

The University of Maine

DigitalCommons@UMaine

Electronic Theses and Dissertations

Fogler Library

Summer 8-2020

Developing a Long-term Perspective on the Sensitivity of Lake Trophic State Changes in Maine

Simona Lukasik

simona.lukasik@maine.edu

Follow this and additional works at: <https://digitalcommons.library.umaine.edu/etd>



Part of the [Terrestrial and Aquatic Ecology Commons](#)

Recommended Citation

Lukasik, Simona, "Developing a Long-term Perspective on the Sensitivity of Lake Trophic State Changes in Maine" (2020). *Electronic Theses and Dissertations*. 3229.

<https://digitalcommons.library.umaine.edu/etd/3229>

This Open-Access Thesis is brought to you for free and open access by DigitalCommons@UMaine. It has been accepted for inclusion in Electronic Theses and Dissertations by an authorized administrator of DigitalCommons@UMaine. For more information, please contact um.library.technical.services@maine.edu.

**DEVELOPING A LONG-TERM PERSPECTIVE ON THE SENSITIVITY OF LAKE
TROPIC STATE CHANGES IN MAINE**

By

Simona M. Lukasik

B.S. SUNY Fredonia, 2018

A THESIS

Submitted in Partial Fulfillment of the

Requirements for the Degree of

Master of Science

(in Ecology and Environmental Science)

The Graduate School

The University of Maine

August 2020

Advisory Committee:

Jasmine Saros, Professor of Paleolimnology and Lake Ecology, Advisor

Jacquelyn Gill, Associate Professor of Paleoecology & Plant Ecology

Amanda Klemmer, Assistant Professor of Food-web Ecology

**DEVELOPING A LONG-TERM PERSPECTIVE ON THE SENSITIVITY OF LAKE
TROPIC STATE CHANGES IN MAINE**

By Simona M. Lukasik

Thesis Advisor: Dr. Jasmine Saros

An Abstract of the Thesis Presented
in Partial Fulfillment of the Requirements for the
Degree of Master of Science
(in Ecology and Environmental Science)
August 2020

Even with similar external drivers, lakes can respond differently because of key ecosystem features that determine lake sensitivity. Identifying factors that determine lake sensitivity are important for successful lake management. The purpose of this research was to determine patterns of algal community change over the past 400 years in lakes with watersheds that vary in surficial geology, specifically the glacio-marine clay layer in Maine known as the Presumpscot formation. Responses to two external drivers, major land use changes and climate change, were assessed. Four lakes were chosen based on their surficial geology and land use history: Unity Pond, Webber Pond, Highland Lake, and Long Lake. Unity and Webber ponds are located on the marine clay layer, while Highland and Long lakes are not. Due to poor diatom preservation and constrained access to the core in recent months, results for Webber Pond are not included here. Unity Pond had higher relative abundances of nutrient-indicating diatoms prior to land use changes compared to the other two lakes. Unity Pond was more sensitive to land use change than Highland and Long lakes, which could be due to disturbance of nutrient-rich soils releasing more P into the lake. In contrast, Highland and Long lakes had no major changes in the

sedimentary diatom or pigment profiles following land use change, but did change in the early 20th century, corresponding to warming climate. The results suggest that the more productive lake was more sensitive to major land use changes, whereas the less productive lakes appeared more sensitive to climate change. This work can be used to gauge sensitivity of lakes to external drivers based on key ecosystem features, to inform management decisions in the future.

ACKNOWLEDGEMENTS

I am thankful for the MAFES program, who funded this research. I would like to thank my advisor, Dr. Jasmine Saros, for her support and guidance throughout my entire project. I am very grateful for the opportunities for field work and lab work that she facilitated. I also thank my committee members, Dr. Jacquelyn Gill and Amanda Klemmer for providing thoughtful input that helped strengthen my thesis. Thank you to the Saros lab members, past and present, who always gave their time and support to contribute to field work, sample analyses, study group discussions, and for the constant laughs: Benjamin Burpee, Kate Warner, Rachel Fowler, Edna Pedraza Garzón, Joe Mohan, Vendy Hazuková, and Matthew Farragher. A special thank you to my many students during my teaching assistantships for always allowing me to go onto tangents about my research during lab sessions.

I am grateful for assistance with algal pigment analysis and interpretation from Rachel Fowler, who was always willing to answer my millions of questions and guide me through each step of the process. I also want to thank Tamara Levitsky for always being able to help me each week setting up and running pigments, for being supportive of my decisions, and for being a listening ear for every situation going on in my daily life. Thank you to L. Brian Perkins for allowing us access to his lab, for his constant help with the HPLC, and for keeping me company while identifying peaks. Thank you to Ben Peierls, through the Lake Environmental Association, for allowing us to use a pontoon to core Highland and Long Lakes, and for his assistance in the coring process. Thank you also to Ben Burpee for helping me core the lakes, and for his constant help with R programming. Thank you to Andrea Nurse for her feedback and insight on my project when I first started. Thank you also to Scott Lariviere for always being available to help

with many details of the research, such as changing out Nitrogen gas tanks, helping me with the LOI process for Unity Pond, and helping me label slides. Thank you to Dr. Jacquelyn Gill for allowing me to use her lab for the LOI process. Thank you to Mike Handley for his assistance and always being able to answer my random questions. Thank you to Becky Addressi and Betty Lee in the Climate Change Institute for their administrative and technical assistance. Thank you to Linda Bacon at the Department of Environmental Protection for allowing me to sub-sample her cores from Unity Pond, Webber Pond, and Sebasticook Lake (not used in this project). Thank you to Jeremy Deeds for sending me his files about the surficial geology in Maine and helping me decipher which lakes were on the glacio-marine clay layer. Thank you to James Deaton for conducting the dating process on Unity Pond. Thank you to Kevin Spigel for his help with the history of Unity Pond. I am also grateful to the School of Biology & Ecology for teaching assistantships.

Thank you to my friends and family back home for your patience and support as I pursued this research. Thank you for being willing to read the multiple rough drafts of my paper, for the random video chats, and for reminding me that you will always be there, no matter where life takes me. Thank you especially to my new friends in Maine. The late night talks, weekly lunch and dinner “dates”, assistance with the details of my research, and for being willing to watch Scooby Doo with me. You all made the move to Maine so much easier and I will never forget the many memories we made together. Thank you also to my roommates, past and present, for making my apartment-living experience enjoyable. Thank you to my church family back in New York, and my new church family in Maine, for all of your prayers and constant love and support. Thank you also to my undergraduate research advisor, Dr. Courtney Wigdahl-Perry, for always being willing to listen, give advice, and for encouraging me to keep moving forward.

TABLE OF CONTENTS

ACKNOWLEDGEMENTS.....	ii
LIST OF TABLES.....	v
LIST OF FIGURES.....	vi
CHAPTER ONE: INTRODUCTION.....	1
CHAPTER TWO: HISTORY AND DESCRIPTION OF STUDY LAKES.....	5
CHAPTER THREE: METHODS.....	9
Sediment core collection and dating.....	9
Algal Pigments.....	10
Data Analyses.....	10
CHAPTER FOUR: RESULTS.....	12
CHAPTER FIVE: DISCUSSION.....	27
CHAPTER SIX: CONCLUSION.....	32
BIBLIOGRAPHY.....	34
BIOGRAPHY OF THE AUTHOR.....	39

LIST OF TABLES

Table 1.	Select characteristics of the four study lakes.....	7
Table 2.	PCA Axis 1 and 2 loadings for each lake.....	22

LIST OF FIGURES

Figure 1.	Location of the four study lakes in Maine, USA, selected for this study.....	6
Figure 2.	Stratigraphic plot of ^{210}Pb activities and sediment chronology in the sediment core from Unity Pond.....	13
Figure 3.	Stratigraphic plot of ^{210}Pb activities and sediment chronology in the sediment core from Highland Lake.....	14
Figure 4.	Stratigraphic plot of ^{210}Pb activities and sediment chronology in the sediment core from Long Lake.....	14
Figure 5.	Relative abundances of major diatom taxa in the Unity Pond core.....	16
Figure 6.	Relative abundances of major diatom taxa in the Highland Lake core.....	18
Figure 7.	Relative abundances of major diatom taxa in the Long Lake core.....	20
Figure 8.	Extracted pigment concentrations, % organic material, and PCA axis 1 in the Unity Pond sediment core.....	21
Figure 9.	Extracted pigment concentrations, % organic material, and PCA axis 1 in the Highland Lake sediment core.....	23
Figure 10.	Extracted pigment concentrations, % organic material, and PCA axis 1 in the Long Lake sediment core.....	25
Figure 11.	Squared chord distance (SCD) for each lake.....	26

CHAPTER ONE

INTRODUCTION

The trophic status, or level of productivity, of a lake is one of its most important features that shapes key aspects of a lake, including water clarity and harmful algal blooms. Changes in trophic states can indicate the sensitivity of a lake to changes in land use and climate; this sensitivity may depend on key ecosystem features. The information collected from studying these factors can be used in management decisions and will also be relevant to drinking water facilities in Maine.

Based on water quality data collected over the past 20 years, there is debate as to whether or not the trophic states of Maine's lakes are changing. Total phosphorus is often an important driver of lake trophic state. Stoddard et al. (2016) found that total phosphorus has increased in surface waters of some northern regions and Huser et al. (2018) showed that total phosphorus has decreased in other surface waters. Satellite data were used to assess whether water clarity changed in Maine's lakes over the past 20 years. Water clarity is important because it can be an indicator of the standing crop of algae, which is one way to quantify lake productivity (McCullough et al. 2013). These authors concluded that water clarity declined in that time period and therefore suggested that this was as a result of changing trophic status. The data were re-evaluated with additional integrated data, including various in-lake measurements, and this re-analysis indicated that water clarity, and hence lake trophic state, were not changing in Maine's lakes (Canfield et al. 2016).

A point of view that is not always integrated into management decisions is the paleoecological one; this would show natural rates of change, variability, and responses to past

land use and climate changes. The research on changing Maine lake trophic states that has been conducted in previous studies was based solely on data from the past twenty years (Hamilton 2010, Davies & Jackson 2006, Tilburg et al. 2015). Paleolimnological data were used during the acid rain debate of the 1980s. Diatom-based reconstructions revealed that pH started to decline in lakes around 1940, with the rate and magnitude of change unusual compared to those in previous centuries, which pointed to anthropogenic sources (Charles et al. 1986; Sullivan et al. 1990). In addition to diatom based reconstructions, algal pigments are an important indicator of long-term lake trophic state changes (McGowan et al. 2005; Patoine & Leavitt 2006). Increases in the abundances of cyanobacteria are not just a recent issue, but started in many lakes of the northern hemisphere around the late 1800s. The main driver of these changes was total phosphorus, followed by warming (Taranu et al. 2015). This underscores the utility of paleoecological approaches in understanding baseline conditions and rates of change prior to substantial anthropogenic disturbances. This is important because in temperate areas, lakes naturally become more productive over time.

