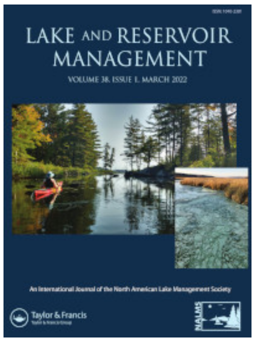

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Assessing the ecological responses of a shallow mesotrophic lake to multiple environmental stressors using paleolimnological techniques

Michael J. J. Murphy[a], Branaavan Sivarajah[a], D. Conrad Grégoire[b,c] and Jesse C. Vermaire[a,d]

[a]Department of Geography and Environmental Studies, Carleton University, Ottawa, ON, Canada; [b]Analytical Chemistry Laboratories (Retired), Geological Survey of Canada, Ottawa, ON, Canada; [c]Environment Committee, White Lake Property Owners Association; [d]Institute for Environmental and Interdisciplinary Sciences, Carleton University, Ottawa, ON, Canada

ABSTRACT

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White Lake, located in southeastern Ontario, Canada is a long, shallow, multibasin lake that supports recreational activities year-round and has 1538 private and commercial dwelling units. The construction of a dam at Waba Creek in 1845 resulted in the minimum water level at White Lake increasing by ~1.5 m, and several water-level management strategies have been implemented since the mid 20th century. The lake has also been impacted by land-clearance activities and shoreline development along the eastern shores and is presently mesotrophic. More recently, White Lake was colonized by zebra mussels (*Dreissena polymorpha*) in 2015, and annual algal blooms have also been observed. In this paleolimnological study, diatom-based approaches and loss on ignition were used to track past changes in water quality and sediment characteristics, respectively, to inform the White Lake Property Owners Association's ongoing lake monitoring program. The diatom-inferred total phosphorus (DI-TP) reconstruction suggests that White Lake has been mesotrophic (DI-TP range 11.5 – 17 µg/L) over the past ~200 years. Notable changes in the diatom assemblages and declines in sedimentary organic content were recorded around the 1890s in the sediment core retrieved near Hardwood Island. The decrease in sedimentary organic content was likely a response to early land-clearance activities around the lake as erosion often leads to greater inorganic export from the catchment to the lake. The switch among benthic fragilarioid diatom taxa suggests that the light conditions at this shallow lake may have changed in response to higher allochthonous inputs. Continuous monitoring of White Lake is needed to ensure that the cumulative impacts from emerging environmental stressors (e.g., climate-mediated changes, invasion by dreissenids) are considered when making management decisions.

KEYWORDS
Diatoms;
land-use changes;
paleolimnology;
water-level fluctuations

The ecological health of freshwater ecosystems is under threat from a variety of anthropogenic stressors (Ormerod et al. 2010, Reid et al. 2019). Understanding the impacts of multiple environmental stressors on lakes is a necessary first step toward making informed management decisions. However, several environmental stressors (e.g., invasive species, nutrient loading, climatic changes) predate most monitoring programs, and in many circumstances, lake managers are required to make decisions without baseline environmental data. In addition, there is limited understanding of how multiple stressors interact with each other over time to alter lake ecosystems and lead to unexpected ecological responses (Quinlan et al. 2008, Strayer 2010, Jackson et al. 2016). This is especially true for shallow lake ecosystems, where modest changes in water level or turbidity can disproportionally and rapidly alter the ecosystem by modifying what proportion of the benthic environment has sufficient sunlight for photosynthetic organisms to grow (Scheffer and van Nes 2007, Vadeboncoeur et al. 2008, Velghe et al. 2012).

In the absence of direct long-term monitoring data, paleolimnological approaches can be used to infer historical water quality trends (Smol 2008). Specifically, well-preserved biological,

CONTACT Jesse C. Vermaire jesse.vermaire@carleton.ca

chemical, and physical indicators in lake sediments can be used to reconstruct and assess changes in the limnology of lakes and surrounding catchment. Diatoms (Bacillariophyceae, siliceous algae) are one of the most widely used biological proxy in paleolimnology, because they respond rapidly to changes in limnological variables and leave identifiable remains in lake sediments (Battarbee et al. 2001, Whitmore et al. 2018). This has made them excellent paleo-indicators to investigate the impacts of nutrient enrichment (Reavie et al. 2014), water-level fluctuations (Laird et al. 2010), acidification (Battarbee et al. 2001), and climate-mediated changes (Rühland et al. 2015). Diatom-based approaches have also been used to investigate the cumulative impacts of multiple environmental stressors in aquatic ecosystems, as taxon-specific shifts can be used to disentangle the impacts of various stressors (e.g., Sivarajah et al. 2018, Stager et al. 2018). For instance, diatom-based approaches were used to investigate the effects of land-use changes, water-level changes, and climate-mediated changes on the water quality of multiple basins of Lake Simcoe and Lake of the Woods (Rühland et al. 2010, Hawryshyn et al. 2012, Reavie et al. 2017).

