Characterizing current and past hydroecological conditions in shallow tundra ponds of the Hudson Bay Lowlands

by

Jerry Edward White

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AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Due to accelerated climatic warming during the past fifty years, interest and concerns have been growing about changes in the ecological integrity of shallow freshwater ponds that dominate the landscape of the Hudson Bay Lowlands (HBL). Climatic warming is altering the hydrological processes that influence the water balances in these ponds, but knowledge remains insufficient to determine the effects these changes have on ecological conditions in the ponds. To address this knowledge gap, this study examines the relations between current hydrological and limnological conditions and recently deposited sedimentary assemblages of diatoms and photosynthetic pigments in 23 shallow ponds in the HBL. The knowledge from the contemporary studies will be used to inform paleolimnological reconstructions using multiple proxies at two ponds to assess how hydroecological conditions have changed during the past several centuries in response to climatic variations.

Water samples were collected three times in 2010 to explore the relations between current hydrological and limnological conditions. The climatic conditions in 2010 provided an excellent opportunity to assess the effects that continued climate warming may exert on hydrolimnological conditions in the study ponds as the warm, dry conditions during the early thaw-season (May- mid-July) followed by extremely wet conditions for the remainder of the thaw-season are likely representative of future climate scenarios. The analysis revealed that the water chemistry in the ponds evolved along one of four different ‘trajectories’ throughout the thaw-season in 2010. These seasonal patterns of limnological conditions closely aligned with similar patterns identified in a study of contemporary hydrological conditions (Light, 2011; Wolfe et al., 2011). The patterns identified in both the hydrological and limnological studies were attributed to differences among ponds in catchment characteristics and hydrological connectivity with adjacent basins.

Surface sediment samples were collected in 2010 to determine if hydrolimnological conditions are reflected in the distribution of recently deposited diatom and pigment communities. It was determined that diatom community composition was most highly influenced by the availability of microhabitat in the ponds which did not appear to be controlled by hydrological conditions. Nitrogen availability was determined to be indirectly influencing diatom community composition as the nitrogen-limited conditions in the ponds favoured the proliferation of N_2-fixing cyanobacterial mats that provided a

large amount of epiphytic habitat in the majority of the study ponds. This resulted in the complete domination of diatom assemblages by one diatom specie that was closely associated with these algal mats in the surface sediments of most ponds. Aphanizophyll, a photosynthetic pigment representative of nitrogen-fixing cyanobacteria, was also in the highest concentration in most of the study ponds as a result of the nitrogen-limiting conditions that allowed these organisms to dominate. Ponds located in the tundra ecozone were also found to have the highest overall pigment concentration which was related to a longer growing season due to the small size of these ponds that resulted in earlier ice-off conditions. The longer growing season of ponds in the tundra ecozone may also be due to high winds that cause a decrease in snow cover, lower surface albedos and an earlier onset of the spring thaw.

The paleolimnological reconstruction of two of the ponds revealed similar shifts in diatom community composition in the stratigraphic record even though patterns of past change in their basin hydrology, as explored though the analysis of the $\delta^{18}O_{PW}$ record archived in the aquatic cellulose contained in the pond sediments, was very different. The water balance of “Left Lake” was found to be highly influenced by increased evaporation associated with recent warming trends as it is a relatively small basin that becomes hydrologically isolated after the melt period. However, “Erin Lake” was not as susceptible to evaporation during the recent warming trend due to its larger catchment and hydrological connections to other ponds. Both of these ponds experienced marked changes in the diatom assemblages. The changes were characterized by a shift from assemblages containing both small, adnate, benthic taxa that prefer mineral grain substrates and epiphytic taxa that are associated with the cyanobacterial mats covering the pond bottoms to assemblages entirely dominated by epiphytic taxa. The shift in diatom community composition occurred ~1820 in Left Lake, but the timing cannot be determined with any degree of confidence in Erin Lake as no diatoms are observed in the sediment record during the period when the change occurred (~1550 to 1850) due to preservation issues. Analysis of fossil pigments indicates that nitrogen-fixing cyanobacteria have been important to the ecology of the ponds over the entire sediment record. However, there is a trend towards lower concentrations of pigments representative of N_2-fixing cyanobacteria in the most recent sediments. The trend in cyanobacterial pigment concentrations coincides with inferred changes in nitrogen availability from the geochemical analysis of the pond sediments by Light (2011). This recent shift in nutrient status may be the result of a number of factors including the increased atmospheric deposition of anthropogenically-derived nitrogen or changes in biogeochemical cycling in the ponds.

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Table of Contents

List of Figures

List of Tables

Chapter 1

General Introduction

1.1 Tundra lakes and ponds and the effects of climate warming

Tundra lakes and ponds dominate many northern landscapes and are important aquatic ecosystems that provide productive habitat and refuge for microorganisms, plants and animals in regions that experience extreme environmental conditions (Smol and Douglas, 2007).These aquatic habitats exist due to the presence of permafrost, which creates an impermeable barrier that limits melting ice and snow and rain from entering the subsurface (Rouse et al., 1997). Instead, water flows over land surfaces or through the shallow active layer, pooling in depressions to form shallow ponds (Woo and Winter, 1993). In areas of thermokarst development, shallow ponds also form as ice-rich permafrost thaws and the ground subsides creating depressions. This localized thawing of permafrost has been attributed to disturbances such as fire as well as a prolonged thaw-season and above average temperatures in response to high-latitude climate warming (Plug et al., 2008).

Climate warming in Canada's North is not only anticipated to affect local temperatures but is also expected to alter regional precipitation patterns (Rouse et al., 1997). Changes in precipitation patterns are expected to be the result of a decreased influence of cold, dry arctic air masses and increased influence of warmer moisture-laden air masses (Pacific, Atlantic, Hudson Bay, southern) (Rouse et al., 1997). As the climate warms, the active layer in permafrost areas will deepen, which could also result in the disappearance of many tundra lakes and ponds as subsurface connections develop and lakes drain (Woo et al., 1992). With continued climate warming, lake ice cover will also be thinner resulting in a longer thaw-season (Rouse et al., 1997). As evapotranspiration typically exceeds precipitation during the thaw-season, early break-up will lead to a more negative water balance.

The combined effects of precipitation change (timing and amounts) and temperature increases will have a substantial effect on water balance, which is critical in determining hydrological and limnological conditions in tundra lakes and ponds (Rouse et al., 1997).Tundra ponds are typically shallow and, due to their high surface area to depth ratio, are highly sensitive to changes in climate patterns as periods of increased evaporation and drawdown can lead to rapid and radical changes in pond conditions (physical, chemical and biological conditions including habitat availability; Smol and Douglas, 2007). Preliminary results from the Old Crow Flats, a thermokarst landscape in continuous permafrost terrain in the Yukon Territory, highlighted that the hydrological processes influencing lake water balances regulate limnological conditions and biological communities (Balasubramaniam, PhD thesis in preparation).

The scientific community has provided a large amount of evidence to substantiate claims that human-driven alterations to climate are greatly affecting the energy balances of arctic and subarctic regions through alterations in hydrological processes (ACIA, 2005). Important climate-driven changes to the hydrological cycle include changes in the amount and duration of snow and ice cover, the proportions of precipitation that fall as rain or snow, evaporation to precipitation ratios and the quality and quantity of water inflows to lakes and rivers due to accelerated glacier retreat and permafrost thaw. Although it is well understood that water balance largely influences limnological conditions, the effect of climate-driven hydrological change on the limnology in shallow tundra lakes and ponds has not been explored to a large degree. It is important to explore these linkages because changes in limnological conditions (longer growing season, warmer water temperatures) can affect the chemical, mineral and nutrient status of lakes, which can have a large effect on biological communities (Rouse et al., 1997).

1.2 The Hudson Bay Lowlands and climate change

The Hudson Bay Lowlands (HBL) is a subarctic landscape located along the south-western shores of Hudson Bay dominated by shallow tundra ponds. As developed more fully in Chapter 2, this thermokarst landscape is an ecologically significant wetland that provides extensive shoreline habitat for nesting waterfowl, shorebirds and other animals, and it has received increasing attention because it has been identified as an area that has undergone some of the greatest increases in summer temperatures during the last 50 years (Kaufman et al., 2009).

A number of climate change models have been developed to simulate both current and future climate regimes in the HBL. Gagnon and Gough (2005) determined that climate models with the greatest accuracy incorporate both sea-ice cover and permafrost into the climate change scenario due to the substantial effects these two variables have on the climate in the region. These models predict that the greatest greenhouse warming will occur from October to April over Hudson Bay. This warming will result in decreased thickness and duration of sea-ice on Hudson Bay allowing for an increased transfer of heat to exposed waters as a result of an extended thaw-season. As water temperatures increase and the thaw-season lengthens, Hudson Bay will have a larger influence on continental climate leading to less extreme winter temperatures and an earlier spring snowmelt. The extended period during which the ground surface is exposed will allow it to absorb a greater amount of radiant energy, further increasing spring temperatures. This ice-albedo positive feedback mechanism is largely responsible for the amplification of the climate warming signal observed at higher latitudes (Perovich et al., 2007).

Climate models of the HBL, as reviewed by Gagnon and Gough (2005), also predict that there will be an increase in summer temperatures over land surfaces due to a reduction in soil moisture conditions. Climate models indicate there will be an increase in the amount of precipitation in the region with climate warming. However, higher evaporation rates in the summer will result in drier soil conditions. The anticipated decrease in soil moisture will result in a larger percentage of solar radiation observed as visible heat instead of latent heat of evaporation, which will further increase continental surface temperatures.

The predicted increase in ground warming is expected to lead to the retreat of permafrost northward with an increase in the depth of the active layer in areas where permafrost persists (Gagnon and Gough, 2005). Since permafrost acts as an impermeable barrier to water flow, overland water flow will decrease as water infiltrates through a deeper active layer. Also, lake drainage may occur as taliks under tundra ponds penetrate through the permafrost (Rouse et al., 1997). In areas of the HBL where lake drainage due to thawing permafrost is not occurring, desiccation due to increased evaporation may be of a greater consequence as large changes in limnological conditions have been observed in other northern landscapes in response to lake level evaporative drawdown (Smol and Douglas, 2007).

Macrae et al. (2004) found that during the thaw-season, large precipitation events resulted in rapid declines in pH and increases in dissolved inorganic carbon concentrations in the tundra ponds of the HBL due to inflows of dissolved substances from the catchments. It was also discovered that during wet periods, depending on sediment composition and hydrological connectivity with their catchment, these ponds became supersaturated with CO_2. Increases in precipitation as predicted by climate change models, will result in more frequent and possibly prolonged shifts in water chemistry that would not only affect the biota residing in these ponds but also result in evasion of CO_2 back into the atmosphere, further increasing atmospheric concentrations of greenhouse gases.

Macrae et al. (2010) conducted a study to better understand the effects that recent warming trends have had on climatic conditions in the Churchill area of the HBL. They also employed regional climate models to simulate the effects that continued warming will have in the future. In the study, air temperatures were found to have increased in all seasons, with the greatest increase observed in winter (Figure 1.1a). This trend was expected to continue throughout the 21st century. Annual precipitation was also found to be increasing, but snowfall has actually decreased (Figure 1.1b). The climate models anticipated that these trends in precipitation patterns will continue until the end of the century. The study found that early and late thaw-season precipitation was also switching from snowfall to rainfall as changes to break-up and freeze-up dates continued to increase the length of the thaw-season (Figure 1.1c). Lengthened thaw-seasons and increased temperatures have resulted in increased open water evaporation, which is expected to continue throughout the 21st century (Figure 1.1d). The increase in late thaw-season precipitation over the last ~70 years has off-set the increases in evaporation and overall the annual pond water balance is becoming more positive. However, since precipitation has not been observed to have increased during the early thaw-season, the ponds experience drier conditions in June and July, which has resulted in a greater number of ponds becoming ephemeral. These temporal changes in climate patterns are hypothesized to result in shifts in limnological conditions throughout the thaw-season that will likely be manifested through alterations in pond ecology.