Even with similar land use and climate changes, lakes can respond differently to external forcing because of key ecosystem features that determine lake sensitivity. To help explain why lakes respond uniquely to changes in climate, a conceptual model was developed with two specific filters: a Landscape Filter and an Internal Lake Filter. This model contains many features including lake morphology, lake history, and food webs (Blenckner 2005). Blenckner concluded that key features in and surrounding a lake determine lake sensitivity to changes in climate. Understanding which features are important in lake responses to different drivers will aid in better informing lake management decisions.

A key ecosystem feature that could determine sensitivity to changes in land use is surficial geology, considering the role that it plays in controlling phosphorus (P) inputs and availability. In Maine, the presence or absence of the glacio-marine clay layer, also known as the Presumpscot formation, in a watershed is one of the strongest controls on lake P concentrations. This glacio-marine clay layer is the result of marine and glacial flooding of downwarped areas left behind by glacial recession; the deposited glacial-marine clay layer releases P into lakes on the surface, leading to lakes with naturally moderate concentrations of P (Amirbahman et al. 2003). In contrast, other areas in Maine are naturally lower in P. Norton et al. (2011) demonstrated that many lakes in Maine have low phosphorus because soils that developed after the last glaciation accumulated aluminum and iron hydroxides. These captured P in soils and ultimately reduced P supplies to lakes. Surficial geology may thus shape lake sensitivity to nutrient perturbations.

The main questions of this study focus on how lake features shape sensitivity to external drivers, specifically major changes in land use (primarily the initial large land clearance events associated with European settlement) and climate. Responses to these drivers will be compared across areas of differing surficial geology. Have land clearance and climate change affected lakes similarly across different surficial geology in Maine, specifically regarding the presence of the glacio-marine clay layer in the watershed? Does fossil evidence suggest similar nutrient disturbance as a consequence of major land clearance events? The overall goal of this study is to determine patterns of algal production over the past 400 years that vary in key features to inform management decisions. Through paleolimnological techniques, I aim to discover how major land use changes over the past 400 years might have affected lakes in Maine. Understanding the ecosystem features that are determining sensitivity in lakes will result in better understanding of

how lakes will respond to a particular driver, in this case land use and climate. This information will better inform management and drinking water facilities of the risk associated with these changes.

The objective of this study is to determine patterns of algal production over the past 400 years in lakes that vary in key features, which will then be used to inform future management decisions. The first hypothesis is that prior to major land use changes, lakes that are over the marine clay layer will contain higher relative abundances of nutrient-indicating diatoms when compared to lakes without the marine clay layer. Over time, the moderate P loading to lakes situated over the marine clay layer will sustain higher relative abundances of nutrient-indicating diatoms in these lakes compared to those that are not situated over the marine clay layer. The second hypothesis is that changes in land use will promote greater algal community turnover in lakes without the marine clay layer compared to those over it. Because of moderate, sustained background P loading to lakes that are over the marine clay layer, these lakes will be less sensitive to additional nutrient inputs as a result of changes in land use compared to lakes without the marine clay layer. The third hypothesis is that over time, climate-driven changes in lake thermal structure will elicit stronger responses in algal communities of lakes over the marine clay layer than in those without it. The P enrichment from the marine clay layer alleviates a major resource constraint on algal abundance and community structure, elevating the importance of light as a constraining resource. Lake thermal structure strongly controls light availability, hence climate-mediated changes in lake thermal structure and light availability will more strongly affect algal response in these lakes.

CHAPTER TWO

HISTORY AND DESCRIPTION OF STUDY LAKES

This research focused on four study lakes (Fig. 1, Table 1) that were chosen based on their surficial geology and land use history. After considering those factors, lakes were selected based on their size and depth, with a focus on larger and deeper lakes. Lakes that have an area greater than 500 hectares will have wind-driven mixing, and lakes with a depth of greater than 8-10 meters will be plankton dominated. Lakes that fit within these two parameters were chosen to eliminate confounding variables of other types of mixing in the lakes, and allow the focus to be primarily on the plankton communities.

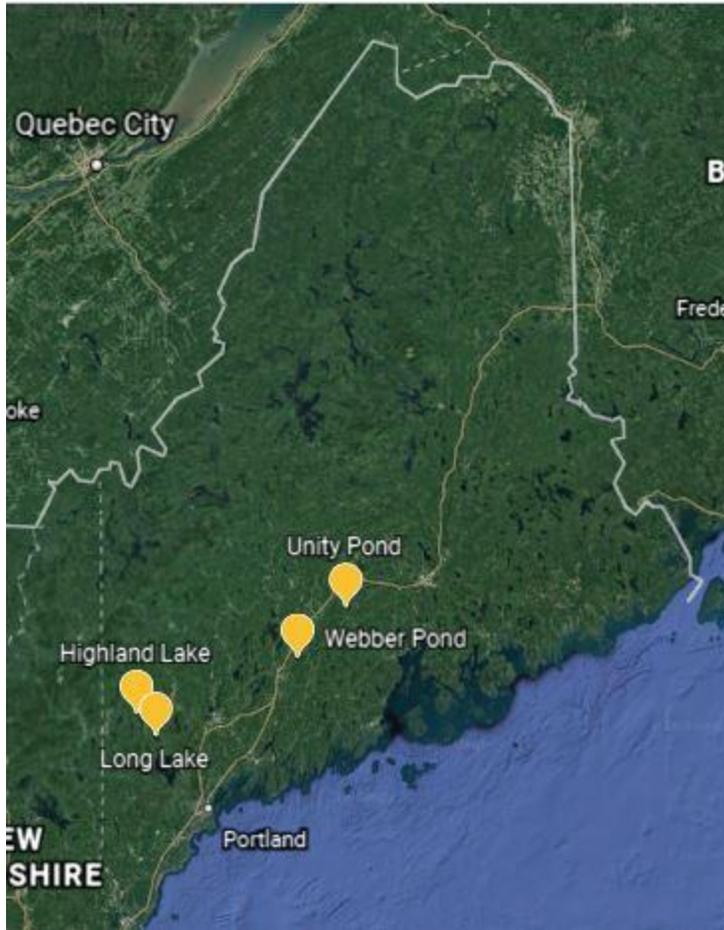


Figure 1. Location of the four study lakes in Maine, USA, selected for this study.

Lake	Surface Area (ha)	Max Depth (m)	Total Phosphorus ($\mu\text{g/L}$)	Avg Secchi (m)	Marine Clay Layer
Unity Pond	1040	12.5	21	2.2	Yes
Webber Pond	499	12.5	18	3.6	Yes
Highland Lake	540	15.2	6	6.7	No
Long Lake	2143	18	4	6.1	No

Table 1. Select characteristics of the four study lakes. Average secchi is based on multiple sampling dates in 2017-2018 (data from Lakes of Maine, www.lakesofmaine.org).

Unity Pond is located in Waldo County, Maine within the towns of Unity, Burnham, and Troy. It has a surface area of 1040 hectares and a maximum depth of 12.5 meters (Table 1). Europeans first settled the area around Unity Pond in 1785. A massive deforestation event occurred in 1830, to clear the land for agricultural practices (<http://www.lakewinnecook.com/docs/unitypondhistory.pdf>). The principal fisheries found here are brown trout, largemouth and smallmouth bass, white perch, and chain pickerel (Maine Department of Inland Fisheries and Wildlife 1989).

Webber Pond is located in Vassalboro, Maine. It has a surface area of 499 hectares and a maximum depth of 12.5 meters (Table 1). Research shows that settlements in the area began as early as 1760 (<https://minerdescent.com/2010/09/10/vassalboro/>). The main occupations of residents were lumbering and farming. The principal fisheries found here are brown trout, smallmouth and largemouth bass, white perch, and chain pickerel (Maine Department of Inland Fisheries and Wildlife 1989).

Highland Lake is located in Bridgton Maine. It has a surface area of 540 hectares and a maximum depth of 15.2 meters (Table 1). Settlement in Bridgton began in 1768 (Mitchell & Hartford 1905). Early businesses included mills, packing plants, and factories (Ward 1974). Principal fisheries found in this lake are brown trout, splake, smallmouth and largemouth bass, white perch, and chain pickerel (Maine Department of Inland Fisheries and Wildlife 1999).

Long Lake is located in Bridgton, Maine. It has a surface area of 2143 hectares and a maximum depth of 18 meters (Table 1). Land use history is the same as it was for Highland Lake. Principal fisheries found in this lake are landlocked salmon, smallmouth and largemouth bass, and chain pickerel (Maine Department of Inland Fisheries and Wildlife 1984).

All four study lakes are located in the Southern Interior climate division in Maine. Between 1600 to 1850, Maine was still in the Little Ice Age, a cooler period. Since 1850, Maine has experienced a period of warming. This had an effect on lakes, including spring ice-out dates. Hodgkins et al (2002) found that lake ice-out dates in New England have become two weeks earlier since 1850, with air temperature being the most important variable in determining this annual trend (Tramoni et al. 1985). The rate of warming in Maine has been increasing over the past century, particularly since 1980 (Jacobson et al. 2009).

CHAPTER THREE

METHODS

The focus of this project was to use sedimentary diatom assemblages and algal pigments from each study lake to test the hypotheses.

Sediment core collection and dating

Sediment from Unity Pond and Webber Pond was sub-sampled from cores collected by Linda Bacon (Department of Environmental Protection). Those cores were extruded in 0.5-1.0 cm increments, and frozen immediately after collection. Lake sediment cores were taken from Highland Lake and Long Lake using a gravity corer and sectioned into 0.5 cm intervals. Sediment sub-samples were then frozen for pigment analysis. Highland Lake and Long Lake had one core collected in 2014 and one core collected in 2019. Loss on ignition (LOI), age, and diatom profiles were determined on the first cores for each lake collected in 2014. The second set of cores, collected in 2019, was used for pigment analysis only. When using the cores from 2014, sediment was collected in 0.5 cm increments, although LOI was only measured on every other interval. The cores from 2019 had sediment for every 0.5 cm increment, which was used for pigment analysis. LOI data were thus interpolated using averages of adjacent intervals as needed for pigment concentration calculations.

Sediment cores were dated based on ^{210}Pb analysis (Appleby & Oldfield 1978), and age models were constructed by assessing linear versus polynomial fits to the data.

Sediment from each core interval was processed for diatom analysis. The sediment was weighed into scintillation vials; 10% HCl was added to remove carbonates, and 30% H_2O_2 was added to remove organic material. The samples were placed onto coverslips and mounted onto

slides with Naphrax®. A minimum of 300 diatom valves were counted for each sample under oil immersion on an Olympus BX-51 microscope with differential interference contrast at x 1000 magnification. Diatom taxonomy was based on Krammer and Lange-Bertalot (1986-1991) and Camburn and Charles (2000).

