind (Bennion et al., 1995). Actual measured TP values in Lake Opeongo can vary annually by 34 µg/L (1984) with a standard deviation of 13 µg/L (Appendix IV)(assuming that these water chemistry data are accurate), which compares reasonably to the bootstrapped estimates of the sample-specific standard deviations (Est Se_{pred}). It is probable that the errors of such diatom-TP models cannot be reduced more (Bennion et al., 1995) because of the difficulty of measuring TP over the year for the calibration set lakes and because the factors controlling diatom abundances are obviously multivariate, rather than just TP alone. According to Birks (pers. com.), the bootstrapped estimates of the sample-specific standard deviations (Est Se_{pred}) of the WA inferred TP values may be underestimates.

In order to properly assess the validity of using diatom-based WA inferences to reconstruct paleoenvironmental variables, diatom-based WA inferences must be compared to actual instrumental data. Unfortunately, historical records of environmental variables are rather rare, but a few such data

sets are being made. Renberg and Hultberg (1992) applied a diatom-pH WA transfer function to a sediment core from Lake Lysevatten, Sweden, a lake with an oxygenated hypolimnion. Water chemistry data existed for a roughly 30 year period for this lake. They found that trends in the WA inferred values compared well with trends in the measured data, although the model underestimated the extremes of shifts. They attributed the discrepancies to at least three factors: (1) bioturbation and physical mixing, (2) redeposition of older sediments from shallower water and (3) no analogs for the postliming flora in the calibration set. They caution that because of bioturbation and physical mixing sudden changes will have a date older than the real date of change. Bennion et al. (1995) applied a diatom-phosphorus WA transfer function to a sediment core with annual varves from Mondsee, Austria. Water chemistry data existed for an18 year period for this lake. They found that the WA inferred values compared well with the measured data, although the model underestimated values for a portion of the core which had poor analogs in the calibration set. The diatom-inferred values compared much better to the measured data than did the sediment TP profile. Lotter (1998) applied a diatom-phosphorus weighted-averaging partial least squares (WA-PLS) transfer function (a method related to weighted averaging) to a sediment core with annual varves from Baldeggersee, Switzerland. Water chemistry data existed for a 29 year period for this lake. He found that the diatom-inferred values compared well with the measured data. However, reconstructions seem to underestimate the actual values. For instance, Little et al.'s (2000) Gravenhurst Bay diatom-inferred TP values had a range of 2.7 - 24.3 µg/L but measured values had a range of 18 - 52 µg/L during the same time period. Also deshrinking methods can distort the actual values (see below). In conclusion, the calibration function approach is valid and the trends in the WA-inferred TP values are credible, but actual WA-inferred TP values probably are not.

Another contribution to WA's underestimation is that averages are taken twice, causing a shrinkage of the range of inferred values (Birks et al., 1990; ter Braak and van Dam, 1989). To correct this, a simple linear deshrinking is done by regression of either the initial inferred values for the training set on the observed values (classical deshrinking) or vice-versa (inverse deshrinking). Classical deshrinking deshrinks more than inverse deshrinking, taking reconstructed values further away from the mean and it

minimizes the bias in the model residuals (Line and Birks, 1994). Inverse deshrinking minimizes the root mean squared error in the training set. All three of the calibration sets used classical deshrinking. This greater stretching effect of classical regression explains the negative values produced by the TP reconstruction based on Reavie and Smol's training set. The reconstruction based on Wilson et al.'s calibration set can be seen to give underestimates of the actual TP values, although without a bootstrap estimate of the error of the actual reconstruction, it is difficult to tell. The reconstruction based on Hall and Smol's training set matched current TP averaged values fairly well (see Appendix II for TP values from 1980-1989). Note: because of turbation in the core, dates cannot be assigned precisely to the top 4 cm of the core (see below for a discussion).

A final problem with the TP calibration function approach is that the amount of phosphorus biologically available to the diatoms may not be accurately measured by epilimnetic TP concentrations. Total phosphorus (TP) is the concentration of all phosphorus in the water, including dissolved phosphorus and phosphorus in plankton. It is an overestimate, sometimes by orders of magnitude, of the phosphorus biologically available to the diatoms (Hudson et al., 2000). However, the actual biologically available phosphorus is extremely difficult to measure and rapidly changes, particularly in the epilimnion where rapid phosphorus recycling is often relied on to sustain productivity as the thermocline isolates the underlying nutrient-rich waters (Hudson et al., 2000). Also the concentration of phosphate (PO_4^{-3}) is often too small to be actually estimated with chemical methods (i.e., if less than 2 µg/l) (Horne and Goldman, 1994). Despite these apparent difficulties, trends in TP, if not actual values, can be reconstructed using diatoms as evidenced by studies that have used diatom-inferred TP transfer functions and cross-checked these with actual limnological measurements (e.g., Bennion et al. 1995; Lotter, 1998).

How reliable are the ^{210}Pb dates? The plot of ^{210}Pb activity versus depth (Fig 2a) shows that there has been some turbation in the top 4 cm of core 3, hence the dates associated with that portion of the core are unreliable. The turbation could be due to sediment slumping or to biological activity. The deepest waters are currently oxygenated. Benthic invertebrates were observed in a core taken nearby and divers have observed burbots digging trenches in the deepest portions of the lake (D. Jackson, pers.

com.). Sediment redistribution due to wind-driven mixing was observed during a surficial coring exercise in June 2000 (Douglas, pers. com.). However, the activity profile of the rest of the core shows a classic exponential decay; and hence, the dates associated with this portion of the core, including the diatom community shift between 8.0 and 8.25 cm, can be used in interpretations.

There are several recent paleolimnological studies of lakes in southern-Ontario that are useful for comparison purposes. In the first study, Hall and Smol (1996) surveyed 54 lakes in the nearby Muskoka-Haliburton region that mainly lay on Precambrian bedrock. Postindustrial pH and TP changes were inferred from surface and pre-1850 sediment diatom assemblages. Diatom-inferred TP suggests that in almost all lakes, present day TP is not higher than before European settlement. They felt that several factors could account for declining lake TP, including lake or watershed acidification processes that reduce P loading or increase P loss rates; and reduction in nutrient loading from watersheds as reforestation occurs.

On the other hand, a number of studies of lakes nearby to Lake Opeongo have shown a definite shift towards eutrophication starting precisely with European settlement at the middle of the last century. However, in comparison to Lake Opeongo, all these lakes experienced a denser settlement in their watershed and thus were more severely impacted. Little et al. (2000) compared historical records to diatom-inferred TP and chironomid-inferred hypolimnetic oxygen levels, and found a clear case of eutrophication at nearby Gravenhurst Bay, Lake Muskoka. Water quality deteriorated significantly following European settlement at the town of Gravenhurst in the mid-1800's around the shores of the bay and reached a low in the first half of the twentieth century. Surface water TP dramatically declined to near oligotrophic levels after the building of a sewage treatment plant in 1972. Gravenhurst Bay's diatom communities were similar to those of Lake Opeongo. Pre-impact, the oligotrophic community consisted of *C. stelligera* complex, *T. flocculosa IIIp* and some *A. formosa.* The eutrophic community consisted of a decreased percentage of *C. stelligera* complex and less *T. flocculosa IIIp* but an increased amount of *A. formosa*, and *A. subarctica, A. ambigua* and *F. crotonensis* become important. After lake recovery, *C. stelligera* complex increased and *A. formosa, A. subarctica, A. ambigua* and *F. crotonensis* all greatly