In this study, diatom-based paleolimnological approaches were used to investigate the impacts of multiple environmental stressors on a heavily used recreational lake in southeastern Ontario, Canada. White Lake (Figure 1) is a shallow, multibasin system that has been impacted by water-level manipulation and nutrient loading from catchment activities. These anthropogenic activities have resulted in the moderate deterioration of water quality at White Lake, destruction of fish spawning habitat, and occurrences of nuisance and potentially toxic blue-green algal blooms over the years (Grégoire and Overholt 2020a, b). While water levels were stabilized in the 1990s, problems with nutrient loading persisted, and more recently the invasion of zebra mussels (*Dreissena polymorpha*) has also been identified as an emerging stressor (Grégoire and Overholt 2020a). Recognizing that these stressors pose a serious threat to White Lake with respect to water quality and recreational use, the White Lake Property Owners Association has taken important steps to monitor the lake (White Lake Property Owners Association 2021). However, the monitoring program started after 2014 with only sporadic limnological data available prior to this period. This paleolimnological study aims to contribute historical limnological perspectives to place the findings from the contemporary monitoring program into long-term environmental context. Specifically, this study has 2 objectives: (1) establish limnological baselines (i.e., conditions prior to major anthropogenic activities in the catchment) and (2) assess the limnological consequences of multiple environmental stressors after the mid 1800s. This paleolimnological study will provide multidecadal perspectives on limnological changes at White Lake, which can be used to inform lake management practices and assist with the implementation of ongoing monitoring efforts by the White Lake Property Owners Association.

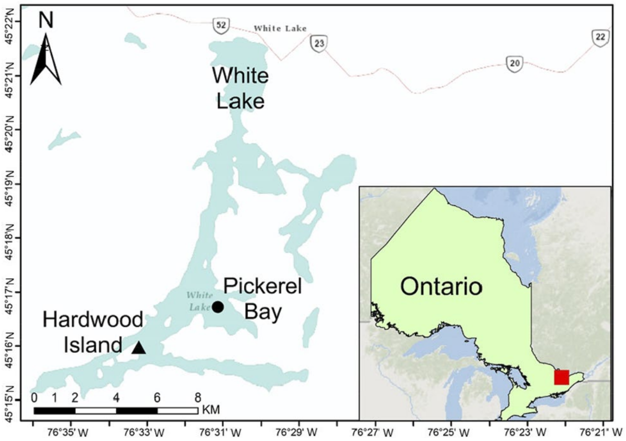

Figure 1. Map of White Lake, with the circle indicating the location of Pickerel Bay and the triangle indicating Hardwood Island. The square in the inset map identifies the location of White Lake within the province of Ontario, Canada.

Study site

White Lake is a long (~16 km), shallow (mean depth of 3.1 m and maximum depth of 9.1 m), mesotrophic lake with a surface area of 22.5 km^2 and 97.9 km of shoreline. This relatively large lake is located ~70 km west of Ottawa (southeastern Ontario, Canada) and within the Lake Simcoe–Rideau Ecoregion (Crins et al. 2009). This lake has a watershed area of 211 km^2, abuts the Precambrian Shield on its western shore, and is underlain by calcareous rocks, which also form much of the geology of the remaining shoreline. White Lake is a popular recreational site as it is easily accessed by roads and supports a range of activities throughout the year. Specifically, the eastern part of the lake has several embayments (Figure 1) with extensive shoreline development (e.g., permanent residences, recreational resorts, trailer parks, and campgrounds). In 2018, there were 1538 permanent and commercial dwellings, and the number of permanent residences had increased from 59 in 1985 to 209 in 2018. White Lake supports both cool (walleye [*Sander vitreus*] and pike [*Esox lucius*]) and warm (perch [*Perca*], bass [*Micropterus*], sunfish [Centrarchidae]) water fish species (Grégoire and Overholt 2020a), which has made this lake an attractive destination for year-long recreational fishing.

The single outflow from White Lake, Waba Creek, is located at the north end of the lake. Starting in 1845, this outflow was dammed for logging operations (Ontario Power Generation 2009). The construction of the dam raised the minimum water levels by ~1.5 m at this shallow lake (Mathers and Kerr 1998). In 1968 the dam was rebuilt, and a water-level management plan was implemented based on the seasons (spring, summer, fall/winter; Ontario Power Generation 2009) with sustained high water levels maintained during the summer months to accommodate boating activities (van Rosen 1989). An earlier fisheries assessment of this lake suggests that shortly after the construction of the dam, fish spawning habitat was negatively impacted, and midsummer algal blooms were observed (von Rosen 1989). In 1977 a new regime was implemented with midsummer drawdowns, which helped to improve water quality and fish spawning habitat (van Rosen 1989). The 1977 regime was slightly modified in 1981 to retain part of the spring freshet throughout the summer (van Rosen 1989). The current water-level regime, which began in 1998, is an intermediate to the previous strategies, with summer levels of approximately 1.5 m above pre-dammed levels and slower drawdowns to winter levels of 1 m above pre-dammed levels (Figure 2A).

Although White Lake was not intensively monitored prior to 2014, water clarity measurements (i.e., Secchi disk depth) were recorded annually between 1969 and 1995 (Figure 2B). During this period, the Secchi disk depth ranged between 1.8 and 3.8 m, and the mean was 2.8 m with no directional change (Mathers and Kerr 1998). However, after the arrival of zebra mussels in 2015 the mean Secchi disk depth increased to 4.5 m (Figure 2B). Greater water clarity in recent years has also increased the amount of lake bottom exposed to sunlight (increasing from ~27% in 2015 to ~90% in 2017; Grégoire and Overholt 2020a), thus opening more habitat for benthic primary producers. Ice phenology data was collected between 1979 and 2020; the lake ice cover varied between 98 and 160 d (mean: 133 d), and an overall declining trend was observed (Grégoire and Overholt 2020a, Figure 2C). Since 2014, total phosphorous (TP) measurements have also been collected at White Lake throughout the open-water season (Figure 2D) as part of the Lake Partner Program (Ontario Ministry of Environment, Conservation and Parks). Near Hardwood Island, the TP concentrations ranged between 7 µg/L and 21 µg/L during the open-water period and were indicative of mesotrophic conditions (Figure 2D). The TP levels were generally lower in samples collected in May and September, while samples from July and August recorded higher concentrations (Figure 2D).

