ASSESSMENT OF ALGAL AND ELEMENTAL CHANGES IN THREE KAWARTHA LAKES: A PALEOLIMNOLOGICAL ASSESSMENT (June 2018)

Prepared for: Kawartha Lake Stewards Association (KLSA)

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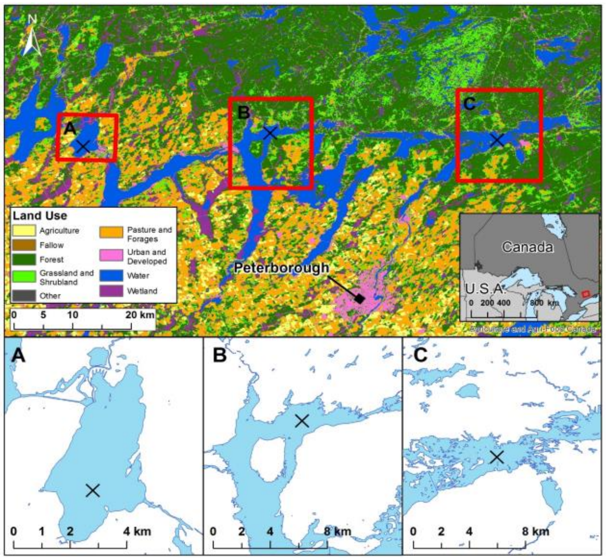

ACKNOWLEDGEMENTS:

We would like to thank: Dr. Frost (Trent University) for his assistance with lake selection; Dr. Peter Leavitt and Deirdre Bateson for analyzing the sediments for pigments; K. Moir for undertaking a brief literature search on the behavior and sources on Mn in water and sediments; B. Tregunno (Aquatic Biologist at Kawartha Conservation), W. Napier and M. Dolbey for constructive comments on an earlier version of document; and to W. Napier for initiating, helping design and for managing this project. Funding for this project was arranged by the Kawartha Lake Stewards Association (KLSA) through the Stony Lake Heritage Foundation, the City of Kawartha Lakes, Kawartha Conservation and private donors.

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EXECUTIVE SUMMARY

Eutrophication of lakes is one of the most pervasive issues impacting freshwater ecosystems around the world. The Kawartha Lake Stewards Association (KLSA) has long been concerned with changes in lake nutrients that could result in increased algal production and undesirable algal blooms, including cyanobacteria (blue-green algae) or other nuisance algae. This study was developed to assess changes in algal production over the past several hundred years to establish baseline conditions and variability in algal production prior to European settlement and recent trends over the last ~100 years (Cumming and Napier, 2017). The paleolimnological data was also put into the context of total phosphorus (TP) measurements collected through the Lake Partner Program from 2002-2016 AD.

Three lakes were selected for this study: Cameron Lake, Pigeon Lake and Stony Lake. Sediment cores were retrieved from each lake on May 10th or 11th in 2016. Analysis of the radioisotopes in the sediment cores allowed the establishment of sediment chronologies for the past century. Time-depth profiles of the cores for ages greater than ~100 years are based on extrapolation, and therefore only provide only a coarse chronological framework that needs to be interpreted with caution. Indicators of changes in algal production in this study were based on the species composition and abundance of diatoms in the cores and changes in the concentration of algal pigments preserved in the sediment cores.

Changes in the sediment cores are summarized by diatom-based estimates of TP. The post-2000 AD diatom-based estimates of TP were compared to the Lake Partner dataset of TP measurements from 2002-2016 to provide an evaluation of the accuracy of the paleo-inferred estimates of TP. The calibration dataset for the diatoms is based on summer water chemistry. The diatom-inferred TP for Cameron Lake falls within the average summer TP measurements. For sediment cores for Pigeon and Stony lakes, the average post-2000 diatom-inferred TP values are higher than the average peak summer measurements, but fall within the maximum values for both of these lakes. The correspondence of diatom-based TP estimates with the measured values provides a valid technique to provide a longer-term context of changes in nutrients in these lakes and baseline conditions of pre-settlement conditions that can be used as realistic remediation targets. However, changes in diatom assemblages are also influenced other environmental variables, that often result in an interplay between a number of factors, including nutrient and climate interactions.

Early changes in lake-water production

Early European settlement in the Kawartha area began in 1818 AD, with extensive logging and agricultural cultivation occurring in the mid-to-late 1800s, such that by 1913 AD the region was described as barren from

clear cuts or cultivated (Conolly, 2017). Additionally, damming for movement of logs and for the development of the Trent-Severn Waterway was also occurring in the early-to-mid 1800s that increased lake levels and the potential area for littoral habitat. These multiple disturbances would be stressors on both the terrestrial and aquatic environments. Remedial action activities in the 1920s and 1930s resulted in the regrowth of forests within the Kawartha region (Conolly, 2017). In Cameron Lake, a distinct increase in eutrophic planktonic diatoms occurred in the 1800s with diatom-based estimates of total phosphorus (TP) increasing from an average of ~14 µg/L to ~20 µg/L. Post ~1940 AD estimates of TP decline, potentially initially related to regrowth of the forests. The drainage area of Cameron Lake is mostly in the north and thus regrowth of forests could affect the overall nutrient load to the lake. In Pigeon Lake, a similar increase in eutrophic planktonic diatoms occurred during the 1800s that also included an increase in hypereutrophic planktonic diatoms. Diatom-based estimates of total phosphorus (TP) increased from < 15 µg/L to ~24 µg/L by the mid-1930s. Unlike Cameron Lake, there was no decline in TP post ~1940 AD, coincident with reforestation in the region, but rather TP estimates remained relatively high and stable until ~1990 AD. The drainage area of Pigeon Lake contains a larger portion of the southern agricultural region than the other two lakes, thus any potential influence of forest regrowth may have been offset by inputs from agriculture. The Stony Lake sediment record has a somewhat different history, with increases in eutrophic planktonic diatoms much earlier (deeper in the sediments), with diatom-based estimates of TP increasing from an average of ~17 µg/L to ~21 µg/L. The TP estimates further increased to ~25 µg/L in the mid-to-late 1800s, where it remained around 23-25 µg/L until the early 2000s. Although the majority of the drainage basins of Stony Lake are in the north, the watershed is overall much smaller than the other two study lakes.

Recent (post-1990 AD) changes in lake-water production

All lakes indicated increases in algal production and alteration of the algal community in the post ~1990 AD sediments. The recent alterations in the lakes are most probably being driven by the interaction of nutrients and climate, but timing also corresponds to when zebra mussels invaded this region. However, the changes in the diatom species seen in the sediment records are not easily be related to the establishment of the zebra mussels. There is some potential that changes in the other algal groups, particularly increases in blue-green algae may in part be related to the zebra mussels. However, recent increases in temperature and resultant changes to the aquatic environment could have resulted in enhanced algal production and associated shifts in the composition of the phytoplankton community. Eutrophication of lakes is also often characterized by both increased pelagic (planktonic) production and changes to the makeup of the phytoplankton community. Both climate warming and nutrients can result in conditions that may be more conducive to cyanobacteria (blue-green algal) blooms. The study lakes vary in their susceptibility to changes in nutrients, and also would vary in their response to climate forcing depending on changes occurring in the watersheds and morphological features of the lakes. One of the most dramatic changes to temperate lakes with warming climate is a decrease in the duration of ice cover, which affects the physical, chemical and biological seasonal dynamics of lakes, and can thereby have profound impacts on the lake ecosystems, in addition to the warming water and any nutrient inputs from the watershed. Similar to many temperate lakes around the world, shorter periods of ice cover have been noted for the lakes in the Kawartha region (Monkman, 2015).

A major conclusion of this study is that the diatom species that comprise the post 1990 AD sediments of all three lakes consist of a unique composition of species that include species that were rare prior to ~1990 AD. Whether these changes are primarily related to nutrient changes (influenced by changes in nutrient pathways related to changes in nutrient inputs, zebra mussels and internal phosphorus loading) or other factors more related to climate is difficult to discern. It is likely that both changes in diatom assemblages are related to both nutrients and climate. The analysis of the pigments preserved in the sediments also indicates changes to the overall algal composition. In all lake cores there is evidence of recent increases in cyanobacteria (blue-green algae). Increases in cyanobacteria pigments are most pronounced in Pigeon and Stony lakes, but the pigment history of both lakes suggests these lakes have been conducive to having cyanobacteria blooms prior to European settlement. Cyanobacteria pigments are also present in the Cameron Lake prior to settlement but at lower concentrations.

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INTRODUCTION

Cultural eutrophication is one of the most pervasive environmental issues impacting freshwater ecosystems, and is normally the result of increases in nutrient loading to aquatic systems as the result of human activities. The ramifications of cultural eutrophication include increases in algal production and biomass, including potentially toxic cyanobacteria. In order to assess how nutrients in lakes have changed over time, and if the changes now exceed the range of natural variation, a long-term perspective spanning decades to centuries is required. Modern data collection (i.e. monitoring) provides a snapshot of trends since the onset of monitoring programs. The use of proxy data to infer changes in production from sediment cores provides a longer-term perspective of on the variability and timing of changes in lake production (Napier and Cumming, 2017). The analysis of long-term trends from lake sediment cores forms the basis of paleolimnology, a science that uses the physical, chemical and biological information preserved in sediment cores to reconstruct past ecological and environmental conditions. The use of both recent monitoring and paleoecological techniques provides insights on the dynamics of variation in lake production within the Kawartha Lakes, and if the lakes have become more productive in comparison with the variation that these lakes experienced in the past.

In the Kawartha region, lake-water phosphorus concentrations generally increase from west to east, as water flows from Balsam Lake to Lovesick, except when nutrient-poor water enters from Upper Stony Lake. Changes in phosphorus loading from the watershed and increases in algal production are normally associated with land-use practices. However, changes in climate can also alter algal production and assemblage composition with influences of warmer conditions, longer growing seasons, and changes in precipitation (e.g. Karmakar et al 2015).

The Kawartha Lake Stewards Association (KLSA) has long been concerned with changes in lake nutrients that could result in enhanced amounts of algal production that could lead to undesirable algal blooms. This study was initiated in early 2016 by the KLSA in conjunction with Dr. Brian Cumming at Queen's University and Dr. Paul Frost at Trent University. The main focus of this study was to analyze indicators preserved in lake sediment cores, to infer how lake production has changed over the past ~100 to 150 years in three lakes that were selected across the Kawartha region including Cameron Lake (A), Pigeon Lake (B) and Stony Lake (C) (Fig. 1). The length of the records analyzed was several hundred years. This long time frame provides a clear indication of how much variability occurred under natural conditions and what baseline conditions were prior to European settlement.

Indicators of changes in lake production of primary producers were reconstructed from analysis of algal indicators in the sediment cores from: changes in the species composition and abundance of diatom assemblages, one of the most widely accepted techniques for tracking changes in lake-water nutrients (Cumming et al., 2015); and changes in the concentration of algal pigments present in the sediments (Leavitt & Hodgson 2001).

STUDY SITES AND RECENT PHOSPHORUS MEASUREMENTS

Lakes were selected so that a gradient in production was covered, and that the lakes of interest would have characteristics from which relatively undisturbed lake sediments could be retrieved. Based on bathymetric maps (i.e. contours of the lake bottom), coring locations were identified where relatively undisturbed sediments may be present. Three lakes were cored (Fig. 1): Cameron Lake (A), an upstream lake with values of total phosphorus (TP) typically < 10-12 µg/L, but with concentrations as high as ~15 µg/L; Pigeon Lake (B), downstream of Cameron and Sturgeon lakes, a lake with seasonal TP values typically between 10 and 22 µg/L, but with measured TP values reaching ~25-30 µg/L; and Stony Lake (C), located downstream of Pigeon, Buckhorn and Lower Buckhorn, with seasonal TP values typically between 10 and 20 µg/L, with measured TP values exceeding 25 µg/L.

Table 1. Total phosphorus values are based on the Lake Partner dataset. Average TP is based on all Lake Partner sites analyzed (Fig. 1) which can vary between sampling sites, particularly in Pigeon and Stony lakes. Lake physical data is from White (2006)

Lake Name	Average TP (range) µg/l 2002-2016	Lake Order	Lake Area (ha)	Max Depth (m)	Mean Depth (m)
Cameron	10.8 (7.1-14.9)	2	1303	18.3	6.3
Pigeon	16.2 (6.9-31.6)	3	5344	17.4	3
Stony	14.4 (6.4-33.8)	3	2825	32	5.9

Lake partner total phosphorus measurements

To provide some context to the paleo-study, total phosphorus measurements were downloaded from the Lake Partner program dataset, which is coordinated by the Ontario Ministry of the Environment and Climate Change since 2002 through the Dorset Environmental Science Centre (DESC). Lake-water samples were collected by volunteers, and analyzed for TP by the DESC Water Chemistry Laboratory. Data was available for all three of the study lakes from 2002-2015 for Cameron Lake and 2002-2016 for Pigeon and Stony lakes. Data from five sampling sites for Cameron Lake consisted of 38 samples, but seasonal data was only available for five years (2002, 2003, 2005, 2006 and 2013). Data from Pigeon Lake was downloaded from 7 sites with a total of 378 samples and Stony Lake from 4 sampling sites for a total of 276 samples. Both of these sites had multiple years of seasonal data, although the years and consistency of sampling every month from May to October varied (see Supplemental Table S1). Other sources exist for TP measurements, such as the Provincial Water Quality

Monitoring Network (PWQMN) and government reports, For this report, we focused on the easily accessible Lake Partner dataset with consistent methodology and a focus on lake sampling, as opposed to the PWQMN which focuses on samples from rivers and streams. We also examine earlier TP measurements in a MOE-MNR (1976) report that studied the Kawartha region. While there may be other government reports that have lake data post 1976, it is outside of the scope of this study to do an exhaustive search for such data.

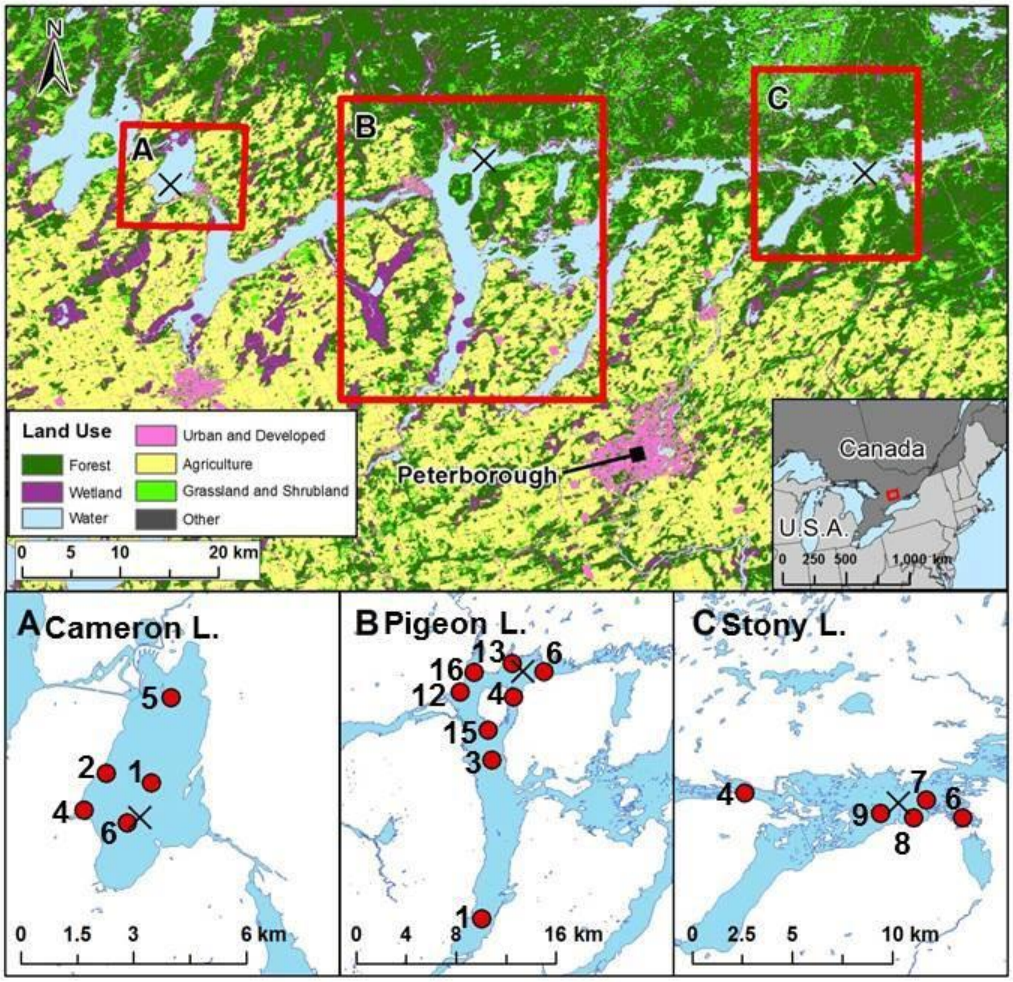


Figure 1. Map showing the location of the three Kawartha study lakes used in this study (A. Cameron, B. Pigeon, C. Stony). Cores were removed from each lake between May 10th and 13th, 2016. The coring location in each of the lakes is marked by an ' X'. Lake partner sites and ID # are labelled by a solid circle.

COLLECTION AND SAMPLING OF LAKE CORES

Sediment cores were collected from ~10-m water depth from Cameron and Pigeon lakes and from ~12-m water depth Stony Lake on May 10th and 11th 2016 by Dr. Brian Cumming, Cecilia Barouillet and Brett Elmslie. The cores were 58, 52 and 56 cm in length respectively and were sectioned into ~0.5-cm intervals. The cores were stored at 4° C in the P.E.A.R.L. coldroom at Queen's University. The weight of all samples was measured to determine the total wet weight of sediment prior to sub-sampling for analyses. Samples were prepared for gamma spectroscopy every 2 cm for the length of the core to provide the data to develop a chronology model for each core. Sediment samples were also sub-sampled for diatoms, pigments, spectral chlorophyll *a*, organic matter and elemental analyses.

PALEOLIMNOLOGICAL METHODS

Chronology models

Chronology for each of the study lakes was developed from gamma analyses (see Napier and Cumming, 2017). The concentration of radioisotopes in sediment samples provides a means to determine when the sediment was deposited and the sedimentation rate over time. The most common technique to date the recent history archived in cores is through the measurement of ^{210}Pb, an isotope with a half-life of ~22 years (i.e. it loses half of its activity every 22 years). Another useful isotope can be ^{137}Cs, which can peak in sediment cores, corresponding to the peak of atmospheric testing of atomic bombs in ~1963, before this practice was banned.

Sediment samples were dried in the freeze drier at PEARL (24 hr. cycle) and the dry weight and percent moisture in each sample was determined. Dried sediment was weighed into a plastic tube for gamma spectroscopy and the weight was recorded. The samples were then sealed with epoxy and allowed to sit for two weeks in order for ^{214}Bi to equalize with ^{226}Ra, the parent isotope of ^{210}Pb. Activities of ^{210}Pb, ^{137}Cs and supported ^{210}Pb (via ^{214}Bi) were determined for each sample using gamma dating following the procedures and similar equipment outlined in Schelske et al. (1994). Unsupported ^{210}Pb activities were used to estimate the chronology of the cores using the constant rate of supply (CRS) model (Appleby & Oldfield 1978) using the MATLAB programme ScienTissME (M. Scheer, unpublished) calibrated to IAEA standards 312 & 385. The chronological models are based on cumulative dry weight of sediment over time, because percent moisture generally declines in an exponential nature with core depth. Gamma analysis is non-destructive, thus after dating models were developed, the samples were used for spectral chlorophyll *a* analysis and finally for estimates of organic matter content, which is a destructive process.

Diatom preparation and enumeration

Samples for diatom analysis were prepared every 2 cm for the length of the cores using standard techniques (Cumming et al., 1995). Briefly, a small amount of wet sediment (~0.2-0.3 g) was suspended in a 50:50 (molar) mixture of sulfuric and nitric acid in a 20-ml glass vial for 24 hr. prior to being submersed at 70˚C in a hot water bath for approximately 7 hr. The remaining sediment material was settled for a period of 24 hr, at which time the acid above the sample was removed. The sample was rinsed with distilled water and allowed to settle once again for 24 hrs. The procedure was repeated approximately 8 times until the sample was acid free (litmus test). The samples were settled onto coverslips in a series of four 100% dilutions, which when dry, were mounted onto glass slides using the high-resolution mounting media Naphrax®. For each sample, at least 400 diatom valves were enumerated with a Leica DMRB microscope equipped with DIC optics at 1000X magnification (Numerical Aperature of objective = 1.3). Dr. Kathleen Laird identified and enumerated the diatom values. These analyses were based primarily on the references of Krammer and Lange-Bertalot (1986, 1988, 1991a, b), Cumming et al. (1995), Lange-Bertalot and Melzeltin (1996), Camburn and Charles (2000) and Fallu et al. (2000).

The concentration of diatoms was determined for all samples analyzed using methods outlined in Battarbee & Keen (1982). Briefly, each sample was spiked with a known concentration of microspheres prior to plating the diatoms onto coverslips. The microspheres were enumerated along with the diatoms and used to calculate estimates of # diatom valves per gram dry weight. Total diatom concentration (#/g dry weight x 10^7) provides a means of assessing whether there were any changes in total diatom production.

Diatom-based reconstructions of total phosphorus

Inferences of total phosphorus (TP) from the diatom assemblages in the core are based on a phosphorus model developed from 268 freshwater lakes from across British Columbia. Lake-water chemistry was collected over the summer months (June to early September) and thus TP inferences best represent summer conditions. This dataset includes lakes from several regions within British Columbia. This model is based on estimates of the optima of taxa from weighted-averaging regression on non-transformed relative percentage data. The coefficient of determination (r^2) of this model is 0.62, and the bootstrapped r^2 is 0.51 (Cumming et al., 2015). Given the large size of this lake data set, the model can be used in other regions as common diatom taxa are cosmopolitan. Estimates of TP from the diatom assemblages in this study were based on those taxa which were present in the cores at an abundance of 1% or greater.

The diatom-based inferences of total phosphorus were critically assessed to determine if they tracked the main directions of variation in the diatom species assemblages. If the diatom-based phosphorus reconstructions match the main direction of variation in the diatom assemblages in the core, then we can be fairly confident that the diatoms are tracking changes that are related to phosphorus, or correlated variables. If the correlation between the main direction of variation and the diatom-inferred phosphorus values is weak or nonexistent, then other environmental variables (e.g., water column stability, water depth, conductivity, turbulence, etc), or interactions between environmental variables, are likely responsible for the observed changes in diatom assemblages.

Determination of the main directions of variation

The main directions of variation in the diatom assemblages in the study lake cores were determined from the axis-1 and 2-scores from a principal components analysis (PCA) using non-transformed species percent abundance data. The PCA scores were based on the same datasets used to calculate the diatom-inferred TP estimates. A PCA is a mathematical analysis that transforms a large set of variables (in this case diatom taxa) into a smaller set, which represents the main trends of the larger dataset. The first axis or principal component accounts for the most variability, with each successive component accounting for the increasingly less variation. PCA was chosen to represent the main directions of variation of the diatom assemblages because the gradient length in an initial detrended correspondence analysis was < 1.5 standard deviation units. Correlations between the estimated total phosphorus (TP) and the first two PCA axes were determined. The strength of the eigenvalues for each PCA axis, which can range from 0 to 1.

Cluster analysis

Cluster analysis provides a means of grouping diatom assemblages that are most similar to each other. The programs, TILIA and TGVIEW 2.02 (Grimm, unpublished), were used to determine similar zones down-core based on the diatom assemblages. The cluster analyses were stratigraphical constrained in order to group the assemblages according to core depth (or core age) using a squared Euclidean similarity coefficient. As with the other analyses, species data was not transformed. Cluster analysis was used to define zones for the diatom assemblages, pigment analyses, as well as the elemental analyses.

Organic matter analysis

Organic matter content in lakes sediments is the result of organic production within the lake and from inputs of terrestrial and wetlands ecosystems. Organic matter content for each of the cores was determined every 2 cm for the length of the cores from the dried sediments used in the gamma analysis, as the latter is a non-destructive process. Gamma tubes were cut open and sediments were initially used for spectral analysis of Chlorophyll *a* (see below), another non-destructive analysis, prior to weighing ~0.15-0.3 g of dried sediment into a dried and weighed crucible. Samples were dried in the muffle furnace at 105 ˚C to ensure all moisture was eliminated. Samples were placed in a desiccator to cool down and then re-weighed to determine dried weight of samples. Percent organic matter was then determined using standard loss-on-ignition (LOI) methods (Heiri et al. 2001). Briefly, the dried sediment (recorded to four decimal places) was heated to 550˚C for 4 hours. The difference between the dry weight of the sediment and the weight of sediment remaining after ignition was used to estimate the percent of organic matter in each sediment sample.

Pigment analysis

Concentration of pigments in the sediment cores was analyzed on 20-22 samples per core spanning the last ~ 200-300 years, in Professor Peter Leavitt's laboratory at the University of Regina. Cameron Lake was sampled every 1 cm for the top 14 cm and then every 2 cm to 26 cm. Pigeon and Stony lakes were sampled every 2 cm for the top 38 cm and 42 cm, respectively.

Sedimentary pigments were extracted from freeze-dried sediments in acetone under an argon atmosphere at −20°C for 24h. During pigment extraction, samples were kept in the dark and on ice at all times to avoid pigment deterioration from light, heat, and oxygen (Leavitt & Hodgson 2001). Samples were then centrifuged, the supernatant was decanted and filtered (0.2 μm), and finally placed into sealed glass vials. Sample extracts were analyzed using a High Performance Liquid Chromatography (HPLC) system equipped with a photo diode array (Waters Model 2996) and a multiwavelength fluorescence detector (Waters Model 2475). The HPLC produces chromatograms in a specific sequence based on the properties of the compounds being analyzed. The resulting retention time and absorbance spectra were compared to purified reference materials to quantify the sedimentary pigments.

Sedimentary pigments

Chlorophyll *a* (Chl*a*)– a general pigment found in plants and algae.

B-Carotene - a general pigment found in plants and algae, and in some phototrophic bacteria.

Alloxanthin – pigment found in all Cryptophyta (i.e. cryptonomads).

Diatoxanthin – pigment found in chrysophytes, diatoms, and dinoflagellates.

Lutein – pigment found in Chlorophyta (green algae), Euglenophyta (green flagellated algae) and plants.

Chlorophyll b - pigment found in Chlorophyta (green algae), Euglenophyta (green flagellated algae) and plants.

Canthaxanthin – pigment found in colonial cyanobacteria (blue-green algae).

Echinenone - pigment found in cyanobacteria (blue-green algae).

Myxoxanthin - pigment found in colonial cyanobacteria (blue-green algae), such as *Microcystis*.

UV index - scytonemin derivatives which arise from cyanobacteria when exposed to damaging levels of high energy irradiance (UV).

Chl*a*/pheoa – indication of pigment preservation.

Spectrally-inferred chlorophyll *a*

Spectrally-inferred chlorophyll *a* (Chl*a)* content for each of the cores was determined every 2 cm for the length of the cores from the dried sediments used in the gamma analysis. Spectral analysis of chlorophyll *a* and associated degradation products in sediment samples was conducted following the methods outlined in

Michelutti et al. (2010). Sediments were freeze dried and then sieved through a 125-µm mesh, and placed into glass vials. All reflectance spectra were obtained using a Model 6500 series Rapid Content Analyzer (FOSS NIRSystems Inc.). The sedimentary chl*a* concentration was inferred using an equation derived based on a 35-lake calibration dataset by Michelutti et al. (2010). Spectral chl*a* provides another estimate of overall algal production that can be compared to chl*a* estimates from pigment concentrations in the sediments, providing confirmatory support if trends are similar.

Elemental analyses

Sediment cores were sampled for elemental analysis spanning the past ~200-300 years of history. The suite of elements analyzed were aluminum (Al), arsenic (As), barium (Ba), beryllium (Be), cadmium (Cd), calcium (Ca), chromium (Cr), cobalt (Co), copper (Cu), iron (Fe), lead (Pb), magnesium (Mg), manganese (Mn), molybdenum (Mo), nickel (Ni), potassium (K), selenium (Se), silver (Ag), sodium (Na), strontium (Sr), thallium (Tl), tin (Sn), titanium (Ti), uranium (Ur), vanadium (V) and zinc (Zn). Only the most dominant elements were graphed for each core.

Samples were prepared using USEPA Method 200.7 (*Trace Elements in Waters, Solids and Bio-solids by Inductively Coupled Plasma-Atomic Emission Spectroscopy*) and 200.8 (*Determination of Trace Elements in Waters and Wastes by Inductively Coupled Plasma-Mass Spectroscopy*) with minor modifications. Briefly, samples were air-dried and ground to a fine powder with a mortar and pestle. Approximately 0.5 g of powdered sample was added to a plastic digestion tube along with 2 mL nitric acid, 6 mL hydrochloric acid and 7 mL distilled deionized water in a Digiprep digestion block and heated at 90-95°C for 300 minutes. A known subsample was filtered through a Whatman No. 40 filter paper and analyzed using an Agilent 7700x inductively coupled plasma-mass spectrometer. Samples were analysed in batches of up to 48, which comprised of up to 40 samples, 2 blanks, 4 duplicates and 2 reference materials (NRC MESS-4).

RESULTS

Lake Partner total phosphorus data

The average TP for each of the lakes, calculated from all values from 2002-2015 for Cameron Lake and 2002-2016 for Pigeon and Stony lakes, was 10.8, 16.2 and 14.4 µg/L, respectively. A similar yearly pattern occurs for the average TP for each lake over time with Pigeon Lake being the highest, Stony Lake slightly lower and Cameron Lake typically 4-5 µg/L lower (Fig. 2). No large changes in the average yearly TP were apparent in any of the three study lakes, similar to the previous findings of across the Kawartha region (Dolbey, 2015). The most recent values for Cameron Lake (2014 & 2015) are higher than the other years, but are based on only 1 sample from 1 sampling station, so any apparent increase must be interpreted cautiously.