declined. In another nearby study, Clerk et al. (2000) found that Peninsula Lake, a small, shallow lake on the Precambrian Shield, shifted to a more meso-eutrophic state concurrent with European settlement (ca. 1870). At this time, the diatom community shifted from an oligotrophic one dominated by *C. stelligera* to one characterized *by F. crotonensis, C. bodanica* var *lemanica, A. formosa, Aulacoseira ambigua* and *A. subarctica*. Starting in the mid-1960's there was a shift back towards more oligotrophic conditions shown by an increase in the abundance of *C. stelligera*. Karst and Smol (2000) found a modest diatom-inferred TP increase in Lake Opinicon, a shallow, macrophyte-dominated lake, starting ca. 1860. However, this response to the heavy settlement in Lake Opinicon was much less than that observed at Upper Rideau Lake, another similarly-disturbed deep lake in the area (Christie and Smol, 1996). Lake Opinicon was also flooded by the building of the Rideau Canal (1828-1832). Interestingly in comparison to Lake Opeongo, diatom assemblages changed with respect to those observed in the pre-settlement sediments. Planktonic *Aulacoseira ambigua* decreased from 14% to less than 5%, while benthic diatoms (primarily *Fragilaria construens* and *F. pinnata*) increased by 15% to nearly 95% of the assemblages. This rise in benthic taxa may be due to an expansion of the littoral zone of Lake Opinicon, a very shallow lake. In contrast, surprisingly the diatom communities of Lake Opeongo showed no changes when the dam at Annie Bay was constructed, perhaps because it was initially a deep-water lake with less littoral zone than lakes Opinicon and Upper Rideau, and remained one after flooding. Although not in southern Ontario, Brugam (1988) surveyed core tops (present conditions) and core bottom (pre-European settlement conditions) in 15 Washington State lakes that showed eutrophication due to Euro-American settlement. He found that *A. formosa* and *F. crotonensis* tended to increase in the core tops and that species that decreased with human disturbance (i.e., those found preferentially in the core bottoms) included *A. distans, A. italica* and *C. stelligera* complex. Fritz et al. (1993) studied four oligo-mesotrophic lakes in Michigan and found that three of them showed a trend towards higher phosphorus concentrations at the time of logging and settlement during the last century and a subsequent decline.

What are the possible causes for the apparent eutrophication of Lake Opeongo? There are a number of possible ways that this could have occurred, acting singly or jointly. Among these are direct

human impact on the watershed; increased North American post-war fertilizer use; increased nutrient run-off as a result of fires in the watershed; global warming, including changes in the thermal conditions of the lake; and trophic level changes caused by human manipulation of the fish community of the lake. Because of the recent date of the diatom community shift, increased nutrient release into the lake from the construction of the dam (ca 1867) and from logging (Bormann and Likens, 1979), heaviest from 1860-1900, are unlikely main causal factors.

It is possible that the lake has received a higher input of nutrients due to increased direct human impacts on the watershed. However, intuitively, it seems that earlier logging, railway and road-building would have had a greater impact than the current levels of tourism, mainly campers and anglers, and that the shift should have occurred earlier (ca 1870), as at Gravenhurst Bay (Little et al., 2000) and Pennisula Lake (Clerke et al., 2000). As well, such eutrophication due to human impact is not typical of the area as shown by Hall and Smol's (1996) survey of 54 nearby lakes with comparable settlement history and similar bedrock geology. They observed only five lakes that had increased their diatom-inferred TP by more than 2 µg/L over the past 150 years. However, it is also possible that the impacts of the last century were simply delayed or were not quite enough to reach a threshold which was recently crossed.

After World War II, inorganic fertilizer use increased greatly in North America until today more nitrogen is fixed by human activities (140 Tg N/yr) than by biotic processes (90-130 Tg N/yr) (Jefferies and Maron, 1997; Vitousek, 1994). Inorganic phosphate fertilizer use has also greatly increased during this time period (Vitousek, 1994) and significant proportions of these fertilizers are transported long distances in the atmosphere and redeposited (Langan, 1999). Wolfe et al. (2000) report unprecedented changes beginning in 1950 in the diatom communities, from oligotrophic species to more mesotrophic ones (including *A. formosa* and *F. crotonensis*), in alpine lakes in Rocky Mountain National Park, Colorado, USA due to airborne nitrogen deposition from agricultural and industrial activities in adjacent lowland regions. Stoermer et al. (1990) found that there have been significant changes in the diatom community in McLeod Bay, Great Slave Lake, Canada during the post-industrial era. Because there is little apparent anthropogenic modification of the bay's drainage basin or biotic communities, they

implicated atmospheric transport of nutrients from remote sources. It is possible that the mesotrophic shift in Lake Opeongo is due to increased nutrient levels in the lake from deposition of airborne nitrogen and phosphorus. The 1956 timing of the shift is suggestive. Unfortunately, the two nearby lakes, Gravenhurst Bay of Lake Muskoka (Little et al., 2000) and Pennisula Lake (Clerke et al., 2000), for which diatom stratigraphies with ^{210}Pb dates exist cannot be used to confirm this possibility. Both lakes had fairly heavy settlement on their shores and consequently were eutrophic by 1960 as indicated by diatom-inferred TP and by 1969 by instrumental TP measurement for Gravenhurst Bay. Remediation efforts, including sewage treatment, beginning in the 1970's have significantly improved water quality in both lakes. The initial meso-eutrophic conditions in the 1960's and the reduction of the strong local impacts could have masked any nutrient enrichment from airborne deposition of fertilizers. However, if nutrient enrichment due to long-distance airborne transport of fertilizer were occurring, a greater number of Hall and Smol's (1996) 54 lakes would be expected to be more eutrophic, unless acid deposition has masked this trend (Langan, 1999; Hall and Smol, 1996). An examination of Lake Opeongo's sediment for d^{15}N values would reveal whether there has been deposition of airborne fertilizer (Wolfe et al., 2000).

Increased nutrient run-off into the lake as a consequence of forest fires is a possible cause of a eutrophic trend (Moser, 1996). However, several studies have demonstrated that fires have little effect on nutrient levels in lakes (e.g. Schindler et al., 1980; Bayley et al., 1992), and that limnic diatoms show little, if any, response to the slight nutrient increases after a fire (Bradbury, 1986; Moser, 1996). Although Rhodes and Davis (1995) and Korhola et al. (1996) found that lake water pH increased significantly after a fire, their study lakes were very small (less than 2 ha in area), and the impact of a fire would have been greater than in large Lake Opeongo. According to records from the Algonquin Forestry Authority (K. Fletcher, area forester, pers. com.) there was only one fire in the Lake Opeongo watershed during the period of 1920-1978. That fire occurred in 1943 and any nutrient run-off would have flowed into Annie Bay and out of the lake, not into the South Arm. However, this record appears to be incomplete as Guyette and Dey's (1995) dendrochronological study identify a fire in 1936 in Costello Creek which eventually flows into the South Arm and M. Ridgway (Harkness Fisheries Laboratory, pers. com.) reports

a large fire in 1953 at the North Arm. If the amended record is accepted, it seems unusual that the significant change in the diatom community that has persisted for more than 40 years be due to fire. This is especially true since the 1953 North Arm fire was not extensive enough to be included in the Park records and since limnological effects from fire have been found to last from anywhere from 2-3 years (Bayley et al., 1992) to 30-40 years (Korhola et al., 1996). Furthermore, it seems unlikely that no other fire of similar magnitude has occurred in the Lake Opeongo watershed over the approximately 400 years represented by the diatom stratigraphy, yet the diatom record records no other similar events.