Increases in temperature have already resulted in many changes in hydrological conditions in the ponds of the Hudson Bay Lowlands (HBL) (Wolfe et al., 2011). A reconstruction of the paleohydrology of four ponds (unofficial names: ‘Left Lake’, ‘Larch Lake’, ‘Puddle Pond’, ‘Erin Lake’) using the cellulose oxygen isotope composition of autochthonous organic matter contained in pond sediments indicated that hydrological conditions were similar and relatively stable until the early part of the 20th century in the HBL. After which, hydrological conditions of the four ponds diverge due to differences in hydrological connectivity among the ponds (Figure 1.2). Two of the ponds, Left Lake and Larch Lake shared very similar responses to climate warming beginning in ~1920 in which the cellulose-inferred pond water records trended towards more enriched values due to increased evaporation on the water balance in the ponds. In ~1960 the cellulose-inferred $\delta^{18}O_{PW}$ records of Left Lake and Larch Lake diverge. The $\delta^{18}O_{PW}$ record of Left Lake continued towards more enriched values as evaporation continued to play a greater role on pond water balance, while Larch Lake began to trend towards more depleted $\delta^{18}O_{PW}$ values indicating that the pond had established a hydrological connection with an adjacent basin that allowed for sufficient inflows to offset the effects of evaporation. Comparison of the contemporary water balances of these ponds supported this interpretation as Left Lake was determined to be hydrologically isolated throughout the thaw-season, which resulted in declining water levels during drier periods, while Larch Lake received subsurface flows that maintained water levels under similar conditions. The Erin Lake cellulose-inferred pond water record was much different that these two ponds and had remained relatively constant over the past ~90 years in response to climate warming. This is largely due to the fact that Erin Lake is hydrologically connected to a much larger adjacent pond through an outlet channel. This connection, along with a large catchment act to offset the effects of increase evaporation due to climate warming and maintain pond-water levels in Erin Lake. Puddle Pond $\delta^{18}O_{PW}$ values begin to decline ~1920 and trended towards values very similar to the isotopic composition of precipitation. Analysis of contemporary conditions in Puddle Pond indicated that the pond develops an outlet channel as the active layer deepens throughout the thaw-season and water isotope values reflected those of rainfall due to the short residence time of water within the pond. Although climate models indicate that the water balance of the ponds is becoming more positive due to increases in annual precipitation, paleohydrologic reconstructions of four ponds indicate a more complex and divergent hydrological response due to the differences in hydrological connectivity with the surrounding landscape.

Current climate models indicate that the changes due to climate warming in the HBL will be complicated and multifaceted due to the complex roles that both sea-ice and permafrost play in regional climatic conditions (Gagnon and Gough, 2005). A number of studies have assessed the effects climate warming will have on hydrological processes (Gagnon and Gough, 2005; Light, 2011; Wolfe et al., 2011); however, very few studies address how physical, chemical and biological conditions in lakes and ponds will respond to these alterations in hydrological processes. As hydrological processes are probably the main drivers of limnological conditions in remote northern landscapes (Rouse et al., 1997), this further highlights the importance of using contemporary limnological studies (water chemistry, surficial sediment samples) to better understand the hydroecology of these small tundra ponds. The knowledge gained from the study of current hydroecological conditions in the HBL can then be used to inform a paleolimnological reconstruction of the changing hydroecological conditions in these shallow ponds from pond sediment cores. Policy makers can use the long-term records of past hydroecological variations in these shallow ponds to help better predict future limnological and biological responses of the ponds to continued alterations in hydrological processes as a consequence of climate warming. This information can then be used to guide future management decisions.

1.3 Relationships between hydrological and limnological conditions and biological community composition

In previous studies of circumpolar freshwater ecosystems, it has been found that water chemistry varies predictably with changes in hydrological conditions. For example, in the HBL, Macrae et al. (2004) established a relationship between biogeochemical cycling (pH and CO_2 concentrations) and the hydrological connectivity of ponds with their drainage basin in the HBL. This relationship was also influenced by the type of sediments observed in the ponds (mineral or organic). It was determined that pond pH increased and CO_2 levels declined during the post-melt period when the ponds became hydrologically disconnected from their basins. During this period, ponds with organic sediments sequestered atmospheric CO_2, whereas ponds with mineral sediments evaded CO_2 to the atmosphere but at a lower rate than during periods of hydrological connection. Following large precipitation events, pond pH declined and CO_2 evaded to the atmosphere at increased rates in ponds with both sediment types as the ponds became reconnected with their terrestrial drainage basins.

Current hydroecological conditions in lakes can be effectively monitored through the analysis of algal community composition, because algae are highly sensitive and responsive to changing environmental conditions (Dixit et al., 1992). One type of algae contained within the biogenic fraction of lake sediments are diatoms. These microscopic, unicellular, eukaryotic organisms are preserved in the paleolimnological record due to their siliceous cell wall that resists dissolution. It is this siliceous cell wall, the diatom frustule, which also allows for the characterization of diatoms into various taxonomic groups based on observable differences in morphology (Battarbee et al., 2001). Diatoms are useful biological indicators because the distribution of species within an aquatic system is dependent upon the environmental conditions. This allows limnologists to infer hydrolimnological conditions based on the presence of indicator species in surficial lake sediments. This method of developing relationships between limnological conditions and diatom indicator taxa has proven effective in estimating physical parameters such as light, temperature and turbulence or chemical parameters such as pH, dissolved organic carbon, nutrients and salinity (Battarbee et al., 2001).

Another useful biological indicator of limnological conditions contained within lake sediments is photosynthetic pigments. Photosynthetic pigments are produced by all photosynthetic organisms and are deposited within lake sediments following plant senescence and death (Wetzel, 2001). The type of pigment can be used for taxonomic identification and determining algal community composition (Reuss et al., 2010). Photosynthetic pigment concentrations also provide a comprehensive marker for whole lake monitoring and are often used to infer lake primary production (Cuddington and Leavitt, 1999).

The technique of linking biological indicator taxa to hydrological conditions was attempted previously in the HBL. Macrae (1998) explored relations between the diatom community composition contained within pond sediments and carbon accumulation rates in a set of ponds in the HBL. Macrae (1998) attempted to determine if diatom autoecology correlated to the factors that influenced carbon accumulation rates in order to develop transfer functions to assess whether these rates have fluctuated over the period of time captured within pond sediment cores. Although floristic changes in the pond sediment record suggested some environmental change had occurred, the environmental variables included in the study only explained a small fraction of the variation in diatom community composition within the study ponds, which made it difficult to draw conclusions concerning carbon accumulation rates from the data set.

Previous studies have examined the relationships between hydrological inputs and biogeochemical cycling in the ponds in the HBL, as well as the distribution of diatoms in relation to carbon accumulation rates (Macrae, 1998; Macrae et al., 2004). This thesis builds on the knowledge gained from these previous studies to develop a more comprehensive understanding of the relationships between algal community composition and hydrolimnological conditions within the ponds of the HBL.

1.4 Paleolimnology

Historical records are often too short and incomplete to develop a clear understanding of past hydroecological responses to environmental change. These records can be greatly extended by using relationships between current hydrolimnological conditions and biological and geochemical proxies in recently deposited sediments to interpret past hydroecological responses to environmental changes that are archived in lake sediment records. These long-term records can then help predict ecological responses to ongoing and future warming. Paleolimnology is a branch of science that attempts to reconstruct past environmental conditions from the physical, chemical and biological information archived within lake sediments cores (Frey, 1968). The data archived in lake sediments are like the "flight recorders" on an aircraft and can be used to provide a long-term record of environmental change that can be used to guide future management practices (Smol, 2008). Paleolimnology has been found highly effective in the study of other northern landscapes to link biological and geochemical proxies to more accurately interpret information contained in the lake sediment record (Hay et al., 1997, 2000; Hall et al., 2004; Sokal et al., 2008). This thesis uses loss-on-ignition (LOI), radiometric dating and biological indicators (diatoms, pigments) to interpret the paleoenvironmental conditions from pond sediment records. These proxies are then combined with the results from geochemical chemical analysis conducted by Light (2011) to assess past hydroecological conditions using a multi-proxy paleolimnological approach.

1.4.1 Loss-on-ignition

Sequential loss-on-ignition (LOI) is used to determine sedimentary moisture, organic matter, inorganic matter and carbonate content, which can be used to assess changes in lake productivity and mineral matter supply (Dean et al., 1974). LOI data also aid in cross-correlation of sediment core chronologies from one dated sediment core to other undated cores. LOI analysis can also be conducted on surface sediment samples to allow for the characterization of the lake bottom substrates (mineral, organic), which in previous studies in the HBL were determined to have an effect on the biogeochemical cycling within the ponds (Macrae et al., 2004).

1.4.2 Sediment core chronology

In order to link biological proxies to past environmental events, an accurate sediment chronology is of crucial importance. One of the most reliable methods for dating recent sediments is through the decay of a naturally occurring radioisotope of lead (Appleby, 2001). ^{210}Pb has a half-life of 22.3 years and can be used to accurately date sediments from about 0-150 years old (Appleby, 2001). ^{210}Pb occurs naturally through the decay of its parent isotope ^{226}Ra, both of which are radionuclides derived through the natural decay of ^{238}U (Appleby, 2001). The total ^{210}Pb inventory of lake sediments consists of two components: supported and unsupported. The supported ^{210}Pb component is derived from the decay of ^{226}Ra in the lake sediments (Appleby, 2001). ^{226}Ra and ^{210}Pb would be in equilibrium if it were not for the deposition of ^{210}Pb from the atmosphere and watershed representing the unsupported component observed in the lake sediments (Appleby and Oldfield, 1984). The unsupported ^{210}Pb component in lake sediments is derived from ^{222}Rn, a gaseous intermediate isotope in the ^{238}U decay series, which diffuses from the soils into the atmosphere. In the atmosphere, ^{222}Rn decays to ^{210}Pb through a number of short-lived radionuclides, and is deposited onto the lake surface by precipitation or dry deposition (Appleby, 2001). It is the concentration of unsupported ^{210}Pb contained within the sediment intervals that can be used to determine the age of the sediments based on decay laws (Appleby and Oldfield, 1984).

Two models commonly used to develop chronologies for the lake sediments are the “Linear” and the “Constant Rate of Supply” (CRS) models (Appleby, 2001). Both models assume that the atmospheric flux of ^{210}Pb has remained constant over time, which has been demonstrated to be reliable on a regional scale (Appleby and Oldfield, 1984). The Linear model is used in situations when a constant rate of sedimentation can be assumed (Appleby and Oldfield, 1984). The CRS model is used in situations where a change in sedimentation rates is likely to have occurred. There are instances when neither of these methods is reliable due to mixing of sediments or alterations in sediment focusing patterns, which requires additional ad hoc models to be applied to develop accurate sediment chronologies (Appleby, 2001).

Validation of a ^{210}Pb-based chronology can be achieved using independent dating markers. In areas receiving fallout from nuclear weapons testing and the Chernobyl reactor accident, the most widely used marker is the artificial radionuclide ^{137}Cs (Appleby, 2001). These events are recorded in ^{137}Cs activity in the sediment profile representing the dates corresponding with peak above-ground nuclear arms testing in 1963 and the Chernobyl nuclear reactor accident in 1986 (Appleby, 2001).