Algal Pigments

Fossil algal pigments were used to assess the history of lake production and primary producer community structure (Strock et al. 2019). Pigments were isolated using the procedure found in Leavitt & Findlay (1994), separating chlorophylls, carotenoids and derivatives by high-performance liquid chromatography. The samples were prepared following the methods in Chen et al. (2001). Sediment samples were stored at -20°C, then freeze-dried and weighed to around 0.2 g. The samples were extracted overnight in a mixture of acetone:methanol:water (80:15:5) at -20°C. Extracts were filtered through 0.22 µm PTFE syringe filters and dried under N₂ gas. The samples were then re-dissolved in a 70:25:5 mixture of acetone, ion pairing reagent (0.75 g tetrabutyl ammonium acetate and 7.7 g ammonium acetate in 100 ml water), and methanol before being injected into the Agilent HPLC unit (Fowler 2019). Pigments were then identified by spectral characteristics and chromatographic mobility, and compared to commercial pigment standards (Chen et al. 2001).

Data Analyses

Data were analyzed using vegan (Oksanen et al. 2013), analogue (Simpson 2007, Simpson & Oksanen 2020) readxl (Wickham & Bryan 2019), rioja (Juggins 2017), and cluster (Maechler et al. 2018) packages in R version 3.5.0 (The R Foundation for Statistical Computing Platform 2018). Stratigraphic diagrams were made for the abundant diatom species ($\geq 5\%$).

Stratigraphic diagrams were also made for pigments found in each lake. Principal component analysis (PCA) was conducted on diatom assemblages and pigment concentrations to show variations over time (Legendre & Birks 2001). A cluster analysis was also conducted on diatom assemblages and pigment concentrations, with the number of clusters determined by the broken stick model. Clusters were formed using within-group squared Euclidean distances between objects in the dataset (Grimm 1987).

The degree of dissimilarity between diatom assemblages at the bottom of the core (prior to land use changes) and those in every subsequent interval was calculated using the squared chord distance (SCD) dissimilarity index (Overpeck et al. 1985). The SCD values range from 0, for assemblages that are perfectly similar, to 2, for assemblages that are perfectly dissimilar (Wengrat et al. 2019). The SCD was calculated for each core interval in each lake, and used to compare lakes over the marine clay layer to those that are not.

CHAPTER FOUR

RESULTS

Webber Pond sediments contained no diatom frustules, but did have preserved pigments. While pigment data have been collected, LOI data were not available, and will require that I re-sample the archived core at the Maine Department of Environmental Protection. With COVID-19 closures, I have not been able to collect that last piece of information, and the timeline for completing that is unclear. The following results address the other three lakes: Unity Pond, Highland Lake, and Long Lake.

Age models were constructed for Unity Pond (Figure 2), Highland Lake (Figure 3), and Long Lake (Figure 4). Polynomial fits were used for Highland Lake and Long Lake because they had a higher R^2 value than linear models. In contrast, a linear fit was used for Unity Pond. The R^2 value of the polynomial model (0.988) was also higher than the linear fit (0.96) for Unity Pond, but the polynomial model started to assign larger time intervals to each slice from 1850 down the core. As a result of this, the year 1600 was reached at 20 cm, and 250 years (a time span of 1850 to 1600) was compressed into about 8 cm. The top 12 cm of the Unity core captured about 1850 to present in both the linear and polynomial models for this lake, so both models were comparable for this period (capturing about 13 years per 1 cm slice on average), and this is the period for which ^{210}Pb is most accurate. In the polynomial model, prior to 1850, each 1 cm slice would represent 31 years of sedimentation, on average. In the linear model, prior to 1850, each 1 cm slice would represent 14 years of sedimentation, on average. Therefore, a linear model was chosen for the whole Unity Pond core because it was more consistent with the sedimentation rate that both the linear and polynomial models indicated for the 1850 to present time period.

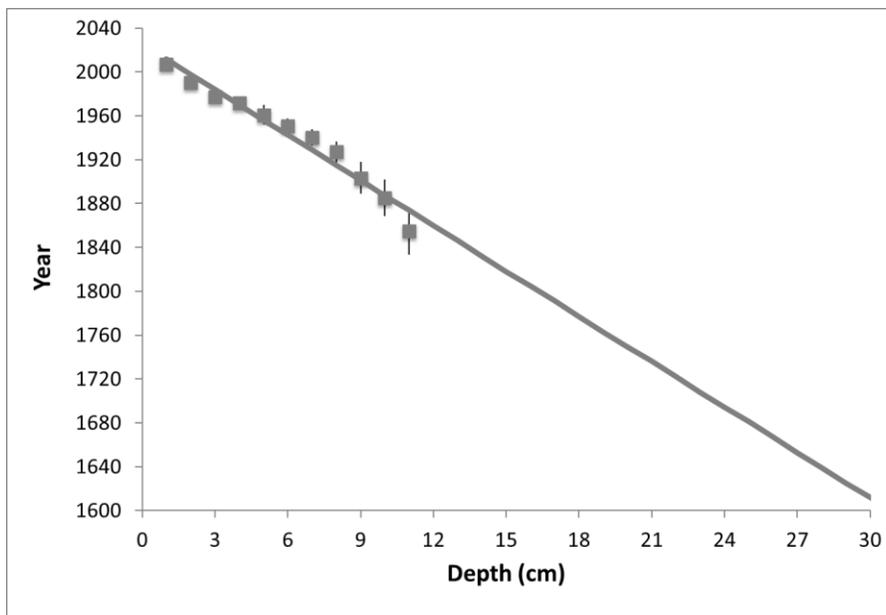


Figure 2. Stratigraphic plot of ^{210}Pb activities and sediment chronology in the sediment core from Unity Pond. Error bars represent the error in the estimated dates.

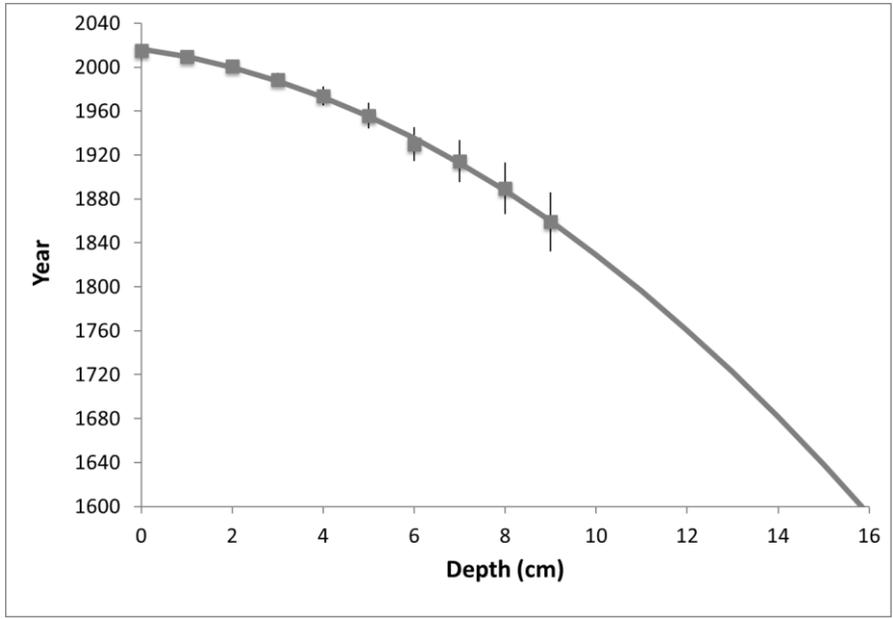


Figure 3. Stratigraphic plot of ^{210}Pb activities and sediment chronology in the sediment core from Highland Lake. Error bars represent the error in the estimated dates.

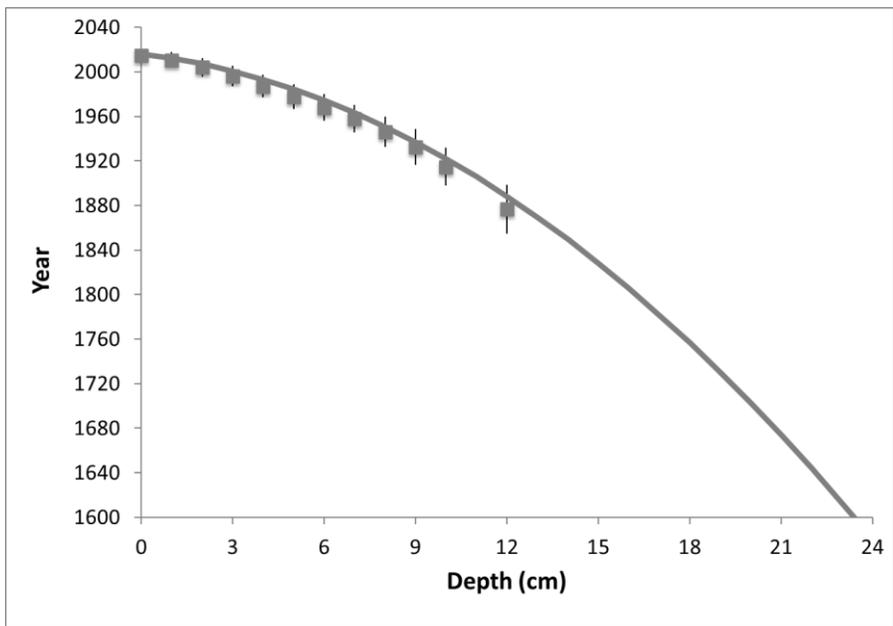


Figure 4. Stratigraphic plot of ^{210}Pb activities and sediment chronology in the sediment core from Long Lake. Error bars represent the error in the estimated dates.

Sedimentary diatom assemblages in Unity Pond varied over the past 400 years (Figure 5). The major diatom taxa present in the core were *Lindavia bodanica* (Eulens. ex Grunow) Nakov, Guillory, M.L. Julius, E.C. Ther. and A.J. Alverson, *Tabellaria flocculosa* (Roth) Kütz, *Fragilaria crotonensis* (Kitton), *Discostella stelligera* (Cleve and Grunow) Houk and Klee, and *Aulacoseira pusilla* (Meister) Tuji and Houk. Cluster analysis of diatom assemblages identified three zones: 1600 to 1780, 1780 to 1840, and 1840 to present. During the 1600s and much of the 1700s, *L. bodanica* and *T. flocculosa*, dominated the record with each generally representing 10-28% of assemblages. *Fragilaria crotonensis*, *D. stelligera* (8-15%), and *A. pusilla* (3-21%) were also abundant during this period. At the end of the 1700s, the percent organic content of sediments started to increase, from 11 to 13%. After 1780, the relative abundances of *L. bodanica* declined to less than 10% of assemblages, while *A. pusilla* and *F. crotonensis* generally increased, with each comprising more than 20% of assemblages. Relative abundances of *T. flocculosa* were relatively high during this period as well as to present. After 1840, the percent organic content of sediments continued to increase and *A. pusilla* further increased while relative abundances of *F. crotonensis* became more variable. *Asterionella formosa* (Hassall) also increased to 5-12% of assemblages after the mid-1800s, with relative abundances prior to this time being less than 5%. PCA analyses suggest the greatest community shifts occurred around 1780, after which trends in PCA axis 1 scores largely reflect the pattern of change in *A. pusilla*.