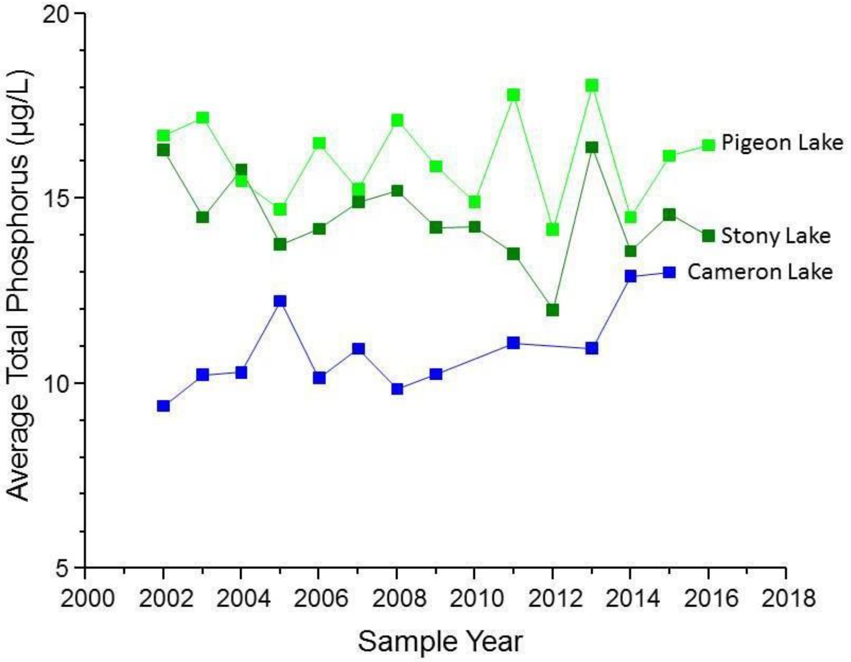


Figure 2. Average yearly total phosphorus measurements based on data from the Lake Partner program for Cameron Lake (2002-2015) and for Pigeon and Stony lakes (2002-2016).

The seasonal range in TP values from May to October is relatively low for Cameron Lake at ~6 µg/L, whereas for Pigeon Lake it is ~20 µg/L, and for Stony Lake typically ~13-16 µg/L but upwards of 20-25 µg/L in a few unusual years (see Supplemental Figs. S1-3). In Pigeon Lake there is a clear and consistent seasonal pattern across five sampling sites that have seasonal sampling in multiple years (see Supplemental Fig. S4). Mean May TP is lowest at ~9-10 µg/L, increasing each month to a peak in August of ~18-19 µg/L, which typically decline in September. A summary of Lake Partner TP data which averaged data from several lakes found a similar seasonal pattern (Mackenzie, 2016). Average October TP for Pigeon Lake was site dependent, with sample sites # 12 & 16 having the highest values at around ~22 µg/L and sites # 3 & 13 lower at around ~16-17 µg/L (see Supplemental Figs. S4-9). There was a slightly different seasonal pattern in Stony Lake, with most sampling sites indicating lowest TP in May (~8-11 µg/L), which increase to July, and then remain relatively stable from August to October at around ~14-15 µg/L (see Supplemental Fig. S10). The Burleigh Locks Channel (site 4, Supplemental Figs. S10-11) not surprisingly has higher TP than other main basin Stony Lake sample sites (Supplemental Figs. S12-14). In Cameron Lake, a seasonal pattern is not as apparent as in the other lakes, likely due to the lower nutrient values in Cameron Lake and the lower number of samples (see Supplemental Fig. S15). There were only 3 years in which seasonal sampling were done, 2002, 2005 and 2013 and typically only at one site, except 2013 when two sites were sampled.

The Lake Partner average TP trends (Fig. 2) were compared to data from the MOE-MNR (1976) report focused on a water quality assessment of lakes in the Kawartha region in 1972 and 1976 (Supplemental Fig. S16). This report generally only documents the average and range of the measured variables sampled over the ice-free season, and not the individual values from each sample date, with the exception of the temperature and dissolved oxygen profiles. The recorded average values of TP in 1972 and 1976 are generally higher than those post 2000 AD from the Lake Partner dataset (Sup. Fig. S16), with a decrease between 1972 to 1976 that are likely related to the reduced phosphorus loading to the lakes as the result of the phosphorus removal programs initiated at the major sewage treatment plants in 1975 (MOE-MNR, 1976). Methodology for TP measurements may have changed between the 1970s and post 2000 AD, and would have to be verified to have complete confidence in comparison between Lake Partner TP data and MOE-MNR measurements, as well as variations related to sampling locations. Many abatement programs for reductions in algal production followed the classic studies carried out at the Experimental Lakes Area (ELA) in northwestern Ontario in the early 1970s which indicated the dominant role of phosphorus in controlling algal production (e.g. Schindler, 1974).

Chronology models

All of the Kawartha lake cores exhibit an exponential decay in total ^{210}Pb with depth, with ~ 91%, 86% and 98% of the variance explained by a simple exponential decay model in Cameron, Pigeon and Stony lakes respectively (see Fig. 4 below). The relatively constant ^{210}Pb activity in the upper sediments of Pigeon Lake suggests potentially higher sedimentation rates that would dilute ^{210}Pb or alternatively may be indicative of recent sediment mixing. As a consequence, some caution is advised on the estimated chronology for the Pigeon Lake core. Background ^{210}Pb activities were reached by ~ 20-cm core depth in Cameron Lake, and at 23 and 27 cm in Pigeon and Stony lakes, respectively. The average rate of sediment accumulation post ~1900 AD was lower in Cameron Lake at 0.14 cm/year compared to 0.20 cm/year in Stony Lake and 0.24 cm/year in Pigeon Lake. In all lakes sediment accumulation was higher at the top of the core versus lower in the core, in part due to the higher water content in the uppermost sediments. Cameron Lake exhibits a distinct peak in ^{137}Cs between 7 and 9 cm, which would represent the peak of atmospheric bomb testing in the early 1960s, and is consistent with the estimated age based on the decay in unsupported ^{210}Pb. The ^{137}Cs at ~11 cm in Stony Lake is reasonably consistent with the ^{210}Pb estimated date, but suggests some upward mobility of ^{137}Cs, likely as the result of the sediments higher in organics in comparison to Cameron Lake (see organic matter section below). Pigeon Lake does not exhibit a distinct peak in ^{137}Cs, which may be due to higher sedimentation or some mixing of the uppermost sediments.

Age models based on the gamma analyses provide an estimate of date with depth (Fig. 3). The error associated with each date increases with depth. As a consequence, only ages since 1900 are noted on the graphs. Pollen analyses can provide an estimate of land clearance by settlers, and can be used to date the settlement horizon in the cores. Carbon dating could also be completed on deeper sediment samples and models developed that incorporated these into the core chronology.

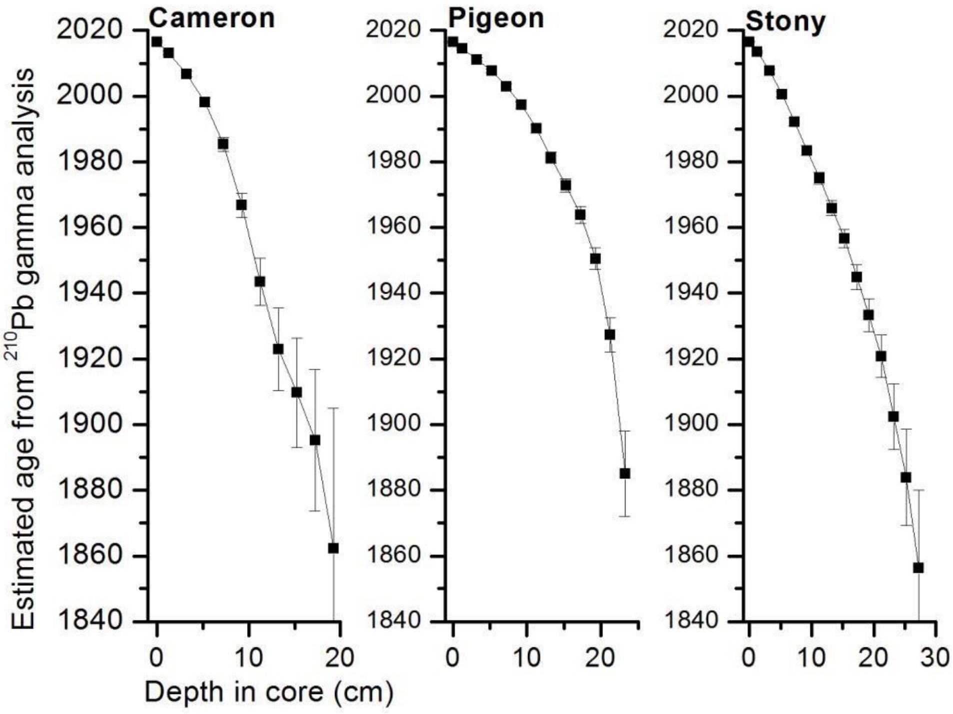


Fig. 3. Age models for the sediment cores from Cameron, Pigeon & Stony lakes based on gamma analysis (Fig. 4 below). Note the increasing error of estimated dates going down core.

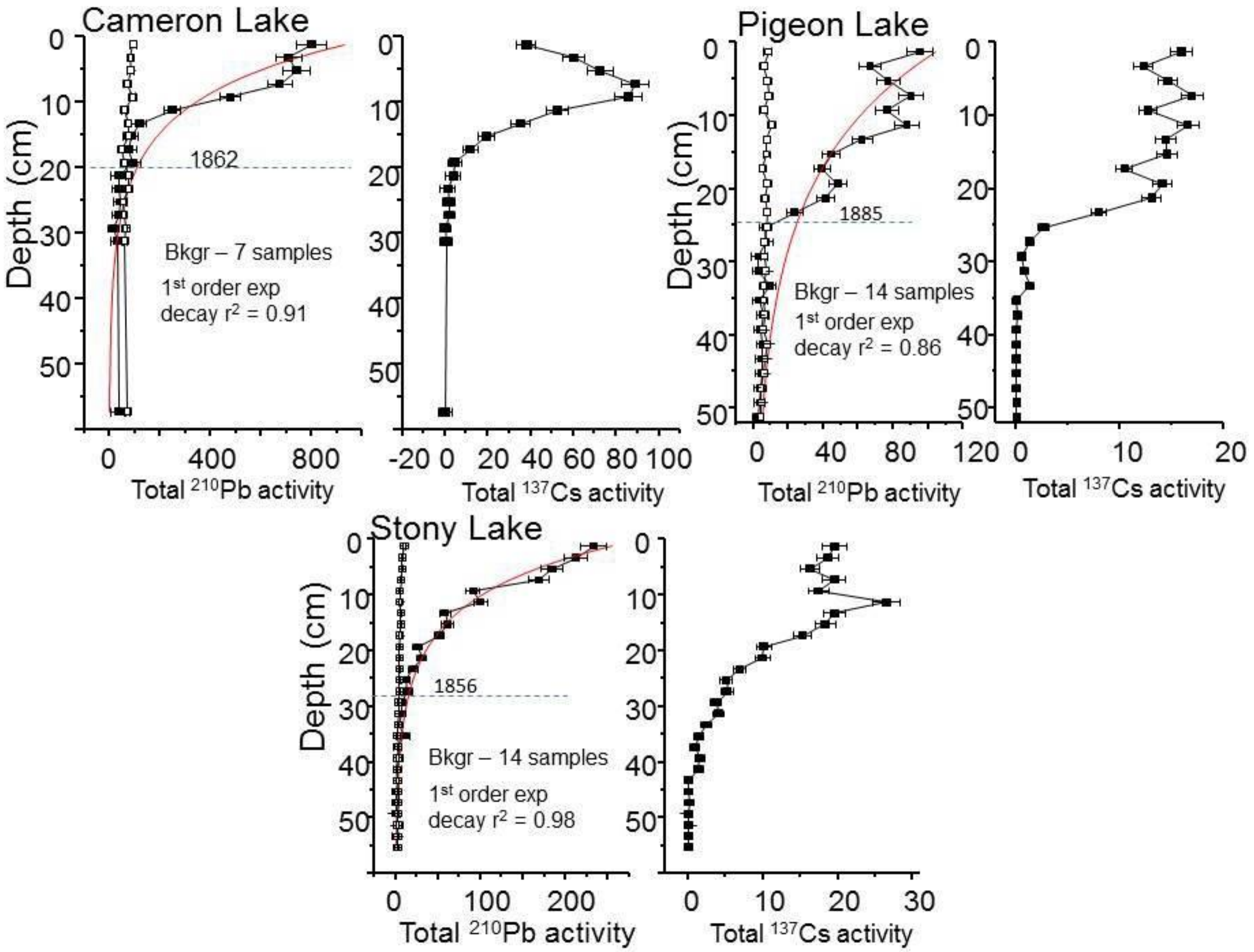


Figure 4. Gamma activities of ^{210}Pb and ^{137}Cs versus depth used to provide a chronology in the sediment core for Cameron Lake, Pigeon Lake and Stony Lake. Background was determined by intercept with ^{214}Bi (open squares) and ^{210}Pb (solid squares). The blue dotted line denotes estimated background and approximate date. The red line denotes a first order exponential decay, with coefficient of determination (r^2) noted. Activities are in Bq/kg.

Diatom assemblage, pigment concentrations and other proxy changes in Cameron Lake

Diatom changes

The diatom taxa documented in the sediment core from Cameron Lake is very diverse, with 285 taxa documented. Most of these taxa (80%) are very rare (< 1% abundance). Benthic and epiphytic taxa comprised ~ 60% of the assemblages (Fig. 5), with small Fragilariaceae, particularly *Staurosirella pinnata*, present at relatively high abundances throughout the core (see Supplemental Fig. S17). The small Fragilariaceae group is split into 3 main genera: *Staurosira*, *Pseudostaurosira* and *Staurosirella*. In this report we often refer to these taxa as small Fragilariaceae because of their similar preference for benthic habitat and the ability to form long chains. There is a diverse array of other benthic taxa in this core, particularly from the genera *Navicula* and

Nitzschia (see Supplemental Fig. S17). The most dominant epiphytic (species which attach to aquatic plants) taxon was *Achnanthidium minutissimum*, along with a large diversity of other epiphytic *Achnanthes* taxa. Benthic taxa are typically associated with bottom substrates (sediments, rocks) in the littoral regions of the lakes that have sufficient penetration of light to allow photosynthesis. The small Fragilariaceae taxa that can form long chains can also be found within the water column.

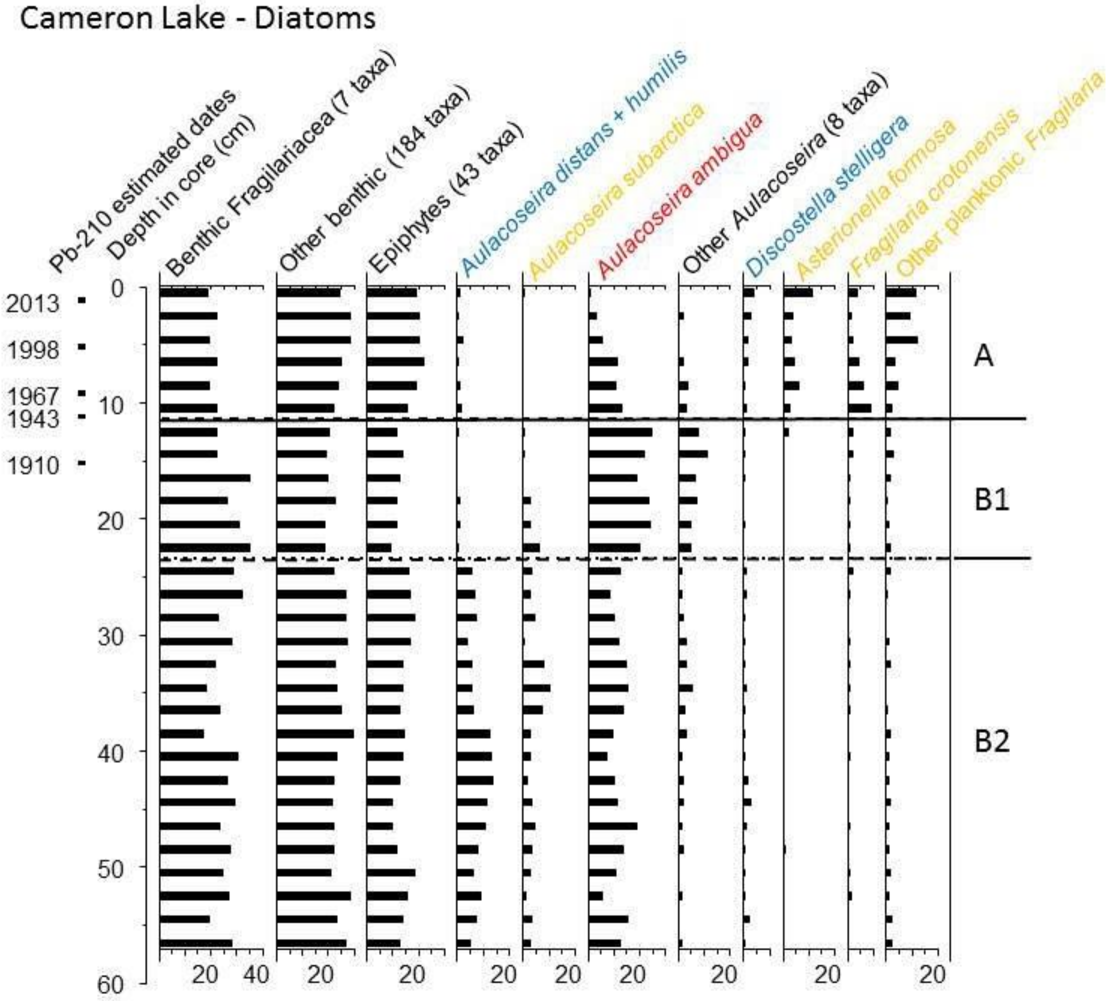


Figure 5. Percent abundance of dominant diatom taxa in the Cameron Lake sediment core versus core depth and estimated dates. Taxa names that are in blue are considered to be more oligotrophic (low nutrient) taxa. Taxa names that are in orange are considered to be within the more mesotrophic (medium nutrient) range, and those in red tending more towards eutrophic (high nutrient) range. Those taxa name in black are groups of different taxa and thus have no specific nutrient range. Zones are based on a depth-constrained cluster analysis.

The planktonic component of the assemblages varied from more oligotrophic *Aulacoseira distans* to mesotrophic taxa that include *Asterionella formosa* and *Fragilaria crotonensis*, as well as high abundances of the more eutrophic taxon *Aulacoseira ambigua* (Fig. 5). The total planktonic component of the assemblages comprised on average ~36%, which varied from ~25 to 46%, with no consistent decreasing or increasing trend through time (Fig. 6). The dominant diatom taxa are well represented in the calibration set, with 70-85% of all taxa being present in the calibration set, ensuring good representation for inferences of TP estimates (Cumming et al., 2015).

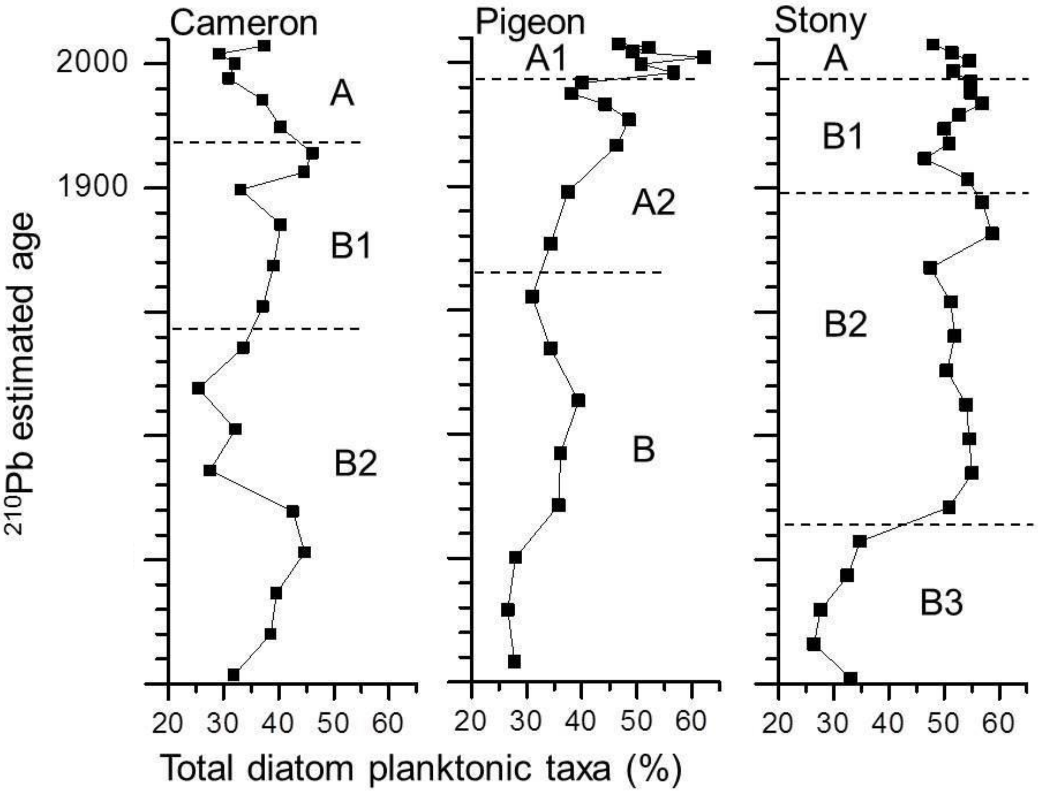


Figure 6. Total % planktonic diatom taxa for Cameron, Pigeon and Stony lakes versus estimated age in the sediment core. Horizontal lines and labels represent the zones based on the diatom assemblages for each core. Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with dates prior to 1900.

Cluster analysis identified two major periods of diatom assemblages in the Cameron Lake core over the past several hundred years (Fig. 5). Proceeding from the past to the present, the period prior to ~ 1800 AD (Zone B2) is dominated primarily by mesotrophic benthic Fragilariaceae (TP optima ~10-15 µgL^{-1}), particularly *Staurosirella pinnata, Staurosira construens* and *Pseudostaurosira brevistriata* (Supplemental Fig. S17). Other benthic taxa, particularly from the genera of *Navicula* sensu lato and *Nitzschia,* along with epiphytic taxa from the genus *Achnanthes* sensu lato also comprised a good proportion of the pre-1800 AD assemblages. Planktonic taxa comprised ~25 to 45% of the pre-1800 AD assemblages, particularly the oligotrophic taxon *Aulacoseira distans* (TP optima ~4-6 µgL^{-1}), the mesotrophic taxon *Aulacoseira subarctica* (TP optimum ~19 µgL^{-1}) and more eutrophic *Aulacoseira ambigua* (TP optimum ~26 µgL^{-1}). In Zone B1 (~1800-1940 AD), a similar composition of benthic and epiphytic taxa still comprised ~50-70% of the assemblage. Changes occur in the planktonic portion of the assemblage in Zone B1, with increases in the eutrophic planktonic taxon *Aulacoseira ambigua,* along with other highly eutrophic *Aulacoseira* including *Aulacoseira granulata*. At this time there were also declines in the more oligotrophic taxon, *Aulacoseira distans* as well as in mesotrophic *Aulacoseira subarctica*.

In Zone A (~1945 - 2015) *Aulacoseira ambigua* decline and a number of pennate mesotrophic taxa, such as *Asterionella formosa* and *Fragilaria crotonensis* increase, along with small increases in the more oligotrophic *Discostella stelligera*. Other planktonic *Fragilaria* sharply increased post ~2000 AD (Fig. 5, Supplemental Fig. S17). The benthic component still comprised ~60-70% of the assemblage, with few apparent changes in the species composition.

Total phosphorus estimates

Diatom-inferred total phosphorus (TP) estimates for the past several centuries indicate mid-summer TP concentrations varied between ~11 to 22 µg L^{-1} (Fig. 7). TP estimates were lowest pre-1800 AD (mean TP ~14 µg L^{-1}), increased during Zone B1 to an average of ~20 µg L^{-1} driven largely by increased *Aulacoseira ambigua* and other eutrophic *Aulacoseira,* and decreased in Zone A from ~ 17 to 12 µg L^{-1} with decline in *Aulacoseira ambigua* and increase in more mesotrophic taxa.

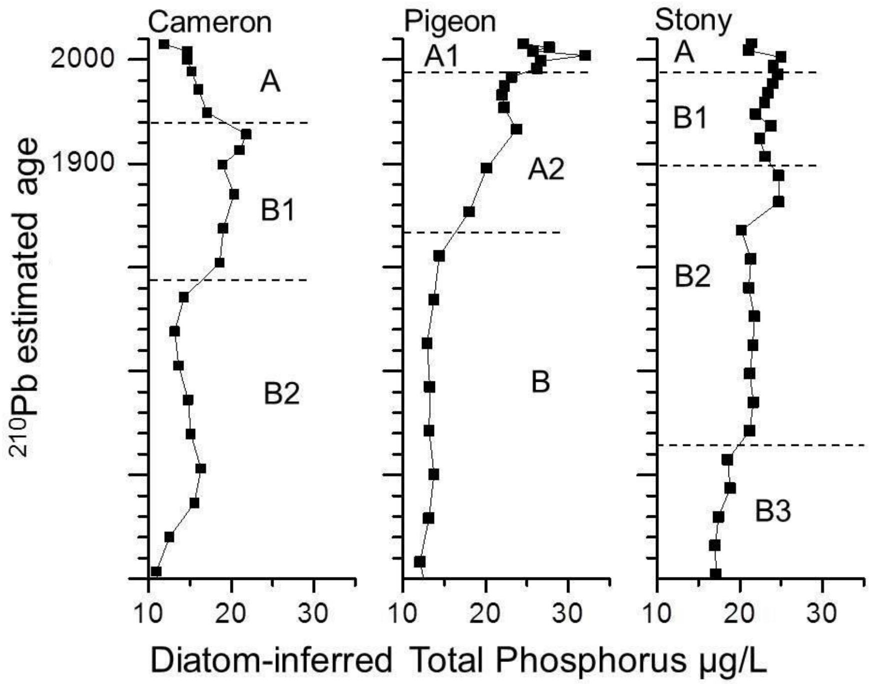


Figure 7. Diatom-inferred total phosphorus for each of the sediment cores versus estimated age. Horizontal lines and labels for each lake core represent the zones based on cluster analysis of the diatom assemblages. Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.

The correlation between PCA axis-1 scores (eigenvalue = 0.40) and diatom-based TP inferences is high (r = 0.70), as is the correlation between PCA axis-2 scores (eigenvalue = 0.24) and TP estimates (r = 0.65) (see Supplemental Fig. S18). PCA axis-1 scores are highly correlated with the % abundance of eutrophic *Aulacoseira ambigua* (r = 0.95) and PCA axis-2 scores are highly correlated with the % abundance of oligotrophic *Aulacoseira distans* (r = 0.93). These findings suggest that the main changes in diatom assemblages are consistent with changes in nutrients. Changes in water column stability (timing, length and strength of stratification), light availability or seasonality changes may also be contributing factors in the most recent changes in the diatom phytoplankton composition.

Diatom and chrysophyte concentration changes

Chrysophyte scale concentration increases ~6 times from ~1940 to 2015, suggesting a change in the algal composition (see Supplemental Fig. S19). Chrysophytes are flagellated golden-brown algae, and thus can be mobile within the water column. Trends in diatom concentration are not apparent, particularly post-1800 AD.

Pigment concentration changes

Cluster analysis identified two major periods of change in pigment concentrations (Fig. 8). The preservation index (Chl*a*/Pheo*a*) is low and constant, except one sample in the middle of Zone A2, indicating any trends in the pigments are likely not related to differential preservation. Analysis of the pigment concentrations since ~1800 AD indicated the largest change at ~10 cm (start of Zone A2), which corresponds to the most recent change in diatom assemblage at the start of diatom Zone A (Fig. 5). Alloxanthin and lutein increased ~2 fold; β-carotine and Chlorophyll *a* also generally have higher concentrations in Zone A2 in comparison to Zone B. Alloxanthin pigments represents the cryptophyta, which are single-celled flagellated algae. The pigment lutein represents green algae and the flagellated euglenophytes. These changes in the pigments, which correspond to a similar time of change in the diatom assemblage and increased chrysophyte scale concentration, suggest flagellated algae have recently increased.

The recent increase in Chlorophyll *a* and alloxanthin in the top two sediment samples (Fig. 8, Zone A1), corresponds to the recent rapid increase in scale concentration (see Summary figure below). Lutein and canthaxanthin (colonial cyanobacteria) also increased in the top two samples. The diatoms also show some change at this time with an increase in planktonic *Fragilaria* (Fig. 5). These combined changes may indicate changes in water column stability and/or changes in seasonality related to recent regional warming.

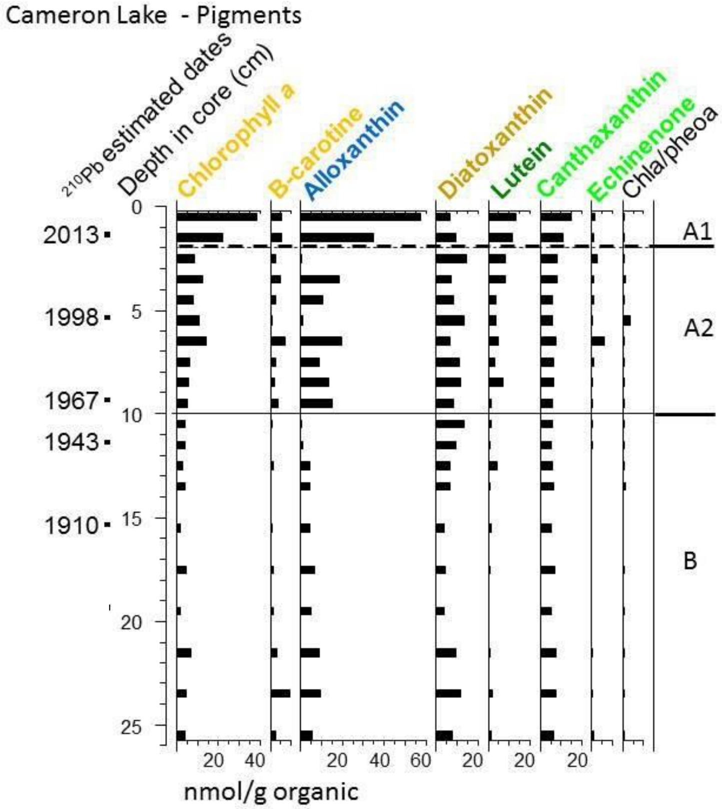


Figure 8. Concentration of the dominant pigments in the Cameron Lake sediment core versus depth in core, and estimated dates. Orange colour of the pigment names denote general pigments found in most algal groups. Pigments in blue denotes cryptophytes, whereas pigment names in brown denotes golden brown algae – chrysophytes & diatoms. Pigments names written in dark green denotes green algae, and light green denotes cyanobacteria (blue-green algae). Zones are based on a depth-constrained cluster analysis of the pigments.

Near infrared spectral analysis of sedimentary chlorophyll a (Chl*a*)

Spectral analysis estimates of Chlorophyll *a* indicate relatively low and stable conditions pre-1800 AD (Supplemental Fig. S20, also see Summary figure below). The zones indicated were based on the cluster analysis of the diatom assemblages. Estimates of Chlorophyll *a* increase slightly between ~1800 to 1900 AD (Zone B1), corresponding to higher diatom-inferred estimates of TP (Fig. 7, period of increased eutrophic planktonic *Aulacoseira ambigua*, Fig. 5). However, when standardized to the amount of organic matter in the sediments (the sediment fraction that contains the pigments), the trend is not apparent (Supplemental Fig. S21). In the recent sediments (Zone A), there is a distinct increasing trend in Chlorophyll *a*, which results in ~doubling of the estimates in the uppermost sediments. This period corresponds to the pigment Zone A (A1 & A2), increased Alloxanthin and Chlorophyll *a* (Fig. 8), particularly in the uppermost sediments and corresponds to the increased chrysophyte scale concentration. Diatom-inferred TP estimates declined during this period as the result of an increase in meso-eutrophic taxa (Fig. 7), a trend that is likely the result of interactions between nutrients and changes in seasonality and lake stratification.