Another possible impact on Lake Opeongo is that from global warming. Recent climate models predict increases in air temperatures and decreases in soil moisture in much of North America at mid-latitudes as a result of increased greenhouse warming (Harvey, 1999). Schindler et al. (1990) examined twenty years of climatic, hydrologic and ecological records for the Experimental Lakes Area (ELA) of northwestern Ontario. They predicted that with global warming, the boreal ecosystems of northwestern Ontario should be severely affected as the area is already quite warm and arid with thin soils with small water storage capacities, similar to that of Lake Opeongo. They found that over 1969-1988 water renewal rates have decreased and concentrations of most chemicals, including total dissolved nitrogen and more conservative ions, but not phosphorus, increased because of decreased water renewal and forest fires in the catchments. The ratio of total nitrogen to total phosphorus doubled from 25:1 to approximately 50:1 and phosphorus became more limiting. However, in Lake Opeongo, phosphorus (or perhaps another limiting nutrient?) appears to have become more, not less, abundant. It is possible that the conditions that have caused phosphorus to become limiting in the ELA are not present in Lake Opeongo. Lake Opeongo's apparent diatom-inferred increase in nutrients could be due to global warming if water renewal rates have decreased. In the ELA, thermoclines deepened due to increased wind velocities, increased wind exposure in burned catchments and increased transparency. In contrast, King (1997) found that the mid-summer thermocline depths for the South and East Arms of Lake Opeongo have shown a significant ($p<0.05$) shallowing trend during the time that records exist (1958-1996). Unfortunately, thermocline records do not exist for the period prior to the diatom community shift.

This shift in thermal stratification could affect the nutrient dynamics of the lake as well as the light regime experienced by the epilimnetic diatoms. Such shifts, as evidenced by Tilman et al.'s (1977, 1982) experimental work although dealing with different species in part, could effectively shift the dominance of one taxon over another, i.e., *Asterionella formosa* vs *Cyclotella meneghiniana*. However, shallowing of the epilimnion appears counter intuitive to a eutrophication trend. Schindler et al. (1996) found that in the same study area, both climate warming and lake acidification led to declines in the dissolved organic carbon (DOC) content of lake waters, allowing increased penetration of solar radiation, including biologically harmful ultraviolet-B radiation. Unfortunately, little is known about the effect of UV-B on individual diatom species and there is no long term DOC record for Lake Opeongo covering the appropriate time period.

Another possible cause of eutrophication is trophic cascade effects triggered by the introduction of cisco in 1948. After the addition of cisco, a planktivore which filled an underutilized niche in highly oligotrophic and nutrient limited Lake Opeongo (Matuszek et al., 1990), a shift in the diatom community away from the oligotrophic species and towards more mesotrophic species would be expected. The increased planktivory by the cisco is predicted to shift the zooplankton towards smaller species with higher nutrient recycling (Carpenter and Kitchell, 1993; Carpenter et al., 1985). The photic zone often must rely on rapid phosphorus recycling to sustain productivity as it is isolated from underlying nutrient-rich waters by distinct thermoclines except at turn-over events (Hudson et al., 2000). The diatom community does show the predicted increase in nutrient availability in response to this event starting at 8.0 cm. Given Lake Opeongo's oligotrophy and nutrient limitations, any increase in nutrient availability from increased recycling could have a marked impact on the diatom community. Further work could help determine if trophic cascade changes had an effect on the diatoms. Zooplankton fossils, including those of cladocera, copepods and *Chaoborus*, preserve well in sediment and can be used to determine if there is the predicted shift towards smaller zooplankton in response to increased planktivory from cisco. Fossil algal pigments also preserve well in the sediments and could be examined to determine the overall changes in phytoplankton composition (Leavitt et al., 1989). Paleolimnological reconstructions using

diatoms suffer from the fact that diatoms can only be used for inferences about the photic zone. A more complete investigation could also include examining the sediments for chironomid remains which would give information about deepwater oxygen and TP availability.

In conclusion, before European settlement, Lake Opeongo was highly oligotrophic, with the diatom community consisting primarily of *Cyclotella stelligera* complex with subdominants *Tabellaria flocculosa* IIIp and *Aulacoseira distans*. The diatom-inferred paleolimnological record showed no changes when the dam was built linking the three original lakes, during forest clearance or during the first half of this century. A significant change occurs ca. 1956 when the diatom community shifted to the current more meso-eutrophic one consisting of *Asterionella formosa* with lesser amounts of *C. bodanica var lemanica, C. stelligera* complex, *Fragilaria crotonensis* and *T. flocculosa IIIp*. This shift could have occurred due to increased direct human impacts on the watershed; increased North American post-war fertilizer use; global warming, including changes in the thermal conditions of the lake; and trophic level changes caused by human manipulation of the fish community of the lake. Further studies including analyses of the zooplankton proxy record as well as nitrogen isotope records could help to identify the primary cause of this apparent eutrophication trend.

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Appendices

Appendix I: ^{210}Pb dating results. Precision – 1 standard deviation of the activity, CRS – Constant rate of supply, SAR – sediment accumulation rate, STD – standard deviation, @bgnd – at background.