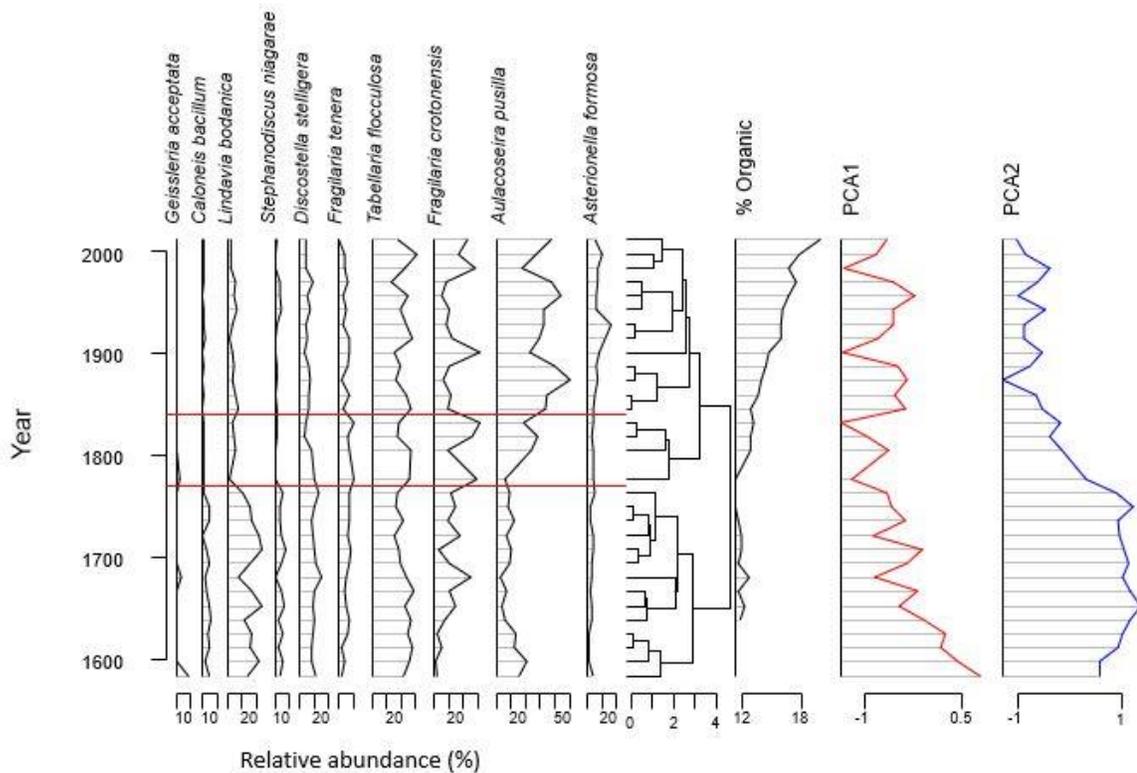


Figure 5. Relative abundances of major diatom taxa in the Unity Pond core, with ^{210}Pb chronology on the left. Cluster analysis, % organic material, and PCA axis 1 and 2 sample scores are plotted on the right. Horizontal red lines represent the breaks from the cluster analysis.

Sedimentary diatom assemblages in Highland Lake also varied over the past 400 years (Figure 6). In contrast to Unity Pond, the Highland Lake record tended to have more diatom taxa at moderate relative abundances as opposed to a few dominant species. Diatom taxa with the highest relative abundances in the core were *Discostella stelligera*, *Aulacoseira ambigua* (Grunow) Simonsen, *Aulacoseira distans* (Ehrenberg) Simonsen, and *Fragilaria exigua* (Grunow). Cluster analysis of diatom assemblages identified four zones: 1600 to 1730, 1730 to 1935, 1935 to 1980, and 1980 to present. From 1600 to 1730, *D. stelligera*, *A. ambigua*, *A. distans*, and *F. exigua* were the most abundant taxa, with each generally representing 8-20% of

assemblages. At the end of this time period, the percent organic material in sediments slightly increased from about 25% to 26-27%. The next cluster of diatom assemblages, spanning from 1730 to 1935, was characterized by slightly higher but variable relative abundances of *D. stelligera* (14-27%) and *L. bodanica* (4-10%), sustained moderate but variable relative abundances of *A. ambigua* (3-21%) and *A. distans* (3-16%), and a decline in the relative abundance of *F. exigua* (<10%). Starting around 1935, *A. ambigua*, *A. distans*, and *L. bodanica* declined, while *D. stelligera* continually increased over this period from about 30-46% of assemblages. After 1980, the relative abundances of all *Aulacoseira* taxa declined further while *D. stelligera* dominated assemblages at 40-50%. *Asterionella formosa* also increased after 1980, ranging from 5 to 11% of assemblages. PCA axis 1 scores suggest that the greatest community turnover occurred during the 20th century.

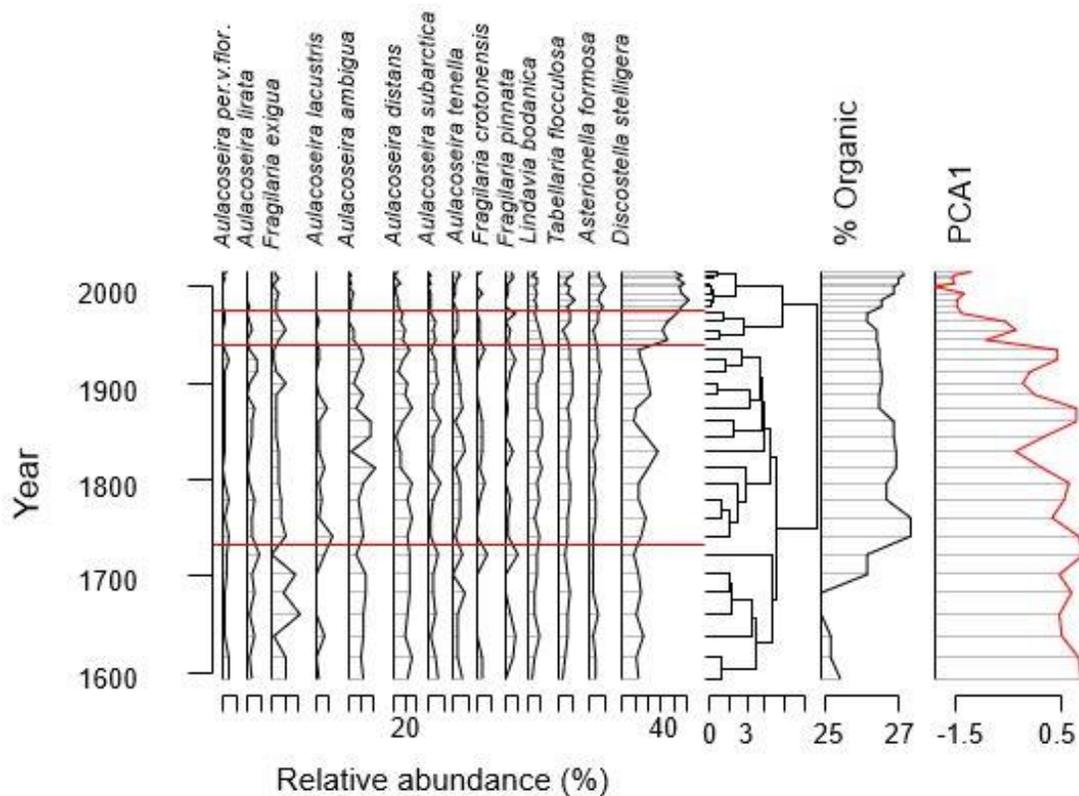


Figure 6. Relative abundances of major diatom taxa in the Highland Lake core, with ^{210}Pb chronology on the left. Cluster analysis, % organic material, and PCA axis 1 sample scores are plotted on the right. Horizontal red lines represent the breaks from the cluster analysis.

Similar to Highland Lake, the Long Lake record had more diatom taxa at moderate relative abundances as opposed to a few dominant species (Figure 7). Diatom taxa with the highest relative abundances in the core were *Aulacoseira subarctica* (O.Müll.) E.Y.Haw , *A. ambigua*, *A. distans*, *T. flocculosa* and *D. stelligera*. Cluster analysis of diatom assemblages identified four zones: 1600 to ~1710, 1710 to ~1890, 1890 to 1987, and 1987 to present. From 1600 to ~1710, *A. ambigua*, *A. subarctica*, *A. distans*, *T. flocculosa*, and *D. stelligera* were the most abundant taxa, with each generally representing 8-22% of assemblages. The next cluster of

diatom assemblages, spanning from 1710 to 1890, was marked by modest increases in the relative abundances of *L. bodanica* (4-10%) and *F. crotonensis* (0-10%) and a general decrease in *Aulacoseira tenella* (Nygaard) Simonsen. The relative abundances of *A. ambigua*, *A. subarctica*, *A. distans*, *T. flocculosa*, and *D. stelligera* remained similar to the first period over this time, while *Lindavia michiganiana* (Skvortsov) Nakov, Guillory, Julius, Theriot & Alverson slightly increased in relative abundance during the second half of this period. At the end of this time period, there was a sharp decrease in % organic material to about 16%; this rebounded shortly thereafter. Between 1890 to 1987, there was a decrease in relative abundances of *A. ambigua* and *F. crotonensis*, with each generally representing 0-7% of assemblages, while *D. stelligera* increased steadily (17-39%). Relative abundances of *A. subarctica* and *A. distans* were more variable over this time, while those of *T. flocculosa* remained similar. In the last time period of 1987 to present, the % organic content of sediments continually increased. Relative abundances of *A. subarctica* and *A. distans* decreased, while those of *T. flocculosa* and *D. stelligera* were higher. PCA axis 1 scores suggest that the greatest community turnover started with the 20th century.