Organic matter

Organic matter content of the sediments in Cameron Lake was low, ranging from ~11 to 16% (Supplemental Fig. S22, also see Summary figure below). Organic matter was low and relatively stable pre-1800 AD, and increased slightly in Zone B2. Organic matter continued to increase slightly post 1860 AD (Zone B1), reaching highest values post 1960 AD (Zone A), were they remained relatively stable.

Elemental analyses

Elemental lead in the sediments can be used as a chronological marker in the sediments as it often follows the increased trend in leaded-gasoline to the 1970s and then declined as lead was removed from gasoline. The peak in lead in Cameron Lake between ~ 8-10 cm roughly corresponds to the 1970s (Supplemental Fig. S23).

Cluster analysis of the suite of elements indicates two zones within the sediments analyzed, although change throughout the core was relatively small for most elements. The largest changes in the core occur in the upper sediments with increases in manganese in particular, as well as some smaller increases in barium and iron. Of potential concern is that the concentrations of Mn (and Fe) in the uppermost sediments exceed the Severe Effect Level (SEL) for sediments outlined by the Province of Ontario (https://www.ontario.ca/document/guidelines-identifying-assessing-and-managing-contaminated-sediments-ontario). The largest sources of Mn are domestic wastewater, sewage sludge, and from refineries and mining, as well as emissions from fuel additives that are comparatively quite small. The movement of Mn in lake sediments can be quite complex (Davison, 1993; Schaller et al., 1997), with enrichment of the uppermost sediments from the pore waters and new inputs from the water column.

SUMMARY OF CAMERON LAKE SEDIMENT CORE DATA

Changes in diatom assemblages indicated a major alteration in Zone B1 (post ~1800 to 1940 AD), with inferred increases in estimated total phosphorus (Fig. 9). Early European settlement in the Kawartha area began in 1818 AD (Conolly, 2017), with extensive logging beginning in the Kawartha regions in the late 1840s. The extensive logging and agricultural development resulted in a majority of the region in 1913 AD being described as either cultivated or barren from clear cuts. Damming to facilitate the movement of logs and the creation of the Trent-Severn Waterways, occurred in the 1830s to the 1880s or later, and led to increased lake levels and associated inputs from the terrestrial environment into the Kawartha lakes (Taillon & Fox 2004, Walters 2007, Lodge 2015). Organic matter content increased in Zone B1, albeit relatively small in terms of absolute degree of change of around 2-3%, and was part of an earlier increasing trend starting in Zone B2 (Fig.9). Pigments data suggests the period which corresponds to diatom Zone B1 was comprised of relatively low and constant concentrations of pigments.

Post ~1940 AD (Zone A), the diatom assemblage indicated another alteration in the dominant taxa, with several taxa that were rare in earlier sediments now comprising ~15-30% of the assemblage (Fig. 5). Diatom-based phosphorus estimates declined with this assemblage change. Initial TP declines may in part represent the regrowth of the forests with the remedial action activities of the 1920s and 1930s (Conolly, 2017), which may have offset influence of cottage development in the mid 1950s. However, the PCA analyses suggest that only a moderate degree of the variation in the diatom assemblages is explained by TP changes over the length of the sediment record. Some of the observed changes in the diatom composition may be the result of changes in internal lake dynamics such as lake stratification and lake turbulence, which in turn can affect light availability and nutrient dynamics. Diatom taxa, such as *Aulacoseira* spp., typically need more mixed conditions to stay in the euphotic zone, and thus the decline may represent more stable water column conditions (i.e. stronger stratification). Other proxies also indicated recent changes (Fig. 9), with sharp increase in chrysophyte scales, increased organic matter, increases in both spectral and pigment Chlorophyll *a* concentrations and increases in the Cryptophyta, which are single-celled flagellated algae (Alloxanthin pigment). All of these changes reflect changes in the internal dynamics of the lake, and are all would be influenced by climate and associated changes in the watershed, as well as other anthropogenic influences. The most recent changes, post ~1990 AD, also correspond to the timing of zebra mussel invasion into the Kawartha lakes in the 1990s (Bill Napier *pers.com*).

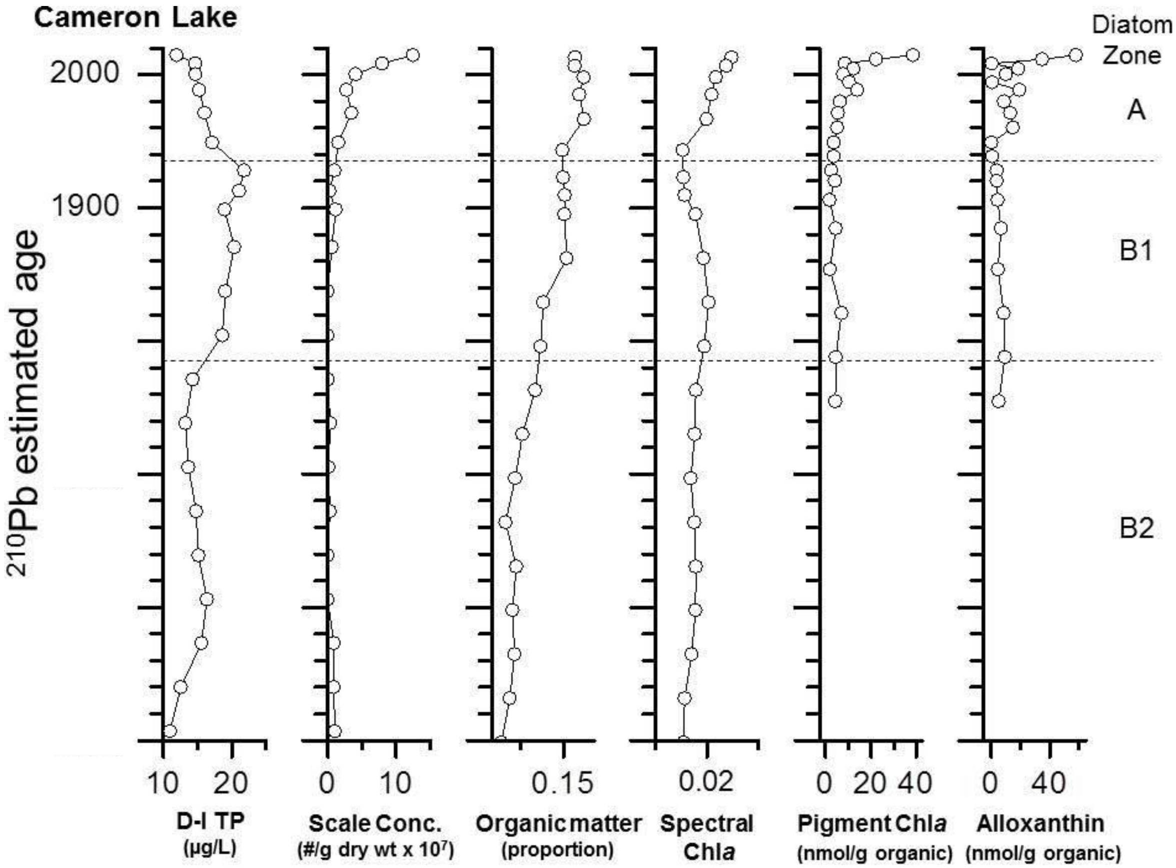


Figure 9. Summary paleolimnological data for Cameron Lake versus estimated age. Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these dates.

The watershed of Cameron Lake is very large, but is mostly comprised of drainage basins in the north on the Canadian Shield which are more forested and much less developed then the southern lands dominated by agriculture (White, 2006). The northern Canadian Shield drainage basins have a much lower contribution of nutrients into Cameron Lake and thus would help maintain lower nutrient levels even though much of the area surrounding Cameron Lake is cultivated (Fig. 1). The large forested region within the Cameron Lake watershed is likely a large reason explaining declined estimates of TP post the 1940s and the continued lower TP measurements documented from the Lake Partner data (Fig. 2). Cameron Lake also has the highest mean depth of the three study lakes (Table 1).

Diatom assemblage, pigment concentrations and other proxy changes in Pigeon Lake

Diatom changes

The diatom taxa documented in the sediment core from Pigeon Lake was diverse, with 204 taxa encountered. Most of these taxa (75%) were very rare (< 1% abundance). Benthic and epiphytic taxa comprise on average ~ 63% of the assemblages, but this varied from ~37% to 87%, decreasing through time with an average of ~72%

in the bottom samples (Zone B) to the most recent samples that consisted of 47% benthic taxa (Zone A1, Fig. 10). As in Cameron Lake, *Staurosirella pinnata* (benthic Fragilariaceae) was present at relatively high abundances throughout the core (Supplemental Fig. S24). The assemblages encountered in the Pigeon Lake core also consisted of a diverse array of other benthic and epiphytic taxa, particularly from the genera *Navicula* and *Achnanthes*. The planktonic component of the assemblages varied from the oligotrophic *Aulacoseira distans* to the mesotrophic *Fragilaria crotonensis* and more eutrophic *Aulacoseira ambigua,* as well as more hypereutrophic taxa including *Aulacoseira granulata* and *Cyclostephanos dubius* (Fig. 10). The total planktonic component of the assemblages comprised on average ~38%, with a distinct increasing trend through time from ~28% of the assemblages in Zone B to ~53% in Zone A1 (Fig. 6). The dominant taxa are well represented in the calibration set, with 80-89% of all taxa being present in the calibration set, thus providing good representation for inferences of TP estimates (Cumming et al., 2015).

Cluster analysis divides the core into two major periods of diatom assemblages (Fig. 10). Proceeding from the past to the present, Zone B, the period prior to ~ 1885 AD was dominated primarily by mesotrophic benthic Fragilariaceae (TP optima ~10-15 μg L^{-1}), particularly *Staurosirella pinnata, Staurosira construens* and *Pseudostaurosira brevistriata* (see Supplemental Fig. S24). Other benthic taxa, particularly from the genus of *Navicula* sensu lato, along with epiphytic taxa from the genus *Achnanthes* sensu lato comprised a good proportion of the pre-1885 AD assemblages. Planktonic taxa comprised ~20 % of the early portion of Zone B, and increased to ~35% of the assemblages, particularly the oligotrophic taxon *Aulacoseira distans* (TP optima ~4-6 μg L^{-1}), and the more eutrophic *Aulacoseira ambigua* (TP optimum ~26 μg L^{-1}) in the upper portion of Zone B. The more hypereutrophic *Aulacoseira granulata* was present at ~5% throughout Zone B.

In Zone A2 (~1885-1990 AD), a similar composition of benthic and epiphytic taxa still comprised an average of ~59% of the assemblage. Distinct changes were evident in the planktonic portion of the assemblage, with increases in the eutrophic planktonic taxon *Aulacoseira ambigua*, along with other highly eutrophic *Aulacoseira* taxa including *Aulacoseira granulate,* and declines in the less nutrient tolerant *Aulacoseira distans* (Fig. 10). Zone A1 (~1990 - 2015) was characterized by small decreases in the percent abundance of *Aulacoseira ambigua* and increase in more hypereutrophic *Aulacoseira granulata* and *Cyclostephanos dubius*. A number of pennate mesotrophic taxa, particularly *Fragilaria crotonensis*, also increased. The benthic component still comprised ~47% of the assemblage, with few apparent changes in the species composition (see Supplemental Fig. S24).

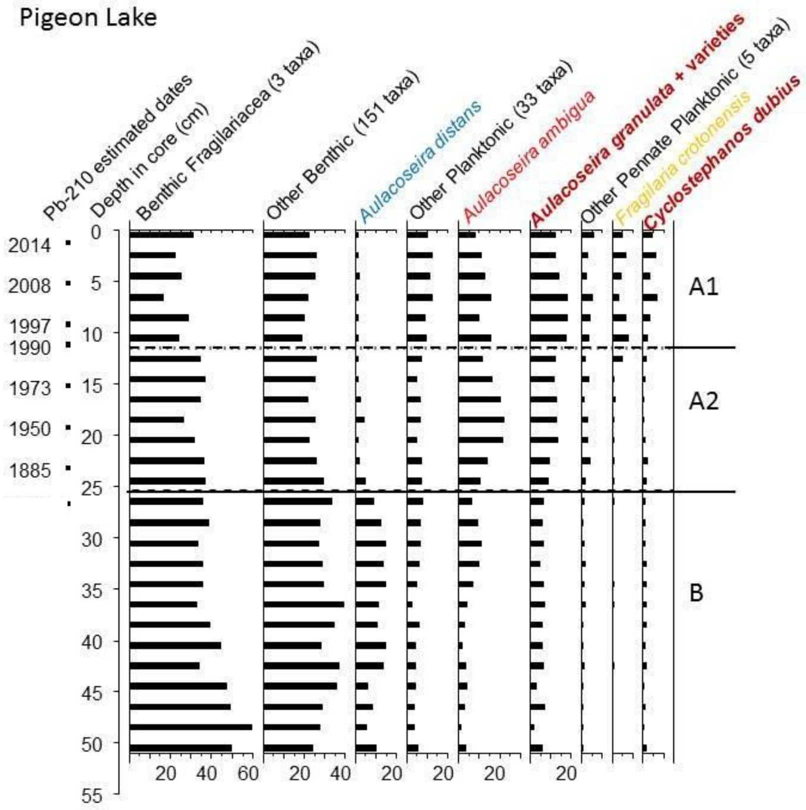


Figure 10. Percent abundance of dominant diatom taxa in Pigeon Lake sediment core versus depth and estimated dates. Taxa names that are in blue are considered to be more oligotrophic (low nutrient) taxa. Taxa names that are in orange are considered to be within the more mesotrophic (medium nutrient) range, and those in red tending more towards eutrophic (high nutrient) range and hypereutrophic taxa (very high nutrients) in bold red. Those taxa name in black are groups of different taxa and thus have no specific nutrient range. Zones are based on a depth-constrained cluster analysis of the diatom assemblages.

Total phosphorus estimates

Diatom-inferred total phosphorus (TP) estimates for the past several hundred years indicate mid-summer conditions varied between ~12 to 32 µg L^{-1} (Fig. 7). TP estimates were lowest in Zone B, pre-1885 AD (mean TP ~13 µg L^{-1}), increased during Zone A2 (~1885 to 1990 AD) to an average of ~22 µg L^{-1}, and further increased in Zone A1 (post 1990 AD) to ~27 µg L^{-1}. The increasing trend in estimates of TP is largely being driven by increases in the eutrophic and hypereutrophic planktonic *Aulacoseira ambigua*, *Aulacoseira granulata* and *Cyclostephanos dubius*.

The correlation between PCA axis-1 scores and TP inferences is very high (r = 0.93, Supplemental Fig. S25). PCA axis-1 scores are highly correlated with the % abundances of hypereutrophic *Aulacoseira granulata* (r = 0.93), as well as *Aulacoseira ambigua* (r = 0.87), as these taxa co-vary (r = 0.7). These findings suggest that the main changes in diatom assemblages are consistent with changes in nutrient changes or by variables which co-vary with nutrients. Changes in water column stability, light availability or seasonality changes may also be factors contributing to the more recent changes in the planktonic diatom composition.

Diatom and chrysophyte concentration changes

Chrysophytes scales were at extremely low concentration pre-1960 AD, increased slightly 1960 to 1980 AD, then rapidly increase post 1990 AD (see Supplemental Fig. S26 and Summary figure below). Chrysophyte scale concentration increased ~2 times post 1990 AD, suggesting a recent change in the algal composition. Diatom concentration was relatively stable in Zone B (pre-1900 AD) and then ~ doubled in Zone A2 (post ~1900 AD) and remained relatively stable to present. The pigment diatoxanthin (diatoms & chrysophytes) increased at the same time as indicated by the diatom concentrations based on the assemblage analyses (Fig. 11).

Pigment concentration changes

Cluster analysis identified two major periods of change in pigment concentrations (Fig. 11). Concentration of pigments in the Pigeon Lake core indicated relative stable conditions for the majority of the history prior to ~1995 (Zone B). Pigment Zone B corresponds to diatom zones B and A2 (Fig. 10). High concentration of the UV-index pigment occur in Zone A2 (~1997-2008 AD) along with increases myxoxanthin (cyanobacteria) and small increases in chlorophyll *a*. Pigment zones A1 and A2 correspond to diatom Zone A1. This period corresponds to a slightly higher percent abundance of the hypereutrophic planktonic diatom *Aulacoseira granulata* at the beginning of diatom Zone A1 (Fig. 10), and corresponds to the initial increase in chrysophyte scale concentrations (Supplemental Fig. S26, also see Summary figure below). Pigment Zone A1 (post 2008 AD) is characterized by the complete disappearance of the UV index, which is based on scytonemin derivatives which arise from cyanobacteria when exposed to high levels of UV irradiance. Increased myxoxanthin in the pigment Zone A2 (~1997-2008 AD) suggests that populations of colonial cyanobacteria (blue-green algae, such as *Microcystis* & *Aphanizomenon*) were highest during this period. If populations of colonial cyanobacteria were sufficiently high in the surface waters they were likely exposed to high UV radiation, potentially explaining the high amounts of scytonemin derivatives. The highest diatom-inferred estimates of TP occurred in diatom Zone A1 (post 1990 AD), which corresponds to pigment zones A2 and A1. The presence of canthaxanthin (colonial blue-green algae) throughout the core also suggests the lake has been conducive to having cyanobacteria blooms.

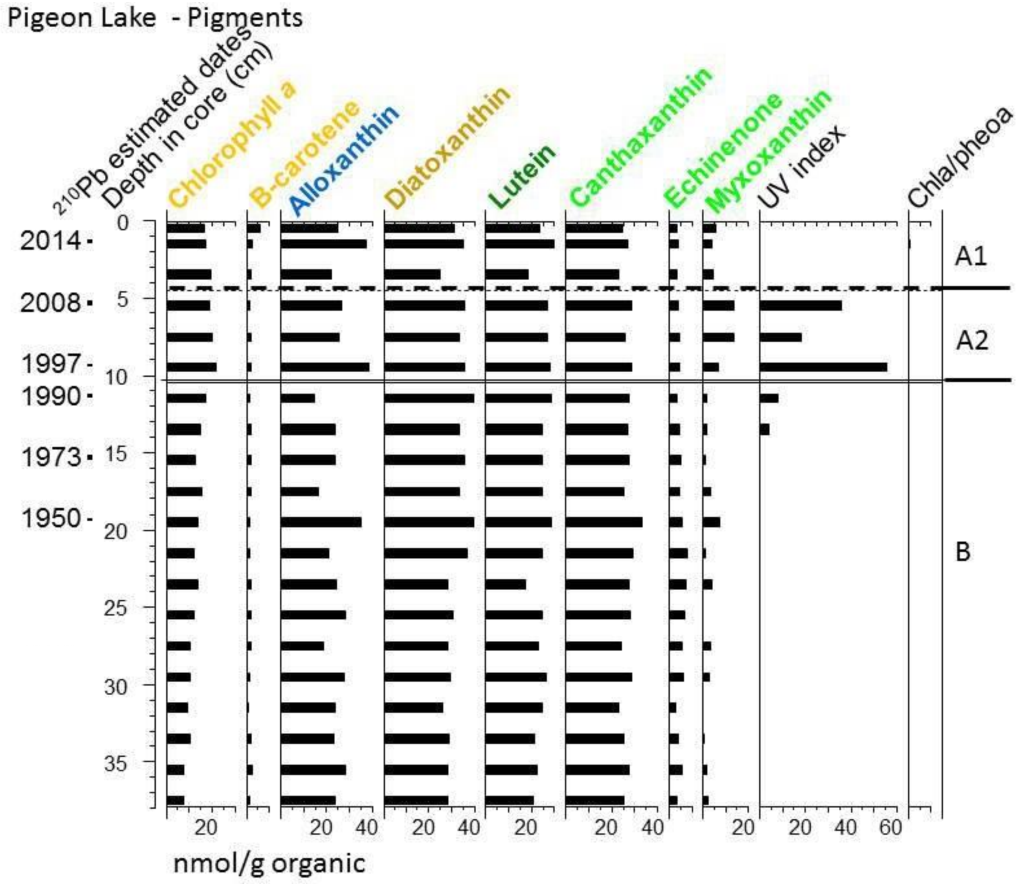


Figure 11. Concentration of dominant pigments in the Pigeon Lake sediment core versus depth and estimated dates. Orange text indicates general pigment that correspond to overall production. Text in blue denotes cryptophytes, whereas text in brown denotes golden brown algae – chrysophytes & diatoms. Dark-green text denotes green algae. And light-green text denotes cyanobacteria (blue-green algae). Zones are based on a depth-constrained cluster analysis.

Near infrared spectral analysis of sedimentary chlorophyll a **(Chl*a*)**

Spectral analysis estimates of Chlorophyll *a* indicates relatively low and stable production pre-1900 (Supplemental Fig. S20). The zones indicated are based on the cluster analysis of the diatom assemblages. Estimates of Chlorophyll *a* increase post 1800 to 1985 AD (Zone A2), corresponding to a period of increasing diatom-inferred estimates of TP that is driven by the increase in the eutrophic planktonic diatom *Aulacoseira ambigua* and decreased in the oligotrophic planktonic diatom *Aulacoseira distans* (Fig. 10). In the most recent sediments (Zone A1, post 1985 AD), there was a distinct increase in Chlorophyll *a* to a level approximately double the baseline pre-1900 AD estimates, which corresponds to the highest diatom-inferred TP estimates.

Organic matter

Organic matter content of the sediments in Pigeon Lake over the past ~500 years was relatively high, with low variability ranging from ~38 to 41% (see Supplemental Fig. S22). Organic matter content was more than double the organic matter content in the Cameron Lake sediments, suggesting generally higher production in Pigeon Lake. Higher spectral Chlorophyll *a* and higher estimates of diatom-inferred TP beginning in Zone A2, also suggests a more productive lake (see Summary figure below).

Elemental analyses

Elemental lead in the sediments can be used as a chronological marker in the sediments as it often follows the increasing trend in leaded-gasoline to the 1970s and then declining as lead was removed from gasoline. There is an early rise in lead at ~23 cm (estimated age of 1885 AD), which then becomes constant to the surface of the Pigeon Lake core at ~21 cm (see Supplemental Fig. S27), and thus cannot be used as a chronological marker.

Cluster analysis of the suite of elements indicated two zones within the sediments analyzed, although change throughout the core was relatively small for most elements. The only obvious change in the core, besides lead, occurred in the upper sediments, when manganese increases to levels that exceed the SEL. The enrichment of manganese in the uppermost sediments of Pigeon Lake is not correlated with iron (r = 0.08) which is unlike the trends in both Cameron and Stoney lakes where manganese and iron are highly correlated in the cores (r = 0.84 and 0.86 in the cores from Cameron and Stony lakes respectively), suggesting that the signal in the Pigeon Lake core may not be related to changes in redox. As mentioned previously in this report, the movement of manganese in lake sediments is complex (Davison, 1993; Schaller et al., 1997), and may be attributable to an external source, possibly related to sewage treatment. Alternatively, under reducing conditions, manganese can bind to carbonates, which has been suggested as an explanation for recent increases of manganese in sediments (Schaller et al., 1997).

SUMMARY OF PIGEON LAKE SEDIMENT CORE DATA

Changes in the diatom assemblages indicate a major alteration in Zone A2, with gradual increases in estimated total phosphorus until ~1940 AD, at which time it has remained relatively stable (Fig. 12). The oligotrophic taxon *Aulacosiera* declined and were replaced by more eutrophic *Aulacoseira* (Fig. 10). As outlined in the Cameron Lake summary, early European settlement in the region began in 1818 AD, with extensive logging beginning in the late 1840s and along with agricultural development that resulted in the majority of the region being either cultivated or barren from clear cuts by 1913 AD (Conolly, 2017). European settlement also led to damming of inflows into lakes in order to facilitate the movement of logs, as well as the creation of the Trent-Severn Waterways, all beginning around the 1830-1840s into the 1880s or later, which led to increased lake levels and further inputs from the terrestrial environment into the Kawartha lakes (Walters 2007, Lodge 2015).

Diatom concentration almost doubled during Zone A2, and both spectral Chlorophyll *a* and pigment Chlorophyll *a* concentrations increased (Fig. 12).

Post ~1990 AD, the diatom assemblage indicates another alteration in the dominant taxa, with two taxa in particular (mesotrophic *Fragilaria crotonensis* and hypereutrophic *Cyclostephanos dubius*) increase. These taxa were rare in earlier sediments but comprised ~8-12 % of the assemblage in Zone A1 (Fig.10). Total phosphorus estimates increased with this assemblage change as well as increases in the abundance of the hypereutrophic taxon *Aulacoseira granulata*. Scale concentration and spectral Chlorophyll *a* increased in Zone A1 (Fig. 12). Pigment concentrations were varied during this period, with initial increases Chlorophyll *a*, myxoxanthin and a pigment indicative of high UV (UV-index, Leavitt & Hodgson 2001), which all decline in the most recent sediments (post-2008 AD). *Aulacoseira granulata* also declined resulting in slightly lower diatom-inferred TP estimates.

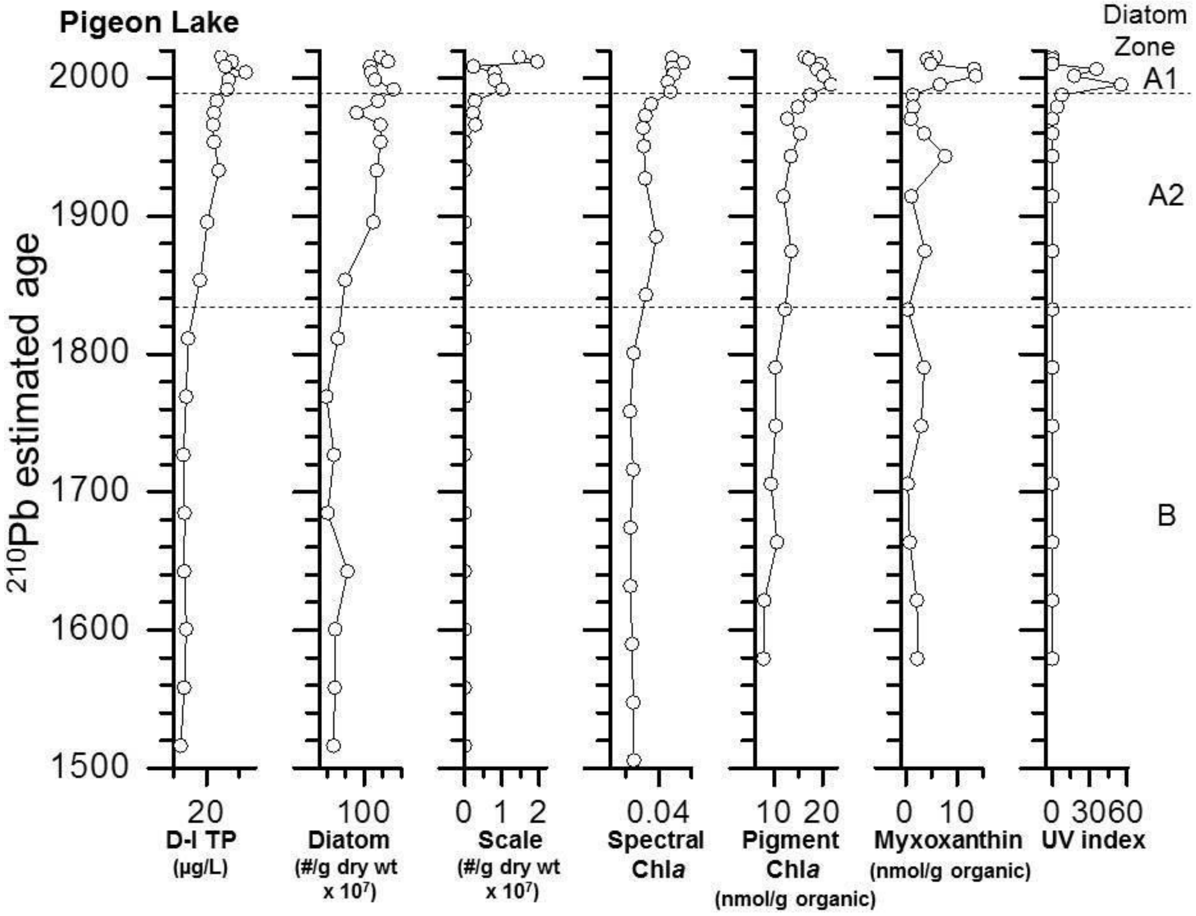


Figure 12. Summary paleolimnological data for Pigeon Lake versus estimated age. Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.

Regrowth of the forests with the remedial action activities of the 1920s and 1930s (Conolly, 2017) are not apparent in the sediment record in Pigeon Lake, TP estimates only decreased slightly post 1940s, and have increased post 1990. A larger portion of the drainage area of Pigeon Lake is dominated by the southern agricultural region, in comparison to both Cameron and Stony lakes (White, 2006), and thus any nutrient remediation from re-forestation may have been offset by nutrient inputs from agricultural development within the watershed. The high correlation between diatom-inferred TP estimates and the main direction of variation of the assemblages (PCA axis-1 scores) indicate that the main changes in diatom assemblages is consistent with changes in nutrients. The low mean depth of Pigeon Lake and the higher phosphorus contribution from the cultivated southern area would make Pigeon Lake highly susceptible to phosphorus loading. There is no clear evidence in the sediment record from the deeper northern area of Pigeon Lake of the decreased phosphorus export post 1970 with upgraded sewage treatment (Dolbey, 2017), nor any apparent decrease associated with the invasion of zebra mussels in the 1990s. The continued estimated increase in TP may be due to the accumulation of phosphorus laden sediments and internal nutrient dynamics within the lake, as well as potentially by periods of spring bypass and other storm runoff events that have been documented by the KLSA. These types of extreme events are projected to increase with continued climate warming (see Monkman, 2015 for a Kawartha Lake perspective). While the estimates of TP based on the diatom assemblages may over estimate values (Table 2), there was a distinct shift in the diatom assemblages, as well as changes in other proxy data post 1990 AD. These distinct changes would be related to the various factors influencing lake processes, including climate, and nutrient dynamics as a result of interactions between different components including, nutrient inputs, potential internal P loading, and potential near-shore influences of macrophytes and zebra mussels.