1.4.3 Biological indicators

Diatoms are particularly informative bioindicators of limnological conditions in northern lakes as they typically form a large portion of the algal community (Lim et al., 2007). It is because of their high abundance and resistance to dissolution that diatoms are the mainstay of many paleolimnological investigations (Wetzel, 2001; Lim et al., 2007). Shifts in diatom community composition are often used to interpret effects of climatic changes because of their sensitivity to physical, chemical and biological conditions and habitat availability (Sokal et al., 2008). Paleolimnological studies often rely on robust inference models developed from quantitative relationships between diatom community composition and limnological conditions developed from a large spatial survey of lakes within a geographical area (Rühland and Smol, 2002; Lim et al., 2007). In the Mackenzie Delta (Hay et al., 1997, 2000), the Peace-Athabasca Delta (Hall et al., 2004) and the Slave River Delta (Sokal et al., 2008), a different approach was utilized. In these studies, relationships were developed between diatom indicator taxa and lake categories that differed in limnological conditions due to their degree of river connection. These indicator taxa were then used to construct a long-term record of the frequency and extent of river flooding by their presence or absence in sediment cores collected from lakes within the deltas.

Fossil pigments contained within the organic constituents in lake sediments are less well studied in high-latitude lakes than other biological markers such as diatoms and chironomids (Reuss et al., 2010). However, pigments show promise in interpreting the ecological responses in lakes to changes in climate and environmental conditions as the factors influencing the phototrophic community become better understood (Reuss et al., 2010). Because the phototrophic community within lakes have been shown to respond to changes in the concentrations of nutrients and dissolved organic carbon (DOC) and climate-related factors like temperature and precipitation, fossil pigments are proving to be a valuable tool that can be used to interpret changes in the lake sediment record from high-latitude lakes in response to climate warming (Reuss et al., 2010). In previous studies, fossil pigments have been used to interpret the stratigraphic record in a sediment core by using relationships developed between sedimentary pigment concentrations and community composition and contemporary limnological conditions in a set of survey lakes (Reuss et al., 2010). This method proved successful in the interpretation of the pigment stratigraphic sequence from sediment cores from northern Sweden (Reuss et al., 2010) and will be incorporated into this study to interpret the relationships between contemporary and fossil pigments and limnological conditions in the shallow tundra ponds of the Hudson Bay Lowlands.

1.5 Research Objectives

The objective of this study is to incorporate contemporary and paleolimnological methods to determine current and past hydroecological conditions in the shallow tundra ponds that dominate the

HBL landscape. The relative importance of various hydrological processes on the water balances of the study ponds was previously assessed by Light (2011) through the analysis of water stable isotope tracers. This study builds on that knowledge by examining the modern limnology of the study ponds and its link to hydrology through the analysis of pond water chemistry during the thaw-season in 2010. A multi-proxy approach was also utilized to assess past hydroecological responses to climate warming in which biological proxy indicators in pond sediments were integrated with the geochemical analysis conducted by Light (2011). The data derived from the study were used to address the following research questions:

1. What are the relationships between the hydrological processes that control the pond water balances and the limnological conditions that are observed in shallow tundra ponds of the HBL that lie within the Churchill Wildlife Management Area?
2. How are the hydrolimnological conditions reflected in recently deposited sedimentary assemblages of diatoms and pigments?
3. How have hydroecological conditions changed over the period represented within the sediment record (generally spanning the past few centuries). In particular, how have conditions changed over the past 50 years in response to climate warming? Can these changes be used to anticipate the future hydroecological conditions in these ponds under current climate change scenarios?

1.6 Thesis Overview

The main goal of this research is to assess the effects that recent shifts in climate patterns have had on the hydrological and ecological conditions in the shallow tundra ponds in the HBL. In order to accomplish this goal, this thesis is divided into seven chapters. The second chapter provides an overview of the study area and background information regarding the processes that influence the hydrology of the study ponds. The third chapter provides an in-depth description of the methods used to conduct both the contemporary and paleolimnological analysis of the study ponds. The next three ‘data chapters’ present results that address specific research objectives. Chapter 4 explores the contemporary limnology in the ponds and the relations between hydrological and limnological conditions. Chapter 5 assesses the relations between hydrolimnological conditions of the study ponds and diatom and the photosynthetic pigment community composition in recently-deposited surface sediments. Chapter 6 presents paleolimnological reconstructions of environmental conditions during the past several centuries at two ponds in the HBL. Specifically, the relations developed in Chapter 5 are used to interpret past hydroecological responses to climatic changes from analysis of fossil diatoms and pigments within the pond sediment records. This chapter also incorporates results from the geochemical analyses of the same pond sediments to better inform the reconstructions (Light, 2011). Chapter 7 uses the important findings from each of the data Chapters (4,5,6) to synthesize an understanding of current and past hydroecological conditions in the ponds. Chapter 7 also addresses the research objectives of this thesis and makes recommendations for future investigations.

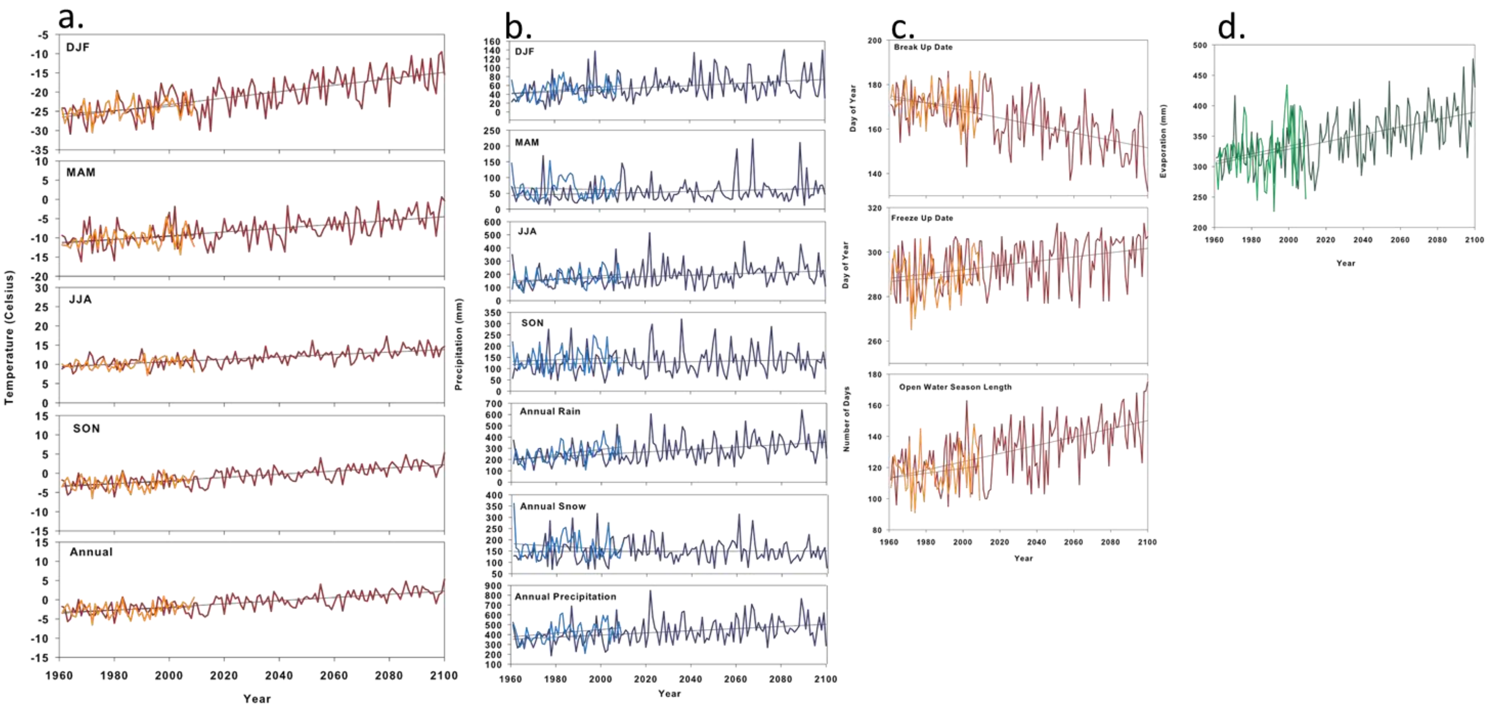

Figure 1-1 Climate data (1943-2009) and climate projections (2010-2100) for Churchill MB. (Macrae et al., 2010). Panel a) displays measured (orange line) and modeled (red line) for seasonal and annual air temperatures. Panel b) displays measured (light blue line) and modeled (dark blue line) for seasonal and annual total precipitation and total annual rain and snowfall. Panel c) illustrates the measured (orange line) and the modeled (red line) changes to freeze-up and break-up dates and length of the thaw-season. Panel d) displays the measured (light green line) and modeled (dark green line) changes to pond water evaporation.

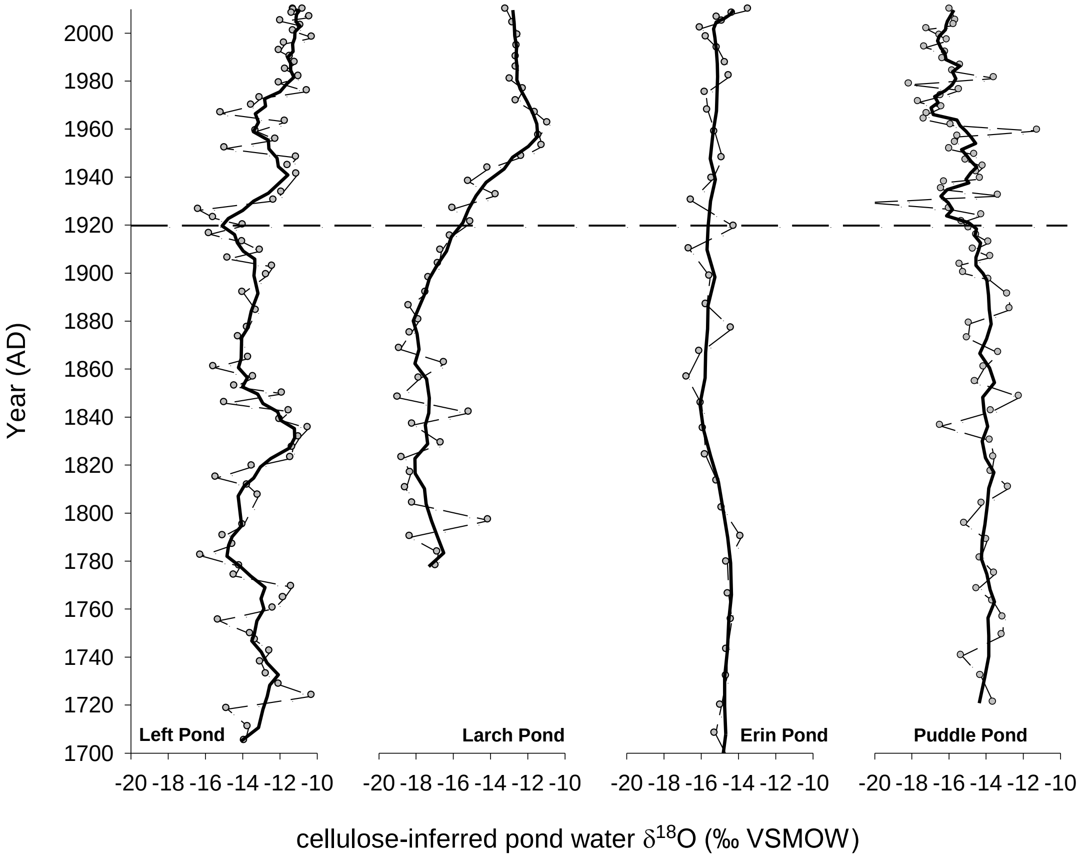

Figure 1-2 Cellulose-inferred pond water δ_{18}O record from sediment cores collected from Left Lake, Larch Lake, Erin Lake and Puddle Pond located in the Churchill Wildlife Management Area (Wolfe et al., 2011).