Materials and methods

Field sampling and radiometric dating

A sediment core was retrieved near the north end of Hardwood Island (45°16’09.7"N, 76°33’12.2"W; Figure 1) from a water depth of 5 m using an ARI gravity corer during the summer of 2014. The 21 cm long core was sectioned on site at 1 cm contiguous intervals, and the sediment samples were stored at 4 C prior to analysis. The chronology for the Hardwood Island sediment core was established using alpha spectroscopy at Flett Research Ltd (Winnipeg, Manitoba, Canada). Specifically, ^{210}Po (proxy for total ^{210}Pb) and ^{226}Ra (proxy for supported ^{210}Pb) activities in dried sediments were measured with alpha spectroscopy and were used to infer unsupported ^{210}Pb activities (Appleby 2001). The

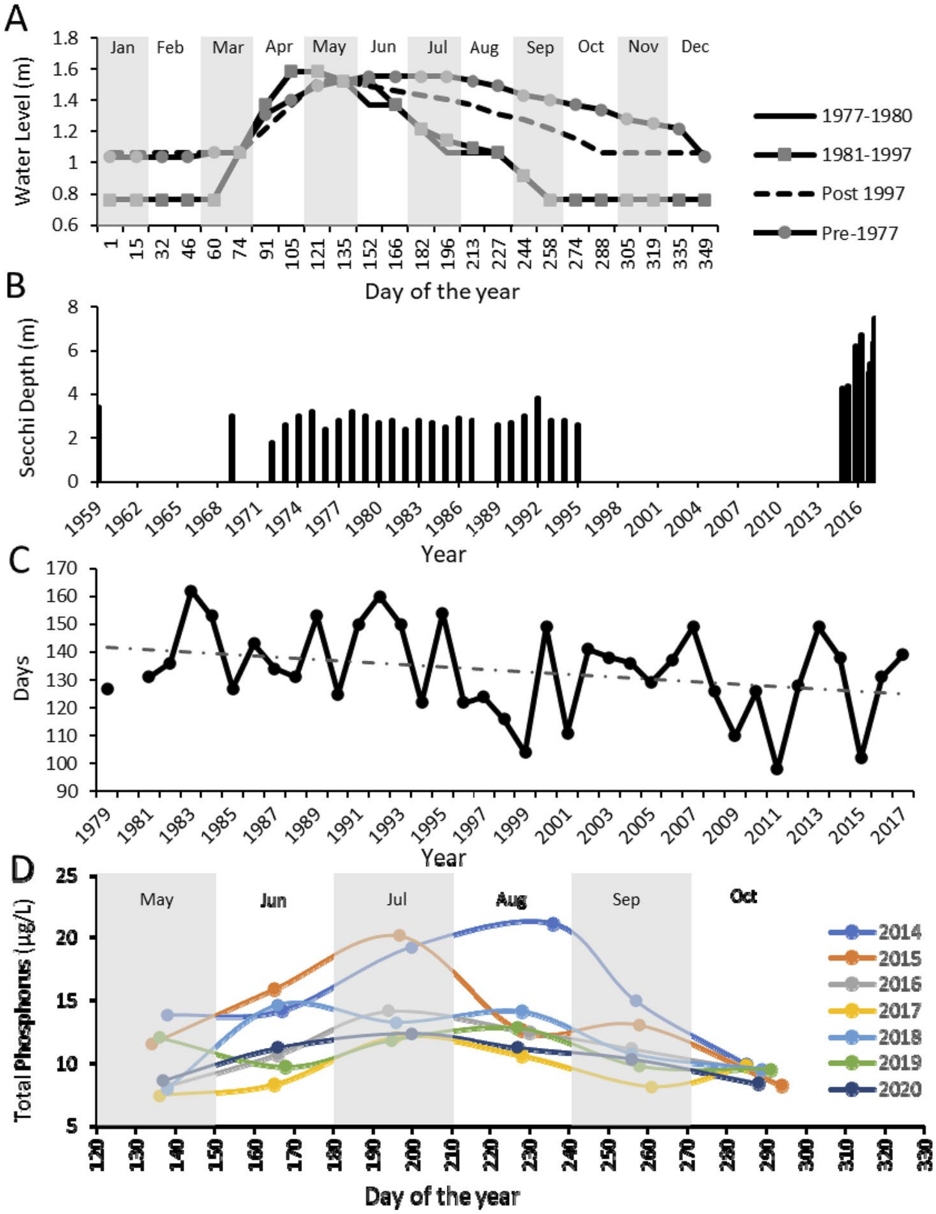

Figure 2. (A) Water-level management strategies at White Lake. The dashed line represents the current management strategy. Values shown only represent target lake levels; actual lake levels fluctuate with the weather. (B) Secchi disk depth (m) measured at White Lake between 1959 and 2017. (C) Ice-cover duration (number of days) was monitored at Pickerel Bay between 1979 and 2020. (D) Total phosphorus concentrations (µg/L) measured at Hardwood Island during the open-water period (May–Oct) between 2014 and 2020.

age–depth model was developed using a constant rate of supply (CRS) model (Appleby 2001).

A 35 cm sediment core was collected from a deeper site, Pickerel Bay (coring depth 9 m), in 2017. Unfortunately, a reliable chronology could not be developed for this sediment core as the ^{210}Pb activities did not decrease with depth, which may suggest that the sediments were accumulated over a very short period of time or have been subjected to mixing. Hence, the timing of changes in the sedimentary proxies cannot be determined for the Pickerel Bay core. However, we are using the information preserved in the sediments of the Pickerel Bay site to examine whether the sediment composition and biological assemblages were different between this site and the Hardwood Island site.