Diatom assemblage, pigment concentrations and other proxy changes in Stony Lake

Diatom changes

The diatom taxa documented in the sediment core from Stony Lake was diverse, with 188 taxa being enumerated. However, most of these taxa (77%) were very rare (< 1% abundance). Benthic and epiphytic taxa comprised on average ~ 48% of the assemblages throughout most of the sediment record collected, with oldest sediments (Zone B3) having a higher abundance of benthic taxa (~70%, Fig. 13). As in the other lakes, *Staurosirella pinnata* (benthic Fragilariaceae) was present at relatively high abundances throughout the core (Supplemental Fig. S28). There was a diverse array of other benthic and epiphytic taxa, particularly from the genera *Navicula* and *Achnanthes*. The dominant planktonic component of the assemblages included mesotrophic *Fragilaria crotonensis* and more the more eutrophic taxon *Aulacoseira ambigua,* as well as the hypereutrophic diatom *Aulacoseira granulata* (Fig. 13). Oligotrophic planktonic taxa were rarer in Stony Lake in comparison to the cores from Cameron and Pigeon lakes, with low abundances of small *Cyclotella* and *Discostella* taxa. The total planktonic component of the assemblages comprised on average ~52% over the last several hundred years, with lower abundances in the Zone B3 at ~30% (Fig. 6). The dominant taxa are well represented in the calibration set, with 81-94% of all taxa being present in the calibration set, resulting in a good representation for inferences of TP (Cumming et al., 2015).

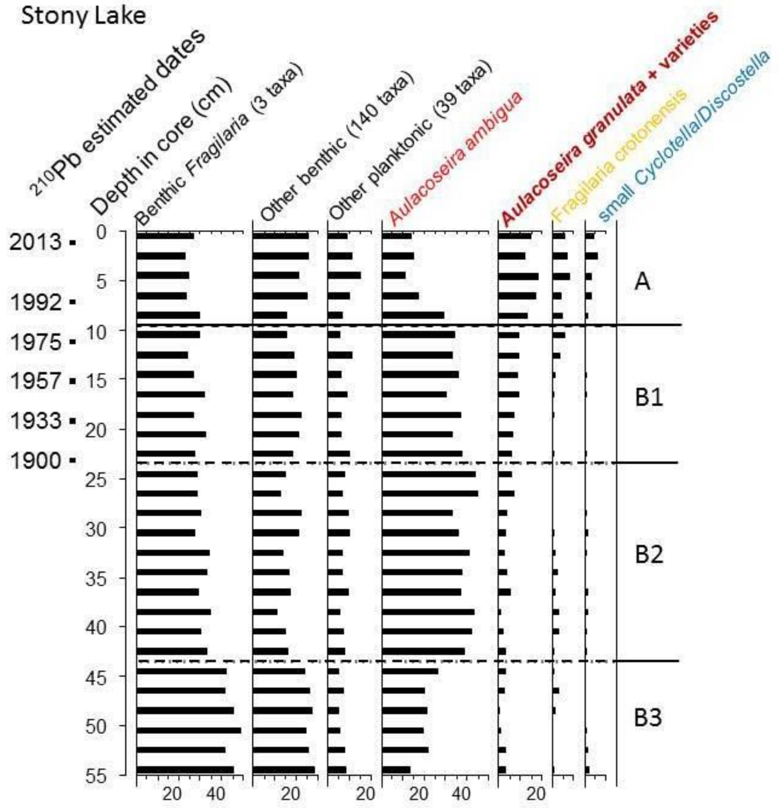


Figure 13. Percent abundance of dominant diatom taxa in the Stony Lake sediment core versus depth in core and estimated dates. Taxa names that are in blue are considered to be more oligotrophic (low nutrient) taxa. Taxa names that are in orange are considered to be within the more mesotrophic (medium nutrient) range, and taxa names in red tend more towards eutrophic (high nutrient) taxa. The names of hypereutrophic taxa (very high nutrients) are in bold red. Those taxa name in black are groups of different taxa and thus have no specific nutrient range. Zones are based on a depth-constrained cluster analysis of the diatom assemblages.

Cluster analysis divides the core samples into two major periods of diatom assemblages (Fig. 13). Proceeding from the past to the present, the oldest sediments (Zone B3) was dominated primarily by mesotrophic benthic Fragilariaceae (TP optima ~10-15 μg L^{-1}), particularly *Staurosirella pinnata, Staurosira construens* and *Pseudostaurosira brevistriata* (see Supplemental Fig. S28). Other benthic taxa, particularly from the genus of *Navicula* sensu lato, along with epiphytic taxa from the genus *Achnanthes* sensu lato, contributed to the high abundance of benthic taxa in Zone B3. The eutrophic planktonic *Aulacoseira ambigua* (TP optimum

~26 μgL^{-1}) comprised ~20 % of the Zone B3 assemblages, increasing to ~38% in Zones B2 & B1 (Fig. 13). The more hypereutrophic *Aulacoseira granulata* was present at low abundances in the older sediments, increasing in Zone B2 and reaching ~9% of the assemblage by the middle of Zone B1 (~1950 AD).

In Zone A (~1985-2015 AD), a similar composition of benthic and epiphytic taxa still comprised an average of ~48% of the assemblage. Distinct changes were evident in the planktonic portion of the assemblage, with a decrease in eutrophic planktonic *Aulacoseira ambigua*, and increases in hypereutrophic *Aulacoseira granulata*, mesotrophic *Fragilaria crotonensis* and oligotrophic *Cyclotella* and *Discostella* (Fig. 13).

Total phosphorus estimates

Diatom-inferred total phosphorus (TP) estimates indicate mid-summer conditions varied between ~15 to 25 μg L^{-1} (Fig. 7 and see Summary figure below). TP estimates were lowest in Zone B3 (mean TP ~17 μg L^{-1}), increased during Zone B2 to an average of ~22 μg L^{-1}, reaching ~25 μg L^{-1} by the top of this zone. TP estimates remained relatively stable in Zone B1 (~1900-1985 AD) at an average of ~23 μg L^{-1} into the beginning of Zone A and then decreased slightly to ~21 μg L^{-1} in the most recent sediments. The increasing trend in estimates of TP is largely being driven by increases in the eutrophic taxon *Aulacoseira ambigua* and the hypereutrophic taxon *Aulacoseira granulata* (Fig. 13).

The correlation between PCA axis-2 scores (secondary direction of variation) and TP inferences is high ($r = 0.83$, Supplemental Fig. S29). PCA axis-2 scores are highly correlated with the % abundances of hypereutrophic *Aulacoseira granulata* ($r = 0.88$), as well as the small benthic Fragilariaceae taxa *Staurosirella pinnata* ($r = 0.90$). The correlation between PCA axis-1 scores (the main direction of variation) and TP inferences is low ($r = 0.39$), with PCA axis-1 scores highly correlated to the abundance of *Aulacoseira ambigua* ($r = 0.99$). These findings suggest that there is a more complex interaction between nutrient changes and main changes in diatom assemblages in Stony Lake. Nutrients likely played a role in the early increase in *Aulacoseira ambigua*, (Zone B3 to Zone B2), but *Aulacoseira ambigua* declined in Zone A, while TP estimates increased as the result of increases in taxa such as the hypereutrophic *Aulacoseira granulata*. The percent abundance of *Aulacoseira ambigua* and *Aulacoseira granulata* did not co-vary as they did in Pigeon Lake. Changes in lake stratification and potential lake warming as the result of climate warming could influence light availability or seasonality changes in diatom taxa and may be one factor in the most recent changes in the phytoplankton composition of the diatoms. The invasion of zebra mussels in the 1990s into the Kawartha lake region could also influence the nutrient and light dynamics in the lakes, although there was no distinct change in the benthic diatoms that might be more clearly related to the introduction of the zebra mussel population.

Diatom and chrysophyte concentration changes

In contrast to Cameron and Pigeon lakes, there is no trend in the concentration of chrysophyte scales in Stony Lake (Supplemental Fig. S30). Diatom concentration was highest in Zone B2 into Zone B1, where they declined, and reached lowest concentration in Zone A.

Pigment concentration changes

Cluster analysis identified two major periods of change in pigment concentrations (Fig. 14). Concentration of pigments in the Pigeon Lake core indicate relative stable conditions for the majority of the history prior to ~1988 AD (Zone B). Zone B2 indicated higher concentrations of diatoxanthin and lutein in the lower sediments, which become less variable in Zone B1. The recent, post ~1990 AD, sediments (Zone A) were distinguished by the appearance and increasing concentrations of the colonial cyanobacteria pigment myxoxanthin (pigment in blue-green algae such as *Microcystis* & *Aphanizomenon*) in the uppermost samples (Fig. 14). Alloxanthin (pigment in cryptophytes, single-celled flagellated algae) increased in the top two samples. Echinenone (cyanobacteria pigment) also increased slightly in the upper two samples. In addition, β-carotene and chlorophyll b increased in Zone A, suggesting increased production. The presence of canthaxanthin throughout the core suggests the lake has been conducive to having cyanobacteria blooms prior to 1900 AD.

Pigment Zone A corresponds to the diatom Zone A, with increases in both hypereutrophic planktonic diatom taxa including *Aulacoseira granulate*, more mesotrophic taxa (*Fragilaria crotonensis*), as well as more oligotrophic taxa (*Cyclotella* and *Discostella*). Changes in the abundances of diatoms from all of these categories result in slightly lower estimates of TP in the upper two samples (Fig. 13, see also Summary figure below). As discussed for the diatom assemblage, there is likely a complex interaction between nutrients and climate that is seen in the pigments as increases in both cyanobacteria pigments and the flagellated algae.

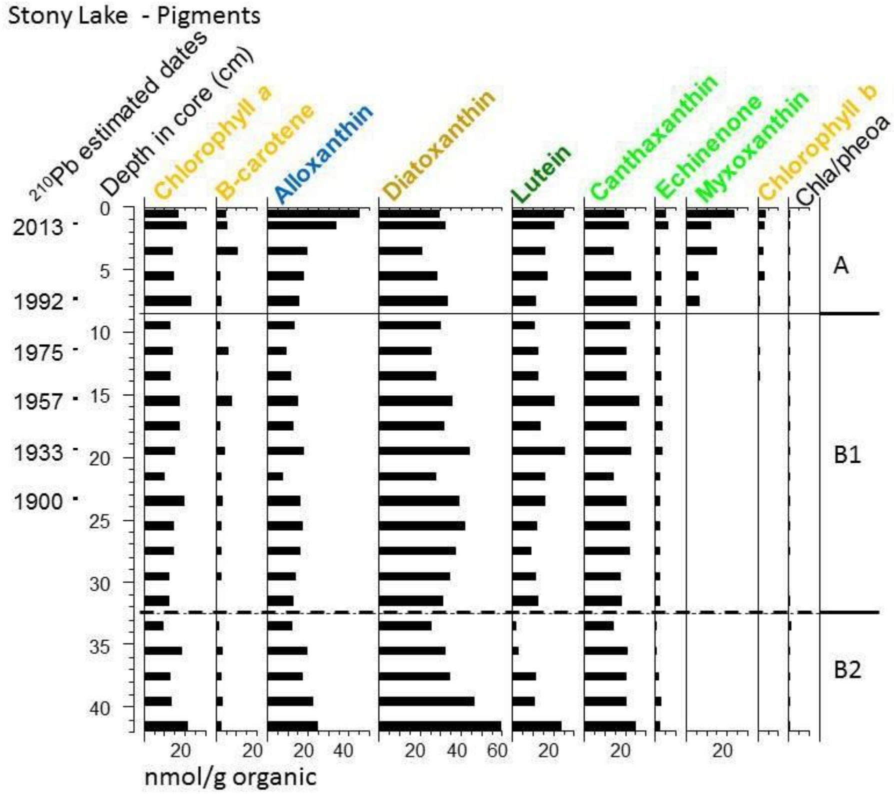


Figure 14. Concentration of dominant pigments in the Stony Lake sediment core versus depth in core, and estimated dates. Text in orange indicated pigments related to overall production. Text in blue denotes the cryptophytes. Text in brown denotes golden brown algae – chrysophytes & diatoms, and text in dark green denotes green algae, and text in light green denotes cyanobacteria (blue-green algae). Zones are based on a depth-constrained cluster analysis of the pigments.

Near infrared spectral analysis of sedimentary chlorophyll a **(Chl*a*)**

Spectral analysis estimates of Chlorophyll *a* indicated large variability pre-1900 AD (see Supplemental Fig. S20 and Summary figure below). The zones indicated are based on the cluster analysis of the diatom assemblages. The peak in the spectral Chlorophyll *a* in the beginning of Zone B2 corresponds to the higher abundance of eutrophic *Aulacoseria ambigua*, however the spectral Chlorophyll *a* declined to very low concentration in Zone B1, whereas *Aulacoseira ambigua* remained relatively constant until recent (Zone A). Estimates of spectral Chlorophyll *a* reached lowest values in Zone B1, whereas sedimentary pigment analysis indicated minimal change (see Summary figure below). Post 1960 AD, there is an increasing trend in Chlorophylll *a* into Zone A (see Summary figure below). These higher values correspond to higher concentration of myxoxanthin and B-carotene, whereas as pigment Chlorophyll *a* remained relatively stable (Fig. 14).

Organic matter

Organic matter content of the sediments in Stony Lake was relatively high, with somewhat higher variability than in Cameron and Pigeon lakes, ranging from ~40 to 49%, with highest peak in Zone B3 (see Supplemental Fig. S22 and Summary figure below). Organic matter content was more than double the organic matter content in the Cameron Lake sediments, and was slightly higher than Pigeon Lake. In Zone B2, organic matter was relatively constant at ~41%, and increased slightly post 1920 AD to ~43% and reached ~45% in the uppermost sediments. These general trends are similar to changes in the diatoms and pigments, and similar to spectral Chlorophyll *a* post 1960 AD (see Summary figure below).

Elemental analyses

Elemental lead in the sediments can be used as a chronological marker in the sediments as it often follows the increasing trend in leaded-gasoline to the 1970s and then declining as lead was removed from gasoline. The peak in lead in Stony Lake between ~ 9-13 cm roughly corresponds to the 1970s (see Supplemental Fig. S31).

Cluster analysis of the suite of elements indicated two zones within the sediments analyzed, although change throughout the core was relatively small for most elements. A number of elements increased post ~1900 to 1930 AD (Zone A2), including potassium, copper, zinc, magnesium, titanium, calcium and barium to varying degrees. Manganese increased in the upper sediments (Zone A1), similar to both Cameron and Pigeon lakes. The correlation between manganese and iron is quite strong (r = 0.86) possibly indicating a connection to redox. Manganese is one of the least toxic elements. The federal 'aesthetic' guideline for manganese in drinking water is 50 µg L^{-1} (values above this can cause discolouration of clothes and plumbing and can cause an undesirable taste) and manganese concentrations taken at Rosedale, Bobcaygeon and Burleigh Falls in 2000, 2001, 2015 and 2016 are well below this guideline (Napier et al., 2018).

SUMMARY OF STONY LAKE SEDIMENT CORE DATA

Changes in diatom assemblages indicate an alteration deep in the core (Zone B2), with increased inferences of estimated total phosphorus (Fig. 15) with the rise in eutrophic *Aulacoseira ambigua* (Fig. 13). Diatom concentration approximately doubled, organic matter declined (albeit small), and spectral Chlorophylll *a* increased. These early changes could be an indication of increased indigenous cultivation and clearing of areas of more permanent villages Stony Lake, near Burleigh Falls (Connolly, 2017), but more precise timing that is beyond the capabilities of the ^{210}Pb analysis will have to be more definitely constrained by pollen analysis to identify the European settlement presence in sediments (e.g. rise in cultivated *Ambrosia*).

TP estimates increased near the end of diatom Zone B2 (Fig. 15) with initial rise in the hypereutrophic *Aulacoseira granulata* (Fig. 13). TP estimates remained high throughout Zone B1, ~1900 to 1990 AD. Diatom concentrations declined during this time, as well as the pigment diatoxanthin post-1960 AD (Fig. 13), while organic matter increased post-1920 AD, and spectral Chlorophyll *a* began to increase post-1960 AD. As in the other two study lakes, the Stony Lake watershed would have been impacted by European settlement throughout much of the mid to late 1800s with extensive logging and damming, as well as by the lock and dam development of Trent-Severn Waterways (Angus 1988). The future analysis of pollen preserved in the cores will provide a chronological marker for the early European settlement enabling a better estimate of timing of changes in the 1800s.

Post-1990 AD (Zone A), the diatom assemblages indicated another alteration in the dominant taxa, with declines in eutrophic *Aulacoseira ambigua*, but increases in hypereutrophic *Aulacoseira granulata*, mesotrophic *Fragilaria crotonensis* and oligotrophic *Cyclotella* and *Discostella* (Fig. 13), resulting in similar estimates of TP, except for uppermost sediments (post ~ 2000 AD) where estimates declined to~ pre-1900 AD levels. *Fragilaria crotonensis* and *Cyclotella*/*Discostella* were rare in earlier sediments and comprised ~6-13 % of the assemblage in Zone A. However, diatom concentration declined to lowest numbers during this period, indicating all taxa decreased in absolute amounts, indicating lower overall diatom production. Spectral Chlorophyll *a* and pigment Chlorophyll *a* increased, with higher concentration of myxoxanthin, a pigment indicative of colonial cyanobacteria, which was previously low. Alloxanthin also increased in Zone A, particularly in the most recent sediments (Fig. 15). Increases in these latter pigments suggest lower diatom production may be related, at least in part, to the increases in these other algal groups.

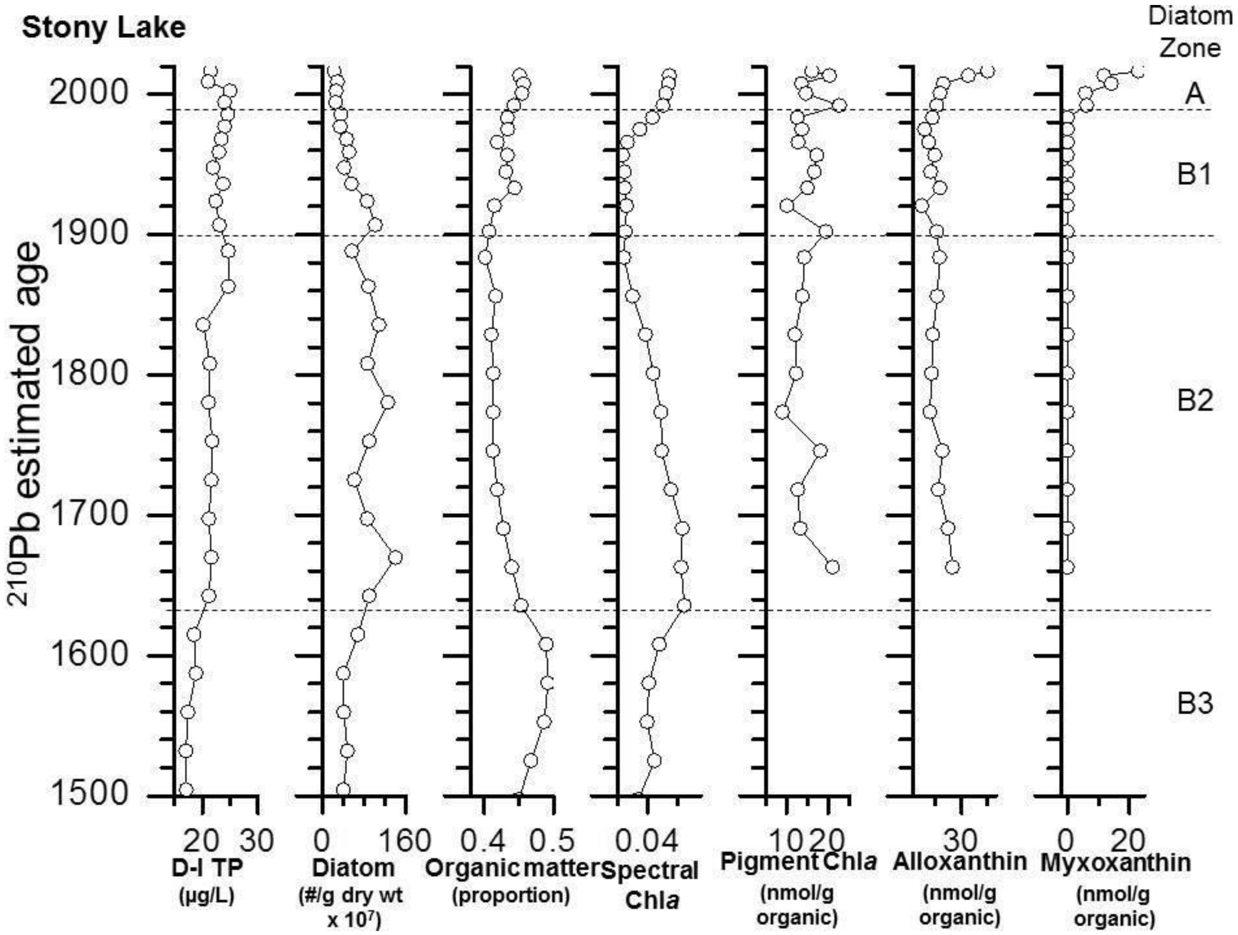


Figure 15. Summary paleolimnological data for Stony Lake versus estimated age. Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.

Stony Lake has a smaller drainage basin than the other two study lakes, with more of it in the northern regions (White, 2006), which may help to explain the relative stability of TP estimates from ~1900 to 2000 AD. Drainage from the more forested northern regions would have lower nutrient inputs in comparison to regions to the south with more agricultural land use. Diatom analyses suggest a complex interaction between estimated phosphorus and the main direction of change in diatom assemblages, with potential changes in lake stratification or seasonality of phytoplankton blooms, as a result of climate influences, may be an additional factor in the recent phytoplankton composition. The timing of the most recent changes also corresponds to when zebra mussels invaded the Kawartha lakes (Bill Napier, *pers. com.*).

Recent TP measurements and diatom-inferred summer phosphorus inferences

Comparison of the recent TP measurements (2002-2016 AD) from the Lake Partner dataset to post 2000 AD diatom-inferred estimates can provide a means of evaluating the potential accuracy of the paleo-inferred TP.

The calibration dataset for the diatom analyses is based on summer water chemistry and thus typically represents the maximum TP values for most lakes, thus the reason for comparison to peak summer measured values of TP for each lake. The lower temporal resolution of the sedimentary record only results in 3-4 samples within this time range. The recent paleo-history of estimated TP for Cameron Lake falls within the average peak summer TP measurements (Table 2). For both Pigeon and Stony lakes the average estimated TP for post 2000 AD is higher than the average peak summer measurements, but fall within the maximum average August values of ~26-28 µg/L for both of these lakes (Table 2).

Table 2. Comparison of average peak summer Lake Partner TP (µg/L) from 2002-2016 AD versus average diatom-inferred summer TP (µg/L) post-2000 AD.

Lake	**Peak summer month**	**Average peak in summer TP measurements (µg/L)**	**Average diatom-inferred TP post 2000 AD (µg/L)**
Cameron	~June	12-15	~14
Pigeon	August (October)	19-22 (max average – 26)	~26
Stony	August	14-16 (23 Burleigh Locks) (max average – 28)	~22

Several factors can influence how well the sedimentary diatom assemblages reflect measured TP values. In the upper sediments, the diatom estimates are based on the accumulation of ~1-2 years, and as such reflect the dominant taxa which bloomed in the various seasons within each lake over that time. Physical factors within a lake, such as light availability, temperature, nutrients, stratification and length of ice cover are dominant drivers of phytoplankton succession. Sommer et al. (2012) review the main drivers of plankton succession over the ice free season in dimictic lakes with turnover twice a year in the spring and fall. The Kawartha lakes are generally shallow and many are likely more polymictic, having multiple turnovers/mixing caused by wind during the ice free season. Nonetheless, the general principles of the phytoplankton succession outlined by Sommer et al. (2012) are still applicable to polymictic lakes, but with the potential of having more diatom blooms or other algae adapted to highly mixed systems. In more oligotrophic systems, such as Cameron Lake, phytoplankton biomass may be greatest in the spring, and somewhat lower in the fall, both lake turnover periods (Sommer et al., 2012). The phytoplankton assemblages during spring and fall turnover are often dominated by diatoms, as they are better adapted to mixing and lower light conditions associated with lake turnover. In contrast, in more eutrophic systems phytoplankton biomass is often greatest in the summer, followed by spring and lowest biomass in the fall. While diatoms certainly are present and persist during the summer, biomass is typically much lower in comparison to other algal groups, but may periodically bloom during more highly mixed conditions during the summer. The more mesotrophic Pigeon and Stony lakes would likely fall somewhere in between this generalization of phytoplankton seasonal dynamics, and will also vary with the strength of stratification and how often wind induced mixing of the water column occurs. Thus the degree of stratification and mixing within the ice free season, which will also influence nutrient and light availability dynamics, could highly influence which diatoms will bloom and deposit in the sedimentary record accumulating at the bottom of the lake.

The degree to which the TP estimates reflect the main direction of change in the diatom assemblages varied between the lakes. For Cameron Lake, the PCA analyses suggest a moderate degree of the variation in the diatom assemblages is explained byTP changes. In Pigeon Lake, analyses indicate the main diatom assemblages can largely be summarized by nutrient changes or by variables which co-vary with nutrients. Internal loading of phosphorus in Pigeon Lake may be a significant proportion of the phosphorus budget. In Stony Lake, PCA analyses suggest a complex interaction between nutrient changes and the main direction of changes in diatom assemblages. Limited TP data is available of lake sampling prior to the Lake Partner Program. The Ministry of Environment and Ministry of Natural Resources did a comprehensive sampling of the Kawartha lakes in 1972 and 1976, including our three study lakes. Based on this data, average TP has declined from the 1970s, with declines being more extensive in Pigeon and Stony lakes (Supplemental Fig. S16). The paleodata does not clearly follow this trend, potentially due to the lower resolution of the latter, and that other lake variables, such as mixing/stratification regimes, which in turn influence both light availability and nutrient dynamics, can also highly influence diatom community dynamics. As such, the changes in diatoms provide a clear indication of alterations within the lakes, but these are not solely driven by nutrients, but rather an interaction of many factors. Furthermore the TP declines from the 1970s are from a higher to lower mesotrophic range in Pigeon and Stony lakes and thus would not have a large influence on the dominant planktonic diatom flora.

Changes in the length of ice cover in temperate lakes with the warming climate effects the seasonal lake stratification regime, light and nutrient availability (Magnuson et al., 2000; Benson et al, 2012; Feuchtmayr et al. 2012; Kirillen et al., 2012) and would influence the recent phytoplankton composition of all of the study lakes to some degree. Warming of lake-surface waters has been documented in many regions around the world, with the lake warming often at a rate faster than the increase in air temperatures (Richardson et al., 2017). This recent survey of 230 lakes across northeastern North America also found that whole-lake warming was greatest in polymictic lakes. As warming can increase algal production in general, and cyanobacteria in particular, climate is likely to be one of the factors influencing the recent changes observed in the algal composition of the study lakes. As in other previous surveys, the strength of thermal stratification was found to have increased in a large number of the lakes that had warmed as the result of climate warming (Richardson et al., 2017).

Strong thermal stratification was documented in all three of our study lakes in 1972 in the MOE-MNR (1976) report over at least portions of the ice-free season. Cameron Lake was stratified on the June 8th sampling date, mixed on June 27th and then stratified on each of the sampling dates of July 18th, August 9th & 29th and completed mixed on September 19th. The north end of Pigeon Lake was stratified on all of the sampling dates, which were approximately every 2 weeks from June 13th to August 30th, but with the thermocline deepening over the season, and then fully mixed on September 21st. Temperature profiles were not available from Stony Lake, but Upper Stoney Lake was stratified at all 7 sampling dates from May 25th to September 26th, with the thermocline deepening somewhat over the season. Low dissolved oxygen or anoxic conditions in the bottom waters were recorded in all of these lakes during the summer, with anoxic conditions being most pronounced at the northern Pigeon Lake site. Anoxic bottom water conditions can often lead to the release of phosphorus from the sediments (Orihel et al., 2017), also known as internal phosphorus loading. As warming of lake waters and increased strength of stratification has been shown in numerous lakes over the past few decades (Richardson et al., 2017), one may surmise that periods of strong stratification and anoxic conditions could have increased in our study lakes since the measurements in 1972, thereby increasing the likelihood of internal phosphorus loading becoming a larger portion of the phosphorus budget.

Potential influence of zebra mussels

Zebra mussels invaded the Kawartha lake region in the 1990s (Bill Napier *pers. com.*), dispersal rates from the Laurentian Great Lakes into neighboring lakes were rapid (Cha et al., 2013) and were documented in Lake Simcoe in 1994 (Gudimov et al., 2015), and noted to have been established in Cameron and Pigeon in 1996 and Stony in 2002 (W. Napier *pers. com.*). Increased water clarity has been noted in many affected lakes after the establishment of large populations of zebra mussels due to the large filtering capacity of well-established colonies of zebra mussels (Zhu et al., 2006; Higgens & VanderZanden, 2010; Brown et al., 2012). The increased water clarity has been related to declines in chlorophyll *a* (Chl*a*) and total phosphorus concentrations (Cha et al., 2013). While many affected lakes and rivers exhibit these responses, not all have shown associated subsequent declines in Chlorophyll *a* and TP, and increased water clarity (Higgens & VanderZanden, 2010; Cha et al., 2013; Gudimov et al., 2015). In Lake Simcoe, TP has not declined with the invasion of zebra mussels, thought to be related to a large portion of the filtered particles (>85%) being returned to the water column as faeces and internal P loading from the sediments (Gudimov et al., 2015). Various interactions occur within both the pelagic (profundal) and benthic (littoral) nutrient pathways that involve zebra mussels and thus it is often difficult to determine what the outcome will be with the zebra musssel introduction (Higgens & VanderZanden, 2010).

Secchi depth data from the Lake Partner Program since ~1995 AD for the three study lakes indicates a fairly large range in measurements across the different years and between different lake sampling sites (Figs. 16-18). Data from the 1976 MOE-MNR report was used as a comparison to the post 1995 AD Lake Partner data to examine the degree of change in water clarity. For Cameron Lake there is no clear trend in water clarity based on the Secchi depth data between the 1970s and post 1995 AD (Fig. 16). Average Secchi depth measurements for 1972 AD and 1976 AD fall within the range of average Secchi depth post 1995 AD. For Pigeon Lake, any trend in Secchi depth is dependent on the lake sampling site (Fig. 17a). If all sites are analyzed together, there is an overall significant declining trend in Secchi depth since 1995 AD (Fig. 17b), which is being driven primarily by sites 12, 15 and 16 (Fig. 17a). Comparison to the 1970s data also indicates variation between lake sampling sites. Site locations from the MOE-MNR (1976) report were approximately corresponded to the current Lake Partner (LP) sampling sites for the few sites that were similar between the two datasets. The LP site closest to the northern Pigeon coring site (site 13) compared to the 1970s MNR P-17 site indicates no distinct change in water clarity based on this limited data (Fig. 17c), whereas the more southern LP site (site 3) has an apparent increase in Secchi depth (although the MNR site used for comparison, P-20, is quite a bit further south than LP-3, and would be much shallower and higher macrophyte coverage). LP site 15, just south of Boyd Island, is graphed as a comparison. For LP site 12 (MNR site P-16) there is also an apparent increase in clarity between the 1970s and the early 2000s, however this increase has steadily declined, approaching the 1970s measurements (Fig. 7d). LP site 16 also has a steadily declining trend post 2000 AD (Fig. 17d). As in Pigeon Lake, if all sites for Stony Lake are examined together there is a significant decline in Secchi depth post 1995 AD (Fig. 18). There were few sampling sites in Stony Lake in the MOE-MNR (1976) report, and none near our coring location. As such, the only comparison is the Burleigh Locks LP site 4 (MNR site S-27), which indicates increased clarity post 1995 AD in comparison to the 1970s (Fig. 18c). LP site 7, near the coring location, is provided as reference.