Chapter 2

The Hudson Bay Lowlands

2.1 The Hudson Bay Lowlands

The Hudson Bay Lowlands (HBL; Figure 2.1) are defined as the region extending inland from the southwestern shores of Hudson Bay and James Bay with an elevation of less than 200 m above sea level (Rouse, 1991). It is the world's second largest contiguous wetland, which encompasses an area of approximately 475,000 km^2 between the latitudes of approximately 51^o and 65^o north (Rouse, 1991, Griffis et al., 2000). Climate normal data (1971-2000; Environment Canada, 2010) indicate the mean annual temperature is -6.9 ^{o}C. The warmest average monthly temperature occurs in July (12.0^{o}C) and the coldest in January (-26.7 ^{o}C). Mean annual precipitation is 431.6 mm, with rainfall comprising 264.4 mm of this value (61.3%).

Gagnon and Gough (2005) describe the region as unique because it is the southernmost region in North America with permafrost (Figure 2.2). This is largely the consequence of the high concentration of sea ice present in the southern and western areas of Hudson Bay from late October until August, which acts to negate the moderating effects that a large body of water like Hudson Bay would have on climate. The region also contains the boundaries for both continuous and discontinuous permafrost (Rouse, 1991). The presence of permafrost in the HBL acts to impede the penetration of surface water, which results in the large number of bogs, fens and ponds that cover the landscape. Due to their shallow depth (typically <1.5 m), the ponds freeze to the bottom in the winter and are well-mixed and isothermal throughout the thaw-season (Stewart and Rouse, 1976). The Arctic treeline boundary also occurs within the HBL, where the terrestrial vegetation shifts from boreal forest in the south to arctic tundra in the north (Duguay and Lafleur, 2003).

2.2 Study Site Description

2.2.1 Location

Twenty-seven ponds were selected for limnological analyses in this study, and they are located in the Churchill Wildlife Management Area (CWMA) in the northwest region of the Hudson Bay Lowlands (Figure 2.3; Table 2.1). These ponds were selected because they represented a wide range of hydroecological conditions, based on information provided by Macrae (1998), and due to accessibility provided by roads near the town of Churchill and the Churchill Northern Studies Centre (CNSC). These ponds were used to explore relationships between the current hydrological and limnological conditions based on analyses of water isotope composition (Light, 2011), water chemistry and biological indicators contained in surficial sediments.

2.2.2 Geology

Dredge and Nixon (1992) describe the geology underlying the 27 study ponds as silty, calcareous till derived from Paleozoic limestone deposited by the Hudson ice sheet during the Wisconsin glaciations (Figure 2.4). Marine silts and sands were deposited above this layer in most of the study area while it was inundated by the Tyrell Sea, following the retreat of the Hudson ice-sheet approximately 7 200 years ago. With the immense mass of the ice sheets no longer present, the region has experienced very rapid glacio-isostatic uplift which has resulted in the disappearance of the Tyrell Sea and the current water levels observed in Hudson Bay. Two of the ponds (West Twin Lake, East Twin Lake), however, are kettle lakes located in an area of glaciofluvial deposition in which the geomorphology consists of kames, eskers and moraines. Thick organic peats (up to ~2 m thick) have accumulated over the past centuries to millennia due to climatic conditions in the HBL. Summer conditions are adequately warm and wet to generate substantial organic accumulations while overall climatic conditions are sufficiently cold to promote the formation of permafrost and prevent the rapid decay of peat deposits which has resulted in the extensive peatlands that now cover the area (Figure 2.5). Coastal fens cover between 55-75% of the land area in regions close to the shores of Hudson Bay. Areas further inland are intermixed with fens and bogs with these features covering between 55-75% of the surface area (Tarnocai et al., 2000). Continuous permafrost underlies most of the study area with the active layer typically reaching a depth of 0.5-1.5 m (Duguay and Lafleur, 2003; Figure 2.2).

2.2.3 Climate

Climate in the study area during the 2010 field season displayed similar trends as described by Macrae et al. (2010) when compared to climate normal data (1971-2000; Environment Canada, 2011). Temperatures were higher than the climate normal with the greatest differences observed during the winter (Figure 2.6a). Precipitation was lower during most times of the year in 2010 compared to climate normal data except in late thaw-season (Figure 2.6b). A marked increase in precipitation occurred in August due mainly to a large rain event on 24-August (~105 mm). Humidity was higher during all periods of the year compared to climate normal data except in May and June (Figure 2.6c). The drier conditions experienced in the early thaw-season combined with decreased precipitation and higher temperatures resulted in the complete desiccation of two ponds by July 2010 (Figure 2.7). Water isotope data also showed that in late July, about a quarter of the study ponds had values exceeding δ_{SSL} , which indicated that the ponds had entered a negative water balance (evaporation > inflow) (Light, 2011; Figure 2.8).

2.2.4 Vegetation zones

The 27 study ponds span the range of ecological zones (tundra, forest-tundra and open forest) observed in the region (Duguay and Lafleur, 2003; Figure 2.6; Table 2.1). The open-forest ecozone is characterized by a mixed stand of black spruce (*Picea mariana*) and tamarack (*Larix laricina*) (Lafleur et al., 1997; Figure 2.6a). The forest floor topography consists of hummocks and depressions with groundcover provided by lichens in drier areas and mosses in wetter areas. Birch (*Betula sp.*), willow (*Salix sp.*) and Labrador tea (*Ledum sp.*) are sparsely distributed throughout the ecozone (Lafleur et al., 1997). Within the forest-tundra transition zone, sparse tree cover consists of white spruce (*Picea glauca),* black spruce (*Picea mariana*) and tamarack (*Larix laricina*) with the density of tree cover decreasing northward (Brook, 2001; Figure 2.6b). In drier areas, ground cover consists of lichens (*Cladina stellaris, Cladina mitis, Cladina rangiferina*) and shrubs (*Betula glandulosa*, *Dryas integrifolia*) (Brook, 2001). In wetter fen areas, sedges (*Carex sp. Eriphorum sp.*) provide the dominant ground cover (Brook, 2001). The tundra ecozone is characterized by low-lying shrubs (Salix sp.), graminoids (*Arctagrostis latifolia*, *Arctophila fulva*, *Poa arctica*, *Carex sp*), lichens and bryophytes (*Selaginella selaginoides*) (Brook and Kenkel, 2002, Light, 2011; Figure 2.6c).

2.2.5 Pond Hydrology

Light (2011) used water isotope tracers to examine the roles of the main hydrological processes (rainfall, snowmelt, evaporation, subsurface and overland flows) that influence the water balances of the ponds in the study area during 2009-2010. The relationship between vegetation and the water balances of the ponds was also explored due to relationships that were identified in another study done in Old Crow Flats, Yukon Territories between ecozone and the main hydrological processes that controlled pond water balance (Turner et al., 2010).

Analysis of the isotopic composition of input water values (δ_I) indicated that the main hydrological input to most of the ponds was rainfall (Light, 2011). At the end of the previous thaw-season, pond water balances were recharged by late summer and early fall rain events. This water froze over winter under bankfull conditions. Evaporation and rainfall were the main influence on the water balances in the ponds during the thaw-season (Macrae, 1998; Light, 2011). Although the input to ponds was dominated by rainfall, isotope data indicated that snowmelt during the spring freshet was also an important source of water to some ponds.

Analysis of water-level data in both study years and frequent monitoring of water isotope composition in three of the study ponds in 2010 showed that the ponds' water levels were high in the spring and began to draw down due to evaporation in late spring/early summer. Late summer rainfall resulted in higher water levels by the end of the thaw-season, which indicated a systematic wet-dry-wet pattern in pond water evolution in both 2009 and 2010 study years. However, this trend was more extreme in 2010 due to an earlier onset of the thaw-season, high summer temperatures and low early summer relative humidity. Based on the predicted changes in climate patterns (Macrae et al., 2010), this trend will become even more pronounced in future years.

Light's (2011) analysis of water isotope tracers in the study ponds identified that variability in evaporation to inflow (E/I) ratios was attributed to surface area, depth, volume, catchment size and connectivity between basins. Smaller and shallower ponds had higher E/I ratios than larger ponds which were also typically deeper. Ponds that had small catchments and that were hydrologically isolated had higher E/I ratios. Ponds that had larger catchments or that were hydrologically connected to other basins were least affected by evaporation and typically had E/I ratios <1. One pond was hydrologically connected to another pond and drained into the adjacent basin due to differences in elevation. This resulted in pond water isotopic composition that was very similar to rain water due to the short residence time of the water.

The effects of terrestrial vegetation on pond water balance could not be adequately addressed in the study due the limited number of ponds included from the open-forest ecozone. The ponds in the open-forest ecozone were also typically larger than the ponds in the other two ecozones, which did not allow for an appropriate comparison of the hydrological processes influencing pond water balances.

2.2.6 Selection of ponds for paleolimnological study

Seven ponds (“Erin Lake”, “Godwit Lake”, “Golf Lake”, “Larch Lake”, “Left Lake”, “Puddle Pond”, “Strange Lake”) of the 27 ponds were selected for sediment core collection and paleolimnological reconstruction by our research group, and two of them are included in this thesis (Figure 2.10). The seven ponds range in size from 181.0 to 13525.9 m^2 in area, and in depth from ~0.2 to 0.63 m (Table 2.2; Light, 2011). These seven ponds were selected to span the range of hydroecological conditions, based on studies conducted by Macrae et al. (2004). The selection of two ponds for paleolimnological reconstruction in thesis was based on ecozone location, water-level fluctuation data (Figure 2.11) and isotope hydrology data (Light, 2011; Figure 2.12). Left Lake and Erin Lake were selected for analysis of diatom assemblage and fossil pigment concentrations as these ponds represent different hydroecological conditions. Left Lake was located in the tundra ecozone (Figure 2.6c). Left Lake experienced a large degree of evaporative isotopic enrichment during the thaw-season in 2009 (Figure 2.12) and water levels appeared to be highly influenced by precipitation events (Figure 2.11). Erin Lake was selected due to its location south of the treeline in the forest-tundra transition zone (Figure 2.6b). Erin Lake experienced a moderate amount of evaporative isotopic enrichment during the thaw-season (Figure 2.12) and water levels appeared not to be substantially influenced by precipitation events (Figure 2.13).