Laboratory analyses

The organic content in the sediments were measured using loss-on-ignition (LOI) analyses following the standard protocol described by Heiri et al. (2001) to assess whether sediment composition has changed at White Lake. Briefly, a 1 cm^3 subsample of wet sediment was taken from each interval and weighed before being placed in a furnace at 110 C for a minimum of 12 h to be dried. The dried sediments were weighed and then placed in a furnace at 550 C for 4 h to burn all organic matter present in the sediment. Samples were then weighed for a final time and the water and organic contents of the sediments were calculated.

Diatom slides were prepared following standard procedures outlined by Battarbee et al. (2001). Briefly, 0.1 g of wet sediment was transferred into 20 mL glass scintillation vials and 10 mL of 10% hydrochloric acid with 10 mL of deionized water was added to the sediment matrix. Samples were left to settle overnight. Subsequently, ~10 mL of the supernatant was pipetted from the vials, without disturbing the settled diatoms, and 10 mL of deionized water was added. This rinsing procedure was repeated 7 times, with the samples being allowed to settle overnight between each rinse. After the seventh rinse, 10 mL of 30% hydrogen peroxide was added to the sample and placed in a water bath (at ~70 C) for 8 h. After heating, the samples were left overnight to cool. Once cool, the rinsing procedure just described was repeated 7 times. Afterward, a few drops of 10% hydrochloric acid were added to each sample to aid diatom dispersion on coverslips. One mL of sample was transferred to a test tube, and 9 mL of deionized water was added and the sample was thoroughly mixed. Each well-mixed diatom slurry was pipetted onto a clean glass coverslip. A one-half dilution series was repeated until 4 concentrations from each sample were made. In a fume hood, dried coverslips were mounted onto slides using Naphrax. At least 400 diatom valves were counted per sample using a Leica DM2500 microscope at 1000× magnification. Taxonomic guides by Krammer and Lange-Bertalot (1986, 1988, 1991a, 1991b) were used to identify diatoms, and the taxonomy was updated using more recent publications and online databases (e.g., Spaulding et al. 2021).

Statistical analyses

Species counts were transformed into relative abundances. Stratigraphic plots showing the relative abundance of the most common diatoms (2% in at least one sample) were produced for both sediment cores. Taxa with a relative abundance of less than 2% in at least one sample were removed from constrained hierarchical clustering analysis and principal component analysis (PCA) to reduce the influence of rare taxa. For the PCA and clustering analysis, the relative abundance data were square-root transformed to reduce the influence of abundant species. Constrained hierarchical clustering analysis with a broken stick model was applied to the downcore diatom data to identify the most important stratigraphic zones using the "rioja" package (Juggins 2017) developed for the R statistical environment (R Core Team 2017). PCA was performed to summarize the multispecies diatom data using the "prcomp" function in R (R Core Team 2017). Ellipses showing 95% confidence intervals of the centroid of the mean of the group were added to PCA plots to denote the significant clusters determined by the constrained hierarchical clustering, to visualize how the clusters plotted relative to each other in ordination space.

Total phosphorus measurements have been collected at White Lake since 2014; however, historical changes in TP are not known. In the absence of direct long-term TP measurements, diatom-based inference models can be used to infer past trends in TP, as diatoms are sensitive to changes in nutrient concentrations and various taxa have specific optima for TP (e.g., Reavie and Smol 2001, Rühland et al. 2010, Reavie et al. 2014). In this study we applied a diatom-based TP model that was developed using calibration data from 64 alkaline lakes in southeastern Ontario, and the model covers a TP gradient of 4 to 54 µg/L (mean 14.1 µg/L; Reavie and Smol 2001). The diatom-inferred TP (DI-TP) model using weighted averaging regression and classical deshrinking had a bootstrapped R^2 of 0.478. This DI-TP has been used to reconstruct TP changes in southeastern Ontario, including the Rideau Canal lakes that are located south of White Lake (Forrest et al. 2002). The DI-TP model was only applied to the Hardwood Island core; it was not applied to the Pickerel Bay core, as this did not have a reliable chronology and any reconstructed TP concentrations could not be placed into appropriate long-term context without dates. The diatom taxa from the Hardwood Island core were well represented in the model because on average 87.6% (range: 80 – 94.4%; median: 87.1%) of the assemblages from each sample were included in the reconstruction.

Results

Radiometric dating

The total ^{210}Po (a proxy for total ^{210}Pb) activity displayed an exponential decay with depth and reached background activities around 9 cm in the Hardwood Island core (Figure 3A). Based on the CRS model, the top 9 cm of this sediment core represents the last ~130 years, approximately encompassing the period between 1884 and 2014 (Figure 3B). The CRS sedimentation accumulation rate for the Hardwood Island core ranged between 0.0026 and 0.0054 g/cm^2/yr, and this was lower compared to other lakes examined from southeastern Ontario (e.g., Forrest et al. 2002, Simmatis et al. 2020). As indicated earlier, a reliable chronology could not be obtained for the sediment core from Pickerel Bay, so the biological and geochemical changes observed in this core have been reported based on core depth.