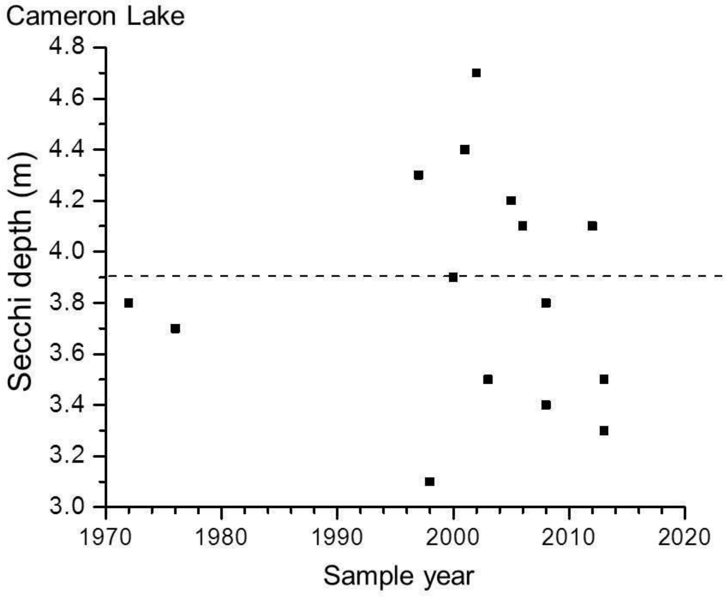


Fig. 16. Average secchi depth data from MOE-MNR (1976) report and Lake Partner Program for Cameron Lake.

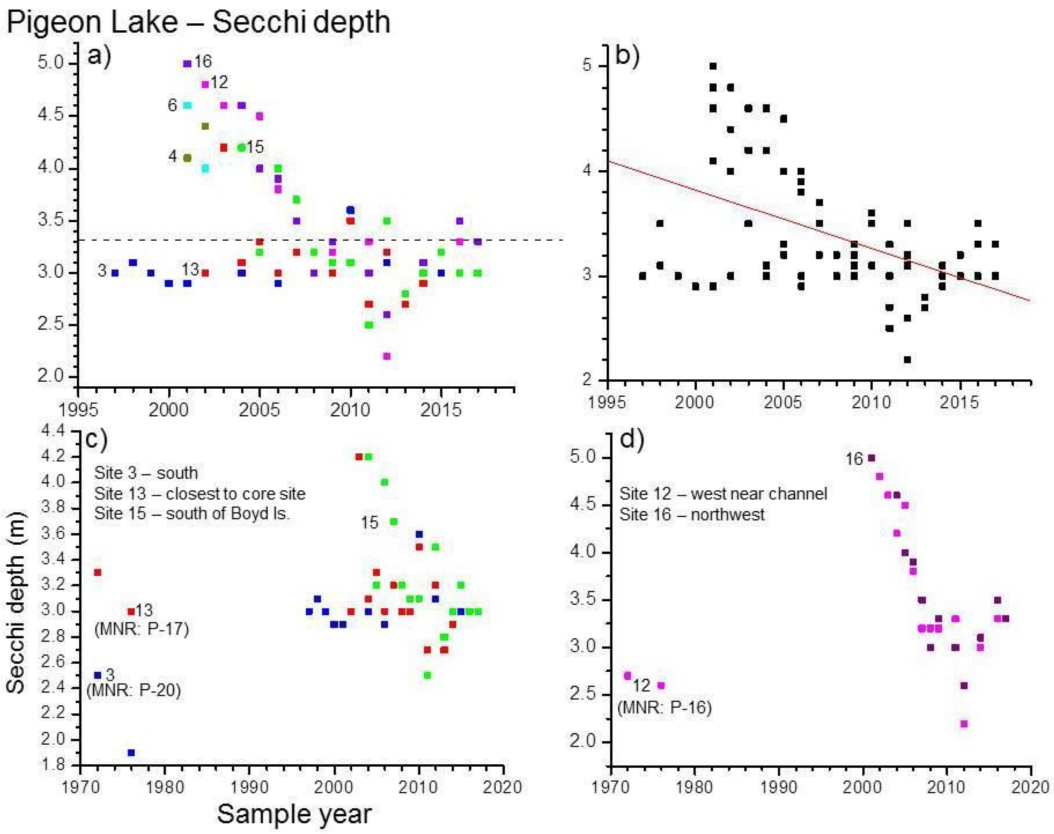


Fig. 17. Average Secchi depth data from Lake Partner Program for Pigeon Lake sites and from MOE-MNR (1976) report in c & d.

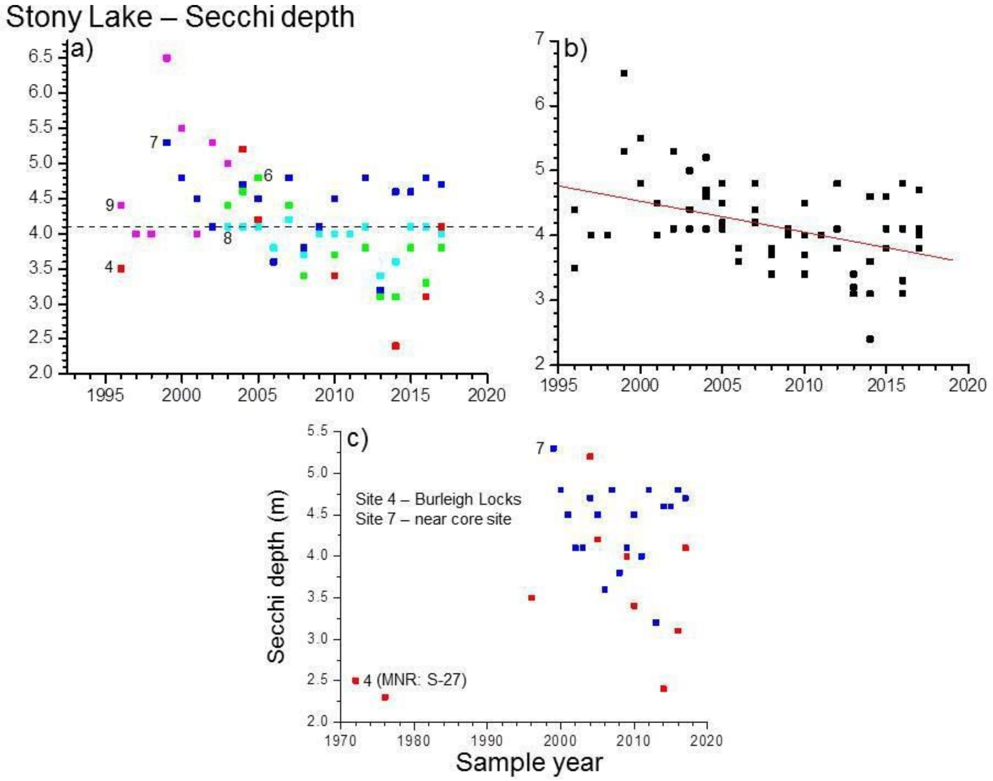


Fig. 18. Average Secchi depth data from Lake Partner Program for Stony Lake sites and from MOE-MNR (1976) report in c & d.

How much of an influence the invasion of zebra mussels had on the diatom and other algal communities is difficult to tease out from the other watershed and climate influences since the 1990s. Certainly there is no apparent influence on the diatom benthic component in any of the cores, as it might be expected that potential increased clarity would result in a greater proportion of benthic taxa in the cores. The main changes in all of the lake-sediment cores were in the planktonic diatom taxa, which are difficult to directly relate to the zebra mussel colonization. In terms of the changes in the algal community based on the sedimentary pigments, the increase in the colonial blue-green pigment myxoxanthin seen in the post-1990 AD sediments of Pigeon and Stony lakes could be in part related to the zebra mussel invasion, as in a few studies, the blue-green colonial algae *Microcystis*, *Aphanizomenon* or *Anabaena* biomass increased following zebra mussel invasion (Higgens & VanderZanden, 2010). The mechanism behind these rises in some lakes and not others with zebra mussels is not clearly known, but in at least one study *Microcystis* has been shown to survive being passed through the zebra mussel digestive tract (Vanderploeg et al., 2001).

DISCUSSION

Eutrophication is a major issue for many inland lakes due largely to human activities. Reduction of nutrient inputs into lakes has had a varied result in both the recovery rate and the degree of achieving baseline conditions (McCrackin et al., 2017). Recovery time under eutrophication management plans varied widely from less than a year to several decades or longer, and in some cases achieving baseline conditions may not be possible. Long recovery time or not achieving baseline conditions was related to various factors, including the legacy of past practices and the degree of watershed disturbance, internal phosphorus loading of phosphorus from the sediments, non-native species invasions and climate-driven processes (McCrackin et al., 2017). Climate warming, known to be occurring across many regions of the Earth (IPCC 2013), and likely also in the Kawartha region (e.g. Monkman, 2015), can also have profound influences on lake dynamics. Changes in the duration of ice cover, driven largely by the warming climate, affects the physical, chemical and biological seasonal dynamics of lakes (Magnuson et al., 2000). Although shorter periods of ice cover in the Kawartha lakes have only been anecdotally noted (Monkman, 2015), numerous peer-reviewed studies have clearly indicated this is common across the Northern Hemisphere (Magnuson et al., 2000; Benson et al., 2012; Kirillen et al., 2012; Magee & Wu, 2017), including in the Great Lakes and surrounding region (Jensen et al., 2007; Wang et al., 2012). Warming of lake-surface waters can also have profound effects on lake dynamics and has been related to the increasing air temperatures (Richardson et al., 2017). Recent changes in lakes are often the result of complex dynamics between anthropogenic-driven nutrient changes and climate change. For example, the lack of recovery in the large, shallow southern basin of the Lake of the Woods despite decades of reduced nutrient inputs, based on a recent paleolimnological study, is thought to be related to changes in lake stratification as a result of climate warming that may be increasing the release of phosphorus from the sediments (Edlund et al., 2017; Reavie et al., 2017). The results presented in this study focused on the long-term estimates of changes in production to provide a context for any recent alterations. Climate influences on phytoplankton assemblages must also be taken into consideration.

Within each of the lakes, the various sedimentary proxies of change were often synchronous (see lake summaries above, Figs. 9, 12 & 15). In particular, the time of the diatom zones based on cluster analysis were the same or very similar to those based on cluster analysis of the concentration of pigments. The timing of change in the other proxies (scale & diatom concentration, organic matter and spectral Chlorophyll *a*) within each lake also generally corresponded to the diatom zonation. This synchrony provides further support to the conclusions based on the patterns observed within each lake. In addition to the general synchrony of the proxy data within each lake, there is some coherence in timing of change across all of the lakes. This is most apparent in the recent timing of change post-1990 AD in both Pigeon and Stony lakes. In Cameron Lake, the dominant change in the diatoms and pigments was post-1940 AD, but further change in both proxies also occurred in the recent post ~2000 AD sediments (Fig. 9).

All lakes indicated increases in algal production with increased spectral and pigment Chlorophyll *a* in the post 1990 AD sediments (Figs. 9, 12 & 15). All lakes also indicated alteration of the algal community, in varying ways, with changes in scale and diatom concentration, and increased alloxanthin (Cameron Lake) or myxoxanthin (Pigeon Lake) pigment or both (Stony Lake). These changes are associated with increased inferences of TP in Pigeon Lake, but declines in Cameron and Stony lakes. Increases in surface waters of lakes associated with warming (Schneider & Hook, 2010; O'Reilly et al., 2015; Richardson et al., 2017) would generally increase algal growth and could result in shifts in the composition of the phytoplankton community (Carey et al., 2012). In particular, increased lake-water temperatures and increased strength of stratification can shift the dominance to favouring cyanobacteria (blue-green algae) (Jöhnk et al., 2008; Carey et al., 2012; Kosten et al., 2012; Taranu et al., 2012). Eutrophication of a lake is also often characterized by increased pelagic (planktonic) production, and can also lead to changes in the composition of the phytoplankton community from one that is diatom dominated to one that is dominated by cyanobacteria or green algae (Davidson & Jeppeson, 2013). Pigeon Lake indicated a recent (post ~1990 AD) increase in the planktonic component of the diatom assemblage from ~30-40% pre-1990 AD to ~50-60% (Fig. 6) providing further evidence of recent eutrophication, although the Lake Partner data do not indicate increases in TP post 1995 AD. Stony Lake shows an earlier (pre-1900 AD) increase in planktonic diatoms, whereas Cameron Lake has varied around 30-40% over the analyzed period of the sedimentary record. Recent alterations in algal production and species composition in the study lakes are likely driven by the interaction of both climate and nutrients. The timing of these changes also corresponded to the invasion of zebra mussels into the Kawartha lakes.

Deciphering which are the primary driving forces on the recent post-1990 AD changes in the diatom composition of the lakes and also in the algal flora based on the sedimentary pigment analysis is difficult, as all potential influences (e.g. nutrients, water clarity, lake-water temperature, stratification strength) could interact. All of the study lakes vary in their susceptibility to nutrient changes, dependent on the degree of cultivated and developed lands in their watersheds, sewage treatment plants, zebra mussel nutrient cycling and would also vary on degree of influence from internal loading of phosphorus. Individual lakes also vary in response to climate forcing dependent largely on watershed dynamics, as well as morphological features of the lakes (Magee & Wu, 2017). The apparent increase in water clarity in some of the lake sampling sites and decreased average measured lake TP since the 1970s may be related to reductions of phosphorus inputs due to sewage treatment plants in the 1970s, as well as the 1990s zebra mussel invasion. However, without more data between 1976 and the late 1990s it is very difficult to determine the influence of each of these factors. Some studies suggest that in systems that were highly turbid prior to zebra mussel invasion (such as the shallow, windy Karwatha lakes) that water clarity may not increase (Higgins & VanderZanden, 2010). Furthermore, complex nutrient recycling pathways influenced by zebra mussels and internal P loading from the sediments may not result in declining TP, as was shown in Lake Simcoe (Gudimov et al., 2015). The overall decline in clarity post 2000 AD in both Pigeon and Stony lake, although driven by particular lake sample sites, occurred while TP has remained relatively stable. Whether the extent of internal loading of phosphorus has increased with lake warming and potential longer periods of lake stratification are not known. The recent changes in the diatom assemblages clearly indicate changes in the lakes post 1990 AD. In both Pigeon and Stony lakes the percent abundance of the more eutrophic *Aulacoseira* (*granulata* vs *ambigua*) increased, while at the same time other diatom taxa that were previously rare increased. *Aulacoseira* species in general need more turbulent conditions compared to *Fragilaria crotonensis*, and the smaller *Cyclotella*, *Discostella* and *Cyclostephanos* which characterized the changes in the recent lake histories. One potential scenario is that the length and strength of stratification have increased in the lakes due to lake warming, enabling the blooms of *Fragilaria crotonensis, Cyclotella*, *Discostella* and *Cyclostephanos*, and at the same time more phosphorus may have been released from the sediments due to increased anoxic conditions during stratification, such that when the lakes do turnover, the hypereutrophic *Aulacoseira granulata* and *Cyclostephanos* may have had more conducive conditions for blooms during the mixed periods. The post 1990 AD increase in colonial blue-green myxoxanthin pigment in both Pigeon and Stony lakes may also reflect a climate-nutrient interaction. The rise in myxoxanthin in our lakes is similar to the distinct sharp increases in myxoxanthin that were documented in all 6 cores from the shallow southern basin of the Lake of the Woods thought to be exacerbated by the interaction of climate warming and internal phosphorus loading (Reavie et al., 2017). Although as noted in the zebra mussel section above, the myxoxanthin rise may also reflect rises in the colonial blue-green algae, such as *Microcystis*, *Aphanizomenon* or *Anabaena* that have increased in some lakes after the establishment of zebra mussels.

One of the strengths of paleo data is the ability to clearly define when changes occurred in the lakes prior to any monitoring, particularly when based on multiple proxy data, such as in this study. Deciphering the exact mechanisms behind these changes can be more difficult, particularly in systems such as the Kawartha lakes, where multiple influences have impacted the lakes, particularly since European settlement. What is clear in all of the study lakes is that the most recent diatom and algal floras are much different than the prior lake history analyzed. While estimates of TP based on the diatom analyses are not 100% reflective of measured TP they can provide a valid comparative context through time. In terms of a temporal context, post-1990 AD estimated TP in Pigeon Lake is far outside of the range of the pre-1900 AD estimates (Fig. 12). In Cameron Lake, estimates were higher in the past pre ~1900 AD (Fig. 9). For Stony Lake, post-1990 AD estimates are generally similar to estimates from ~1900-1990 AD, although the most recent values are approaching ~pre-1900 AD estimates (Fig. 15).

Timing of earlier change in the lakes was more variable, which may be the result of varying degree of intensity and timing of influences from early European settlement, or also in part associated with errors in dating which get more uncertain the deeper in the core. Future analysis of pollen to determine the timing of European settlement will help to refine the ages of the deeper sediments. Carbon dating could also be under taken for the deeper sediments. Nonetheless, clearly the influence of European settlement post ~the 1820s AD would have had impacts on the Kawartha lakes, as large areas were clear-cut by the logging industry and also for agricultural cultivation, all of which would have resulted in nutrient and sediment inputs into the lakes. Furthermore, damming for movement of logs and for the development of the Trent-Severn Waterway in the later 1800s would have resulted in lake-level rises and the creation of a larger littoral zone. The earlier increases in the diatom-based TP estimates are likely related to all of these major alterations which occurred over the mid- to late-1800s within the lake watersheds. Interestingly, the development of the Trent-Severn Waterway in the Kawartha region did not result in the massive compositional change in the diatom assemblages, as was described from a paleostudy of the construction of the Rideau Canal (Christie & Smol, 1996). In the Rideau study, there was nearly a complete species turnover where benthic Fragilariaceae became the dominant diatom group after the construction of the canal. In our study lakes, the early diatom changes were increases in more eutrophic *Aulacoseira* planktonic diatoms, indicating increased nutrients and a potentially larger pelagic zone as a result of the lake-level rises.

The comparison of the post-2000 AD diatom-based estimates of TP to the Lake Partner dataset of TP measurements from 2002-2016 provided an evaluation of the potential accuracy of the paleo-inferred estimates of TP. This evaluation concluded that the diatom-based TP estimates provide a valid means of comparing recent estimates to the past. The estimates provide a long-term context of changes through time and provide potential baseline (pre-1900 AD) remediation targets. However, the recent alterations in the diatom and algal community composition are also being driven by changes in climate, and suggest the recent composition of diatoms is unique and is the result of a complex interaction of both nutrient (including potential zebra mussel pathways, internal P loading, and direct anthropogenic nutrient inputs) and climate influences. The influence of climate may be in opposition to the goals of nutrient targets and to the decline of nuisance algal blooms as these are in part driven by warm waters and changes in stratification regimes.

REFERENCES

Angus, J.T. 1988. A respectable ditch: a history of the Trent-Severn Waterway, 1833-1920. McGill-Queen's University Press, 455 p.

Appleby, P.G. & F. Oldfield. 1978. The calculation of lead-210 dates assuming a constant rate of supply of unsupported ^{210}Pb to the sediment. *Catena* 5: 1-8.

Battarbee, R.W. & M. J. Keen. 1982. The use of electronically counted microspheres in absolute diatom analysis. *Limnology and Oceanography* 27: 184-188.

Benson, B.J., J.J. Magnuson, O.P. Jensen, V. M. Card, G. Hodgkins, J. Korhonen, D. M. Livingstone, K.M. Stewart, G.A. Weyhenmeyer, & N. G. Granin. 2012. Extreme events, trends, and variability in Northern Hemisphere lake-ice phenology (1855–2005). *Climate Change* 112: 299–323.

Brown, M.E., T.M. Curtin, C.J. Gallagher & J.D. Halfman. 2012. Historic nutrient loading and recent species invasions caused shifts in water quality and zooplankton demography in two Finger Lakes (New York, USA). *Journal of Paleolimnology* 48: 623-639.

Camburn, K.R. & D.F. Charles. 2000. Diatoms of low-alkalinity lakes in the Northeastern United States. Academy of Natural Sciences, Philadelphia.

Carey, C.C., B. W. Ibelings, E.P.Hoffmann, D.P. Hamilton & J.D. Brookes. 2012. Eco-physiological adaptations that favour freshwater cyanobacteria in a changing climate. *Water Research* 46: 1394-1407.

Cha, Y., C.A. Stow & E.S. Bernhardt. 2013. Impacts of dreissenid mussel invasions on chlorophyll and total phosphorus in 25 lakes in the USA. *Freshwater Biology* 58: 192-206.

Christie, C.E. & J.P. Smol. 1996. Limnological effects of 19th century canal construction and other disturbances on the trophic state history of Upper Rideau Lake, Ontario. *Journal of Lake and Reservoir Management* 12(4): 448-454.

Conolly, J. 2017. Archaeological and environmental history in the Kawartha lakes. In: Study the Past, Plan for the Future, Kawartha Lake Stewards Association 2016 Annual Lake Water Quality Report (May 2017), pp. 8-10.

Cumming, B & W. Napier. 2017. Kawartha lakes paleolinological study: collection, analysis and age-dating of sediment cores. In: Study the Past, Plan for the Future, Kawartha Lake Stewards Association 2016 Annual Lake Water Quality Report (May 2017), pp. 15-18.

Cumming, B.F., S.E. Wilson, R.I. Hall & J.P. Smol. 1995. Diatoms from British Columbia (Canada) Lakes and their Relationship to Salinity, Nutrients and Other Limnological Variables (with 248 figures, 6 tables and 1041 photos on 60 plates). *Bibliotheca Diatomologica*: 31. Stuttgart, Germany. 207 pp.

Cumming, B.F., K.R. Laird, I. Gregory-Eaves, K.G. Simpson, M.A. Sokal, R.N. Nordin & I.R. Walker. 2015. Tracking past changes in lake-water phosphorus with a 251-lake calibration dataset in BritishColumbia: tool development and application in a multiproxy assessment of eutrophication and recovery in Osoyoos Lake,a transboundary lake in Western North America. *Frontiers in Ecology and Evolution* 3: article 84.

Davidson, T.A & E. Jeppeson. 2013. The role of palaeolimnology in assessing eutrophication and its impact on lakes. *Journal of Paleolimnology* 49:391–410.

Davison, W. 1993. Iron and Manganese in lakes. *Earth-Science Reviews* 34: 119-163.

Dolbey, M. 2015. Analysis of Ontario lake partner program total phosphorus data for the Kawartha lakes. In: The Nature of our Lakes, Kawartha Lakes Stewards Association 2014 Annual Lake Water Quality Report (May 2015), pp. 26-43.

Dolbey, M. 2017. 2015 Kawartha lakes Sewage treatment plants report. In: Study the Past, Plan for the Future, Kawartha Lake Stewards Association 2016 Annual Lake Water Quality Report (May 2017), pp. 36-39.

Edlund, M.B., S.P. Schottler, E.D. Reavie, D.R. Engstrom, N.G. Baratono, P.R. Leavitt, A.J. Heathcote, B. Wilson & A.M. Paterson. 2017. Historical phosphorus dynamics in Lake of the Woods (USA-Canada) – does legacy phosphorus still affect the southern basin. *Lake and Reservoir Management* 33: 386-402.

Fallu, M., N. Allaire & R. Pienitz. 2000. Freshwater Diatoms from Northern Quebec and Labrador (Canada). *Bibliotheca Diatomologica* Band 45. Gebru¨der Borntraeger, Berlin.

Feuchtmayr, H., S.J. Thackeray, I.D. Jones, M. de Ville, J. Fletcher, B. James & J. Kelly. 2012. Spring phytoplankton phenology-are patterns and drivers of change consistent among lakes in the same climatological region? *Freshwater Biology* 57: 331-344.

Gudimov, A., D. Kim, J.D. Young, M.E. Palmer, M. Dittrich, J.G. Winter, E. Stainsby & G.B. Arhonditsis. 2015. Examination of the role of dreissenids and macrophytes in the phosphorus dynamics of Lake Simcoe, Ontario, Canada. *Ecological Informatics* 26: 36-53.

Heiri, O., Lotter, A. F. & G. Lemcke. 2001. Loss on ignition as a method for estimating organic and carbonate content in sediments: reproducibility and comparability of results. *Journal of Paleolimnology* 25: 101-101.

Higgins, S.N. & M.J. VanderZanden. 2010. What a difference a species makes: a meta-analysis of dreissenid mussel impacts on freshwater ecosystems. *Ecological Monographs* 80: 179-196.

IPCC: Summary for Policymakers, p. 3–29. In T. Stocker, D. Qin, G.K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, & P.M. Midgley (eds.), Climate Change 2013: The Physical Science Basis, Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, New York, NY, USA.

Jensen, O.P., B.J. Benson, J.J. Magnuson, V.M. Card, M.N. Futter, P.A. Soranno, & K.M. Stewart. 2007. Spatial analysis of ice phenology trends across the Laurentian Great Lakes region during a recent warming period. *Limnology and Oceanography* 52: 2013–2026.

Jöhnk, K.D., J.Huisman, J. Sharples, B. Sommeijer, P.M. Visser & J.M. Stroom. 2008. Summer heatwaves promote blooms of harmful cyanobacteria. *Global Change Biology* 14: 495–512.

Karmakar, M. P. R. Leavitt & B.F. Cumming. 2015. Enhanced algal abundance in northwest Ontario (Canada) lakes during the warmer early-to-mid Holocene. *Quaternary Science Reviews* 123:168-179.

Kirillin, G., M. Leppäranta, A. Terzhevik, N. Granin, J. Bernhardt, C. Engelhardt, T. Efremova, S. Golosov, N. Palshin, P. Sherstyankin, G. Zdorovennova & R. Zdorovennov. 2012. Physics of seasonally ice-covered lakes: a review. *Aquatic Sciences* 74: 659-682.

Kosten,S. V.L.M. Huszar, E. Becares, L.S. Costa, E. vanDonks, L .Hansson, E. Jeppesen, C Kruk, G. Lacerot, N. Mazzeo, L. Meester, B. Moss, M. Lürling, T. Nöges, S. Romo & M. Scheffer. 2012. Warmer climates boost cyanobacterial dominance in shallow lakes. *Global Change Biology* 18: 118–126.

Krammer, K. & H. Lange-Bertalot. 1986. Bacillariophyceae. 1. Teil: Naviculaceae. In H. Ettl, G. Gärtner, J. Gerloff, H. Heynig & D. Mollenhauer (eds.), Süßwasserflora von Mitteleuropa, Band 2/1, Gustav Fischer Verlag, Stuttgart/New York, 876 pp.

Krammer, K. & H. Lange-Bertalot. 1988. Bacillariophyceae. 2. Teil: Bacillariaceae, Epithemiaceae, Surirellaceae. In H. Ettl, G. Gärtner, J. Gerloff, H. Heynig & D. Mollenhauer (eds.), Süßwasserflora von Mitteleuropa, Band 2/2, Gustav Fischer Verlag, Stuttgart/New York, 596 pp.

Krammer, K. & H. Lange-Bertalot. 1991a. Bacillariophyceae. 3. Teil: Centrales, Fragilariaceae, Eunotiaceae. In H. Ettl, G. Gärtner, J. Gerloff, H. Heynig & D. Mollenhauer (eds.), Süßwasserflora von Mitteleuropa, Band 2/3, Gustav Fischer Verlag, Stuttgart/Jena, 576 pp.

Krammer, K. & H. Lange-Bertalot. 1991b. Bacillariophyceae. 4. Teil: Achnanthaceae Kritische Ergänzungen zu Navicula (Lineolatae) und Gomphonema. In H. Ettl, G. Gärtner, J. Gerloff, H. Heynig & D. Mollenhauer (eds.), Süßwasserflora von Mitteleuropa, Band 2/4, Gustav Fischer Verlag, Stuttgart/Jena, 437 pp.

Lange-Bertalot, H. & D. Melzeltin. 1996. Indicators of oligotrophy. Vol. 2 Iconographia Diatomologica. Koeltz Scientific Books, Königstein.

Leavitt, P.R. & Hodgson, D. A. 2001. Sedimentary Pigments, p. 295-325. in J. P. Smol, H. J. B. Birks & W. M. Last (eds.). Tracking Environmental Change Using Lake Sediments. Volume 3: Terrestrial, Algal, and Siliceous Indicators. Kluwer Academic.

Lodge, M. 2015. The environmental history of the Kawarthas. Maryboro Lodge, The Fenelon Museum. http://www.maryboro.ca/learn/html.

Mackenzie, K. 2016. Phosphorus testing. In: Protecting the Natural Beauty Around Us, Kawartha Lakes Stewards Association 2015 Annual Lake Water Quality Report Report (May 2016), pp. 23-25.

Magee, M.R. & C.H. Wu. 2017. Effects of changing climate on ice cover in three morphometrically different lakes. *Hydrological Processes* 31: 308-323.

Magnuson, J.J., D.M. Robertson, B.J. Benson, R.H. Wynne, D.M. Livingston, T. Arai, R.A. Assel, R.G. Barry, V. Card, E. Kuusisto, N.G. Granin, T.D. Prowse, K.M. Stewart & V.S. Vuglinski. 2000. Historical trends in lakes and river ice cover in the Northern Hemisphere. *Science* 289: 1743-1746.

McCrackin, M.L., H.P. Jones, P.C. Jones & D. Moreno-Mateos. 2017. Recovery of lakes and coastal marine ecosystems from eutrophication: A global meta-analysis. *Limnology and Oceanography* 62: 507–518.

Michelutti, N., J.M. Blais, B.F. Cumming, A.M. Paterson, K. Rühland, A.P. Wolf & J.P. Smol. 2010. Do spectrally inferred determinations of chlorophyll a reflect trends in lake trophic status? *Journal of Paleolimnology* 43: 205-217.

Ministry of the Environment, Ministry of Natural Resources. 1976. The Kawartha Lakes Water Management Study-Water Quality Assessment (1972-1976). MOE 761101.

Monkman, D. 2015. Climate change in the Kawarthas. In: The Nature of our Lakes, Kawartha Lakes Stewards Association 2014 Annual Lake Water Quality Report (May 2015), pp. 10-14.

Napier, W. & B. Cumming. 2017. Paleolimnology: What is it, and why is it useful? In: Study the Past, Plan for the Future, Kawartha Lake Stewards Association 2016 Annual Lake Water Quality Report (May 2017), pp. 11-15.

Napier, W., B. Cumming & K. Laird. Kawartha lakes paleolimnology study: status and update. 2018. In: Healthy Lakes, Healthy Communities!, Kawartha Lake Stewards Association 2017 Annual Lake Water Quality Report (May 2018), pp. 34-38.