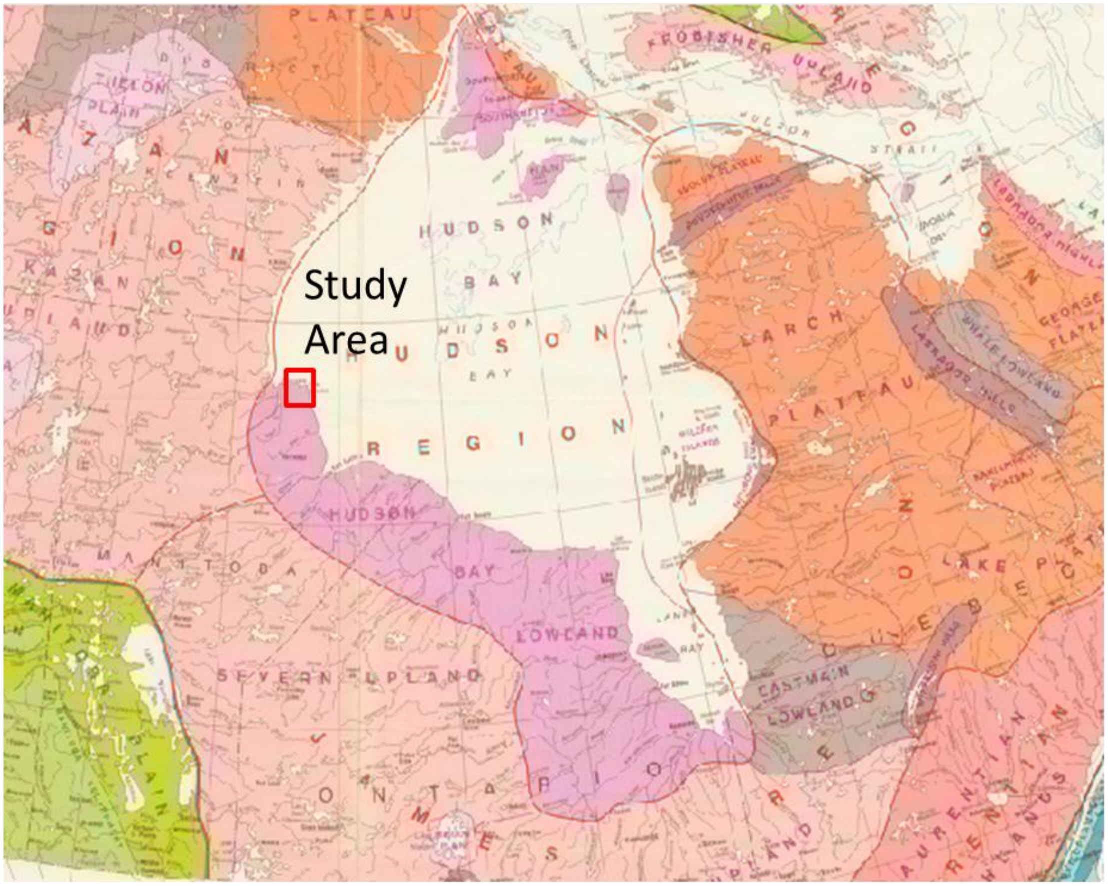

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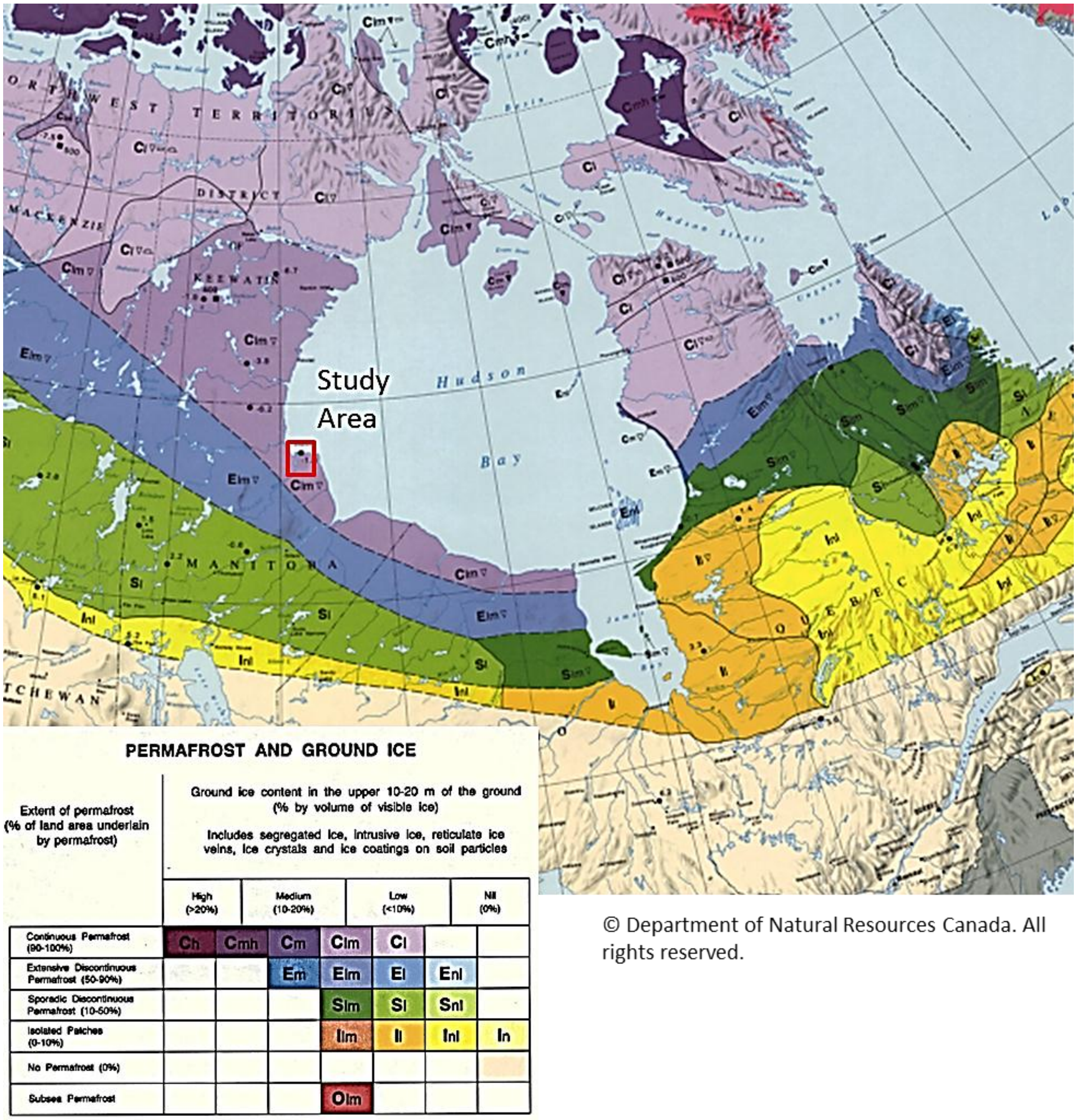
Figure 2-1 Map depicting the location and the land area contained within the Hudson Bay Lowlands. The study ponds are located in northwestern region of the HBL (Bostock, 1970).



Extent of permafrost (% of land area underlain by permafrost)	Ground ice content in the upper 10-20 m of the ground (% by volume of visible ice) Includes segregated ice, intrusive ice, reticulate ice veins, ice crystals and ice coatings on soil particles						
	High (>20%)		Medium (10-20%)		Low (<10%)		Nil (0%)
Continuous Permafrost (90-100%)	Ch	Cmh	Cm	Clm	Cl		
Extensive Discontinuous Permafrost (50-90%)			Em	Elm	El	Enl	
Sporadic Discontinuous Permafrost (10-50%)				Slm	Sl	Snl	
Isolated Patches (0-10%)				Ilm	Il	Inl	In
No Permafrost (0%)							▨
Subsea Permafrost				Olm			

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Figure 2-2 Map illustrating the permafrost boundaries for central Canada. The study ponds are located in the continuous permafrost zone with low to medium ground ice content (NRC, 1995).

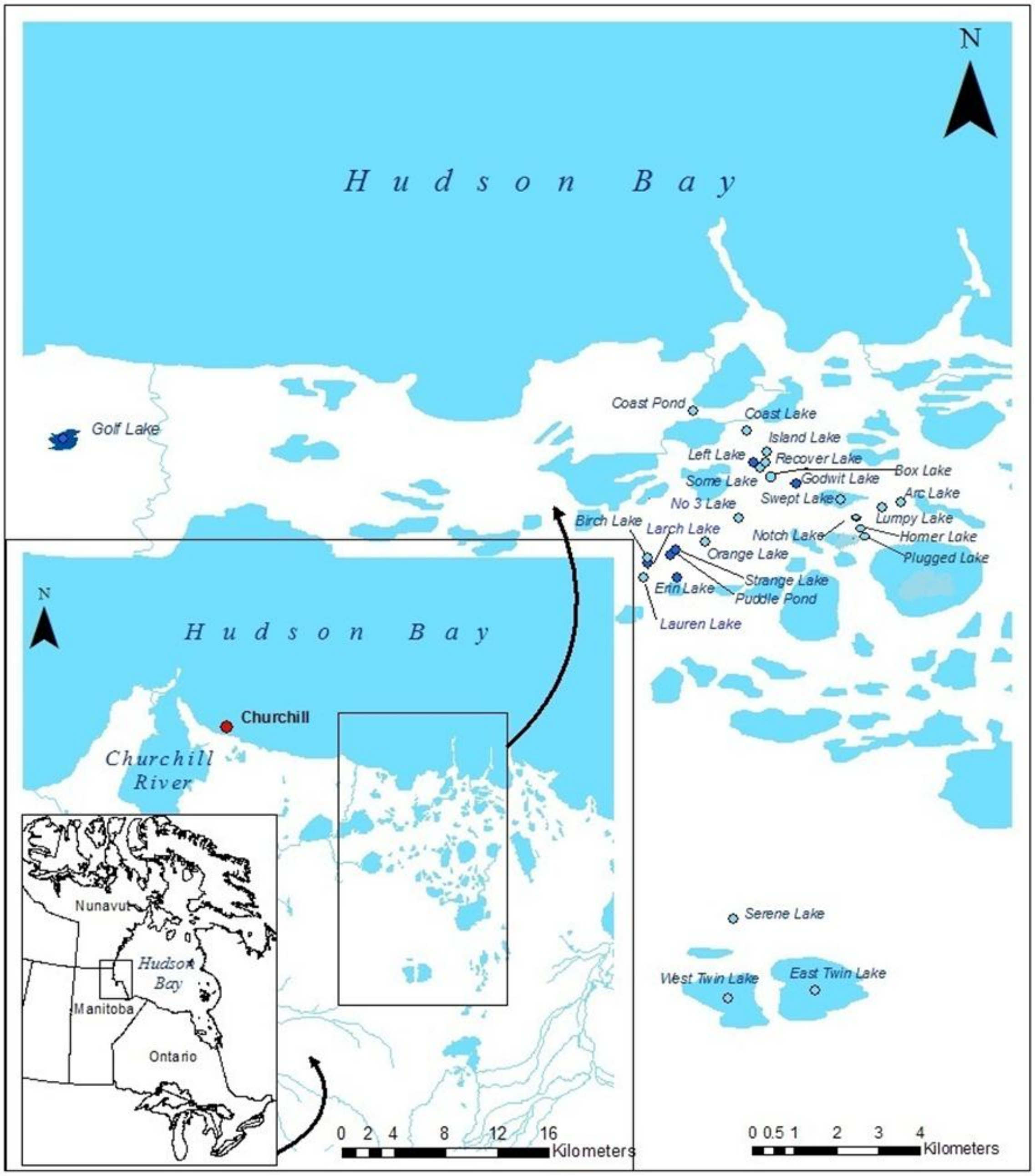


Figure 2-3 Map illustrating the locations of the 27 ponds selected for surface sediment, water chemistry and isotopic analysis in the Churchill Wildlife Management Area. Ponds marked with dark blue circles were selected for sediment core collection. Image modified from Light (2011).