Loss on ignition

The organic matter content in the Hardwood Island core was relatively stable from the bottom of the core to 12 cm (~71 – 74%) and increased subtly between 11 cm and 9 cm (~75 – 77%; Figure 4A). However, the organic content in the sediments began declining around 7 cm and the lowest levels were recorded in the most recent sediments (~60%). Based on the CRS dates, the gradual decline in organic content at Hardwood Island occurred between the 1890s and 2014 (Figure 4A). Organic matter content in the deeper sediments (35 – 22 cm) of Pickerel Bay were higher (~70%), and the content was lower between 22 cm and 6 cm (~64%; Figure 4B). Unlike the Hardwood Island core, the organic content in the most recent sediments (0 – 5 cm) of Pickerel Bay was higher (68 – 73%) and comparable to levels from the deeper sediments (Figure 4B).

Diatom assemblage changes

The sedimentary diatom assemblages at Hardwood Island were dominated by benthic fragilarioid taxa (*Pseudostaurosira brevistriata*, *Staurosira construens*, *Staurosirella pinnata*), while tychoplanktonic *Aulacoseira* taxa and several other benthic taxa were also present in notable abundances (Figure 4A). The cluster analysis with the broken stick model identified 3 distinctive zones in the Hardwood Island diatom assemblages. Zone 1 (19 – 16 cm) was co-dominated (< 20%) by *Aulacoseira ambigua*, *P. brevistriata*, *S.*

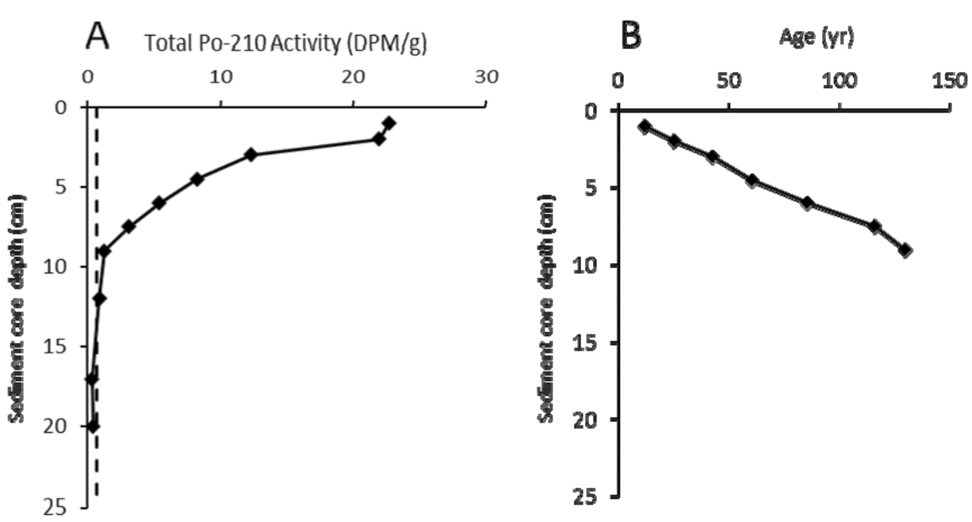

Figure 3. (A) Total ^{210}Po (proxy for total ^{210}Pb) activity over depth for the Hardwood Island core. ^{226}Ra (proxy for supported ^{210}Pb), showing background activity, is displayed by the dashed vertical line. (B) The age of sediments inferred using the constant rate of supply model.