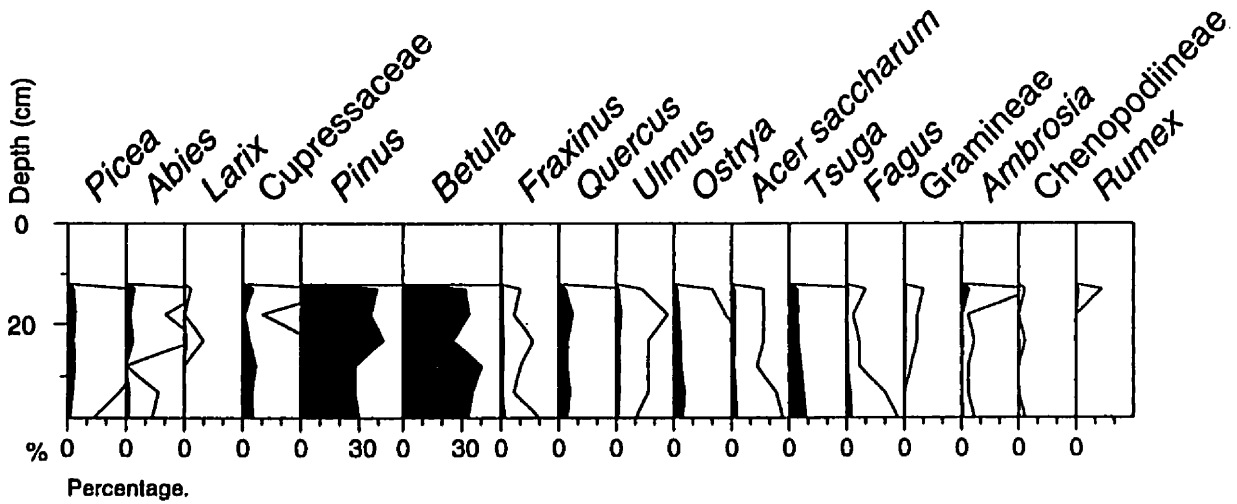
Sample #	Section top (cm)	Section bottom (cm)	^{210}Pb activity (Bq/g)	Precision 1 STD (%)	Age at top of section	CRS SAR ($g/m^2/yr$)	CV in SAR (%)	STD in date (years)
1	0	1	2.214	1.2	2000	233	2.6	0.0
2	1	2	2.484	3.2	1997	194	5.0	0.1
3	2	3	2.067	1.2	1993	205	2.6	0.2
4	3	4	2.203	1.2	1989	167	2.6	0.3
5	4	5	1.364	1.3	1980	210	2.8	0.5
6	5	6	0.958	2.2	1972	232	3.8	1.0
7	6	7	0.827	1.3	1964	211	2.7	1.0
8	7	8	0.594	1.5	1962	277	3.0	1.1
9	8	9	0.485	1.8	1956	282	3.4	1.5
10	9	10	0.425	1.7	1949	266	3.3	1.6
11	10	11	0.342	1.7	1944	281	3.3	1.8
12	11	12	0.292	3.6	1938	276	5.9	3.6
13	12	13	0.240	4.8	1930	270	7.8	5.4
14	13	14	0.222	6.4	1924	245	10.2	7.7
	14	15			1916	227		
15	15	16	0.158	3.7	1908	219	6.3	5.8
	16	19			1900	239		
16	19	20	0.076	4.6	1856	108	9.2	13.2
	20	23			1832	81		
17	23	24	0.036	6.8	1797	67	24.1	48.7
	24	29			1784	87		
18	29	30	0.021	10.5	@bgnd	@bgnd	@bgnd	@bgnd

Appendix II: Pollen diagram for Lake Opeongo, Ontario. In each sample 300 tree pollen were counted together with pollen from grasses and herbs.

Lake Opeongo

opeongo1.pol March 7, 2000. J. McAndrews, analyst.

Samples from J. St. Jacques.



Appendix III: Relative abundances of common and rare species. Also included are the total valves counted per sample and the combined percentages of the taxa which reached at least 1% on at least one level.

Sample depth (cm)	0	0.25	0.5	0.75	1	1.25	1.5	1.75	2
Achnanthes curtissima	0.00	0.37	0.38	0.59	0.00	0.19	0.00	0.00	0.00
Ach. laterostrata	0.17	0.18	0.00	0.00	0.00	0.00	0.18	0.00	0.00
Ach. levanderi	0.00	0.00	0.00	0.39	0.57	0.00	0.18	0.00	0.00
Ach. minutissima	3.23	4.43	3.75	1.58	1.81	2.24	4.16	1.91	1.31
Ach. ricula	0.00	0.00	0.56	0.39	0.00	0.00	0.18	0.00	0.00
Ach. subatomoides	0.34	0.18	0.56	0.20	0.57	0.93	0.00	1.15	0.20
Ach. suchlandtii	0.17	0.74	0.00	0.20	0.00	0.00	0.00	0.38	0.00
Asterionella formosa	18.72	20.78	15.67	19.43	20.25	18.83	17.25	21.01	24.80
Ast. ralfsii var americana	0.00	0.18	0.00	0.89	0.00	0.00	0.18	0.00	0.00
Aulacoseira alpigena	0.17	0.74	0.00	0.20	0.19	0.19	0.36	0.00	0.80
Au. ambigua	0.68	0.18	0.00	0.99	0.00	0.19	0.72	0.38	0.40
Au. crenulata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. distans	5.28	3.69	7.88	3.75	4.97	3.73	5.60	2.48	2.81
Au. lirata	0.00	0.00	0.19	0.00	0.00	0.00	0.36	0.00	0.00
Au. nygaardii	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. perglabra var floriniae	1.02	0.00	0.00	0.00	2.48	0.00	0.00	0.00	0.00
Au. subarctica	0.17	0.92	0.75	0.39	0.38	0.19	0.90	0.76	0.40
Brachysira garrensis	0.00	0.00	0.00	0.00	0.19	0.00	0.18	0.00	0.00
B. type 1	0.51	0.00	0.38	0.00	0.00	0.00	0.18	0.00	0.00
Cyclotella bodanica var lemanica	6.30	7.20	6.75	3.94	5.54	5.22	9.21	4.20	8.43
C. comensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.00
C. kuetzingiana var radiosa	0.00	0.18	0.19	0.00	0.38	0.19	0.36	0.38	1.81
C. michiganiana	3.91	2.95	3.19	2.56	2.10	4.85	2.53	1.72	1.41
C. stelligera complex	12.09	15.14	14.26	21.10	16.62	13.98	13.73	23.69	17.47
C. tripartita	0.00	0.18	0.00	0.39	0.00	0.75	0.00	0.96	0.40
Fragilaria brevistrata	0.51	0.00	0.94	0.99	0.00	2.24	1.26	0.38	0.00
F. brevistrata var inflata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00
F. capucina	0.26	0.00	1.22	0.79	0.00	0.00	0.54	0.19	0.40
F. capucina var gracilis	0.34	0.65	1.22	0.20	0.38	0.47	1.45	1.15	0.00
F. construens var binodis	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.00	0.00
F. construens var venter	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F. crotonensis	11.49	11.82	10.88	8.38	9.65	9.60	8.58	8.69	7.73
F. girdles	1.53	1.85	0.75	1.58	1.53	1.12	1.45	0.38	2.01
F. nanana	1.79	1.11	1.22	0.99	1.05	1.21	2.08	1.34	2.81
F. pinnata	0.00	0.37	0.00	1.18	0.19	1.30	0.00	0.38	0.00
F. pseudoconstruens	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.00
F. tenera	1.62	0.00	0.47	0.69	1.43	1.68	0.45	0.38	2.01
Navicula arvensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. minima	0.00	0.37	0.00	0.00	0.00	0.00	0.00	0.38	0.00
N. minuscula	0.00	0.92	0.00	0.00	0.00	0.00	0.54	0.00	0.00
N. schassmannii	0.00	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. seminuloides	0.34	0.55	0.00	0.39	0.38	0.00	0.36	0.00	1.00
N. submuralis	0.68	0.18	0.94	0.00	0.00	0.00	0.90	0.00	0.00
N. vitiosa	0.00	0.18	0.38	0.99	0.00	0.37	0.00	0.19	0.00
Rhizosolenia eriensis	3.91	3.14	2.63	4.93	2.48	4.75	3.61	3.63	3.92
Tabellaria fenestrata	0.26	0.37	0.00	0.10	0.00	0.19	0.18	0.00	0.00
T. flocculosa	15.49	14.31	16.04	12.52	11.27	14.91	13.37	14.61	10.04
% Total of (>1%) taxa	90.98	94.09	91.18	90.73	84.62	89.28	91.06	91.69	90.16
Valve count	587.5	541.5	533	507	523.5	536.5	553.5	523.5	498