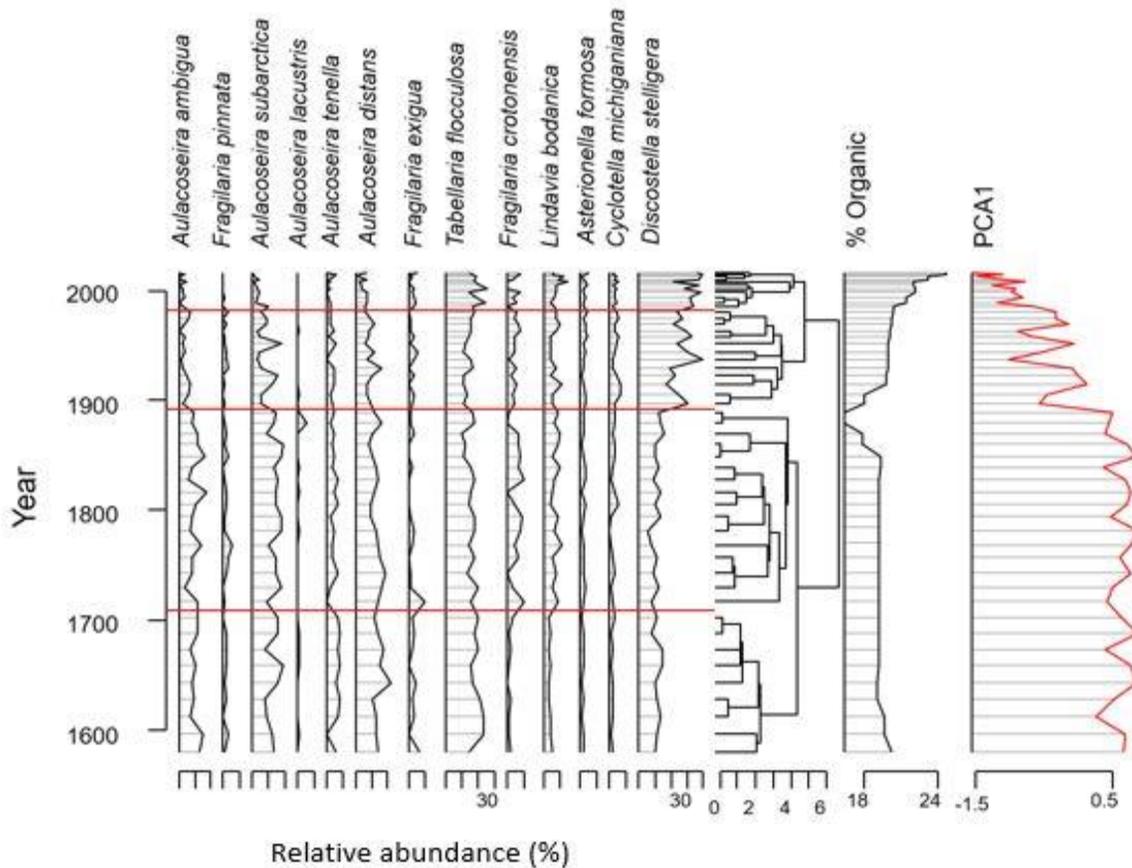


Figure 7. Relative abundances of major diatom taxa in the Long Lake core, with ^{210}Pb chronology on the left. Cluster analysis, % organic material, and PCA axis 1 sample scores are plotted on the right. Horizontal red lines represent the breaks the from cluster analysis.

Sedimentary algal pigments in the Unity Pond core varied over the past 400 years (Figure 8). Alloxanthin (cryptophytes), Diatoxanthin and Fucoxanthin (dinoflagellates, diatoms, and chrysophytes), Lutein and Chl *b* (chlorophytes), Zeaxanthin (cyanobacteria), Canthaxanthin (colonial cyanobacteria), Chl *a*, Phaeophytin *a*, Phaeophorbide *a*, and β -carotene (all algal groups) were all present in the core. PCA axes 1 and 2 were strong predictors of patterns of algal change; axis 1 explained 81% of the variance, while axis 2 explained 10% of the variance. Axis 1 was highly positively related to the concentrations of Phaeophorbide *a* and Phaeophytin *a*, and

weakly negatively related to Zeaxanthin and Lutein. Axis 2 was positively related to the concentrations of Chl *a* and negatively related to Canthaxanthin and Chl *b* (Table 2).

Cluster analysis of pigment concentrations identified a shift at 1780. Between the period of 1600 to 1780, concentrations for all pigments were relatively low. From 1780 to present, there was an increase in concentrations of Lutein, Canthaxanthin, and Phaeophytin *a* pigments. At around 1830, there was a large spike in concentrations of Phaeophorbide *a*, and a steady increase in Diatoxanthin pigments.

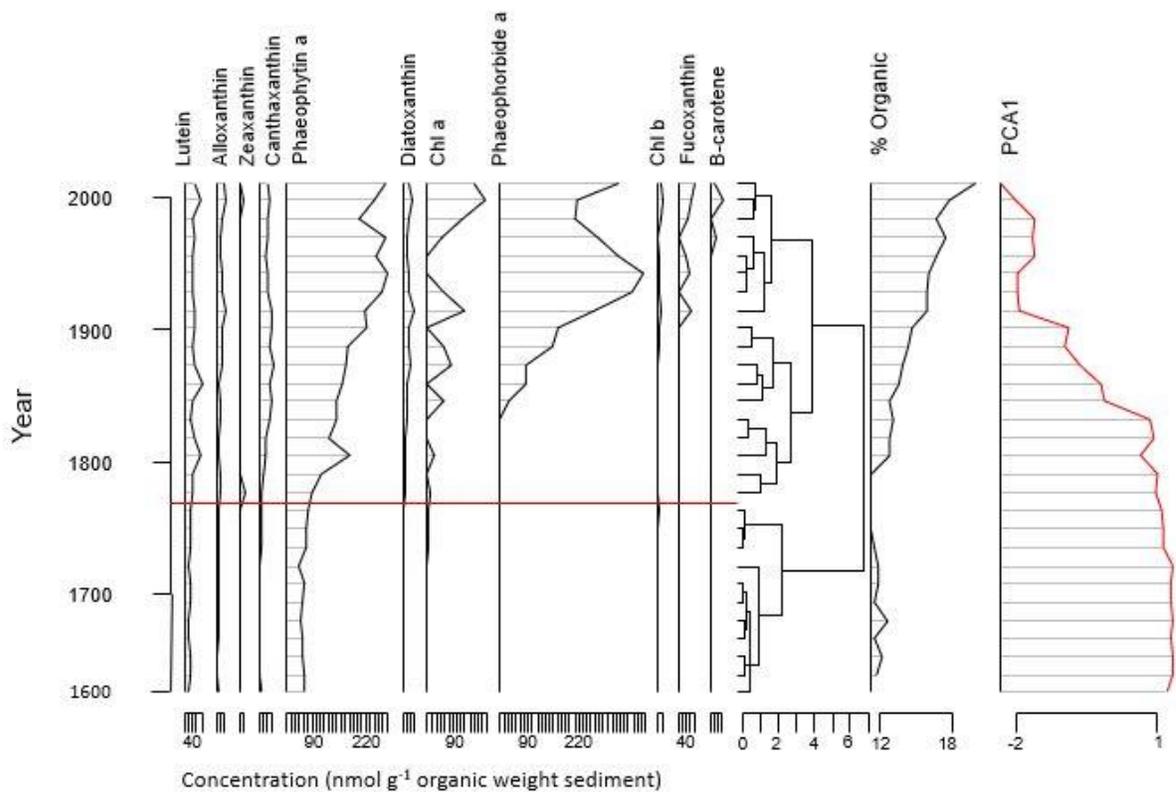


Figure 8. Extracted pigment concentrations, % organic material, and PCA axis 1 in the Unity Pond sediment core. Horizontal red line represents the break from the cluster analysis.

Pigment	Algal Group(s)	Unity Pond		Highland Lake		Long Lake	
		Axis 1	Axis 2	Axis 1	Axis 2	Axis 1	Axis 2
Phaeophorbide <i>a</i>	All algae	-5.23	-0.85	-3.03	-0.59	-1.93	-2.32
Phaeophytin <i>a</i>	All algae	-2.48	-0.18	-2.4	0.67	-2.27	0.99
Chl <i>a</i>	All algae	-1.97	1.97	-2.04	-0.07	-2.01	0.63
β -carotene	All algae	-0.5	0.41	—	—	-0.03	-0.13
Chl <i>b</i>	Chlorophytes	-0.67	0.12	—	—	0.02	0.01
Lutein	Chlorophytes	-0.32	0.13	-1.34	-0.70	-1.29	0.09
Alloxanthin	Cryptophytes	-0.62	0.2	-0.53	-1.72	-0.68	0.26
Zeaxanthin	Cyanobacteria	-0.05	0.25	—	—	—	—
Canthaxanthin	Colonial Cyanobacteria	-0.86	0.12	1.23	-0.58	-0.24	-0.05
Fucoxanthin	Dinoflagellates, Diatoms, Chrysophytes	-1.2	0.21	-2.09	0.56	-0.29	0.08
Diatoxanthin	Diatoms, Dinoflagellates, Chrysophytes	-1.12	0.14	-0.43	0.57	0.65	-1.03

Table 2. PCA Axis 1 and 2 loadings for each lake, with the algal groups indicated to which each pigment corresponds.

In the sedimentary pigment profile of Highland Lake (Figure 9), Canthaxanthin, Alloxanthin, Lutein, Diatoxanthin, Fucoxanthin, Phaeophorbide *a*, Phaeophytin *a*, and Chl *a* were all present in the core. PCA axes 1 and 2 were strong predictors of patterns of algal change; axis 1 explained 64% of the variance, while axis 2 explained 12% of the variance. Axis 1 was highly positively related to the concentrations of Canthaxanthin and Phaeophorbide *a*, and weakly negatively correlated to Alloxanthin and Diatoxanthin. Axis 2 was positively related to the concentrations of Alloxanthin, and negatively correlated to Chl *a* (Table 2).

Cluster analysis of pigment concentrations identified two zones: 1600 to 1950, and 1950 to present. Note that with the age model for the 2014 core applied to this 2019 core, this means that the breaks in the cluster analysis on the pigment record actually occurred just slightly later than the age model indicates. Between the period of 1600 to about 1950, most pigment concentrations were relatively low, with Phaeophytin *a* and Phaeophorbide *a* showing slightly higher concentrations. Starting around 1950, Alloxanthin, Lutein, Diatoxanthin, Phaeophytin *a*,

Chl *a*, Phaeophorbide *a*, and Fucoxanthin concentrations gradually increase. The spike in pigment concentrations at the top of the core may be due to better pigment preservation.