O'Reilly, C. M., S. Sharma, D.K. Gray, S.E. Hampton, et al. 2015. Rapid and highly variable warming of lake surface waters around the globe. *Geophyscal Research Letters* 42: doi:10.1002/2015GL066235.

Orihel, D.M., H.M. Baulch, N.J. Casson, R.L. North, C.T.Parsons, D.C.M. Seckar & J.J. Venkiteswaran. 2017. Internal phosphorus loading in Canadian fresh waters: a critical review and data analysis. *Canadian Journal of Fisheries and Aquatic Sciences* 74: 2005-2029.

Reavie, E.D., M.B. Edlund, N.A. Andresen, D.R. Engstrom, P.R. Leavitt, S. Schottler & M. Cai. 2017. Paleolimnology of the Lake of the Woods southern basin: continued water quality degradation despite lower nutrient influx. *Lake and Reservoir Management* 33: 369-385.

Richardson, D.C., S.J. Melles, R.M. Pilla, A.L. Hetherington, L. B. Knoll, C.E. Willaimson, B.M. Kraemer, J.R. Jackson, E.C. Long, K. Moore, L.G. Rudstam,J.A. Rusak, J.E. Saros, S. Sharma, K.E. Strock, K.C. Weathers & C.R. Wigdahl-Perry. 2017. Transparency, geomorphology and mixing regime explain variability in trends in lake temperature and stratification across Northeastern North America (1975-2014). *Water* 9, 442: doi:10.3390/w9060442.

Schaller, T., H.C. Moor & B. Wehrli. 1997. Sedimentary profiles of Fe, Mn, V, Cr, As, and Mo as indicators of benthic redox conditions in Balderggersee. *Aquatic Sciences* 59: 345-361.

Schelske, C.L., A. Peplow, M. Brenner & C.N. Spencer. 1994. Low background gamma counting: applications for ^{210}Pb dating of sediments. *Journal of Paleolimnology* 10: 115-128.

Schneider.P. & S.J. Hook. 2010. Space observations of inland water bodies show rapid surface warming since 1985. *Geophyscal Research Letters* 37: doi:10.1029/2010GL045059.

Schindler, D. W. 1974. Eutrophication and Recovery in Experimental Lakes: Implications for Lake Management. *Science* 184: 897-899.

Sommer, U., R. Adrian, L. De Senerpont Domis, J.J. Elser, U. Gaedke, B. Ibelings, E. Jeppesen, M. Lürling,J.C. Molinero, W.M., Mooij, E. van Donk & M. Winder. 2012. Beyond the plankton ecology group (PEG) model:

Mechanisms driving plankton succession. *Annual Review Ecology Evolution Systems* 43: 429-448

Taillon, D., & M.G. Fox. 2004. The influence of residential and cottage development on littoral zone fish communities in a mesotrophic north temperate lake. *Environmental Biology of Fishes* 71: 275-285.

Tarenu, Z.E., R.W. Zurawell, F. Picks & I. Gregory-Eaves. 2012. Predicting cyanobacterial dynamics in the face of global change: the importance of scale and environmental context. *Global Change Biology* 18: 3477–3490.

Vanderploeg, H. A., J. R. Liebig, W. W. Carmichael, M. A. Agy, T. H. Johengen, G. L. Fahnenstiel & T. F. Nalepa. 2001. Zebra mussel (Dreissena polymorpha) selective filtration promoted toxic Microcystis blooms in Saginaw Bay (Lake Huron) and Lake Erie. *Canadian Journal of Fisheries and Aquatic Sciences* 58:1208–1221.

Walters, K. 2007. Physical overview of the Kawartha lakes. In: Lake Water Quality 2006 Report (April 2007), pp. 12-17.

Wang, J., X. Bai, H. Hu, A. Clites, M.Colton & B. Lofgren. 2012. Temporal and spatial variability of Great Lakes ice cover, 1973-2010. *Journal of Climate* 25: 1318-1329, doi:10.1175/2011JCLI4066.1

White, M. 2006. Phosphorus and the Kawartha Lakes (Land Use, Lake Morphology and Phosphorus Loading). Kawartha Lake Stewards Association. 68 pp. Available online at http://www. lakefieldherald.com/KLSA/MikeWhitereport.pdf

Zhu, B., D.G. Fitgerald, C.M. Mayer, L.G. Rudstam & E.L. Mills. 2006. Alteration of ecosystem function by zebra mussels in Oneida Lake: impacts on submerged macrophytes. *Ecosystems* 9: 1017-1028.

SUPPLEMENTAL DATA

Table S1. Lake Partner Seasonal water sampling for TP analyses. Site numbers correspond to database.
Pigeon Lake – Years sampled at each site and years of consistent monthly sampling from May to October.

	3. Sandy Pt	4. Back Chan	12. Boyd Is	13. Con17	15. Dead horse	16. Bottom Is
Years sampled	2004-2016	2002	2002-2016	2003-2016	2003-2016	2004-2016
Years with consistent monthly sampling	2004-2016	2002	2007-2009	2003-2016	2012 May-September	2007, 2009, 2013

Stony Lake – Years sampled at each site and years of consistent monthly sampling from May to October.

	4. Burleigh Locks	6. Gilchrist Bay	7. Mouse Is	8. Hamilton Bay
Years sampled	2007-2016	2003-2016	2002-2016	2003-2016
Years with consistent monthly sampling	2007 June-Oct 2010 May-Sept	2013-2014 May-Sept	2002, 2003, 2005,2006, 2009-2016	2005-2007 2009-2016

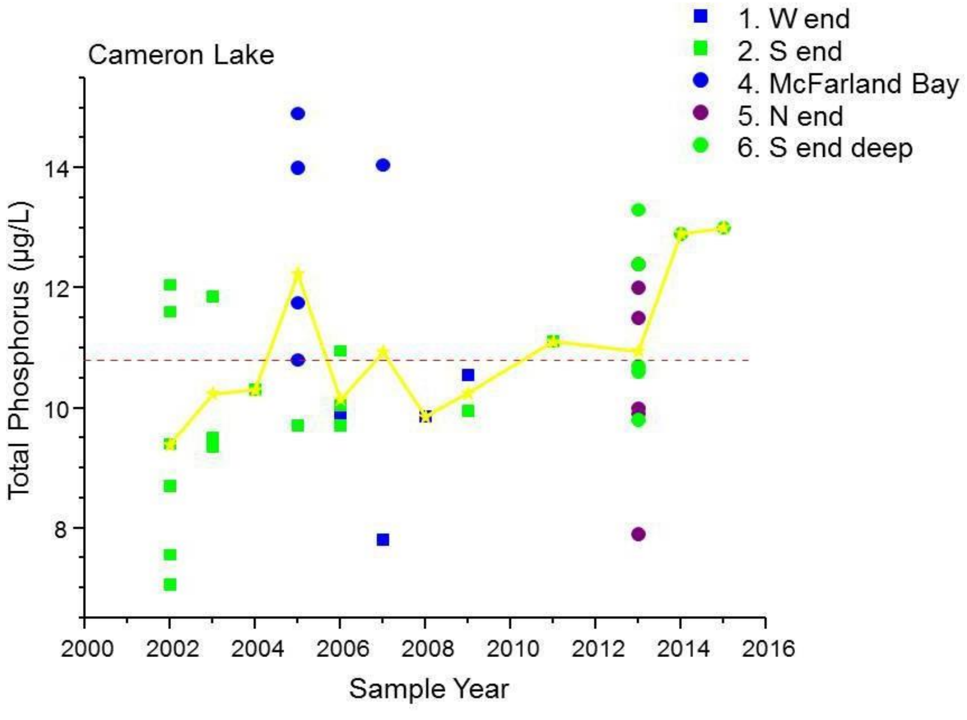


Figure S1. Lake Partner total phosphorus (TP)µg/L for Cameron Lake (2002-2015) from 5 sampling sites. Numbers correspond to site ID of the Lake Partner dataset and those in Fig. 1. The solid yellow line is the yearly average TP.

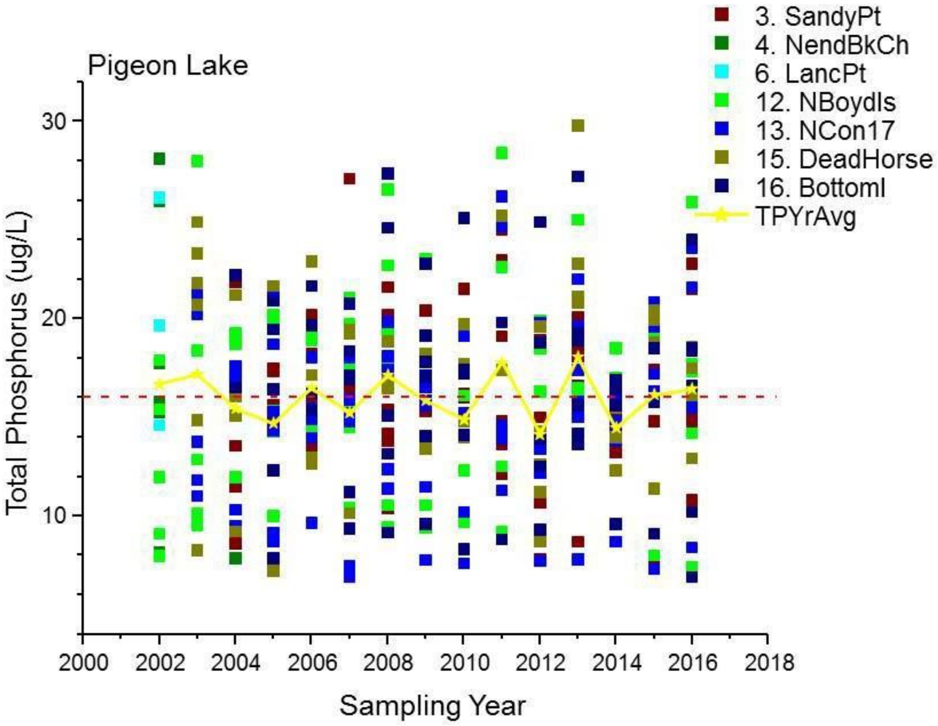


Figure S2. Lake Partner total phosphorus for Pigeon Lake (2002-2016) from 7 sampling sites. Numbers correspond to site ID of the Lake Partner dataset and those in Fig. 1. The solid yellow line is the yearly average TP.

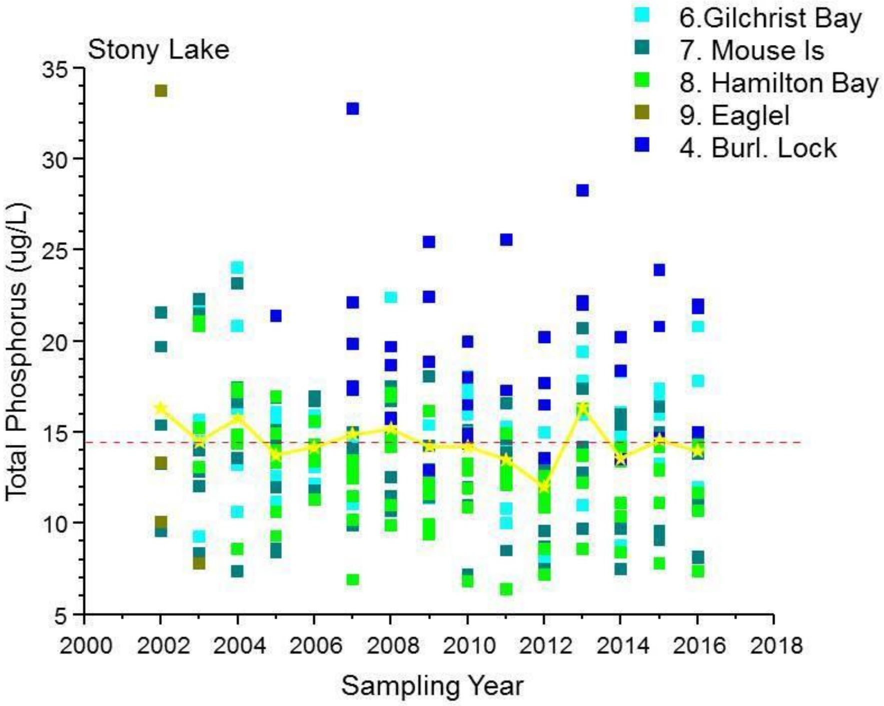


Figure S3. Lake Partner total phosphorus for Stony Lake (2002-2016) from 5 sampling sites. Numbers correspond to site ID of the Lake Partner dataset and those in Fig. 1. The solid yellow line is the yearly average TP.

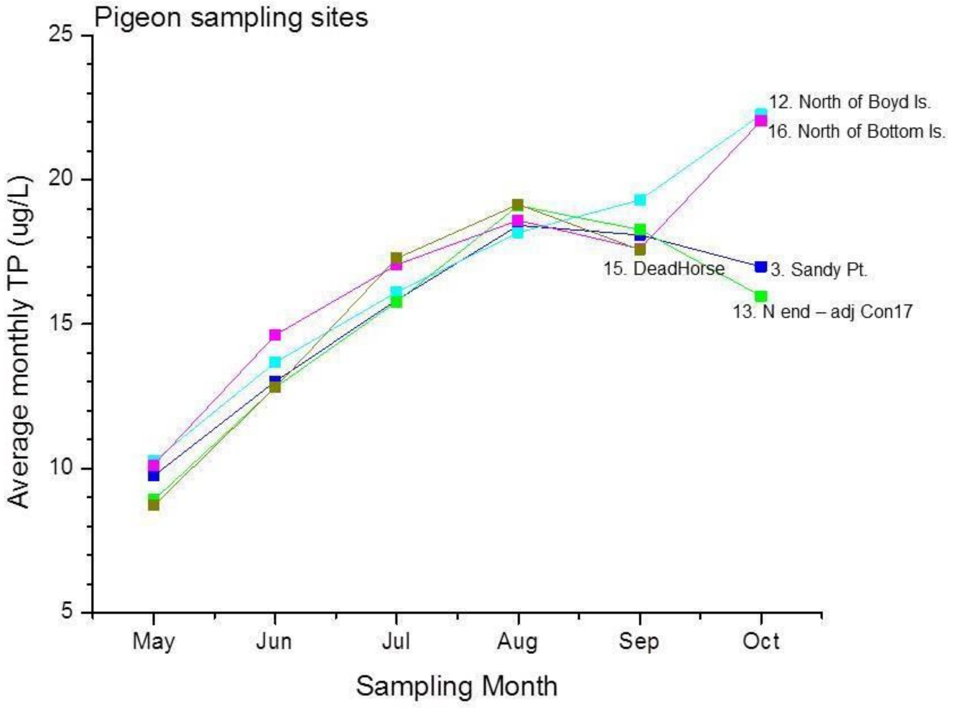


Figure S4. Average monthly Lake Partner total phosphorus for Pigeon Lake from 5 sampling sites with seasonal data within a sampling year. See Table S1 for sampling years for each site.

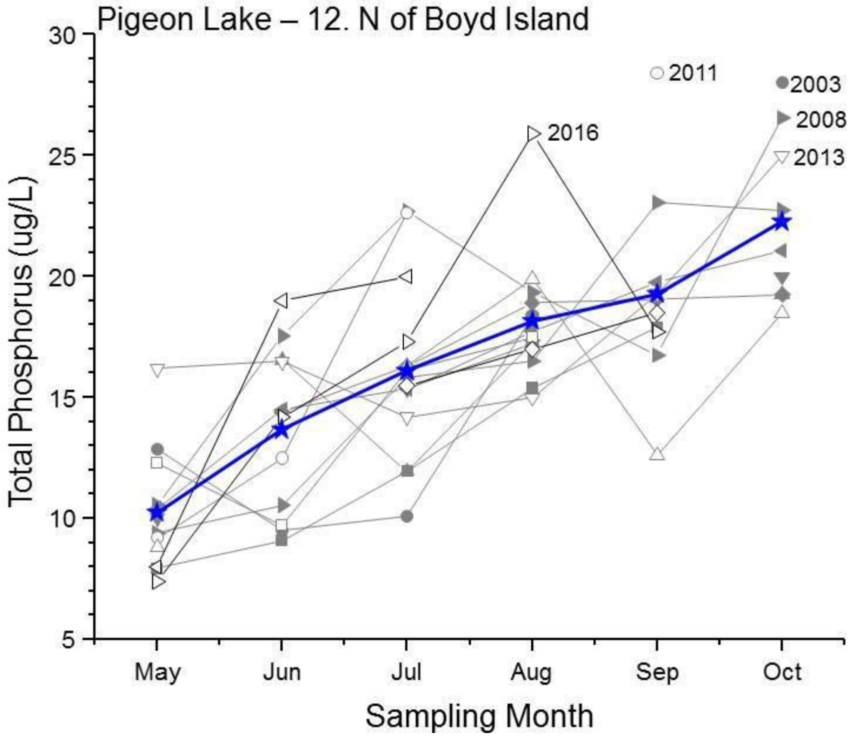


Figure S5. Monthly Lake Partner TP for Pigeon Lake from sampling site 12 with seasonal data within a sampling year. Solid symbols represent years 2002-2009, and open symbols 2010 to 2016. Select years with high TP are labelled. Solid blue line is the average monthly from all years.

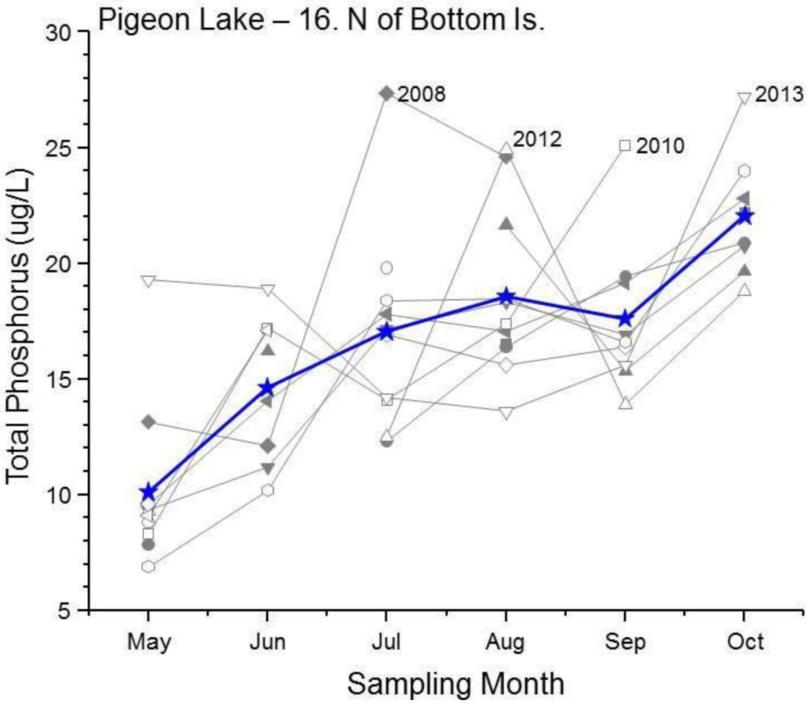


Figure S6. Monthly Lake Partner TP for Pigeon Lake from sampling site 16 with seasonal data within a sampling year. Solid symbols represent years 2004-2009, and open symbols 2010 to 2016. Select years with high TP are labelled. Solid blue line is the average monthly from all years.

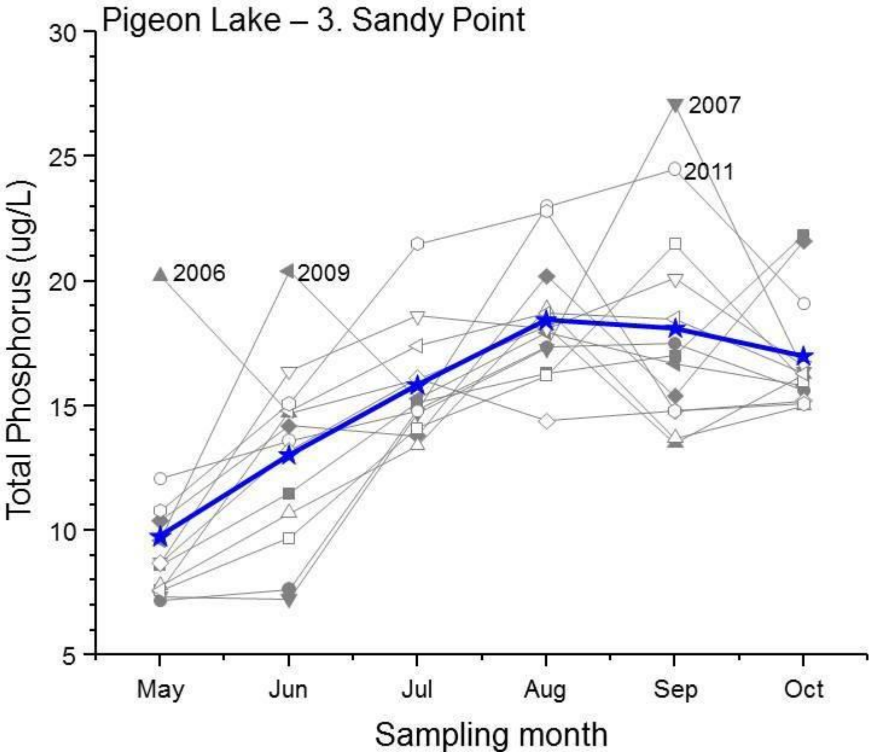


Figure S7. Monthly Lake Partner TP for Pigeon Lake from sampling site 3 with seasonal data within a sampling year. Solid symbols represent years 2004-2009, and open symbols 2010 to 2016. Select years with high TP are labelled. Solid blue line is the average monthly from all years.

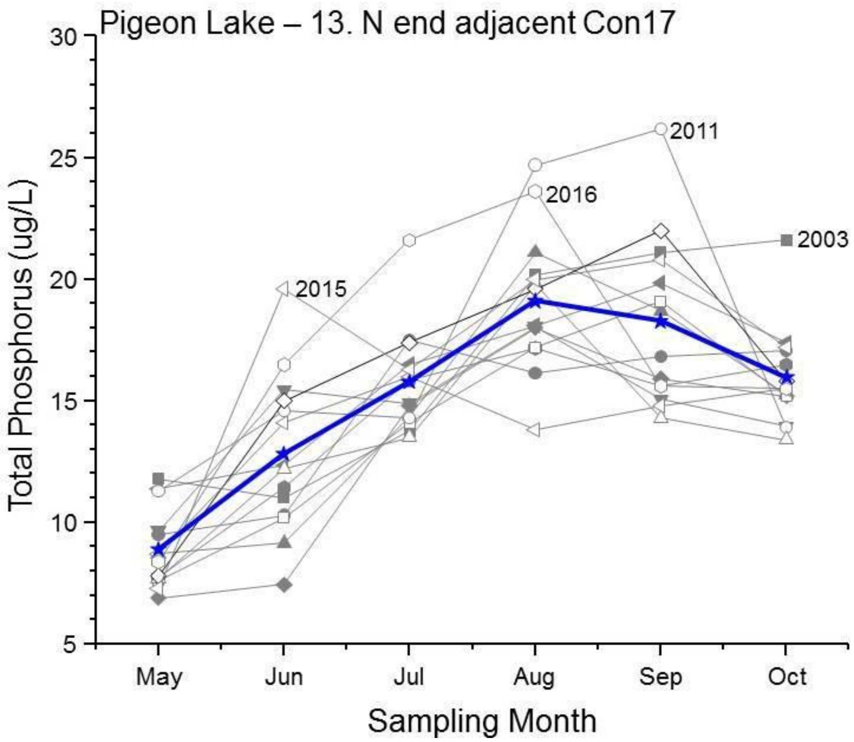
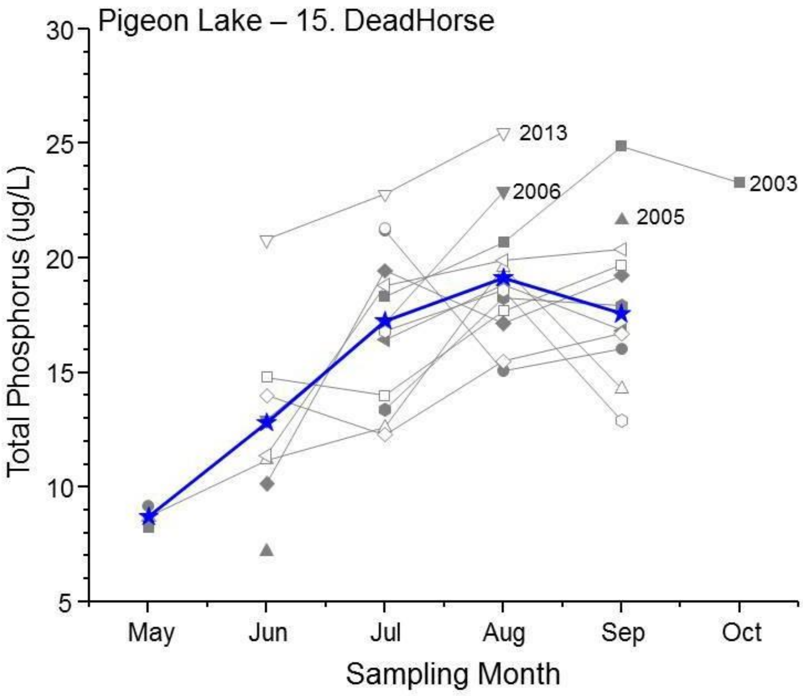


Figure S8. Monthly Lake Partner TP for Pigeon Lake from sampling site 13 with seasonal data within a sampling year. Solid symbols represent years 2003-2009, and open symbols 2010 to 2016. Select years with high TP are labelled. Solid blue line is the average monthly from all years.

Figure S9. Monthly Lake Partner TP for Pigeon Lake from sampling site 15 with seasonal data within a sampling year. Solid symbols represent years 2003-2009, and open symbols 2010 to 2016. Select years with high TP are labelled. Solid blue line is the average monthly from all years.



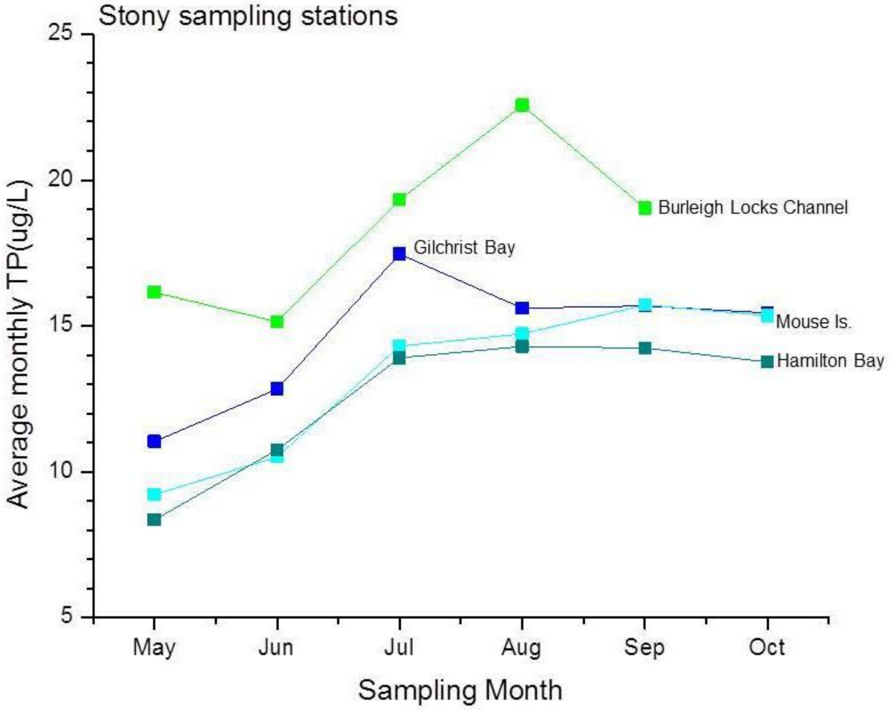


Figure S10. Average monthly Lake Partner total phosphorus (TP) for Stony Lake from 4 sampling sites with seasonal data within a sampling year. See Table S1 for sampling years.

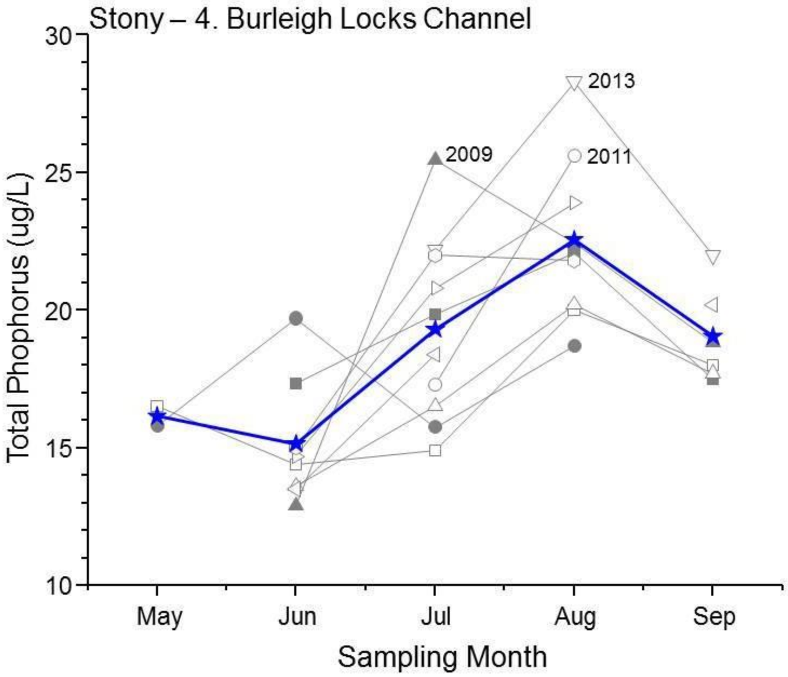


Figure S11. Monthly Lake Partner TP for Stony Lake from sampling site 4 with seasonal data within a sampling year. Solid symbols represent years 2007-2009, and open symbols 2010 to 2016. Select years with high TP are labelled. Solid blue line is the average monthly from all years.

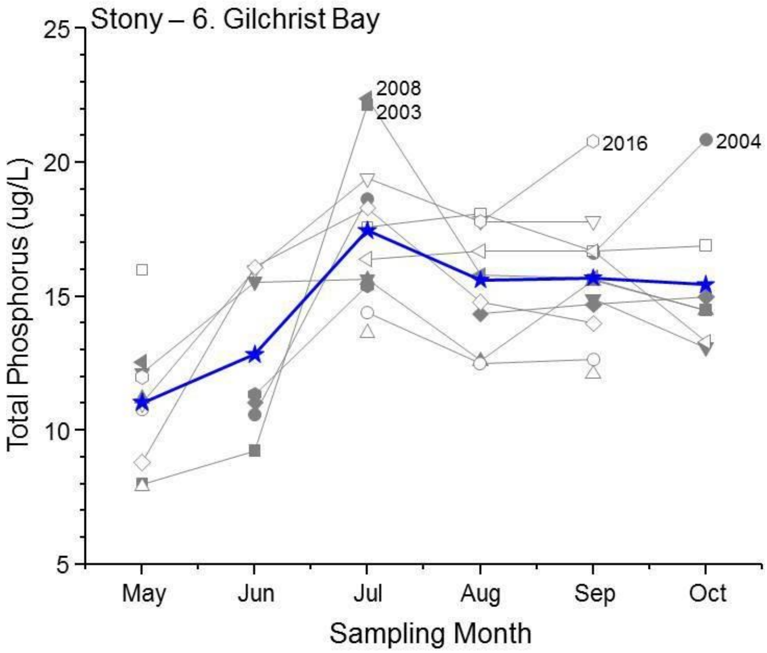


Figure S12. Monthly Lake Partner total phosphorus for Stony Lake from sampling site 6 (Gilchrist Bay) with seasonal data within a sampling year. Solid symbols represent years 2003-2009, and open symbols 2010 to 2016. Select years with high TP are labelled. Solid blue line is the average monthly from all years.