Table 2.1 Latitude, longitude and ecozone of the 27 study ponds located in the Churchill Wildlife Management Area, Churchill, Manitoba.

Pond	Latitude (°East)	Longitude (°North)	Ecozone
Arc Lake	58.73667	93.79161	Tundra
Birch Lake	58.72487	93.84070	Transition Zone
Box Lake	58.74269	93.82088	Tundra
Coast Lake	58.75735	93.83584	Tundra
Coast Pond	58.75690	93.83566	Tundra
East Twin Lake	58.62536	93.81602	Open-forest
Erin Lake	58.72095	93.83947	Transition Zone
Godwit Lake	58.74312	93.81702	Tundra
Golf Lake	58.75279	93.96760	Tundra
Homer Lake	58.73422	93.80011	Tundra
Imposter Lake	58.75710	93.83543	Tundra
Island Lake	58.74797	93.81999	Tundra
Larch Lake	58.72431	93.84403	Transition Zone
Lauren Lake	58.72399	93.84249	Transition Zone
Left Lake	58.74554	93.82204	Tundra
Lumpy Lake	58.73658	93.79311	Tundra
No 3 Lake	58.73196	93.82880	Transition Zone
Notch Lake	58.73616	93.79958	Tundra
Orange Lake	58.72828	93.83379	Transition Zone
Plugged Lake	58.73204	93.79546	Tundra
Puddle Pond	58.72547	93.84042	Transition Zone

Pond	Latitude (°East)	Longitude (°North)	Ecozone
Recover Lake	58.74545	93.82011	Tundra
Serene Lake	58.64030	93.82250	Open-forest
Some Lake	58.74433	93.82076	Tundra
Strange Lake	58.72649	93.83848	Transition Zone
Swept Lake	58.73684	93.80054	Tundra
West Twin Lake	58.62498	93.81886	Open-forest

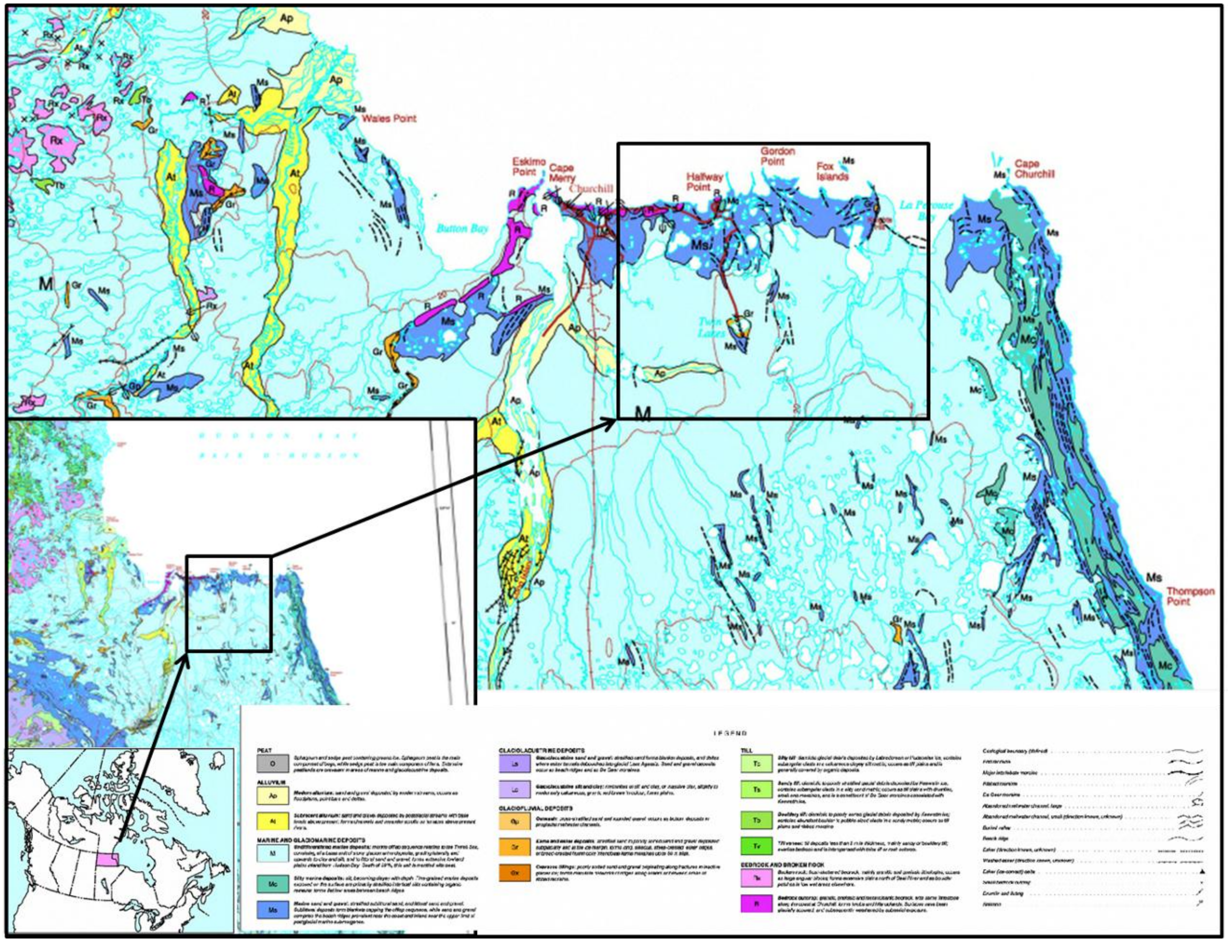
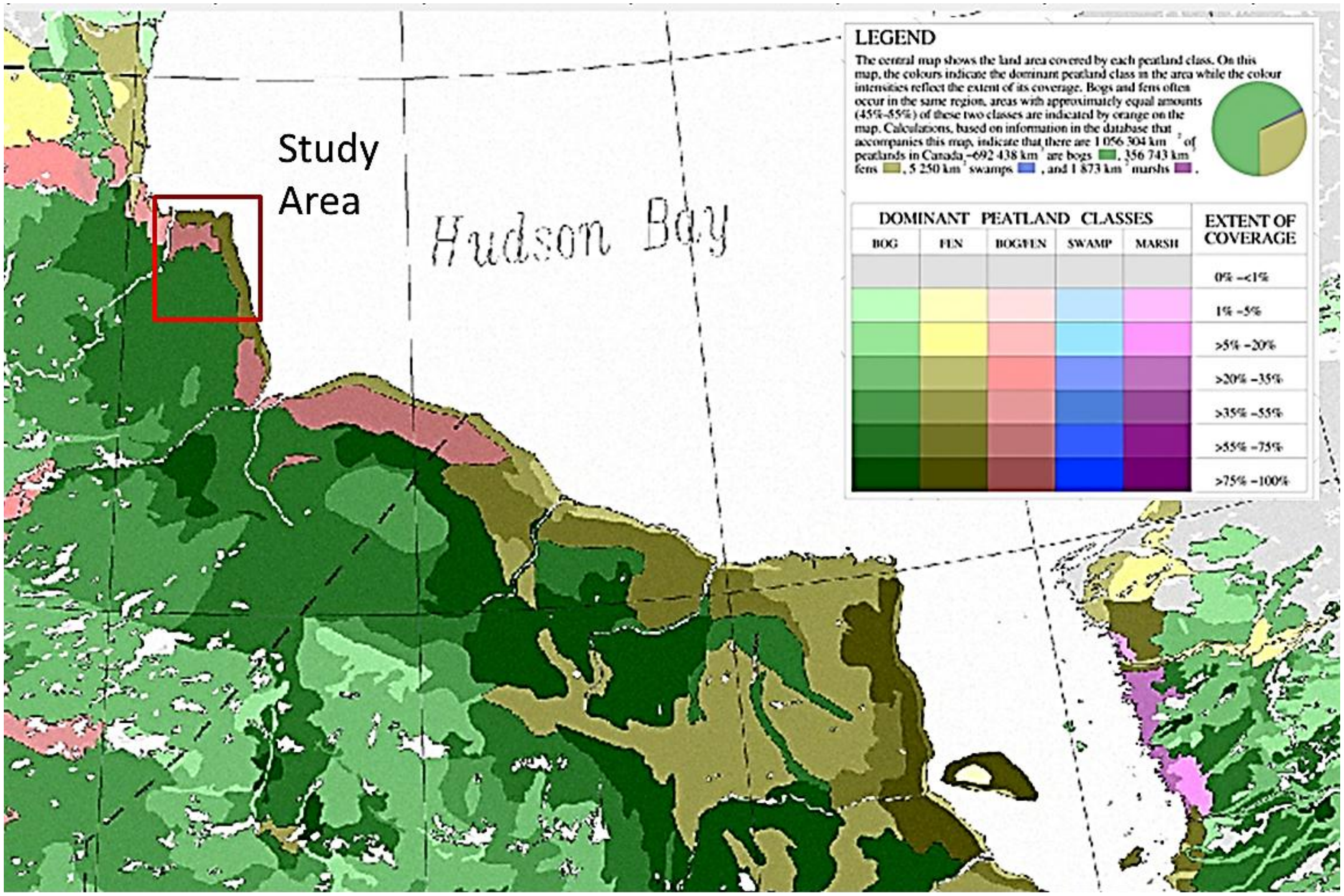


Figure 2-4 Surficial geology underlying the 27 study ponds in the Churchill Wildlife Management Area. 25 of the study ponds are underlain by marine and galciomarine deposits consisting of marine silts, sand and gravel. Two ponds (West Twin Lake, East Twin Lake) are located in an area of glaciofluvial deposits (Dredge et al., 2007).



The central map shows the land area covered by each peatland class. On this map, the colours indicate the dominant peatland class in the area while the colour intensities reflect the extent of its coverage. Bogs and fens often occur in the same region, areas with approximately equal amounts (45%-55%) of these two classes are indicated by orange on the map. Calculations, based on information in the database that accompanies this map, indicate that there are 1 056 304 km^2 of peatlands in Canada – 692 438 km^2 are bogs 🟩, 356 743 km^2 fens 🟨, 5 250 km^2 swamps 🟦, and 1 873 km^2 marshs 🟪.