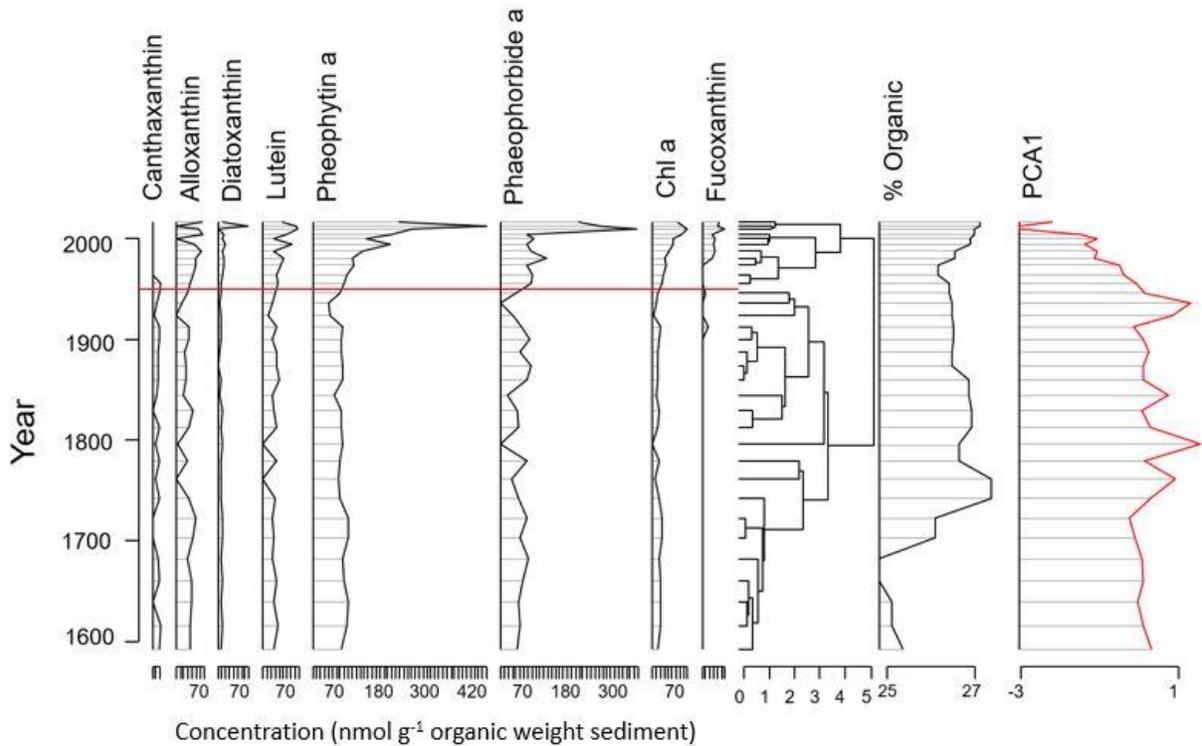


Figure 9. Extracted pigment concentrations, % organic material, and PCA axis 1 in the Highland Lake sediment core. Horizontal red line represents the break from the cluster analysis.

In the Long Lake core (Figure 10), Phaeophorbide *a*, Fucoxanthin, Alloxanthin, Diatoxanthin, Lutein, Canthaxanthin, Chl *b*, Chl *a*, Pheophytin *a*, and β -carotene were all present in the core (Table 2). PCA axes 1 and 2 were strong predictors of patterns of algal change; axis 1 explained 43% of the variance, while axis 2 explained 22% of the variance. Axis 1 was highly positively related to the concentrations of Pheophytin *a* and Chl *a* and weakly negatively correlated with Chl *b* and β -carotene. Axis 2 was positively related to the

concentrations of Phaeophorbide *a* and Diatoxanthin and negatively related to Chl *b* and Canthaxanthin (Table 2).

Cluster analysis of pigment concentrations identified a shift after 1930. As with Highland Lake, the age model for the 2014 core was applied to this 2019 core, and means that the breaks in the cluster analysis on the pigment record actually occurred just slightly later than the age model indicates. From around 1930 to present, there was a steady increase in Alloxanthin, Phaeophytin *a*, and Lutein pigments. There was also a slight decrease in Diatoxanthin during this time period. The spike in pigment concentrations at the top of the core may be due to better pigment preservation.

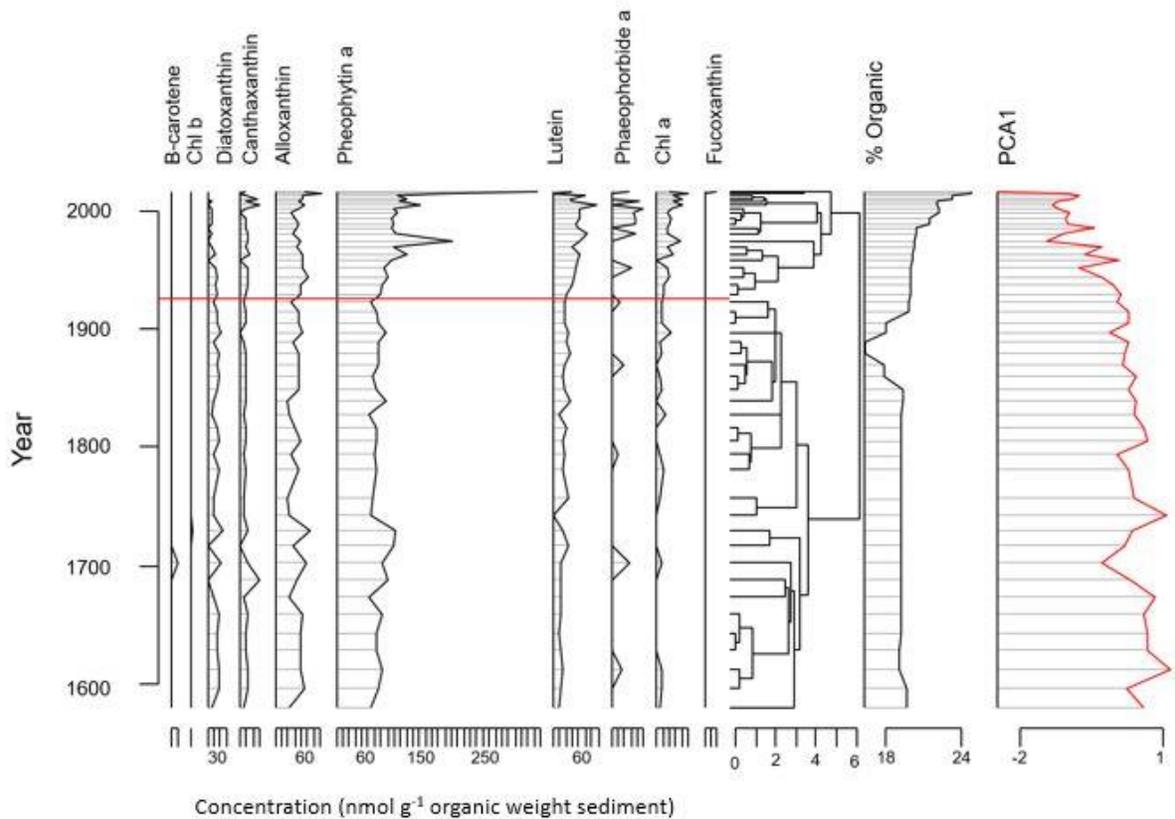


Figure 10. Extracted pigment concentrations, % organic material, and PCA axis 1 in the Long Lake sediment core. Horizontal red line represents the break from the cluster analysis.

The SCD analysis for each lake showed the degree of dissimilarity between the diatom assemblage at the bottom of the core (pre-land use change) to each subsequent assemblage (Figure 11). The same breaks that were identified with cluster analysis are indicated on the SCD plots. The Unity Pond core started off with a low degree of dissimilarity, but gradually increased until 1780, where there was a large increase in dissimilarity. For the rest of the core, the degree of dissimilarity wavered, but still had large peaks post-land use changes. Highland and Long lakes showed similar patterns to each other. In both cores, between the period of 1600 to 1935, diatom assemblages remained similar. Starting around 1935, assemblages in both lakes showed a

large change, with higher degrees of dissimilarity. Around 1980, both lakes showed another increase in dissimilarity. The dissimilarity decreased in Highland Lake at the top of the core, but increased in Long Lake at the top of the core.

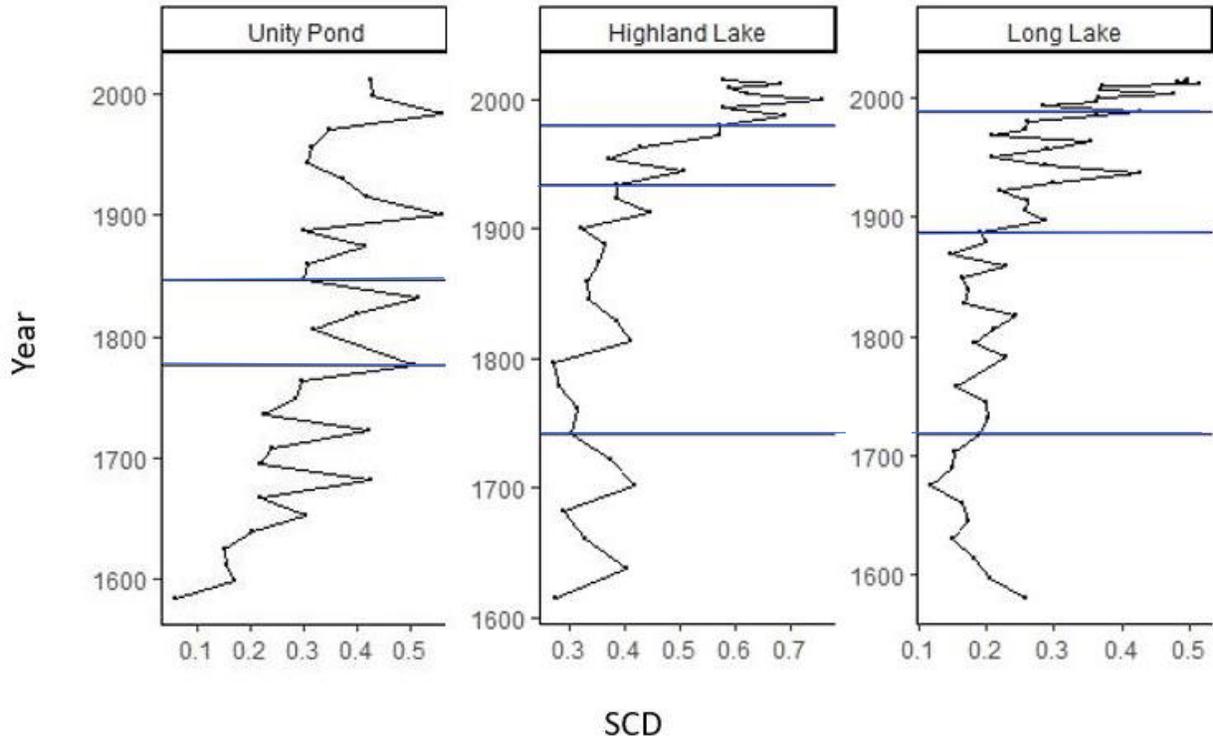


Figure 11. Squared chord distance (SCD) for each lake. Blue lines show the same break points constructed using the cluster analysis for the diatom records. The bottom sample for each lake has a SCD of zero, but is not shown on the plots to improve clarity of the figures.

CHAPTER FIVE

DISCUSSION

The timing of major changes in algal communities differed between Unity Pond, located on the marine clay layer, and Highland and Long lakes, which are not on the marine clay layer. In the Unity Pond core, both diatom assemblages and pigments changed around 1780. During this time, Europeans were starting to settle the area around Unity Pond. There was also a change in diatom assemblages around 1840, which corresponds to a massive deforestation event in 1830 to clear the land for agricultural practices. In the Highland and Long lake cores, changes in diatom assemblages occurred in the early 1700s and at multiple points over the past 110 years. The area around those lakes was settled in 1768 (Mitchell & Hartford 1905), which did not correspond to any changes in the algal records. In both Highland and Long lakes, diatom assemblages changed around 1980, and pigment shifts occurred during the 20th century. Since 1850, Maine has experienced a period of warming, with the rate of warming increasing particularly since 1980 (Jacobson et al. 2009). Collectively, these sedimentary algal records suggest different sensitivity to external drivers in Unity Pond versus Highland and Long lakes.