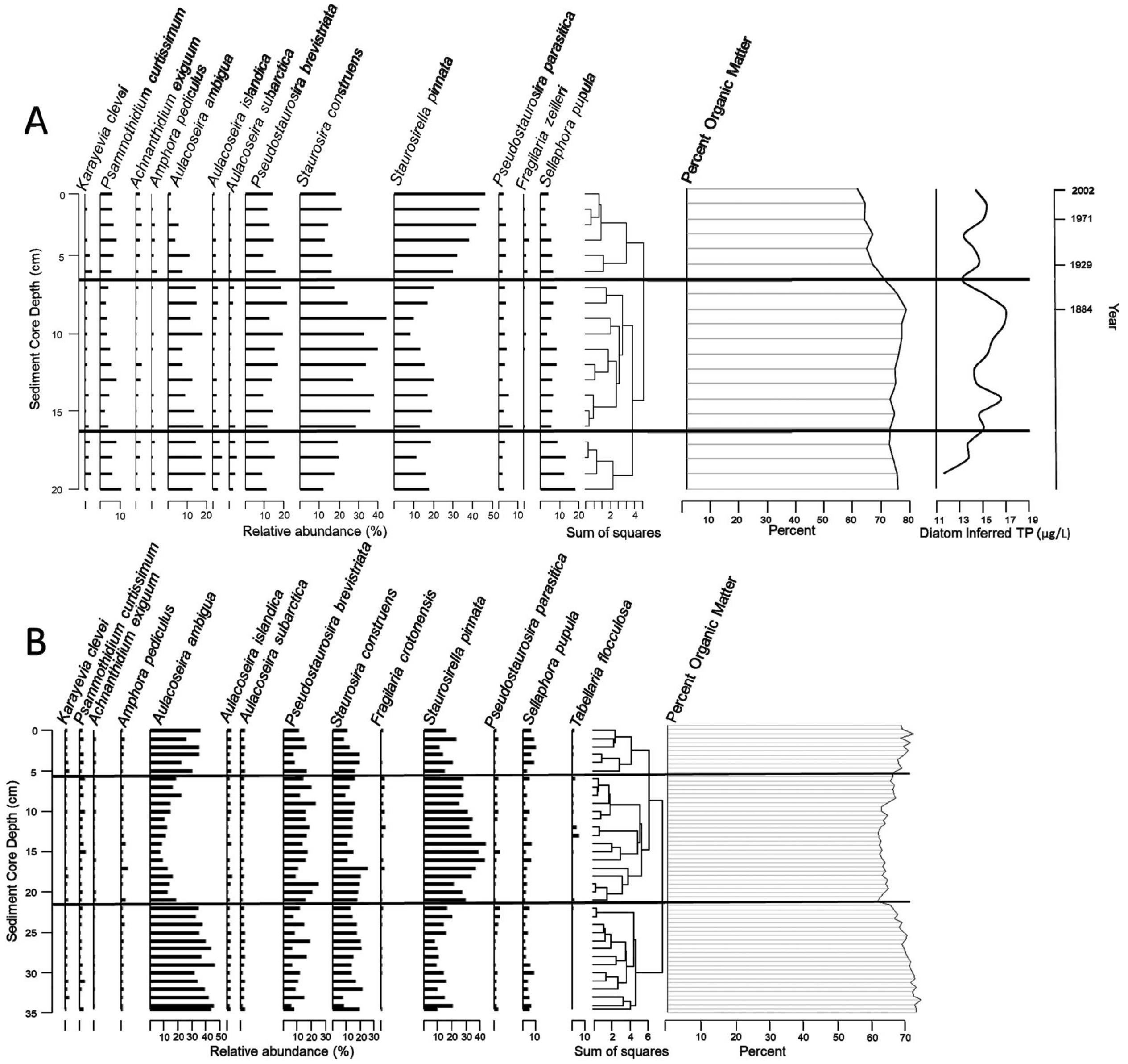

Figure 4. (A) Stratigraphic plot of diatom assemblage changes at Hardwood Island plotted along with loss on ignition and diatom inferred total phosphorus (µg/L). (B) Stratigraphic plot of diatom assemblages in the sediment core from Pickerel Bay, plotted along with loss on ignition. Three significant diatom zones were determined by a broken stick model, and groups are separated by solid horizontal lines.

construens, *S. pinnata*, and *Sellophora pupula* (Figure 4A). In zone 2 (15 – 6 cm), the abundance of *S. construens* (~20 – 40%) increased and dominated the assemblages, while *S. pupula* declined (Figure 4A). Toward the end of zone 2 (7 cm, ca. 1890s), however, the relative abundance of *S. construens* started to decrease and the relative abundance of *S. pinnata* began increasing. In zone 3 (5 – 0 cm; 1929 to the present), *S. pinnata* has continued to increase and has been the dominant taxa during this period (30 – 46%; Figure 4A). Coinciding with the switch in benthic fragilarioid taxa, *A. ambigua* has also decreased in zone 1.

The DI-TP varied between 11.5 µg/L and 17 µg/L at Hardwood Island (Figure 4A), and this range is comparable to the recent monitoring data (Figure 2). The DI-TP increased gradually between 19 and 14 cm (from 11.5 µg/L to 16.5 µg/L). The DI-TP values were generally higher in zone 2, and the highest DI-TP (17 µg/L) was recorded around 8 cm (Figure 4A). Afterward, the DI-TP decreased rapidly to 13 µg/L around 6 cm (Figure 4A). Although DI-TP continued to fluctuate in zone 1, it generally ranged between 13 µg/L and 15.5 µg/L (Figure 4A).

Similar to the Hardwood Island core, 3 important zones were observed in the diatom assemblages of Pickerel Bay (Figure 4B). Since a reliable chronology could not be developed for this core, we could not determine whether the timing of assemblage changes (i.e., the periods encompassed in each zone) was similar across both sites. *Aulacoseira ambigua* dominated (33 – 47%) the diatom assemblages in zone 1 (35 – 22 cm), and benthic fragilarioid taxa (*P. brevistriata*, *S. construens*, *S. pinnata*) were present in notable abundances (generally <20%; Figure 4B). In zone 2 (21 – 6 cm), the relative abundance of *S. pinnata* increased (22 – 45%) and it became the dominant taxon, while the abundance of *A. ambigua* decreased (8 – 23%; Figure 4B). In zone 3 (5 – 0 cm), a recovery in the abundance of tychoplankotnic *A. ambigua* was observed (23 – 36%), along with subtle increases in *S. pupula* (Figure 4B). The abundance of *S. pinnata*, however, decreased in zone 3 (Figure 4B).

The PCA axes 1 and 2 of the Hardwood Island core captured 52% and 24% of the variation in the diatom assemblages, respectively (Figure 5A). The samples belonging to the 3 zones identified by cluster analysis plotted separately in ordination space (Figure 5A). The distribution of sites and species in PCA ordination space indicates that the differences in assemblages were strongly influenced by changes in the abundances of the most abundant species (i.e., *A. ambigua*, *S. pinnata*, *S. construens*, and *S. pupula*; Figure 5A). Furthermore, the clear separation of samples from the 3 zones highlights that the recent assemblages (i.e., zone 3) are not similar to earlier assemblages (i.e., zones 1 and 2, Figure 5A). The PCA axes 1 and 2 captured 59% and 10% of the variation in the diatom assemblages, respectively, from Pickerel Bay (Figure 5B). Consistent with the taxon-specific changes observed at Pickerel Bay, samples from zones 1 and 3 clustered together in the ordination space, where *A. ambigua* dominated the diatom assemblages (Figure 5B). The samples from zone 2, dominated by *S. pinnata*, clustered in the right quadrants of the PCA biplot (Figure 5B).