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Figure 2-5 Map illustrating the distribution of peatlands within the Hudson Bay Lowlands. The region in which the study ponds are located largely consists of coastal fens and inland areas with intermixed bogs and fens (Tarnocai et al., 2000)

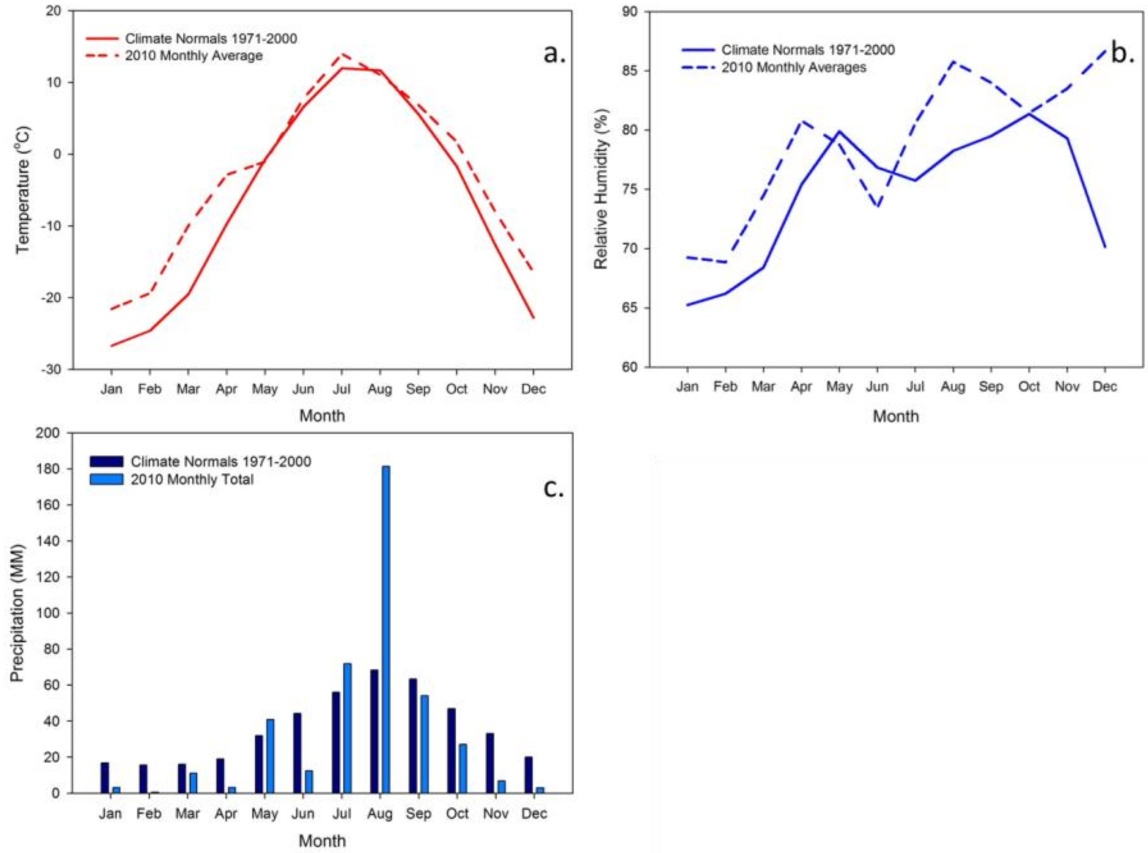

Figure 2-6 Comparison of climate data from the 2010 field season to climate normal data from Churchill, MB (Environment Canada, 2010).

Figure 2-7 Pond conditions observed at Birch Lake (left) and No 3 Lake (right) during sampling in July 2010.

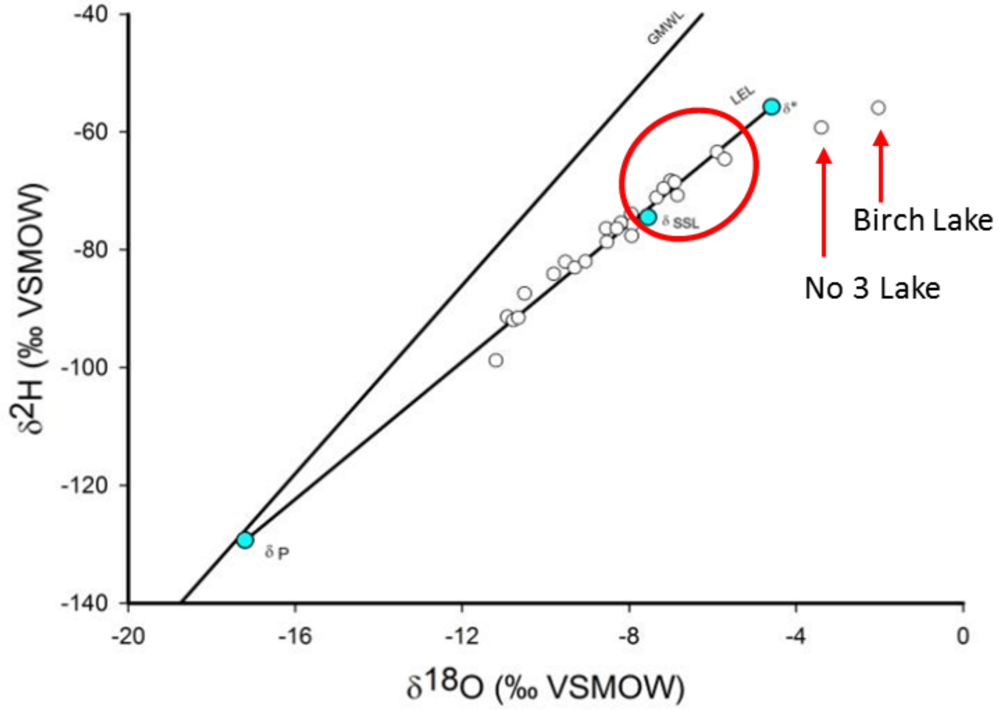

Figure 2-8 Pond water isotope compositions from the ponds in the Churchill Wildlife Management Area based on samples collected in July 2010 (Light, 2011). The red ellipse indicates the ponds that have passed δ_{SSL} on the LEL and have a negative water balance. Birch Lake and No 3 Lake, highlighted in the previous image, plot at the upper end of the LEL due to the extreme evaporation experienced by these two ponds during the thaw-season in 2010.

Figure 2-9 Photographs of vegetation that characterizes the three ecozones in the Churchill Wildlife Management Area, Manitoba in which the study ponds are located: a) spruce and tamarack are observed in the open-forest ecozone; b) a mix of tree cover and ground cover vegetation is observed in the forest-tundra transition zone; and c) low-lying shrubs, lichens and grasses characterize the tundra.

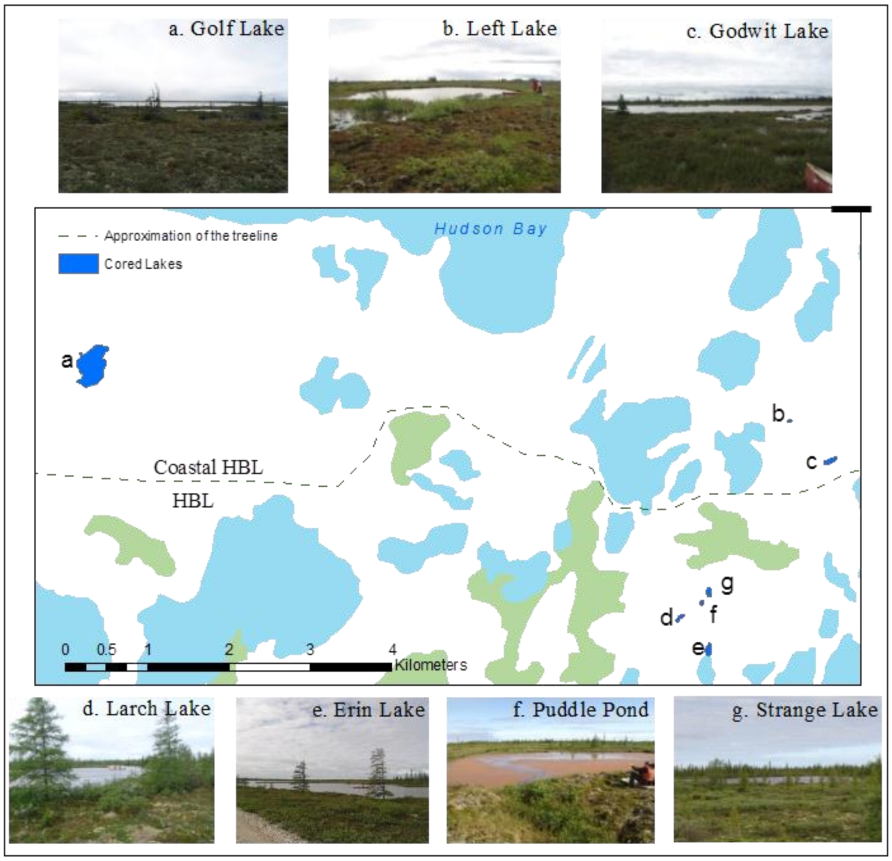

Figure 2-10 Map illustrating the location of seven ponds selected for sediment core retrieval from the Churchill Wildlife Management Area, Manitoba (Light, 2011). Three of the ponds (Golf, Left, Godwit) are located north of the Arctic treeline in the tundra ecozone, while four ponds (Larch, Erin, Puddle, Erin) are located south of the Arctic treeline in the forest-tundra transition zone. Left Lake and Erin Lake sediment cores were selected for the multi-proxy paleolimnological investigation in this thesis and the MSc thesis by Light (2011).

Table 2.2 Pond depth and surface area of the seven study ponds where sediment cores were collected in the Churchill Wildlife Management Area, Churchill, Manitoba. Measurements were taken in August 2009 (Light, 2011).

Pond	Depth (cm)	Area (m^2)
Left Lake	23.7	181.048
Godwit Lake	40.0	884.859
Golf Lake	63.0	13525.912
Erin Lake	34.5	852.131
Strange Lake	42.0	629.547
Puddle Pond	0.0	189.931
Larch Lake	20.0	471.461

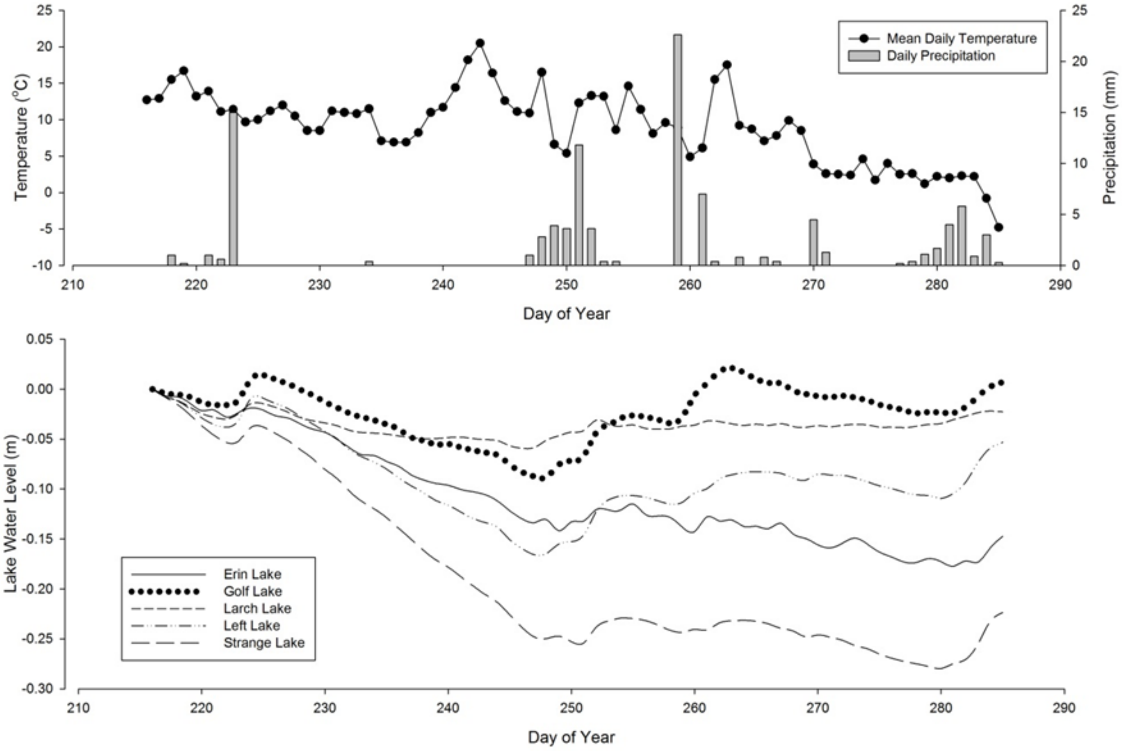

Figure 2-11 Pond water level and climate data for the interval August 4 to October 12, 2009 from 5 of the 7 ponds in the CWMA selected for coring (Light, 2011).