The sedimentary diatom profiles from all three lakes supported the first hypothesis, with higher relative abundances of nutrient-indicating diatoms prior to land use changes in Unity Pond compared to the other two lakes. Prior to major land use changes, Unity Pond diatom assemblages included nutrient-indicating taxa such as *F. crotonensis*, *F. tenera*, *L. bodanica*, and *S. niagarae*. *Fragilaria crotonensis* is abundant in mesotrophic-eutrophic lakes; the growth of this taxon can be stimulated by nitrogen or phosphorus (Saros et al. 2005). *Fragilaria tenera* is also a diatom typically found in mesotrophic lakes, and can also be used as an indicator of nutrient influxes. Tropea et al. (2011) found that abundances of *F. tenera* started to increase in

Ramsey Lake (Sudbury, Ontario, Canada) due to cultural eutrophication in the 1960s. In some lakes, *L. bodanica* tends to be found at moderate P concentrations, in more mesotrophic lakes (Wunsam et al. 1995). *Stephanodiscus* species are also found at moderate P concentrations (Ramstack et al. 2003). Collectively, the moderate relative abundances of these taxa in Unity Pond prior to land use change suggest natural enrichment of the lake with phosphorus. In contrast, in the Highland and Long lake cores, the diatom records showed a different pattern, with lower relative abundances of any nutrient-indicating diatoms prior to land use changes, and assemblages containing various oligotrophic taxa including several *Aulacoseira* species (Tropea et al. 2011).

The results from the Unity Pond diatom and pigment profiles reject the second hypothesis. This lake, located over the marine clay layer, was actually more sensitive to land use change than Highland and Long lakes, not located over the marine clay layer. The soils on the marine clay layer are more nutrient rich, and when these soils are disturbed, this may release P into the lake in strong pulses; this may be the reason why diatoms responded during major land use changes. The soils around Highland and Long lakes are not as nutrient rich, which may be why diatoms did not respond as strongly to land use changes. Sedimentary algal pigments are also an indicator of trophic state changes in the lakes. Riedinger-Whitmore et al. (2005) found that patterns of cyanobacterial pigment change in many Florida lakes strongly corresponded to many sources of eutrophication, such as expansion of agriculture, residential development, and point-source nutrient inputs. In the Unity Pond core, there was an increase in Canthaxanthin pigments, from cyanobacteria, when the land was settled in 1780, further indicating eutrophication in Unity Pond at this time.

As pointed out above, there were many diatom taxa in the Unity Pond core prior to major land use changes that are indicators of P. This supports the claim that there was decent P availability in Unity Pond prior to these land use changes. Post-land use change, relative abundances of species with moderate P declined, while N-indicating species increased. *Asterionella formosa* can respond to N enrichment (Saros et al. 2005); *A. pusilla* (synonym: *Aulacoseira subborealis*) can respond to N in some systems (Williams et al. 2016), and is found in eutrophic waters (Denys et al. 2003). This shift in diatom assemblages may suggest that agricultural practices began supplying the lake with more N, and stimulated an increase in taxa with higher N requirements. I note that the data analysis is based on relative, not absolute, abundances of species; as a result, when some species increase in abundance, relative abundances of others will decline even if their absolute numbers are unchanged. It is thus unclear whether some of the taxa with higher P requirements, such as *L. bodanica* and *S. niagarae*, actually declined with land use change or simply became a smaller percentage of the assemblage because other taxa increased in abundance.

A polynomial age model was applied to the Highland and Long lake cores, whereas a linear age model was applied to the Unity Pond core. The linear model was a poorer fit for Highland and Long lake, but if applied, it would have changed the dates of the major shifts in the record. In the polynomial model for Highland Lake, the break point was at 1730, but the linear model would have put that break point at 1810. For Long Lake, the polynomial model identified a shift at 1710, but the linear model would have identified that shift at 1815. Applying the linear models would have put these shifts about four decades after the major land use change for these lakes. If the linear age model was more accurate, even though the fit is not as good as the polynomial model, diatom assemblages in Highland and Long lakes would have responded more

slowly to land use change than those in Unity Pond. This could be because disturbance here did not lead to as much nutrient release as on the marine clay layer, or the nature of the disturbance (i.e. degree of land use change) may not have been as severe. Unity Pond, the lake over the marine clay layer, responded to land use change immediately, while Highland and Long lakes did not. Regardless of the age model used, Highland and Long lakes were not as sensitive to land use change as Unity Pond.

Patterns of change in the diatom assemblages of Highland and Long lakes reject the third hypothesis. Highland and Long lakes are more oligotrophic-mesotrophic when compared to Unity Pond, and are not located over the marine clay layer. The results showed that diatom assemblages in these lakes were more sensitive to climate change than those in Unity Pond. After the Little Ice Age (post-1850), warming caused lake ice-out dates in New England to become two weeks earlier (Hodgkins et al. 2002). Warming has also caused average wind speed to decline in the region during the 20th century, affecting lake stratification (Boeff et al. 2016). In both the Highland Lake and Long Lake diatom records, there was an increase in the relative abundance of *D. stelligera* post-1850. Boeff et al. (2016) used *D. stelligera* as an indicator of shifting lake thermal stratification, revealing that declining wind speeds over the 20th century led to shallower lake stratification in Tunk Lake, Maine. I can apply this same inference to Highland and Long lakes, as they are also large lakes (greater than 500 ha in surface area) in which wind is the primary driver of lake thermal structure (Fee et al. 1996). My results reveal that diatom assemblages of these two lakes were more responsive to climate than those of the high nutrient Unity Pond. In Unity Pond, the relative abundances of *D. stelligera* declined after major land use changes. The change in climate may have also affected this taxon, and it may have responded favorably to warming, but strong increases in the relative abundances of other species from

eutrophication may have drowned out the climate signal here. In addition, *D. stelligera* can respond to multiple drivers. In higher nutrient systems, like Unity Pond, it has a different ecology than in lower nutrient systems and may not exhibit the same relationship with thermal structure (Saros & Anderson 2015). Overall, these results suggest that sedimentary diatom profiles in oligotrophic-mesotrophic lakes may provide more insight into climate-driven lake ecosystem changes than those from eutrophic lakes.

CHAPTER SIX

CONCLUSION

The results from this research revealed that a lake on the glacio-marine clay layer was more sensitive to land use changes than lakes that are not on this layer. This may be due to a stronger release of nutrients from the P-rich soils of the marine clay layer. My research also demonstrated that the sedimentary diatom assemblages in oligotrophic-mesotrophic lakes responded more strongly to climate-driven changes than those in eutrophic lakes. One question that was raised after this research is why are the lakes on the clay layer more sensitive to land use changes when they have already been exposed to nutrients from the P-rich soils? An experiment could be conducted in a lake over the marine clay layer, introducing nutrients into a mesocosm and assessing the phytoplankton response. A similar experiment could be repeated in a lake not over the marine clay layer, with results between the two lake types compared. Another question that was raised after this research is why did diatom assemblages in the oligotrophic-mesotrophic lakes, not located on the marine clay layer, respond to climate changes but not to land use changes? Future work could investigate other oligotrophic lakes in the Southern Interior division to assess if they also responded the same way to climate change. Their land use history could also be investigated to determine if they responded differently to land use changes than Highland and Long lakes. The extent or degree of land use change around Highland and Long lakes may explain the weaker algal response in these systems.

My purpose with this project was to assess how key ecosystem features of lakes shape their response to external drivers such as changes in land use and climate. The key ecosystem feature examined here, surficial geology, played an important role in determining lake

sensitivity. This knowledge of sensitivity can be used to inform management decisions and to ensure the continued high quality of these ecosystems into the future.

BIBLIOGRAPHY

- Amirbahman, A., A.R. Pearce, R.J. Bouchard, S.A. Norton & J.S. Kahl. 2003. Relationship between hypolimnetic phosphorus and iron release from eleven lakes in Maine, USA. *Biogeochemistry* 65 (3): 369-386.
- 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.
- Blenckner, T. 2005. A conceptual model of climate-related effects on lake ecosystems. *Hydrobiologica* 533: 1-14.
- Boeff, K.A., K.E. Strock & J.E. Saros. 2016. Evaluating planktonic response to climate change across three lakes with differing morphometry. *J Paleolimnol* 56: 33-47. doi: 10.1007/s10933-016-9889-z
- Camburn, K.E. & D.F. Charles. 2000. Diatoms of low-alkalinity lakes in the northeastern United States. Special Publication 18. Academy of Natural Sciences, Philadelphia, PA.
- Canfield Jr., D.E., R.W. Bachmann, D.B. Stephens, M.V. Hoyer, L. Bacon, S. Williams & M. Scott. 2016. *Inland Waters* 6: 11-27. doi: 10.5268/IW-6.1.864
- Charles, D.F. & S.A. Norton. 1986. Paleolimnological evidence for trends in atmospheric deposition of acids and metals. In J. Gibson (chairman). *Acid Deposition: Longterm Trends*. National Academy Press, Washington, D.C.: 335-506. doi:10.1016/S0146-6380(00)00194-7
- Chen, N., T. S. Bianchi, B. A. McKee & J. M. Bland. 2001. Historical trends of hypoxia on the Louisiana shelf: Applications of pigments as biomarkers. *Organic Geochemistry*, 32(4): 543-561.
- Davies, S. P. & S. K. Jackson. 2006. The Biological Condition Gradient: A Descriptive Model for Interpreting Change in Aquatic Ecosystems. *Ecological Applications*, 16(4): 1251-1266.
- Denys, L., K. Muylaert, K. Krammer, T. Joosten, M. Reid & P. Rioual. 2003. *Aulacoseira subborealis* stat. nov. (Bacillariophyceae): a common but neglected plankton diatom. *Nova Hedwigia* 77: 407-427. doi: 10.1127/ 0029-5035/ 2003/0077-0407
- Fee, E.J., R.E. Hecky, S.E.M. Kasian & D.R. Cruikshank. 1996. Effects of lake size, water clarity, and climatic variability on mixing depths in Canadian Shield lakes. *Limnol. Oceanogr.* 41(5): 912-920. <https://doi.org/10.4319/lo.1996.41.5.0912>
- Fowler, Rachel. "Assessing the Mechanisms and Implications of Altered Carbon Cycling in Arctic and Boreal Lakes" (2019). Electronic Theses and Dissertations. 2971. <https://digitalcommons.library.umaine.edu/etd/2971>