Discussion

Comparison of diatom assemblages at Hardwood Island and Pickerel Bay

Water depth is an important factor shaping diatom assemblage composition in lakes, as it can strongly influence habitat structure and light availability for these photosynthetic microorganisms (Laird et al. 2010). White Lake has a mean depth of 3.1 m and maximum depth of 9.1 m and the diatom assemblages were composed of mostly benthic taxa that were reflective of the shallow nature of the lake. Particularly, benthic fragilarioid taxa that are observed in shallow waters and attached to various substrates (Lotter and Bigler 2000, Schmidt et al. 2004, Rantala et al. 2017) were abundant in the Hardwood Island and

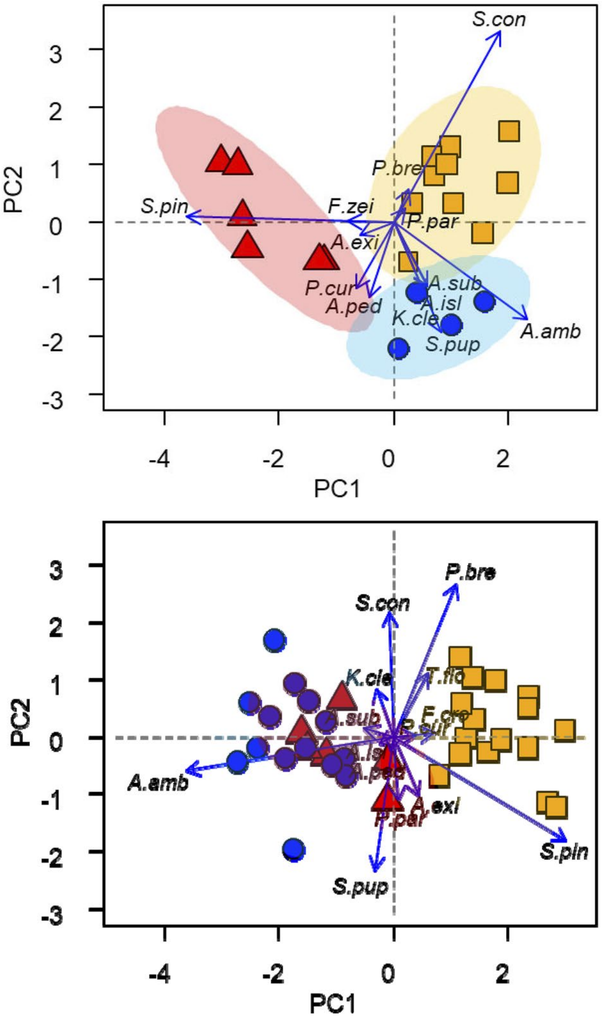

Figure 5. Principal component analysis of diatom assemblages from (A) Hardwood Island and (B) Pickerel Bay. Triangles correspond to zone 3, squares correspond to zone 2, and circles correspond to zone 1.

Pickerel Bay sediment cores. Although several diatom taxa were observed at both sites, the relative contribution of each taxon to the assemblages varied between the 2 sites. Most notably, higher abundances of *A. ambigua* were recorded at the deeper Pickerel Bay (9 m), while it occurred at <20% relative abundance at the shallower Hardwood Island core (5 m). Tychoplanktonic *Aulacoseira* taxa are usually observed at the benthic–planktonic boundary of deep lakes (Kingsbury et al. 2012, Gushulak et al. 2017), where they can maximize light attenuation by remaining suspended in the water column during wind-induced mixing. Hence, the higher abundances of *A. ambigua* at the deeper Pickerel Bay may be influenced by the availability of suitable habitat conditions. This suggests that the differences in site-specific physical characteristics can be important determinants of algal assemblage composition even in shallow multibasin systems. As indicated earlier, we have not assessed whether the timing of diatom assemblage changes coincided at the 2 sites, as a reliable chronology could not be developed for the Pickerel Bay core.

Long-term changes recorded at Hardwood Island

The diatom assemblages and sedimentary organic content prior to the 1890s were relatively stable and only one notable shift occurred around 16 cm. The diatom assemblages in zone 1 (19 – 16 cm) were co-dominated by 3 benthic fragilarioid taxa, *A. ambigua*, and *S. pupula*, all of which have TP optima between 11 and 18 μg/L based on regional surveys of more than 100 lakes from southern Ontario by Reavie and Smol (2001) and Werner and Smol (2005). The mesotrophic range represented by the diatom taxa was reflected in DI-TP reconstruction, as it varied between 11 μg/L and 15 μg/L in zone 1. This indicates that White Lake was mesotrophic before the construction of the dam and major land-use changes.

The relative abundance of *S. pupula* (TP optimum 11 – 13 μg/L) decreased in zone 2, and *S. construens* with a TP optimum between 15 and 18 μg/L increased in abundance. This suggests that the nutrient levels in the lake may have increased subtly during this period, and indeed DI-TP concentrations were slightly higher in zone 2 relative to zone 1. The most notable changes in diatom assemblages and sedimentary organic content occurred around the 1890s. These changes postdated the construction of the dam by nearly 4 decades. However, the nature of geochemical changes was consistent with observations from other lakes impacted by hydro-management and land-use activities (Forrest et al. 2002, Serieyssol et al. 2009). Often construction of dams and land-clearance activities in the watershed lead to excess transport of inorganic material from the catchment, which leads to relative decreases in sedimentary organic content (Serieyssol et al. 2009). However, the delay in the sedimentary geochemical signal at Hardwood Island suggests that construction of the dam may have had limited impact, while land-clearance activities over the subsequent decades have likely led to increased sediment loading into the lake. Furthermore, the highest DI-TP recorded around the late 1800s suggests that allochthonous inputs from land-use changes may have also increased nutrient delivery to the lake.