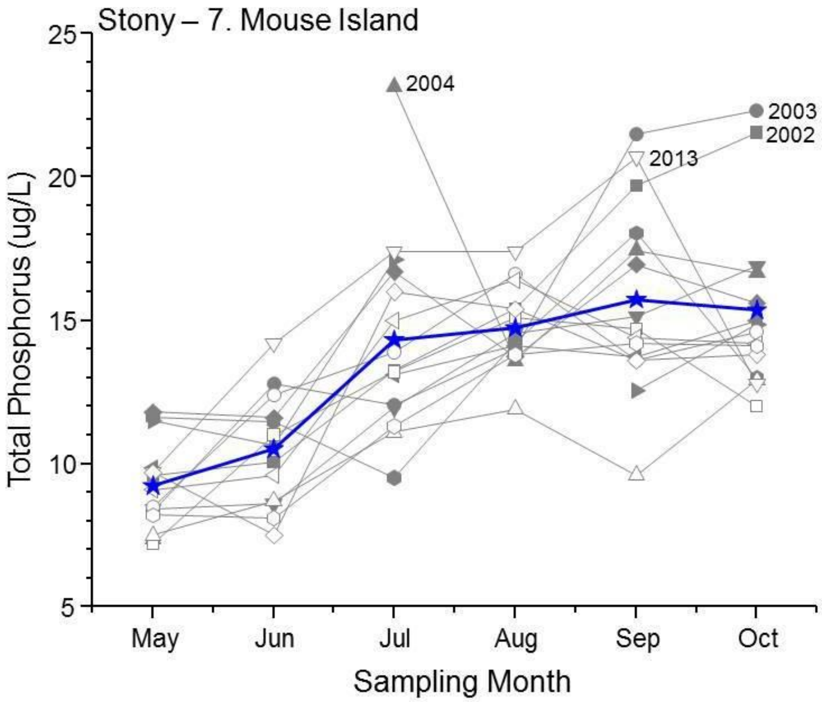


Figure S13. Monthly Lake Partner TP for Stony Lake from sampling site 7 with seasonal data within a sampling year. Solid symbols represent years 2002-2009, and open symbols 2010 to 2016. Select years with high TP are labelled. Solid blue line is the average monthly from all years.

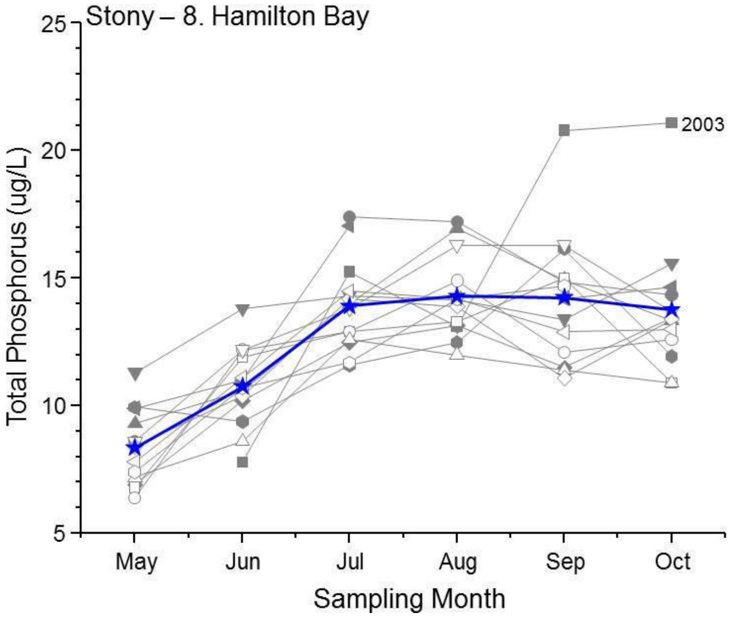


Figure S14. Monthly Lake Partner TP for Stony Lake from sampling site 8 within seasonal data with a sampling year. Solid symbols represent years 2003-2009, and open symbols 2010 to 2016. Select years with high TP are labelled. Solid blue line is the average monthly from all years.

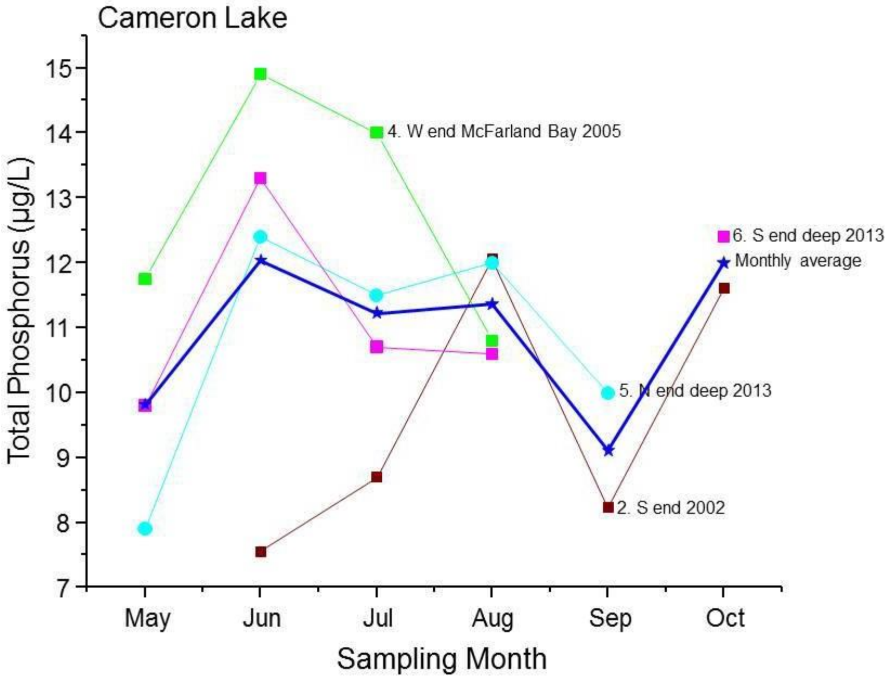


Figure S15. Monthly Lake Partner total phosphorus for Cameron Lake from 4 sampling sites with seasonal data from various sampling years.

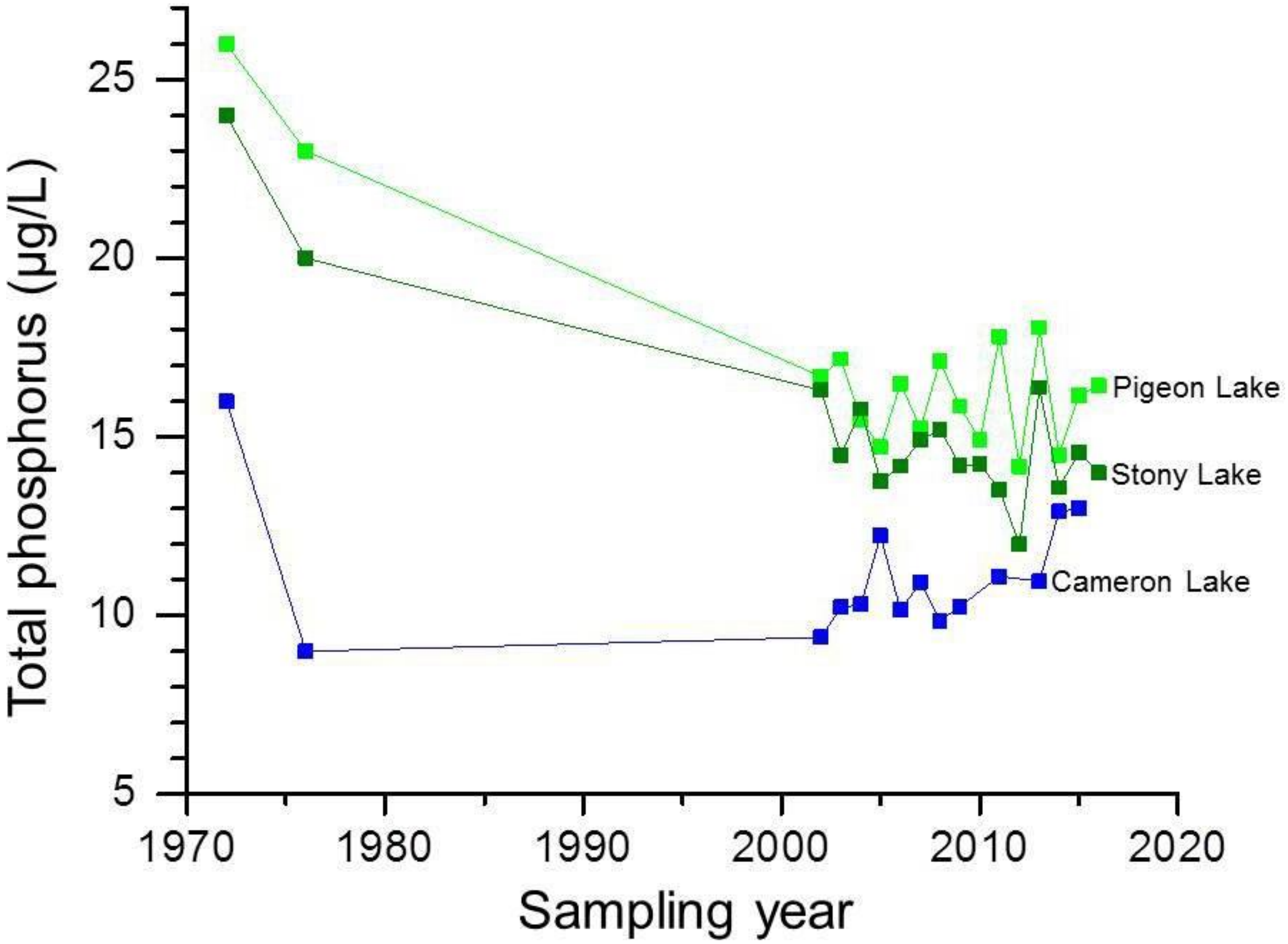


Figure S16. Average yearly total phosphorus measurements based on data from the Lake Partner program for Cameron Lake (2002-2015) and for Pigeon and Stony lakes (2002-2016) in reference to data for 1972 and 1976 from a MOE-MNR (1976) report.

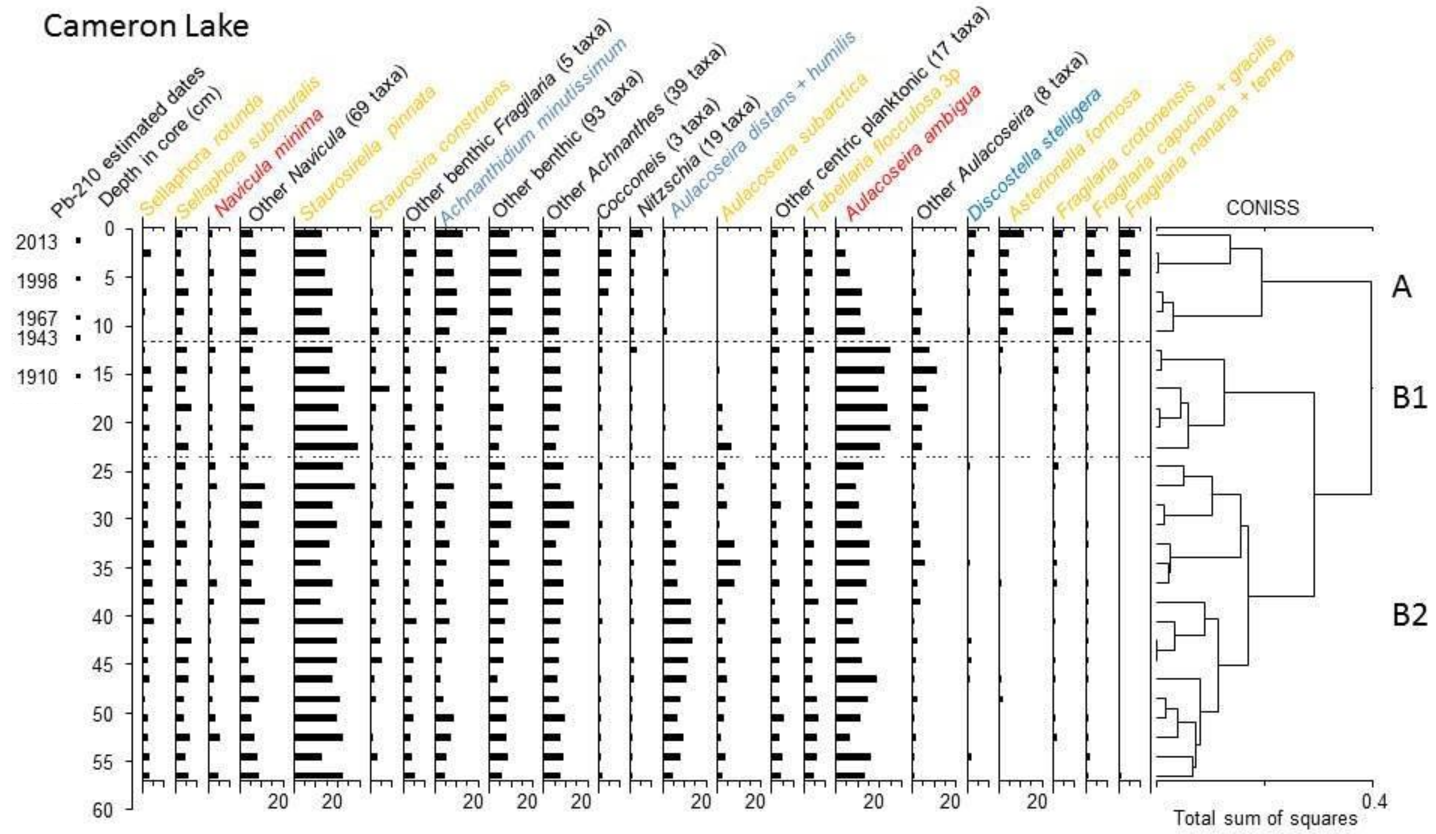


Figure S17. Percent abundance of diatom taxa > 5% abundance in the Cameron Lake sediment core versus depth, and estimated dates noted. Taxa names that are in blue are considered to be more oligotrophic (low nutrient) taxa. Taxa names that are in orange are considered to be within the more mesotrophic (medium nutrient) range, and those in red tending more towards eutrophic (high nutrient) range. Those taxa name in black are groups of different taxa and thus have no specific nutrient range. The depth-constrained cluster analysis shown on the right from which the diatom assemblage zones are based on.

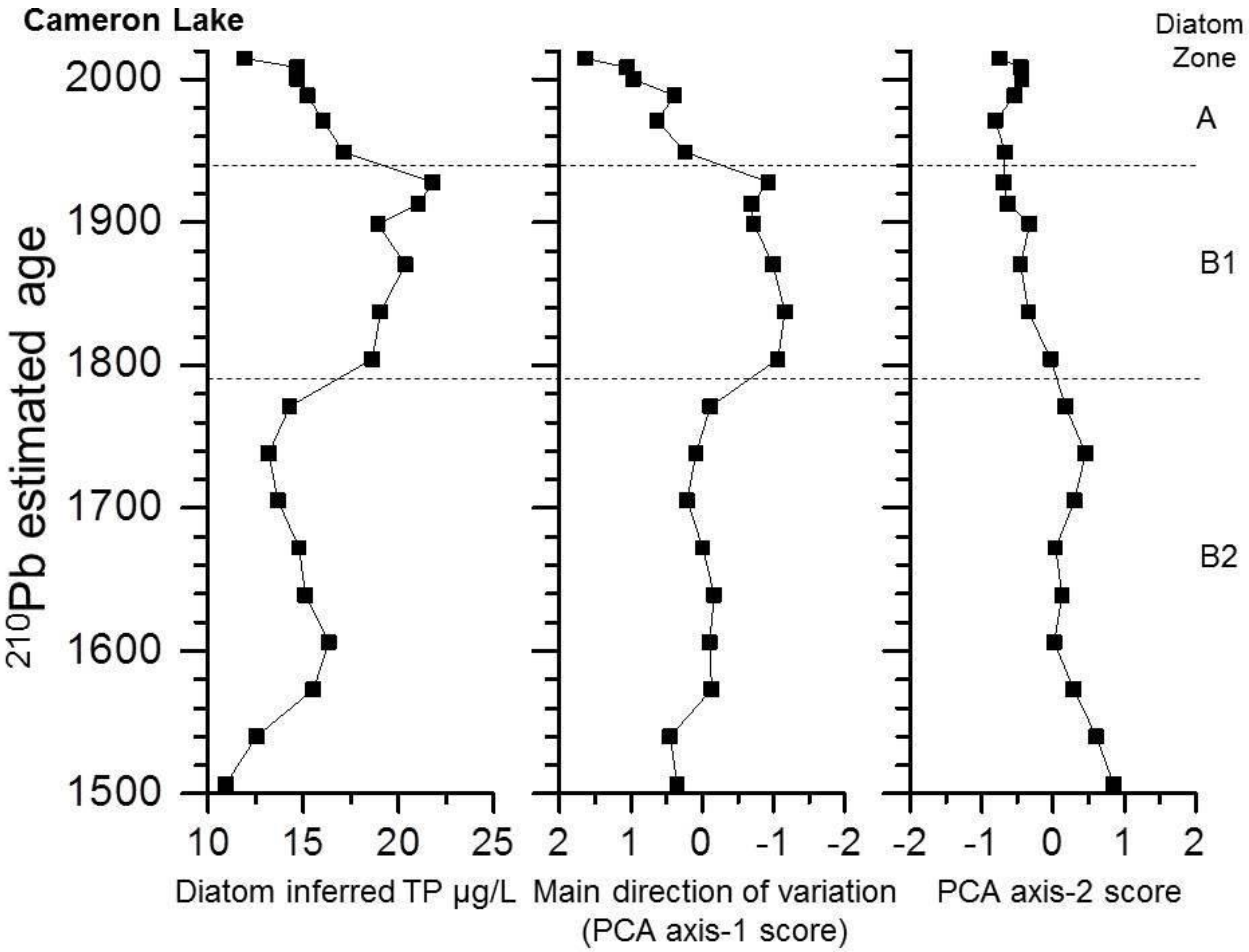


Figure S18. Diatom-inferred total phosphorus (TP), main direction of variation (PCA axis 1 scores) and secondary direction of variation (PCA axis 2 scores) for Cameron Lake versus estimated age. Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.

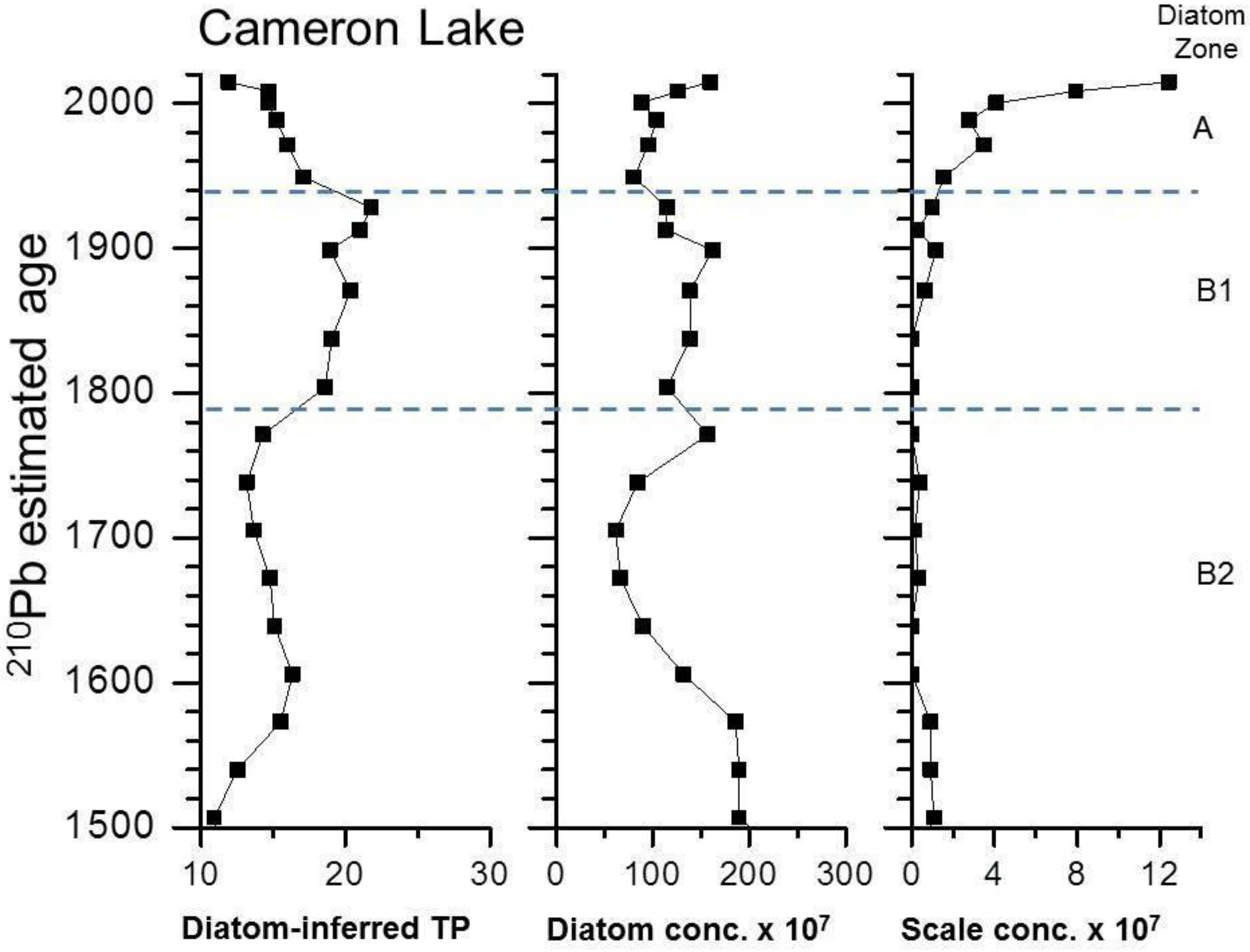


Figure S19. Diatom-inferred total phosphorus (TP), concentration of diatom valves and concentration of chrysophyte scales for Cameron Lake versus estimated age. Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.

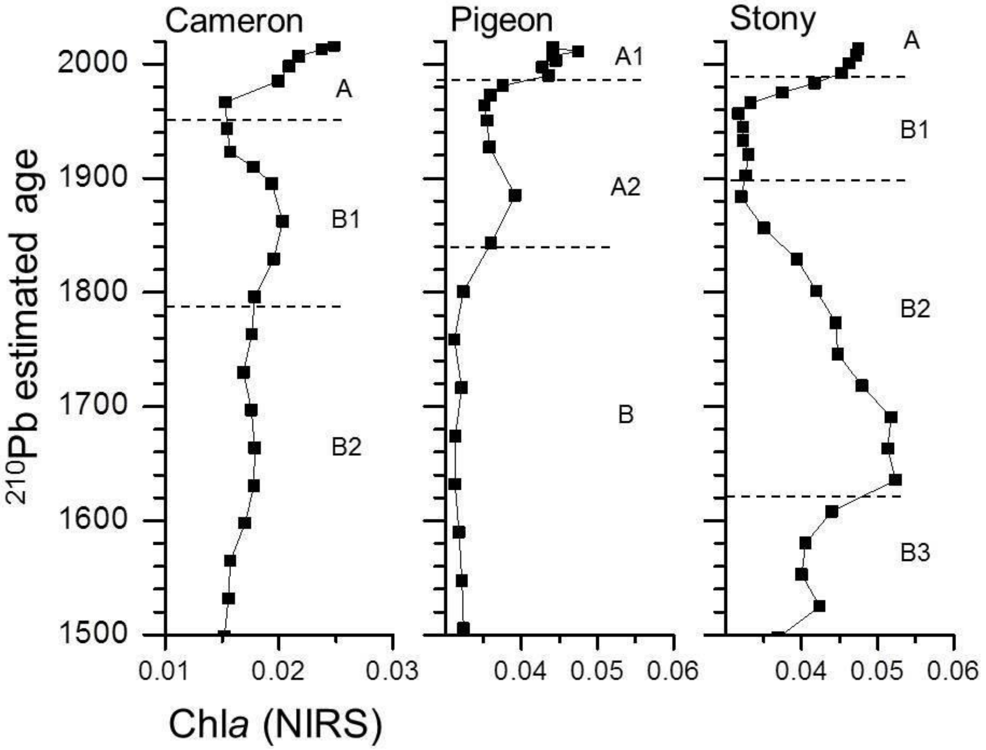


Figure S20. Near-infrared spectral estimates of Chlorophyll a (Chla) versus estimated age for each of the lake cores. Note the different scale for Cameron Lake. Zones are based on the diatom assemblages. *Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.*

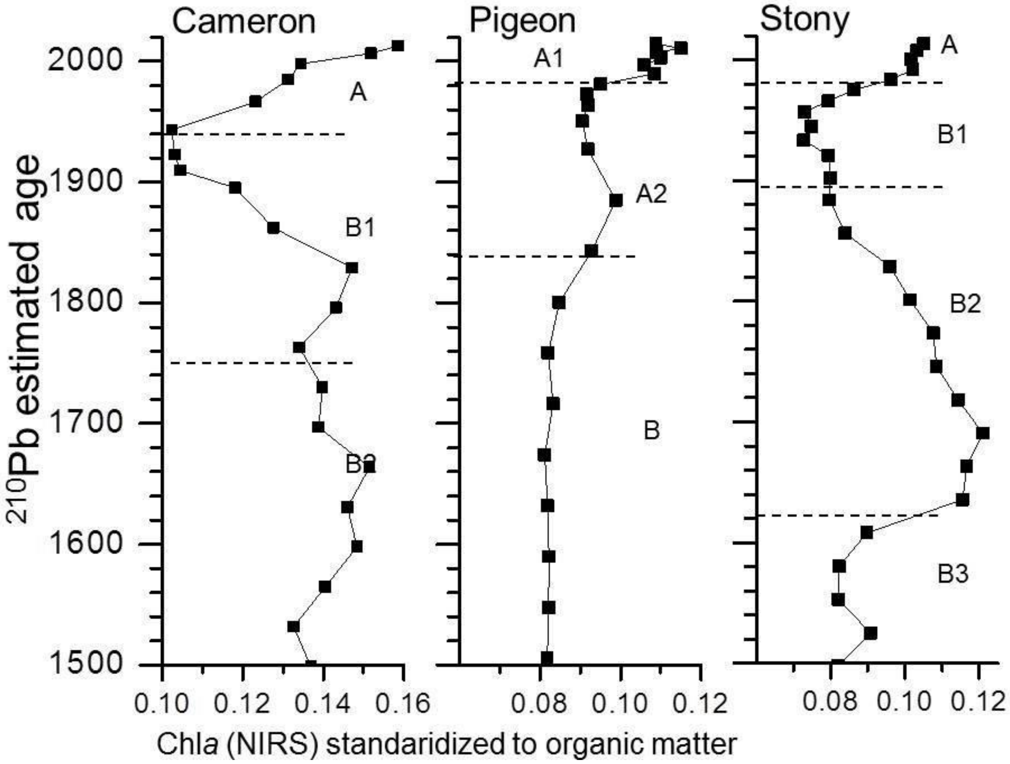


Figure S21. Near-infrared spectral estimates of Chlorophyll a (Chla) standardized to organic matter versus estimated age for each of the lake cores. Note the different scale for Cameron Lake.
Zones are based on the diatom assemblages.
Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.

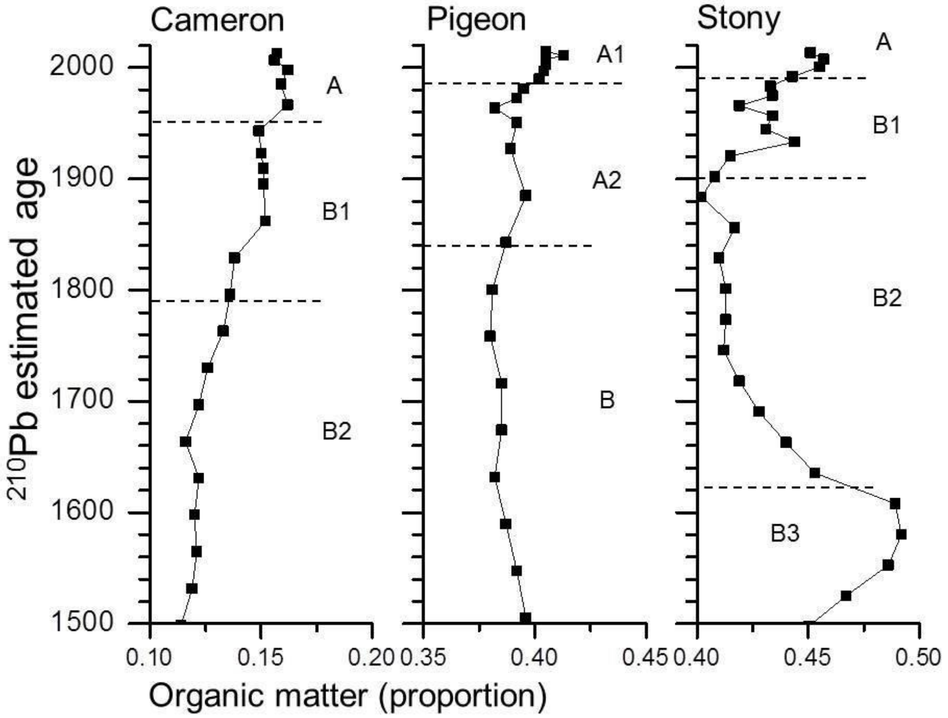


Figure S22. Proportion of organic matter in the sediment samples versus estimated age for each of the lake cores. Zones are based on the diatom assemblages. *Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.*

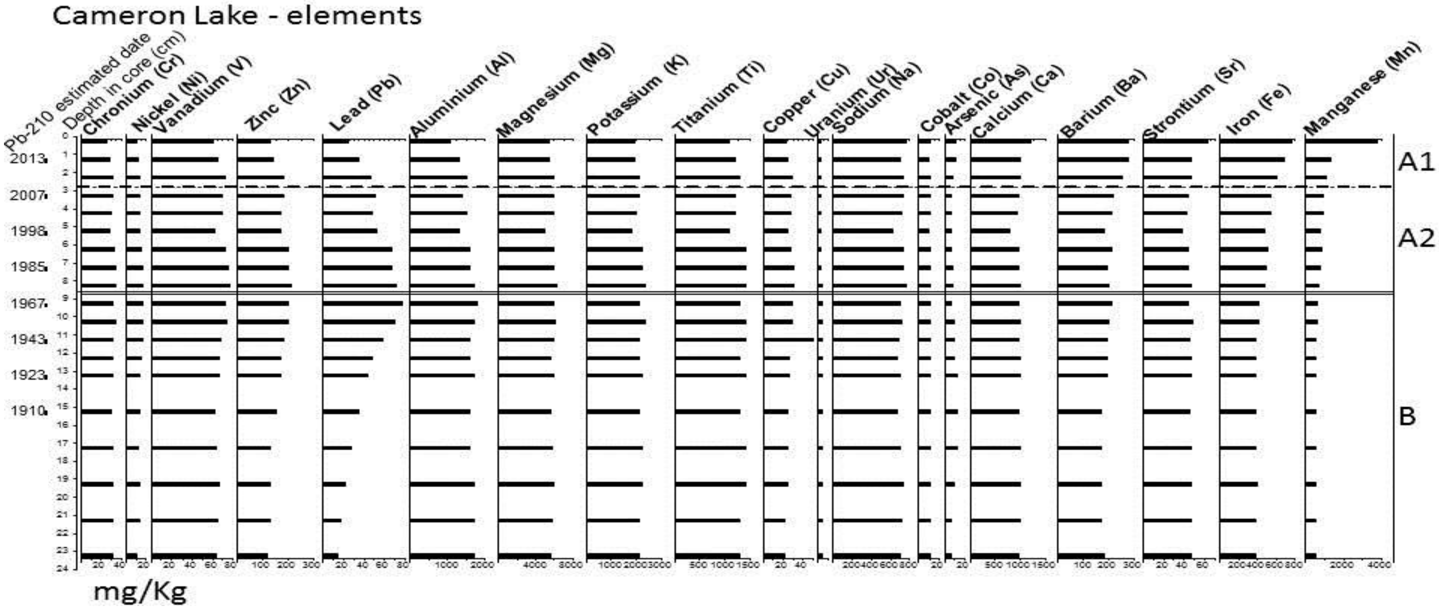


Figure S23. Concentration of elements for Cameron Lake versus depth, with estimated dates noted. Zones based on a depth-constrained cluster analysis of the elements.