- Grimm, E.C. 1987. CONISS: A FORTRAN 77 program for stratigraphically constrained analysis by the method of incremental sum of squares. *Computers & Geosciences*, 13(1): 13-35.
- Hamilton, A., J.D. Stamp & B.G Bierwagen. 2010. Vulnerability of biological metrics and multimetric indices to effects of climate change. *Journal of the North American Benthological Society*, 29(4): 1379-1396.
- Highland Lake. 1999. Maine Department of Inland Fisheries and Wildlife [PDF file]. Retrieved from https://www.maine.gov/ifw/docs/lake-survey-maps/cumberland/highland_lake1.pdf
- History of Unity Pond. Friends of Lake Winnecook [PDF file]. Retrieved from <http://www.lakewinnecook.com/docs/unitypondhistory.pdf>
- Hodgkins, G.A., I.C. James & T.G. Huntington. 2002. Historical Changes in Lake Ice-Out Dates as Indicators of Climate Change in New England, 1850-2000. *International Journal of Climatology* 22: 1819-1827. doi: 10.1002/joc.857
- Huser, B.J., M.N. Futter, R. Wang & J. Fölster. 2018. Persistent and widespread long-term phosphorus declines in Boreal lakes in Sweden. *Science of the Total Environment* 613-614: 240-249.
- Jacobson, G.L., I.J. Fernandez, P.A. Mayewski, and C.V. Schmitt (editors). 2009. *Maine's Climate Future: An Initial Assessment*. Orono, ME: University of Maine. <http://www.climatechange.umaine.edu/mainesclimatefuture/>
- Juggins, S. 2017. rioja: Analysis of Quaternary Science Data, R package version (0.9-21). (<http://cran.r-project.org/package=rioja>).
- Krammer, K., & H. Lange-Bertalot. 1986–1991. Bacillariophyceae. In *Süßwasserflora von Mitteleuropa*. Vol. 2(1–4). (Eds. H. Ettl, G. Gärtner, J. Gerloff, H. Heynig, and D. Mollenhauer), Gustav Fischer Verlag, Stuttgart/Jena.
- Lakes of Maine. 2019. Lake Stewards of Maine. Retrieved from <https://www.lakesofmaine.org/index.html>
- Leavitt, P. R., and D. L. Findlay. 1994. Comparison of fossil pigments with 20 years of phytoplankton data from eutrophic Lake 227, Experimental Lakes Area, Ontario. *Can. J. Fish. Aquat. Sci.* 51: 2286–2299. doi:10.1139/f94-232
- Legendre, P. and Birks, H. J. B., 2012. From Classical to Canonical Ordination. In: Birks H., Lotter A., Juggins S., Smol J. (eds) *Tracking Environmental Change Using Lake Sediments*. *Developments in Paleoenvironmental Research*, vol 5. Springer, Dordrecht

- Long Lake. 1984. Maine Department of Inland Fisheries and Wildlife [PDF file]. Retrieved from https://www.maine.gov/ifw/docs/lake-survey-maps/cumberland/long_lake.pdf
- Maechler, M., P. Rousseeuw, A. Struyf, M. Hubert, K. Hornik. 2018. cluster: Cluster Analysis Basics and Extensions. R package version 2.0.7-1.
- McCullough, I.M., C.S. Loftin, S.A. Sader. 2013. Landsat Imagery reveals declining clarity in Maine's lakes during 1995-2010. *Freshwater Sci.* 32: 741-752.
- McGowan, S., P.R. Leavitt, R.I. Hall, N.J. Anderson, E. Jeppesen & B.V. Odgaard. 2005. Controls of Algal Abundance and Community Composition During Ecosystem State Change. *Ecology* 86, 8.
- Mitchell, B., & C. Hartford. The Bridgton Town Register, 1905. Printed by The H.E. Mitchell Company.
- Norton, S.A., R.H. Perry, J.E. Saros, G.L. Jacobson, I.J. Fernandez, J. Kopáček, T.A. Wilson & M.D. SanClements. 2011. The controls on phosphorus availability in a Boreal lake ecosystem since deglaciation. *Journal of Paleolimnology* 46 (1): 107-122.
- Oksanen, J., Blanchet, F. G., Kindt, R., Legendre, P., Minchin, R. R., O'Hara, R. B., Simpson, G. L., Solymos, P., Stevens, H. H., and Wagner, H., 2013. vegan: community ecology package. R package version 2.0–10. <http://CRAN.R-project.org/package=vegan>
- Overpeck, J.T., T. Webb III & I.C. Prentice. 1985. Quantitative interpretation of fossil pollen spectra: Dissimilarity coefficients and the method of modern analogs. *Quaternary Research* 23 (1): 87-108. [https://doi.org/10.1016/0033-5894\(85\)90074-2](https://doi.org/10.1016/0033-5894(85)90074-2)
- Patoine, A. & P.R. Leavitt. 2006. Century-long synchrony of fossil algae in a chain of Canadian Prairie Lakes. *Ecology* 87 (7): 1710-21.
- Ramstack, J.M., S.C. Fritz, D.R. Engstrom & S.A. Heiskary. 2003. The Application of a Diatom-based Transfer Function to Evaluate Regional Water-Quality Trends in Minnesota Since 1970. *Journal of Paleolimnology* 29: 79-94. <https://doi.org/10.1023/A:1022869205291>
- Riedinger-Whitmore, M.A., T.J. Whitmore, J.M. Smoak, M. Brenner, A. Moore, J. Curtis & C.L. Shelske. 2005. Cyanobacterial Proliferation is a Recent Response to Eutrophication in Many Florida Lakes: A Paleolimnological Assessment. *Lake and Reservoir Management* 21(4): 423-435. <https://doi.org/10.1080/07438140509354447>
- R version 3.5.0 (2018-04-23) -- "Joy in Playing" Copyright (C) 2018 The R Foundation for Statistical Computing Platform: i386-w64-mingw32/i386 (32-bit)
- Saros, J.E. & N.J. Anderson. 2015. The ecology of the planktonic diatom *Cyclotella* and its implications for global environmental change studies. *Biol Rev.* 90(2): 522-541

- Saros, J.E., T.J. Michel, S.J. Interlandi & A.P. Wolfe. 2005. Resource requirements of *Asterionella formosa* and *Fragilaria crotonensis* in oligotrophic alpine lakes: implications for recent phytoplankton community reorganizations. *Canadian Journal of Fisheries and Aquatic Sciences* 62: 1681-1689. doi: 10.1139/F05-077
- Simpson, G.L. 2007. Analogue Methods in Palaeoecology: Using the analogue Package *Journal of Statistical Software*, 22(2), 1--29
- Simpson, G.L. and Oksanen, J. 2020. analogue: Analogue matching and Modern Analogue Technique transfer function models. (R package version 0.17-4). (<https://cran.r-project.org/package=analogue>).
- Stoddard, J.L., J. Van Sickle, A.T., Herlihy, J. Brahney, S. Paulsen, D.V. Peck, R. Mitchell & A. Pollard. 2016. Continental-Scale Increase in Lake and Stream Phosphorus: Are Oligotrophic Systems Disappearing in the United States? *Environ. Sci. Technol* 50, 7: 3409-3415.
- Strock, K.E., J. Saros, S. McGowan, M. Edlund, D. Engstrom. 2019. Response of boreal lakes to changing wind strength: Coherent physical changes across two large lakes but varying effects on primary producers over the 20th century. *Limnology and Oceanography*: 1-15. doi:10.1002/lno.11181
- Sullivan T.J. 1990. In D.F. Charles, ed. *Acidic Deposition and Aquatic Ecosystems: Regional Case Studies*. Springer-Verlag, NY.(in press).
- Taranu, Z.E., I. Gregory-Eaves, P.R. Leavitt, L. Bunting, T. Buchaca, J. Catalan, I. Domaizon, P. Guilizzoni, A. Lami, S. McGowan, H. Moorhouse, G. Morabito, F.R. Pick, M.A. Stevenson, P.L Thompson & R.D. Vinebrooke. 2015. Acceleration of cyanobacterial dominance in north temperate-subarctic lakes during the Anthropocene. *Ecology Letters* 18(4).
- Tilburg, C.E., L.M. Jordan, A.E. Carlson, S.I. Zeeman, P.O. Yund. 2015. The effects of precipitation, river discharge, land use and coastal circulation on water quality in coastal Maine. *R.Soc.opensci*.2:140429. <http://dx.doi.org/10.1098/rsos.140429>
- Tramoni, F., R.G. Barry & J. Key. 1985. Lake ice cover as a temperature index for monitoring climate perturbations. *Zeitschrift für Gletscherkunde und Glazialgeologie* 21: 43–49.
- Tropea, A.E., A.M. Paterson, W.B. Keller & J.P. Smol. 2011. Diatoms as indicators of long-term nutrient enrichment in metal-contaminated urban lakes from Sudbury, Ontario. *Lake and Reservoir Management* 27(1): 48-60. doi: 10.1080/07438141.2011.555935
- Unity Pond. 1989. Maine Department of Inland Fisheries and Wildlife [PDF file]. Retrieved from https://www.maine.gov/ifw/docs/lake-survey-maps/waldo/unity_pond.pdf

- Vassalboro. 2010. Miner Descent. Retrieved from <https://minerdescent.com/2010/09/10/vassalboro/>
- Ward, E.E. 1974. My first sixty years in Harrison, Maine. Page 7. Denmark, Me., Printed by Cardinal Print. Co., 1974.
- Webber Pond. 1989. Maine Department of Inland Fisheries and Wildlife [PDF file]. Retrieved from https://www.maine.gov/ifw/docs/lake-survey-maps/kennebec/webber_pond.pdf
- Wengrat, S., H. Bennion, P.A. de Lima Ferreira, R.C.L. Figueira & D.C. Bicudo. 2019. Assessing the degree of ecological change and baselines for reservoirs: challenges and implications for management. *J Paleolimnol* 62: 337-357. [https://doi.org/10.1007/s10933-019-00090-4\(0123456789](https://doi.org/10.1007/s10933-019-00090-4(0123456789)
- Wickham, H and J. Bryan. 2019. readxl: Read Excel Files. R package version 1.3.1. <https://CRAN.R-project.org/package=readxl>
- Williams, J.J., M. Beutel, A. Nurse, B. Moore, S.E. Hampton & J.E. Saros. 2016. Phytoplankton responses to nitrogen enrichment in Pacific Northwest, USA Mountain Lakes. *Hydrobiologia* 776: 261-276. doi: 10.1007/s10750-016-2758-y
- Wunsam, S., R. Schmidt & R. Klee. 1995. *Cyclotella*-taxa (Bacillariophyceae) in lakes of the Alpine region and their relationship to environmental variables. *Aquatic Science* 57: 360-386. <https://doi.org/10.1007/BF00878399>

BIOGRAPHY OF THE AUTHOR

Simona Lukasik was born in Buffalo, New York on March 7, 1996. She was raised in Perrysburg, New York and graduated from Mount Mercy Academy in 2014. She attended the State University of New York at Fredonia and graduated in 2018 with a Bachelor's degree in Environmental Science and a dual minor in Psychology and Geographic Information Systems. She moved to Maine and entered the Ecology and Environmental Science graduate program at the University of Maine in the fall of 2018. After receiving her degree, Simona will pursue a job in the field of freshwater ecology. She is a candidate for the Master of Science degree in Ecology and Environmental Science from the University of Maine in August 2020.