Both *S. construens* and *S. pinnata* have similar TP optima, 15 – 18 μg/L and 14 – 16 μg/L, respectively. However, toward the end of zone 2 (~7 cm) the relative abundance of *S. construens* decreased, while *S. pinnata* increased. The switch in these 2 benthic fragilarioid taxa was likely driven by factors other than nutrient concentrations. Fluctuations in water levels may have influenced light availability for benthic biota. For instance, the higher water levels maintained after the construction of the dam could have lowered light availability to benthic biota. Although most benthic fragilarioid taxa can tolerate low light conditions (Hall et al. 2004), it is possible that different fragilarioid taxa respond differently to changes in light conditions. The continued success of *S. pinnata* throughout the 20th and early 21st centuries indicates that this taxon can withstand alterations in water levels and shoreline development.

Despite high water levels being maintained during summer months since 1968, the relative abundance of tychoplanktonic *A. ambigua* has steadily decreased beginning around the mid 20th century at the Hardwood Island site. The strengthening of thermal stratification during summer months and weaker water column mixing have led to decreases in the relative abundances of heavily silicified *Aulacoseira* taxa that require turbulent conditions to remain in the water column and maximize light attenuation in thermally stratified lakes across the Northern Hemisphere (Rühland et al. 2015). However, monitoring data collected at Three Mile and Pickerel bays of White Lake do not indicate that the lake stratifies strongly during the summer months (Grégoire and Overholt 2020a). It is likely that the decrease in the relative abundance of *A. ambigua* at Hardwood Island is related to other factors. For instance, recent surveys of the lake have indicated that the macrophyte growth has increased at the lake (Grégoire and Overholt 2021). The higher abundances of macrophytes may be indirectly favoring benthic fragilarioid diatoms that are attached to substrates over tychoplanktonic *A. ambigua*. Additionally, the fluctuations in water levels from the mid to late 20th century may have also altered planktonic habitat availability for *A. ambigua*, thus leading to a decrease in this taxon at Hardwood Island. Interestingly, however, the abundance of *A. ambigua* has increased in the top sediment intervals (5 – 0 cm) of Pickerel Bay. We cannot conclusively determine the mechanism behind the increase in *A. ambigua* at Pickerel Bay, as the timing of this shift cannot be asserted. However, the inter basin differences in the trends for *A. ambigua* warrant further investigations and monitoring.

Implications for management and monitoring

Our paleolimnological data indicate that the limnology of White Lake has changed over the last ~130 years in response to multiple environmental stressors. The most notable and concomitant changes in diatom assemblages and sedimentary organic content occurred around the 1890s, presumably in response to early land-use changes. The biological and geochemical changes induced by damming and land-clearance activities have not been reversed. Specifically, the post-1890s diatom assemblages are remarkably different compared to the previous periods, and the sedimentary organic content has not increased to pre-1890s levels. Although several water-level management plans were implemented between 1968 and 1997, the trajectory of changes induced by 19th century activities was not reversed in both sedimentary proxies.

The recent monitoring data suggest that ice cover and water clarity have been altered at White Lake (Grégoire and Overholt 2020a). The colonization of zebra mussels in 2015 has led to marked changes in water clarity as the Secchi disk depth has increased. The decrease in ice cover is consistent with observations from other lakes in Ontario where warmer temperatures have led to earlier ice-off and later ice-on (e.g., Rühland et al. 2010, Hawryshyn et al. 2012). Algal blooms have also been reported between 2013 and 2020 at several sites across White Lake (Grégoire and Overholt 2020b). This is consistent with the increase in reports of algal blooms in several lakes across southern Ontario (Winter et al. 2011). While nutrient enrichment plays a critical role in the development of cyanobacterial algal blooms, other environmental stressors may also influence the formation of blooms. For example, a growing number of studies are suggesting that climate-mediated changes such as warmer water temperatures and reduced water column mixing can favor certain bloom-forming taxa (e.g., cyanobacteria) in mesotrophic and eutrophic lakes where nutrient levels are already higher (Huisman et al. 2018, Simmatis et al. 2020, Sivarajah et al. 2021). This may be the case at White Lake, where TP measurements from the open water period and historical reconstruction (DI-TP) are indicative of a mesotrophic state (7 – 21 μg/L) and warmer conditions during the past decade have likely provided the ideal setting for cyanobacterial blooms to form.

The ongoing monitoring efforts and this paleolimnological study have provided baseline environmental data and a window into the past, respectively. Specifically, our study suggests that early land-use changes in the catchment and hydro-management activities throughout the 20th century have influenced the limnology of this lake. The recent monitoring programs have reported new issues such as algal blooms and zebra mussel invasions. Therefore, continuous monitoring of White Lake is necessary, as emerging environmental stressors (e.g., invasive species and climatic changes) will likely continue to impact the limnology of this lake. Furthermore, the impacts of future shoreline development around this mesotrophic lake should be assessed within a multiple stressor framework as it is impacted by several co-occurring stressors, including those related to climatic changes and land-use activities. Specifically, to address issues related to algal blooms, both nutrient concentrations and lake thermal properties should be monitored and managed carefully. For instance, annual nutrient budgets from external (i.e., catchment input) and internal (e.g., internal phosphorus loading from sediments) sources should be quantified to assess future changes in nutrient levels. Developing a better understanding of seasonal thermal conditions (e.g., temperature of the epilimnion) will also be useful to determine and predict when cyanobacterial algal blooms could occur. The differences in diatom assemblages between the 2 sites indicate that site-specific variables are influencing biota. Therefore, multiple sites across the lake should be monitored to ensure that the interbasin variability is considered when making management decisions.

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ORCID

Branaavan Sivarajah http://orcid.org/0000-0002-3739-4299

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