Sample depth (cm)	2.25	2.5	2.75	3	3.25	3.5	3.75	4	4.25
Achnanthes curtissima	0.00	0.38	0.19	0.00	0.51	0.00	1.14	0.00	0.00
Ach. laterostrata	0.00	0.19	0.00	0.00	0.00	0.19	0.19	0.00	0.38
Ach. levanderi	0.00	0.00	0.00	0.58	0.34	0.00	0.00	0.00	0.00
Ach. minutissima	1.34	1.53	1.71	1.16	0.68	2.43	0.57	0.35	1.91
Ach. ricula	0.00	0.38	0.19	0.00	0.00	0.00	0.00	0.00	0.00
Ach. subatomoides	1.53	0.38	0.19	0.39	0.68	0.56	0.57	0.17	0.00
Ach. suchlandtii	0.00	0.19	0.00	0.00	0.00	0.56	0.00	0.00	0.96
Asterionella formosa	22.70	26.24	19.53	22.41	13.82	13.58	11.10	14.27	10.71
Ast. ralfsii var americana	0.77	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Aulacoseira alpigena	0.00	0.00	0.19	1.75	0.34	0.19	0.00	0.52	0.00
Au. ambigua	0.19	0.00	0.38	0.78	0.51	0.00	0.19	0.35	0.96
Au. crenulata	0.00	0.00	0.00	1.55	0.00	0.00	0.00	0.00	0.00
Au. distans	2.11	1.72	3.03	0.97	3.92	4.31	3.98	3.83	3.63
Au. lirata	0.00	0.00	0.19	0.00	0.00	0.00	0.00	0.00	0.00
Au. nygaardii	0.00	0.00	0.00	0.00	0.00	0.00	1.14	0.00	0.00
Au. perglabra var floriniae	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. subarctica	0.19	0.19	0.19	0.58	0.00	0.56	0.19	1.22	0.38
Brachysira garrensis	0.00	0.00	0.00	0.78	0.00	0.19	0.00	0.52	0.00
B. type 1	0.00	1.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cyclotella bodanica var lemanica	2.49	7.25	6.82	6.60	5.63	3.93	4.55	9.23	5.16
C. comensis	0.00	0.00	0.19	0.00	1.02	0.00	0.00	0.00	0.19
C. kuetzingiana var radiosa	0.19	0.38	0.76	0.78	1.02	0.00	0.19	0.87	0.19
C. michiganiana	1.15	0.57	1.71	2.91	1.71	2.25	1.33	2.26	0.57
C. stelligera complex	21.26	19.47	29.00	26.38	29.52	33.15	30.74	26.63	32.89
C. tripartita	0.77	0.19	0.00	0.00	1.37	0.19	0.00	0.00	0.19
Fragilaria brevistrata	0.57	0.76	0.00	0.00	0.00	1.31	1.71	3.48	0.57
F. brevistrata var inflata	0.00	0.00	0.19	0.00	0.17	0.00	0.19	0.00	0.00
F. capucina	0.77	0.00	0.00	0.00	0.51	0.00	0.76	0.09	0.48
F. capucina var gracilis	2.97	2.48	0.85	0.00	0.51	0.75	0.19	0.00	0.00
F. construens var binodis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F. construens var venter	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F. crotonensis	9.00	10.69	5.59	4.66	8.62	8.90	11.20	7.75	7.84
F. girdles	1.15	0.00	0.57	1.55	2.05	0.75	0.95	1.39	1.53
F. nanana	1.82	3.72	1.42	3.49	1.88	2.72	2.66	2.52	2.96
F. pinnata	0.38	0.00	0.19	0.00	0.68	0.00	1.52	1.04	3.63
F. pseudoconstruens	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F. tenera	0.19	0.29	0.00	0.97	0.09	0.00	0.00	0.52	0.10
Navicula arvensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. minima	0.00	0.00	0.00	0.19	0.00	0.00	0.00	0.00	0.19
N. minuscula	0.00	0.38	0.38	0.00	0.34	0.75	0.00	0.00	0.19
N. schassmannii	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.00
N. seminuloides	0.00	0.19	0.38	0.97	0.00	0.37	0.00	0.87	0.00
N. submuralis	0.38	0.38	0.19	0.00	0.34	2.25	0.00	0.00	0.19
N. vitiosa	1.53	0.57	0.19	0.00	1.54	0.56	0.76	0.00	0.38
Rhizosolenia eriensis	3.45	3.34	3.51	1.26	2.39	2.81	3.42	1.57	6.69
Tabellaria fenestrata	0.00	0.10	0.00	0.10	0.60	0.19	0.00	0.17	0.00
T. flocculosa	10.82	10.78	13.74	7.95	9.64	9.46	9.11	10.10	6.21
% Total of (>1%) taxa	87.74	93.89	91.47	88.75	90.44	92.88	88.52	89.73	89.10
Valve count	522	524	527.5	515.5	586	534	527	574.5	523

Sample depth (cm)	4.5	4.75	5	5.25	5.5	5.75	6	6.25	6.5
Achnanthes curtissima	0.00	0.00	0.39	0.00	0.00	0.19	0.00	0.00	0.39
Ach. laterostrata	0.18	0.37	0.00	0.00	0.19	0.00	0.00	0.00	0.00
Ach. levanderi	0.00	0.00	0.19	0.00	0.38	0.19	0.39	0.00	0.00
Ach. minutissima	1.28	1.65	1.36	2.27	2.44	0.19	1.74	3.75	3.68
Ach. ricula	0.00	0.00	0.39	0.19	0.75	0.00	0.00	0.20	0.39
Ach. subatomoides	0.73	1.10	0.58	0.57	0.94	0.19	0.39	0.59	1.35
Ach. suchlandtii	0.55	0.00	0.78	0.09	0.19	0.78	0.39	0.79	0.97
Asterionella formosa	8.68	9.07	14.06	13.64	13.25	13.29	11.58	8.97	11.12
Ast. ralfsii var americana	0.00	0.00	0.68	0.00	0.00	0.00	0.00	0.00	0.00
Aulacoseira alpigena	0.37	0.37	1.36	0.76	0.19	3.69	1.35	0.00	0.00
Au. ambigua	0.18	0.55	0.19	0.19	0.75	0.58	0.77	1.38	0.39
Au. crenulata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. distans	6.22	3.11	4.46	2.27	3.95	4.27	3.86	3.75	3.09
Au. lirata	0.37	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. nygaardii	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.00
Au. perglabra var floriniae	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. subarctica	2.38	0.55	1.16	1.52	0.94	1.55	0.77	1.97	1.16
Brachysira garrensis	0.00	0.00	0.29	0.00	0.00	0.00	0.19	0.00	0.00
B. type 1	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19
Cyclotella bodanica var lemanica	5.48	6.78	3.69	3.79	1.88	3.30	3.28	3.55	2.13
C. comensis	0.00	0.18	0.00	0.19	0.00	0.19	0.00	0.00	0.00
C. kuetzingiana var radiosa	0.55	0.18	0.00	0.00	0.38	0.00	0.00	0.00	0.00
C. michiganiana	0.91	2.38	1.36	0.38	1.13	1.94	1.35	0.99	0.39
C. stelligera complex	32.72	31.50	30.84	29.92	37.59	30.65	33.20	34.32	39.46
C. tripartita	0.18	0.18	0.58	0.19	0.00	0.00	0.19	0.20	0.00
Fragilaria brevistrata	0.37	0.00	1.36	1.14	0.00	0.19	2.51	2.76	1.35
F. brevistrata var inflata	0.00	0.18	0.00	0.38	0.00	0.39	0.00	0.00	0.00
F. capucina	0.37	0.09	0.00	0.38	0.94	0.39	0.39	0.69	1.06
F. capucina var gracilis	0.09	0.37	0.00	0.00	0.38	0.19	0.00	0.69	0.10
F. construens var binodis	0.00	0.00	0.00	0.19	0.00	0.00	0.00	0.00	0.39
F. construens var venter	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F. crotonensis	8.68	7.05	5.63	7.77	5.45	7.66	3.67	4.83	4.84
F. girdles	4.39	4.76	3.49	3.22	0.75	0.39	1.35	1.18	1.93
F. nanana	3.11	2.29	4.17	2.75	3.48	2.62	2.03	2.96	1.55
F. pinnata	0.55	0.00	0.00	0.19	0.19	2.13	0.00	0.79	0.19
F. pseudoconstruens	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.00
F. tenera	0.18	0.00	0.58	0.00	0.19	0.00	1.25	0.10	0.00
Navicula arvensis	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.00
N. minima	0.73	0.18	0.00	0.19	0.38	0.00	0.00	0.00	0.19
N. minuscula	0.18	0.00	0.00	0.38	0.94	0.19	0.00	0.00	0.39
N. schassmannii	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.00	0.00
N. seminuloides	0.00	0.55	0.00	0.00	0.19	0.19	1.35	0.00	0.39
N. submuralis	1.28	0.92	0.00	1.14	0.56	0.58	0.00	0.59	2.13
N. vitiosa	0.55	1.65	0.78	0.38	0.00	0.58	0.77	0.39	0.00
Rhizosolenia eriensis	2.74	6.04	3.59	4.73	2.91	5.72	3.09	3.55	3.09
Tabellaria fenestrata	0.27	0.64	0.19	0.09	0.28	0.97	0.19	0.49	0.10
T. flocculosa	6.31	8.15	7.27	8.90	8.46	8.83	9.17	8.68	8.03
% Total of (>1%) taxa	90.77	90.84	89.43	87.78	90.23	92.53	85.23	88.17	90.43
Valve count	547	546	515.5	528	532	515.5	518	507	517