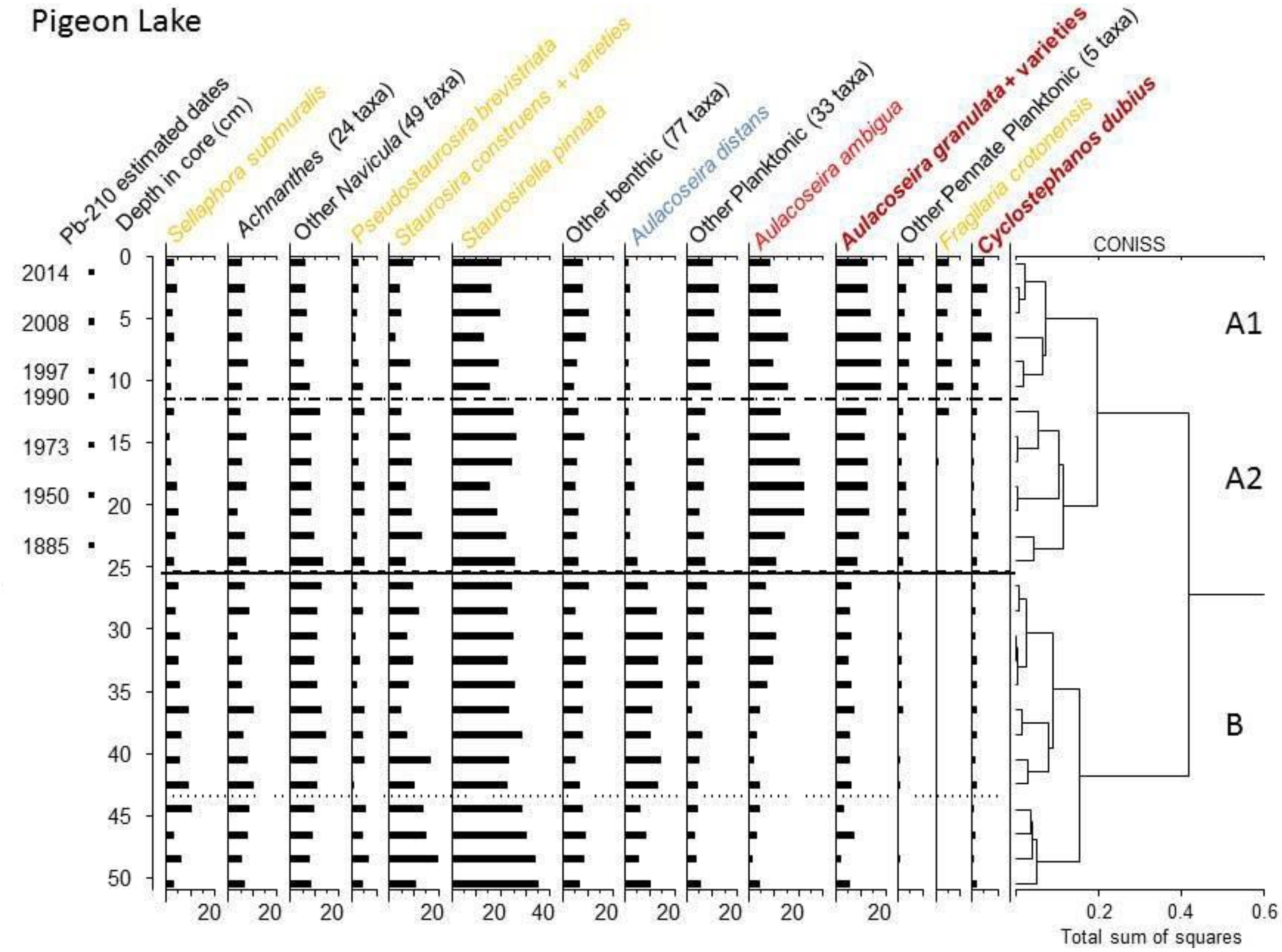


Figure S24. Percent abundance of diatom taxa > 5% abundance in the Pigeon Lake sediment core versus depth, and estimated dates noted. Colour coding of taxa names is same as in Cameron Lake, with the addition of more hypereutrophic taxa (very high nutrients) in bold red. The depth-constrained cluster analysis shown on the right from which the diatom assemblage zones are based on.

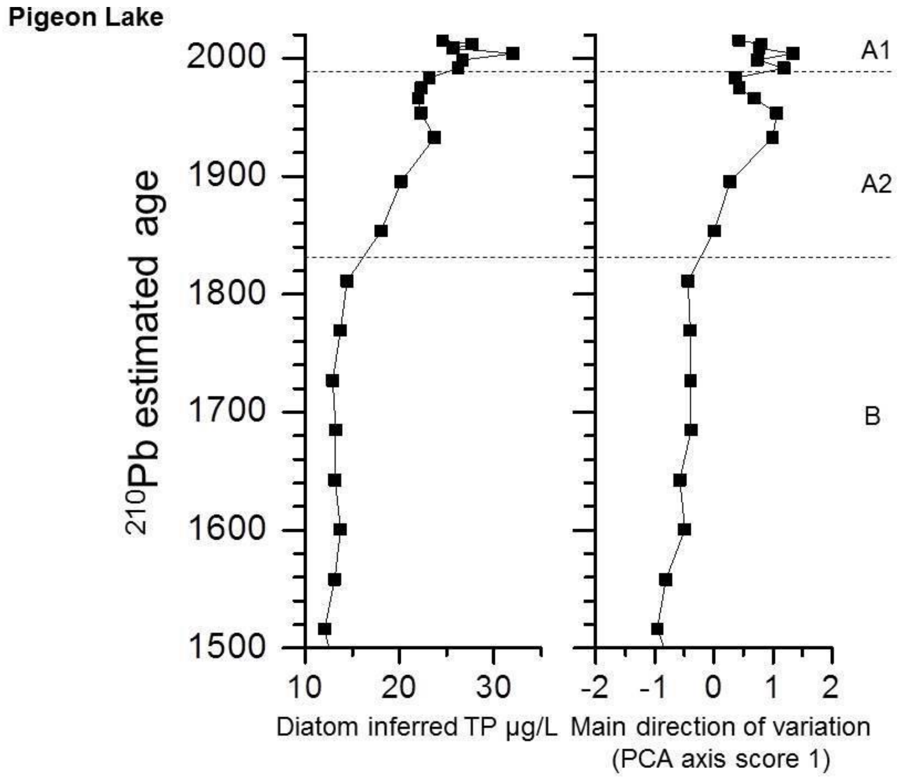


Figure S25. Diatom-inferred total phosphorus (TP), and main direction of variation (PCA axis-1 scores) for Pigeon Lake versus estimated age. *Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.*

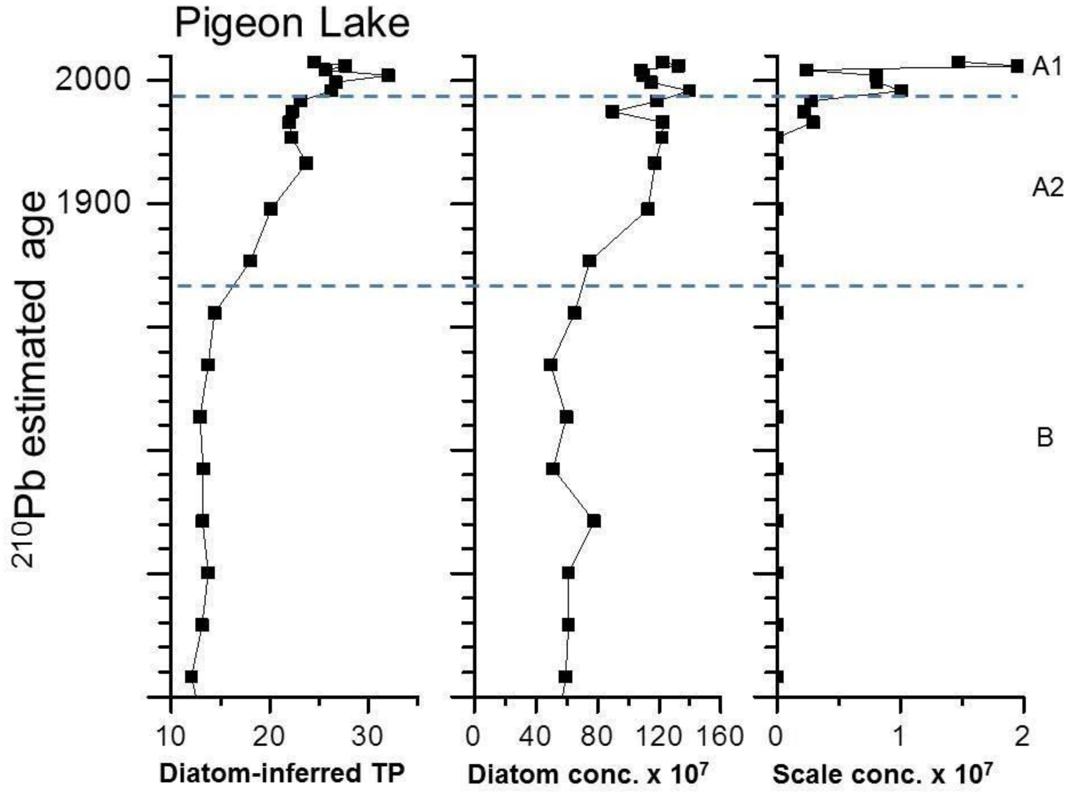


Figure S26. Diatom-inferred total phosphorus (TP), concentration of diatom valves and concentration of chrysophyte scales for Pigeon Lake versus estimated age. *Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.*

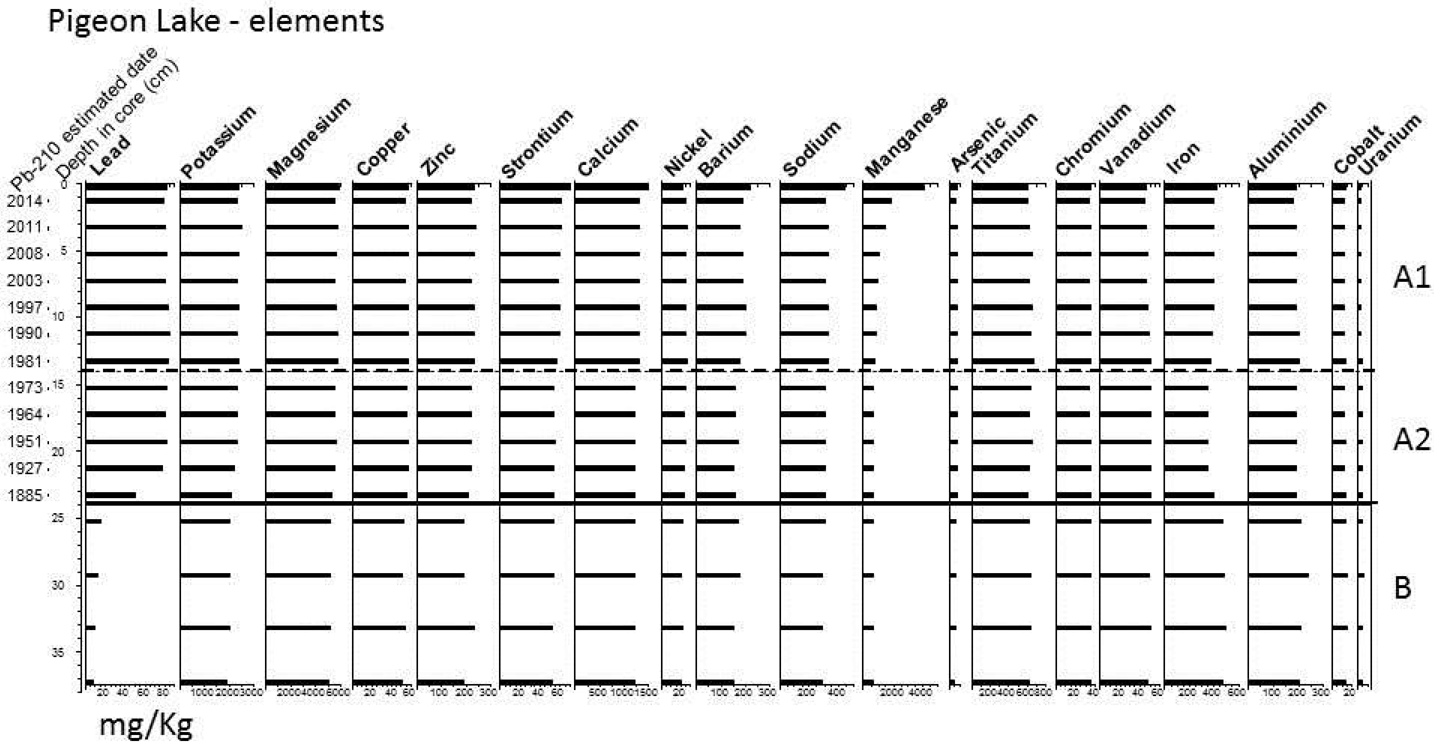


Figure S27. Concentration of elements for Pigeon Lake versus depth, with estimated dates noted. Zones based on a depth-constrained cluster analysis of the elements.

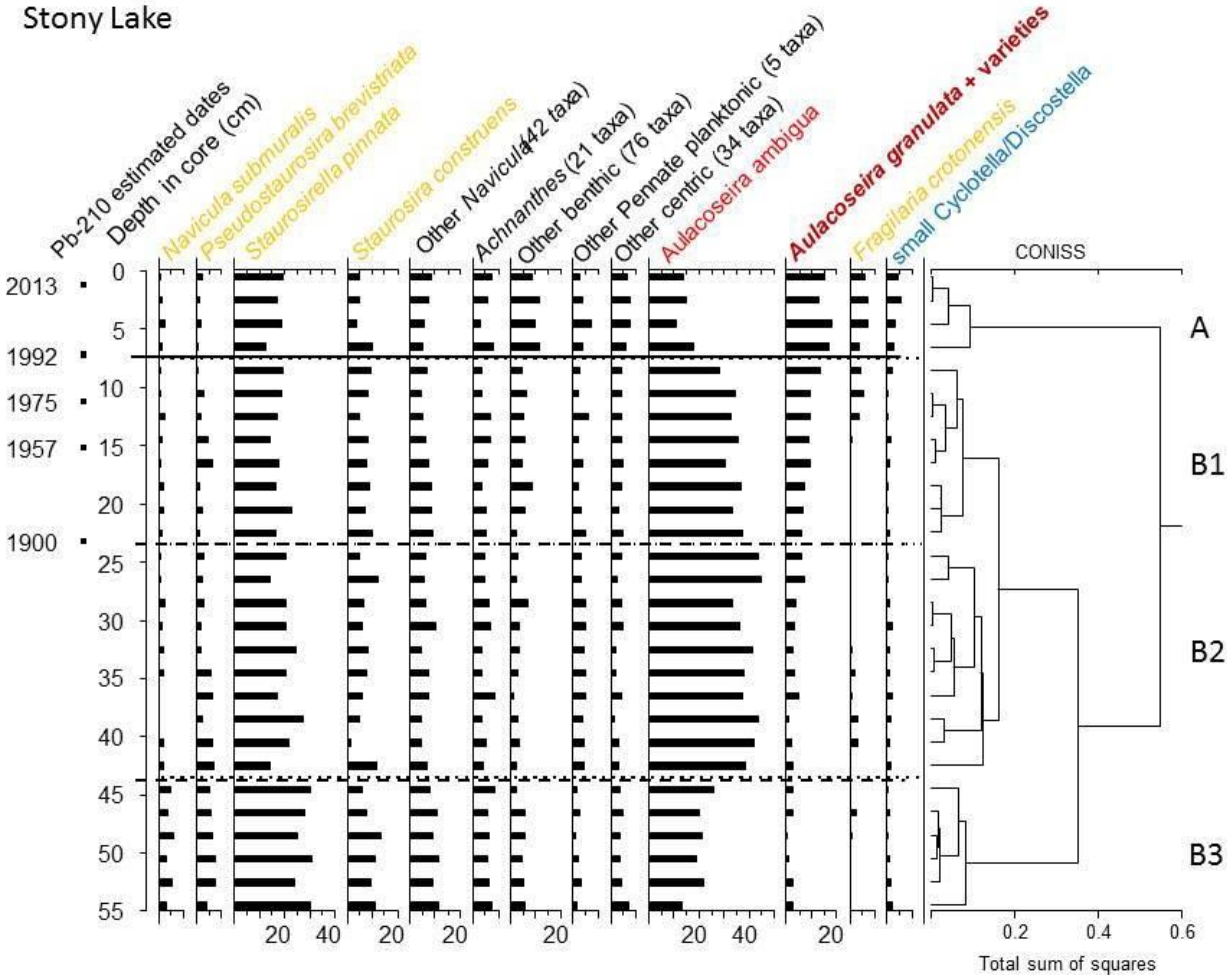


Figure S28. Percent abundance of diatom taxa > 5% abundance in the Stony Lake sediment core versus depth, and estimated dates noted. Colour coding of taxa names is same as in Cameron Lake, with the addition of more hypereutrophic taxa (very high nutrients) in bold red. The depth-constrained cluster analysis shown on the right from which the diatom assemblage zones are based on.

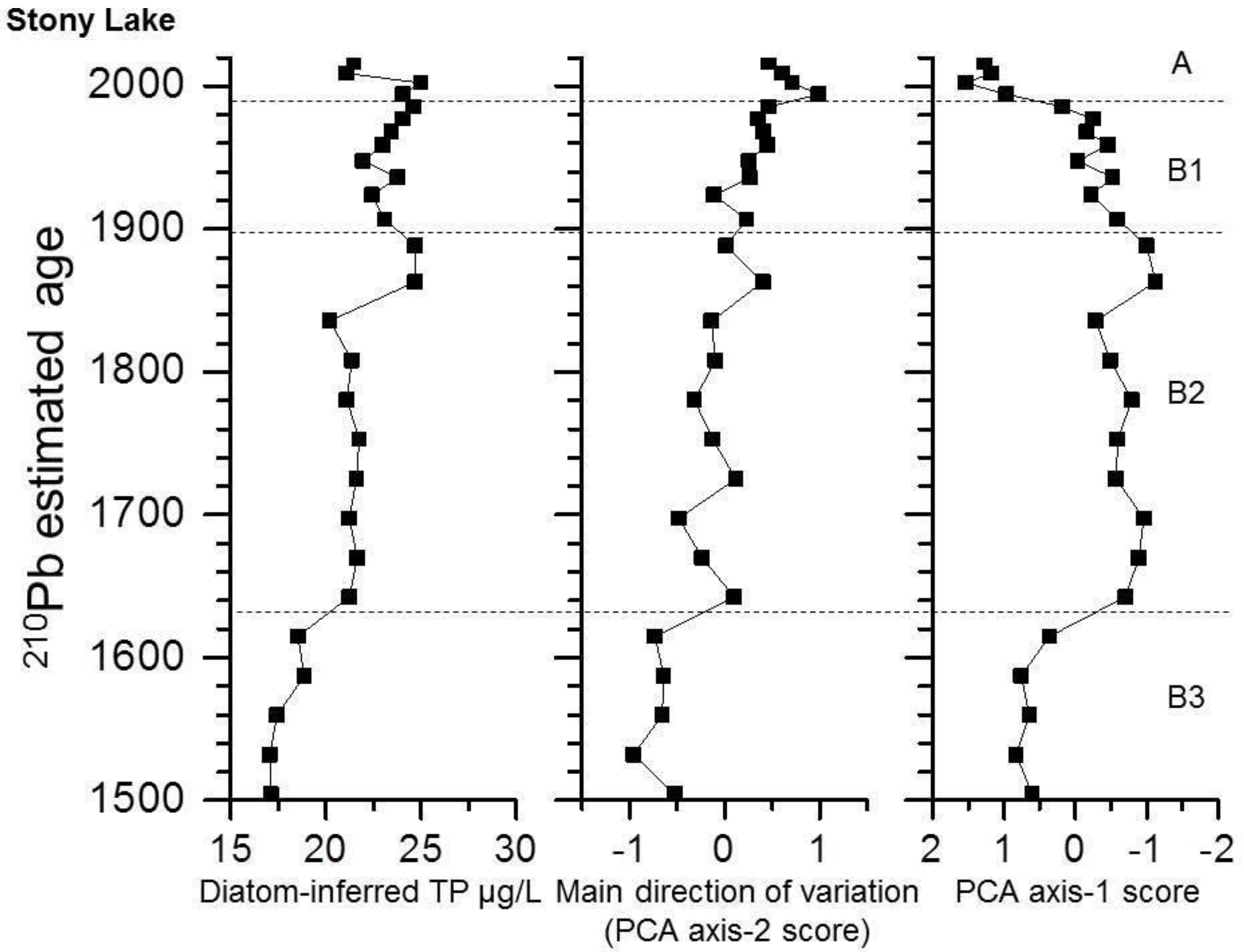


Figure S29. Diatom-inferred total phosphorus (TP), main direction of variation (PCA axis 1 scores) and secondary direction of variation (PCA axis 2 scores) for Stony Lake versus estimated age. Estimated age pre-1900 AD are based on extrapolations of ^{210}*Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.*

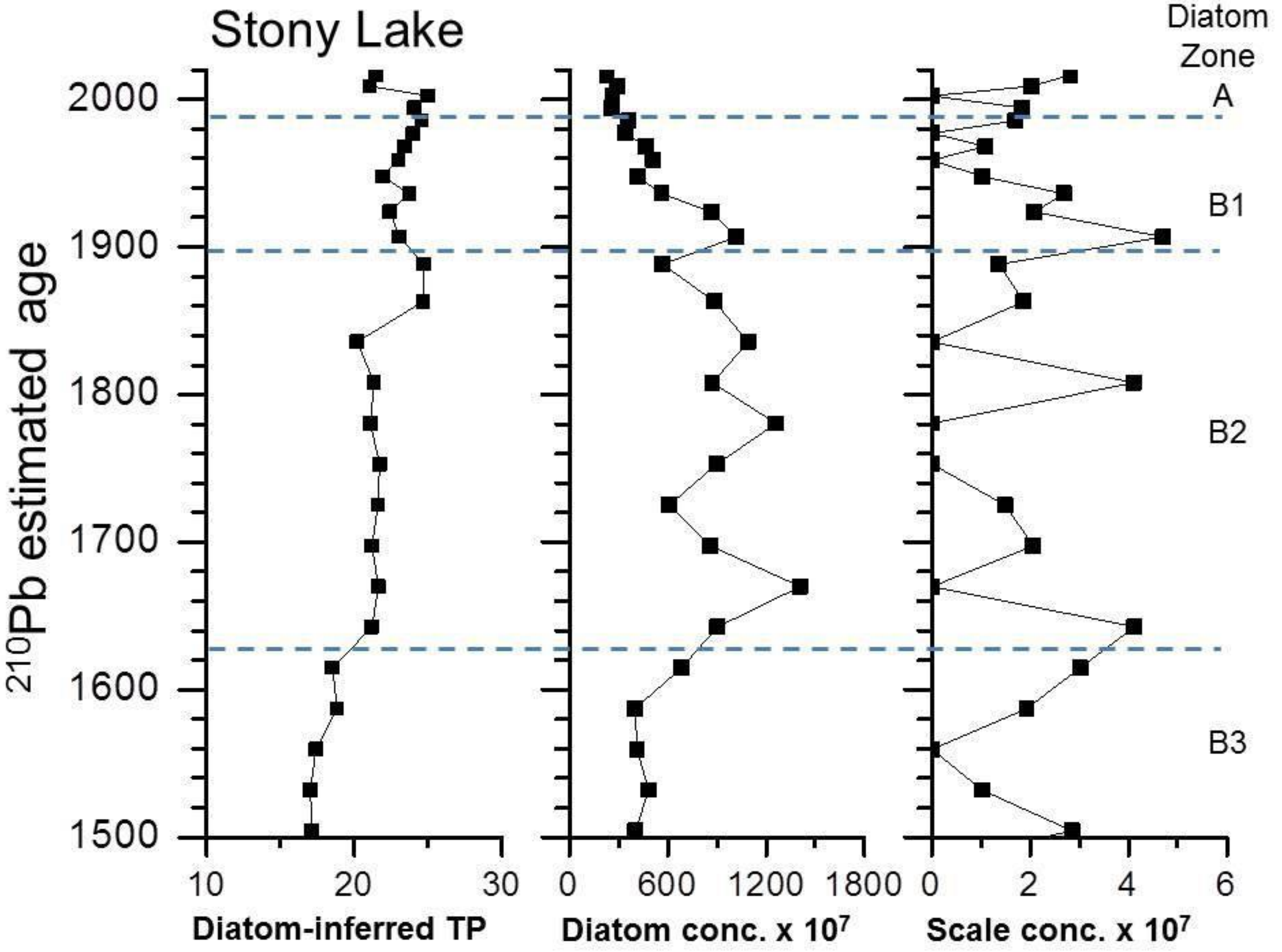


Figure S30. Diatom-inferred total phosphorus (TP), concentration of diatom valves and concentration of chrysophyte scales for Stony Lake versus estimated age. Estimated age pre-1900 AD are based on extrapolations of ^{210}Pb, but are not shown due to large errors associated with these estimates and are just used for graphing purposes.

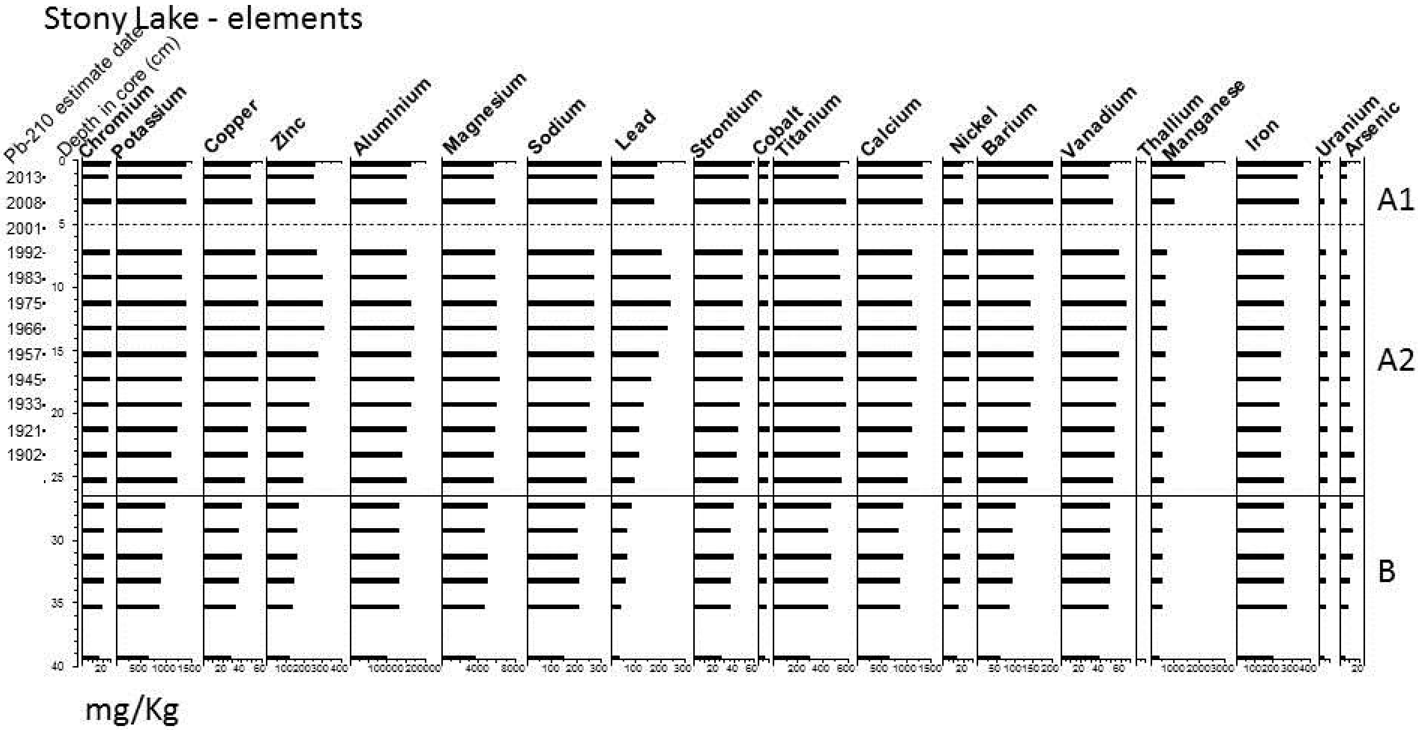


Figure S31. Concentration of elements for Stony Lake versus depth, with estimated dates noted. Zones based on a depth-constrained cluster analysis of the elements.