Sample depth (cm)	6.75	7	7.25	7.5	7.75	8	8.25	8.5	9
Achnanthes curtissima	0.19	0.00	0.00	0.35	0.00	0.00	0.37	0.00	0.00
Ach. laterostrata	0.58	0.00	0.00	0.18	0.10	0.00	0.19	0.00	0.37
Ach. levanderi	0.00	0.58	0.00	0.35	0.00	0.20	0.56	1.18	0.56
Ach. minutissima	1.94	2.14	3.21	3.51	1.73	2.25	2.25	3.94	2.79
Ach. ricula	0.00	0.00	0.00	0.35	0.00	0.00	0.00	0.59	2.05
Ach. subatomoides	0.00	0.39	0.76	1.58	1.15	0.39	0.94	0.00	0.37
Ach. suchlandtii	0.19	0.00	0.38	0.70	1.15	0.78	1.12	0.59	0.19
Asterionella formosa	12.69	4.38	9.07	4.82	7.31	3.52	3.65	3.15	2.33
Ast. ralfsii var americana	0.00	0.39	0.00	0.00	0.00	0.78	0.00	0.00	0.19
Aulacoseira alpigena	0.39	0.78	1.32	0.53	0.96	1.76	0.37	0.59	0.37
Au. ambigua	0.77	1.56	0.95	0.88	0.77	1.56	0.75	1.18	0.74
Au. crenulata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. distans	3.49	4.87	4.73	2.80	3.08	4.49	6.18	4.13	4.66
Au. lirata	0.00	0.00	0.19	0.00	0.00	0.00	0.00	0.00	0.00
Au. nygaardii	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. perglabra var floriniae	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. subarctica	0.97	0.78	1.13	0.88	0.67	2.34	1.69	2.17	0.00
Brachysira garrensis	0.00	0.58	0.00	0.00	0.00	1.07	0.00	0.00	0.00
B. type 1	0.00	0.00	0.00	0.18	0.00	0.00	0.00	1.18	0.37
Cyclotella bodanica var lemanica	3.68	1.75	3.59	1.58	2.50	1.76	0.75	0.59	0.56
C. comensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. kuetzingiana var radiosa	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. michiganiana	0.58	0.19	1.13	0.70	0.96	0.98	0.94	0.59	0.74
C. stelligera complex	34.67	43.43	38.94	45.22	45.43	42.97	47.75	43.50	48.04
C. tripartita	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria brevistrata	1.16	1.17	0.38	1.58	3.08	0.78	0.00	0.98	4.10
F. brevistrata var inflata	0.19	0.00	0.19	0.00	0.77	0.00	0.00	0.00	0.00
F. capucina	0.19	0.58	2.27	0.18	1.06	0.39	0.66	0.39	0.37
F. capucina var gracilis	0.00	0.29	0.38	0.00	0.29	0.00	2.25	0.10	0.19
F. construens var binodis	3.49	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F. construens var venter	0.19	0.00	0.00	0.00	0.19	0.00	0.00	0.00	0.00
F. crotonensis	2.81	2.43	3.59	6.49	0.58	2.34	2.25	0.89	1.68
F. girdles	2.71	2.34	1.32	1.58	0.38	4.49	2.43	3.15	2.05
F. nanana	1.31	2.53	0.47	0.00	0.67	0.59	0.75	0.00	0.00
F. pinnata	0.39	0.39	0.09	0.18	0.00	0.39	0.75	0.39	1.12
F. pseudoconstruens	0.00	0.00	0.00	0.00	0.38	0.00	0.00	1.77	0.00
F. tenera	0.00	1.27	0.00	0.26	0.00	0.39	0.19	0.49	1.40
Navicula arvensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. minima	0.00	0.00	0.38	0.00	0.38	0.00	0.00	0.20	0.19
N. minuscula	0.00	0.00	0.00	1.40	0.00	0.00	0.00	0.79	0.19
N. schassmannii	0.39	0.00	0.19	0.35	0.00	0.00	0.00	0.79	0.00
N. seminuloides	0.00	0.00	0.57	0.35	0.19	2.54	0.19	0.00	0.74
N. submuralis	0.00	0.00	0.19	1.23	0.77	0.00	1.12	1.77	0.93
N. vitiosa	0.39	1.17	0.76	1.05	0.77	0.39	1.12	0.98	0.56
Rhizosolenia eriensis	4.75	4.48	2.27	1.93	1.64	1.46	1.59	1.28	0.84
Tabellaria fenestrata	0.58	0.49	0.09	0.09	0.19	0.49	0.19	0.79	0.19
T. flocculosa	9.88	9.35	9.07	10.52	12.03	10.55	11.05	12.99	12.66
% Total of (>1%) taxa	88.57	88.32	87.62	91.76	89.22	89.65	92.04	91.14	91.53
Valve count	516.3	513.5	529	570.5	519.5	512	534	508	537

Sample depth (cm)	9.25	9.5	9.75	10	11	12	13	14	15
Achnanthes curtissima	0.19	1.13	0.19	0.00	0.66	0.00	0.00	1.17	0.00
Ach. laterostrata	0.19	0.00	0.19	0.00	0.00	0.00	0.00	0.39	0.00
Ach. levanderi	0.56	0.19	0.57	0.18	0.00	0.00	0.38	0.00	0.19
Ach. minutissima	5.83	3.01	3.79	2.29	2.48	1.72	4.21	3.11	3.47
Ach. ricula	0.38	0.00	0.00	0.18	0.33	0.38	0.19	0.00	0.00
Ach. subatomoides	0.38	0.19	0.76	0.37	0.83	0.38	0.77	0.78	1.16
Ach. suchlandtii	0.38	1.32	1.14	0.00	0.17	1.34	0.38	0.78	0.97
Asterionella formosa	5.08	3.20	4.08	4.58	4.37	3.54	1.63	1.84	2.12
Ast. ralfsii var americana	0.00	0.00	0.00	0.09	0.17	0.00	0.10	0.29	0.10
Aulacoseira alpigena	0.94	0.38	0.76	0.18	0.50	0.57	2.11	2.33	0.77
Au. ambigua	2.26	2.07	0.57	0 92	1.16	1.15	0.00	0.39	1.93
Au. crenulata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. distans	2.26	6.59	3.41	6.42	6.27	3.63	9.77	13.01	8.88
Au. lirata	0.00	0.00	0.00	0.37	0.33	0.00	0.00	0.00	0.00
Au. nygaardii	0.38	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.19
Au. perglabra var floriniae	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Au. subarctica	0.94	0.94	1.90	0.92	1.98	0.76	2.49	2.72	1.74
Brachysira garrensis	0.00	0.00	0.00	0.00	0.17	0.38	0.00	0.19	0.00
B. type 1	0.00	0.38	0.00	0.00	0.00	0.00	0.19	0.97	0.00
Cyclotella bodanica var lemanica	2.63	2.45	0.95	1.65	2.48	1.15	1.53	3.50	0.97
C. comensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. kuetzingiana var radiosa	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. michiganiana	0.56	0.38	0.95	0.73	1.49	1.15	1.72	0.78	0.39
C. stelligera complex	38.16	37.29	40.76	33.36	33.83	46.08	39.27	24.66	36.68
C. tripartita	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria brevistrata	0.85	0.00	0.57	4.95	1.49	2.96	0.96	4.08	1.35
F. brevistrata var inflata	1.88	0.00	0.95	0.00	0.00	0.00	0.00	0.00	0.00
F. capucina	0.38	0.38	0.47	0.00	0.00	0.38	0.00	0.39	0.19
F. capucina var gracilis	2.07	0.19	1.04	0.18	0.66	0.57	0.38	0.00	0.00
F. construens var binodis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F. construens var venter	0.19	0.19	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F. crotonensis	1.22	2.26	0.38	0.37	0.41	0.38	0.77	0.78	0.87
F. girdles	1.50	2.64	1.71	3.12	0.99	1.15	3.07	1.94	2.51
F. nanana	1.60	0.19	1.14	0.55	0.00	0.76	0.00	0.39	0.48
F. pinnata	0.75	0.75	1.90	0.00	0.17	0.57	0.38	0.39	0.58
F. pseudoconstruens	0.00	0.00	0.00	0.00	1.16	0.19	1.15	0.00	2.32
F. tenera	0.00	0.47	0.09	1.56	0.58	0.67	1.34	1.07	0.97
Navicula arvensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
N. minima	0.00	1.32	0.00	0.00	0.99	0.00	0.57	0.19	0.00
N. minuscula	0.00	1.32	0.00	0.00	0.00	0.00	0.96	1.55	0.00
N. schassmannii	0.75	1.13	0.57	0.00	0.00	0.00	0.00	0.00	0.00
N. seminuloides	0.38	0.38	0.38	0.37	0.66	1.72	0.57	1.17	0.58
N. submuralis	0.75	1.69	0.19	1.28	2.15	0.00	1.72	0.58	0.58
N. vitiosa	0.75	0.56	0.57	0.00	1.65	0.00	2.30	0.97	0.00
Rhizosolenia eriensis	0.56	0.94	0.95	0.64	1.16	1.24	0.96	0.78	1.35
Tabellaria fenestrata	0.09	0.47	0.47	0.55	0.50	0.38	0.19	1.17	0.48
T. flocculosa	14.85	15.35	15.55	21.45	18.32	15.30	13.03	18.16	19.11
% Total of (>1%) taxa	89.66	89.74	86.92	87.26	88.70	88.53	93.10	90.49	90.93
Valve count	532	531	527.5	545.5	606	523	522	515	518

Sample depth (cm)	16	18	20	22	24	26	28	30	32
Achnanthes curtissima	0.19	0.00	0.39	1.76	0.74	0.00	0.18	0.00	0.00
Ach. laterostrata	0.00	0.00	0.00	0.39	0.00	0.38	0.18	0.19	0.00
Ach. levanderi	0.38	0.00	1.16	0.78	0.37	1.15	0.36	0.77	0.39
Ach. minutissima	3.96	3.73	4.05	3.90	5.72	2.50	5.09	4.45	4.33
Ach. ricula	0.38	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.00
Ach. subatomoides	0.38	0.00	1.54	0.78	0.37	0.00	0.73	1.16	1.18
Ach. suchlandtii	2.07	0.59	1.93	1.95	0.00	1.15	0.55	0.39	0.59
Asterionella formosa	1.32	2.16	4.44	1.17	1.29	2.21	1.27	1.55	0.89
Ast. ralfsii var americana	0.00	0.79	0.29	1.56	1.75	0.38	0.55	0.48	0.49
Aulacoseira alpigena	1.51	0.98	0.39	0.98	0.55	2.12	2.73	2.32	3.15
Au. ambigua	0.94	1.57	1.35	1.37	2.59	0.96	1.27	0.39	0.79
Au. crenulata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00
Au. distans	4.34	9.63	3.28	4.10	9.97	5.77	12.91	6.00	7.09
Au. lirata	0.38	0.00	0.00	0.20	0.00	0.19	0.00	0.00	1.18
Au. nygaardii	0.00	0.79	0.00	0.00	0.00	0.19	1.82	0.00	0.39
Au. perglabra var floriniae	0.38	0.39	0.00	0.00	0.00	0.00	0.00	0.00	0.39
Au. subarctica	0.38	0.79	1.54	1.76	0.55	0.58	2.18	1.94	1.77
Brachysira garrensis	0.19	0.00	0.19	0.00	0.18	0.58	0.91	0.00	0.00
B. type 1	0.75	0.00	0.00	0.00	0.00	0.00	0.55	0.00	0.00
Cyclotella bodanica var lemanica	0.94	1.57	1.93	0.98	0.92	0.58	1.09	3.10	1.58
C. comensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. kuetzingiana var radiosa	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C. michiganiana	0.19	1.38	0.77	0.78	1.29	1.73	1.27	0.00	0.59
C. stelligera complex	40.72	42.44	38.61	35.12	36.57	42.12	29.82	37.37	37.83
C. tripartita	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria brevistrata	3.77	4.72	3.28	2.34	2.77	0.58	3.09	3.10	1.58
F. brevistrata var inflata	0.00	0.20	0.00	0.20	0.00	0.00	0.00	0.58	0.00
F. capucina	2.07	0.59	0.19	3.02	0.55	0.77	0.00	1.36	0.59
F. capucina var gracilis	0.47	0.39	1.06	0.10	0.09	2.02	0.00	0.19	0.00
F. construens var binodis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
F. construens var venter	0.00	0.00	0.00	0.00	0.00	1.15	0.00	0.00	0.00
F. crotonensis	2.36	1.38	0.58	0.00	1.29	0.58	0.00	1.84	0.00
F. girdles	3.02	0.39	0.97	1.56	1.85	0.58	1.27	0.39	0.79
F. nanana	0.00	0.00	0.29	0.00	0.37	0.00	1.00	0.00	0.49
F. pinnata	0.38	0.20	0.19	6.05	0.00	2.31	0.36	0.58	1.58
F. pseudoconstruens	0.00	0.00	0.77	0.00	0.00	0.00	0.00	0.00	0.00
F. tenera	0.75	0.10	0.10	0.29	0.55	0.38	0.91	0.19	0.49
Navicula arvensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.20
N. minima	0.57	0.98	0.39	0.00	0.00	0.00	0.00	0.19	0.00
N. minuscula	0.75	0.00	0.58	0.00	0.55	0.00	1.27	0.00	0.59
N. schassmannii	0.19	0.39	0.00	0.20	0.00	0.38	0.36	0.00	0.20
N. seminuloides	0.19	0.98	0.58	0.98	0.37	0.19	1.27	0.58	0.39
N. submuralis	1.89	1.18	0.97	2.15	0.74	1.15	1.09	2.90	1.58
N. vitiosa	1.32	0.00	0.19	1.17	0.00	1.35	0.00	2.32	0.59
Rhizosolenia eriensis	1.60	0.88	0.97	1.07	1.20	1.06	0.45	0.48	0.89
Tabellaria fenestrata	0.28	0.39	0.29	0.20	0.37	0.10	0.18	0.19	0.10
T. flocculosa	9.43	8.94	12.45	11.51	11.45	12.69	14.27	9.39	13.99
% Total of (>1%) taxa	88.41	88.51	85.71	88.39	85.04	88.27	89.00	84.61	86.70
Valve count	530.5	509	518	512.5	541.5	520	550	516.5	507.5

Sample depth (cm)	34	36	38	40	42
Achnanthes curtissima	0.00	0.00	0.00	0.00	0.57
Ach. laterostrata	0.00	0.00	0.00	1.56	0.38
Ach. levanderi	0.00	0.58	0.00	0.39	1.13
Ach. minutissima	3.24	2.13	4.45	1.17	2.08
Ach. ricula	0.00	0.58	0.37	0.00	0.00
Ach. subatomoides	0.00	1.35	1.11	0.39	0.76
Ach. suchlandtii	0.57	0.77	0.56	0.98	0.38
Asterionella formosa	1.53	1.93	1.85	1.76	1.70
Ast. ralfsii var americana	0.57	0.29	0.19	0.78	0.00
Aulacoseira alpigena	3.62	3.48	3.15	1.17	0.76
Au. ambigua	1.53	1.35	0.56	1.17	1.13
Au. crenulata	0.00	0.00	0.00	0.00	0.19
Au. distans	9.53	8.89	5.93	4.10	6.80
Au. lirata	0.19	0.00	0.00	0.20	0.38
Au. nygaardii	0.19	0.00	0.19	0.00	0.00
Au. perglabra var floriniae	0.00	1.74	0.00	0.00	0.00
Au. subarctica	0.57	0.97	0.74	0.00	2.83
Brachysira garrensis	0.00	0.00	0.00	0.00	0.00
B. type 1	0.00	0.19	0.00	0.00	0.00
Cyclotella bodanica var lemanica	0.95	2.13	1.85	0.98	2.08
C. comensis	0.00	0.00	0.00	0.00	0.00
C. kuetzingiana var radiosa	0.00	0.00	0.00	0.00	0.00
C. michiganiana	0.95	0.39	0.56	0.78	1.32
C. stelligera complex	44.42	36.52	37.44	50.98	37.20
C. tripartita	0.00	0.00	0.00	0.00	0.00
Fragilaria brevistrata	0.00	0.00	2.04	1.95	1.13
F. brevistrata var inflata	0.00	0.00	0.00	0.00	0.00
F. capucina	1.24	0.29	0.83	0.00	1.32
F. capucina var gracilis	0.38	0.39	0.56	0.00	0.28
F. construens var binodis	0.00	0.00	0.00	0.00	0.00
F. construens var venter	0.00	0.00	0.00	0.00	0.38
F. crotonensis	0.38	1.55	0.00	0.78	0.00
F. girdles	0.38	0.00	1.85	1.56	2.64
F. nanana	0.00	0.00	0.00	0.20	0.09
F. pinnata	0.95	0.00	0.00	0.00	0.00
F. pseudoconstruens	0.00	0.00	0.00	0.20	2.27
F. tenera	0.00	2.13	0.37	1.76	0.47
Navicula arvensis	0.00	0.00	0.00	1.95	0.00
N. minima	0.19	0.19	0.37	0.00	0.19
N. minuscula	0.00	1.16	0.00	0.00	0.00
N. schassmannii	0.38	0.39	0.56	0.20	0.19
N. seminuloides	0.95	1.16	0.00	0.78	0.19
N. submuralis	1.33	1.16	2.78	2.15	2.64
N. vitiosa	0.00	0.77	3.15	0.00	1.70
Rhizosolenia eriensis	1.33	0.77	1.02	1.46	0.66
Tabellaria fenestrata	0.00	0.77	0.00	0.49	0.09
T. flocculosa	11.82	11.30	16.87	6.84	15.77
% Total of (>1%) taxa	87.23	85.31	89.34	86.72	89.71
Valve count	524.5	517.5	539.5	512	529.5

Appendix IV: Total Phosphorus (TP) measurements from Lake Opeongo. Average computed using all values over the 10 years. Courtesy of the Algonquin Fisheries Assessment Unit, Ontario Ministry of Environment Water Quality Analysis Laboratory. Details about measuring methods, sites and dates unavailable.

Year	Month/Day	TP (mg/L)	Year	Month/Day	TP (mg/L)
1980	May 7	0.007	**1985**	Mar. 19	0.008
	Aug. 16	0.005		May 7	0.007
	Oct. 22	0.005		July 3	0.006
1981	Mar. 2	0.004		Aug. 27	0.005
	Apr. 28	0.009		Oct. 21	0.012
	Aug. 6	0.005	**1986**	Feb. 25	0.007
	Sept. 15	0.007			
	Oct. 29	0.022	**1988**	Mar. 23	0.008
1982	Mar. 18	0.005		May 3	0.007
	May 4	0.032		June 22	0.006
	June 29	0.006		Aug. 30	<0.002
	Oct. 27	0.007		Oct. 26	<0.004
1983	Feb. 14	<0.003	**1989**	Mar. 21	0.006
	May 9	<0.004		May 8	0.006
	July 6	0.005		June 27	0.006
	Aug. 30	0.005		Aug. 22	0.004
	Oct. 26	0.008		Nov. 7	0.006
1984	Mar. 8	0.002			
	May 11	0.010		**Total**	
	July 10	0.036		**Average**	0.008
	Aug. 21	0.008			
	Oct. 30	0.008			