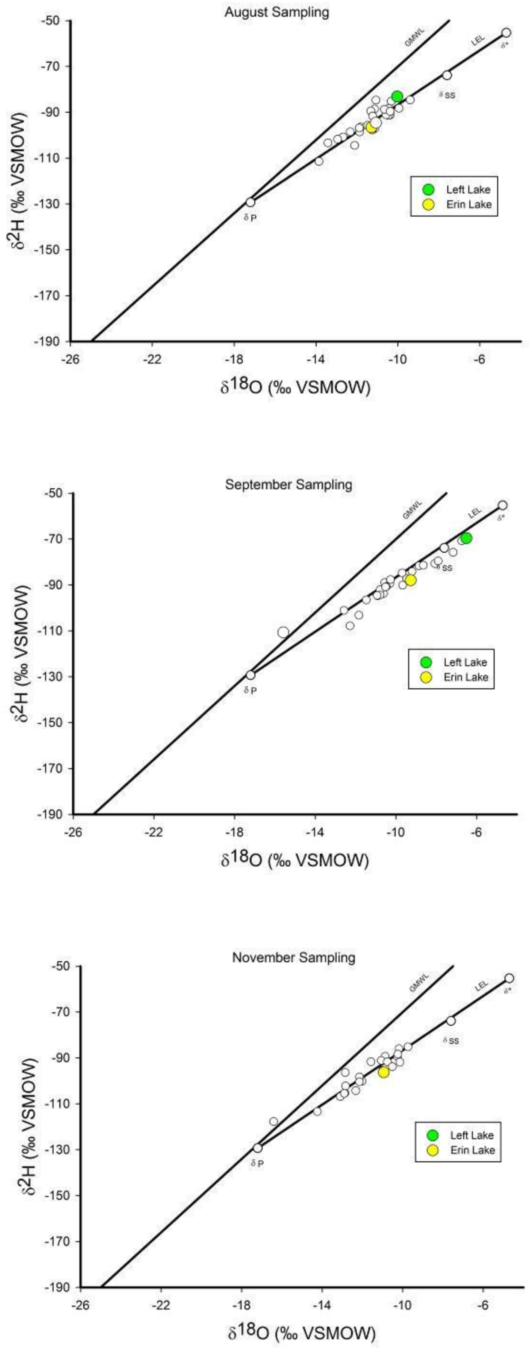

Figure 2-12 Water isotope data from samples collected from 25 study ponds within the study area in August, September and November 2009. The two ponds selected for paleolimnological reconstruction using biological indicators are highlighted. Data for November sampling in Left Lake is unavailable as the pond was ice-covered (Light, 2011).

Chapter 3

Methods

3.1 Exploring linkages between pond hydrology and limnological conditions

This section describes the methods used to explore linkages between the pond water balances that were characterized by Light (2011) using water isotope tracers (^{18}O, ^{2}H) and the limnological characteristics of the study ponds. Water samples for the Light (2011) study were collected during the thaw-season in 2009 and 2010 and measured for deviations in stable isotope concentrations from Vienna Standard Mean Ocean Water (V-SMOW) values. Water-level logger data collected throughout the open –water season on selected ponds were used to help characterize the influence of precipitation and evaporation on pond water balances.

Relationships between basin hydrology and limnological characteristics of the study ponds were investigated using similar methodology employed by Sokal et al. (2008). Modifications to the methods used in data analysis by Sokal et al. (2008) are described in section 3.1.4.

3.1.1 Sampling methods

Water samples were collected in June, July and September in 2010 from the 27 study ponds for analysis of total suspended solids (TSS), chlorophyll-a (chl-*a*) and water chemistry (concentrations of nutrients and ions and pH, as detailed below) in order to assess the current limnological conditions in the study ponds. Water samples were filtered *in situ* through pre-combusted (1 hour @ 550°C) Whatman® GF/C filters (pore size 1.2 microns) for analysis of TSS and through Whatman® GF/F filters (pore size 0.7 microns) to concentrate phytoplankton samples for analysis of chl-*a* concentrations. Filters were frozen and stored in the dark until returned to the University of Waterloo for analysis. TSS was analyzed using standard methods (Lind, 1974) and photosynthetic pigments were analyzed using Reverse-Phase High Pressure Liquid Chromatography (RP HPLC) (Leavitt and Carpenter, 1990) at the Waterloo Environmental Change Research Laboratory (WATER lab) at the University of Waterloo. Chlorophyll-*a* was analyzed using standard fluorometric techniques (Parsons and Strickland, 1963, SCOR/UNESCO, 1966) at the Aquatic Ecology Group Analytical Laboratory at the University of Waterloo. Samples collected were analyzed in the Biogeochemistry Lab at the University of Waterloo for alkalinity and concentrations of total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate and nitrite (NO_3^- +NO_2^-), ammonium (NH_4^+), total Kjeldahl nitrogen (TKN) and most major ions (Cl^-, SO_4^{2-}, Na^+, K^+, Mg^{2+}, Ca^{2+}). Samples were collected in July for analysis at the National Laboratory for Environmental Testing (NLET) in Burlington, ON for additional measurements of concentrations of silica (SiO_2), dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC). Water temperature, pH, conductivity and dissolved oxygen (DO) content were measured *in situ* at each sampling site at the time of water sample collection using a YSI 600XL handheld multi-meter.

3.1.2 TSS determination

Prior to sample collection, GF/C filters were prepared by combusting them in crucibles in a muffle furnace at 550 °C for 1 hour and weighing the ashed filters (in grams) to four decimal places. Filters were individually stored in a small Whirl-pak® bag and the filter number and weight recorded on the bag. At the Churchill Northern Studies Centre (CNSC), water samples were thoroughly mixed prior to filtering and the volume of water filtered was recorded. Filters were placed in numbered crucibles in the drying oven at 90 °C for 24 hours, removed and allowed to cool in a desiccator and then reweighed. TSS was calculated in units of mg/L as (sample weight – initial weight)/ volume filtered. Crucibles and filters were then combusted in muffle furnace at 550 °C for 1 hour, allowed to cool in a desiccator and reweighed. Ash-free Dry Weight (AFDW), which is equivalent to organic suspended solids (OSS), was calculated in mg/L as (sample weight – combusted weight)/ volume filtered. Inorganic suspended solids (ISS) was calculated in mg/L as TSS – AFDW.

3.1.3 Chlorophyll-*a* determination

In low-light conditions, GF/F filters were placed in vials with 20 mL of 90% acetone solution for 18-24 hours to extract chl-*a* pigments. The fluorometer was blanked with 90% acetone solvent prior to analyzing the first sample. If necessary, the extract was diluted until the pigment colour was barely noticeable and the dilution factor was recorded. The extract was then loaded into a fluorometer cuvette, inserted into the fluorometer and the reading recorded. Four drops of 6N HCl was added to sample in the fluorometer cuvette, and the sample was then reinserted into the fluorometer and the reading recorded. This step converted chl-*a* to the breakdown product pheophytin-*a* which can be used as an estimate of algal health by comparing the fluorescence of the unacidified to the acidified samples (Yentsh and Menzel, 1963). Chl-*a* concentration was calculated as:

$$\text{Chl-}a\ (\mu g/L) = F * [\tau/(\tau-1)] * (F_b-F_a) * D * (v/V)$$

$$\text{Phaeo}\ (\mu g/L) = F * [\tau/(\tau-1)] * ((\tau F_a)-F_b) * D * (v/V)$$

where F was the calibration factor, V was the volume of filtered water, v was the volume of extraction solution, τ was the ratio of fluorescence of pure chl-*a* before and after acidification, D was the dilution factor and F_b and F_a were the fluorescence readings before and after acidification.

3.1.4 Numerical analysis

Numerical analysis was conducted using methods modified from Sokal et al. (2008). Initially, limnological variables were assessed to eliminate variables below detection limits from the data set. Soluble reactive phosphorus (SRP) was removed from numerical analysis because values were below detection limits in more than 25% of the study ponds. Study ponds that were missing variables in more than one sampling period were also removed in order to assess limnological changes throughout the open water season. Birch Lake, Imposter Lake and Lauren Lake were removed from the analysis for this reason as sampling error or desiccation resulted in an incomplete data set in more than one sampling period in 2010. Ponds that were missing values in only one sampling period remained in the analysis and average values from the sampling period from all the ponds replaced zero values. No 3 Lake was missing data for a few variables in July 2010 due to desiccation and remained in the data set. Outliers were assessed for validity and removed from the analysis or retested. Puddle Pond was removed from the statistical analysis due to anomalously high concentrations of NH_4^+, chl-*a*, suspended solids and TKN. Although the values obtained were highly plausible given the extreme hydrological and limnological conditions in this pond, the pond was removed from numerical analysis because its relatively high values obscured the ability to assess differences and patterns among the other study ponds. Puddle Pond was also missing values for numerous limnological variables due to insufficient water levels to conduct sampling. Three ponds (Box Lake, Coast Pond, East Twin Lake) had abnormally high TP values in comparison to the other ponds in July. Comparison of other limnological variables between these ponds and the other ponds did not reveal a similar pattern, which indicated that the digestion process may have resulted in contamination of these samples.TP was retested, which provided results that were no longer outliers in the data set. All environmental variables (except SRP) remained in the data set for the numerical analyses except for the direct gradient ordinations of diatom and pigment community composition of the surface sediments in which manual forward selection was used to remove correlated variables with high variable inflation factors. Following this data screening step, 23 of the original 27 ponds included in the study remained in the data set for numerical analysis of limnological conditions. Next, the remaining limnological variables were tested for normal distributions using the Kolmogorov-Smirnov test for normality and normal quantile plots (Q-Q plots) constructed using the computer software SPSS 19.0. All data with a non-normal distributions were transformed using ln (x + (0.5*minimum non-zero value)) to meet assumptions of parametric statistical tests.

Principal Components Analysis (PCA), an indirect-gradient ordination method with underlying linear model, was used to evaluate differences in limnological conditions among the ponds during each individual sampling period. The ponds were coded in the ordination diagram based on vegetation zone (tundra, forest-tundra transition, open-forest) and water temperature and water isotope tracer data were plotted passively in order to assess relationships among basin hydrology, limnological conditions and catchment vegetation. Also, analysis of variance (ANOVA) tests, *post-hoc* tests based on Tukey's test statistic and boxplots were used to identify the limnological variables that differed significantly ($p \leq 0.05$) among the vegetation categories in July 2010.

PCA analysis was also used to explore limnological conditions in the 23 study ponds as pond water chemistry evolved over the thaw-season in 2010 by including the data from all three sampling period in one ordination. Pond categories defined by the "seasonal water chemistry trajectory" (SWCT) identified in this ordination were then used to assess the differences in limnological conditions in the ponds during the three individual sampling periods. The SWCT categories were used to code water stable isotope data in δ-δ plots to further assess the hydrological conditions of the study ponds. CANOCO version 4.5 was used for all ordinations (ter Braak and Smilauer, 2002).

3.2 Linkages between Hydrolimnological Conditions and Biological Communities

This section describes the methods used to explore the relationships between the hydrolimnological conditions in the study ponds and the algal community composition (based on diatoms and pigments) contained in surficial pond sediments (Sokal et al., 2008, Rühland and Smol, 2002). The relationships developed in this section between indicator taxa and hydrolimnological conditions were used to reconstruct past hydrolimnological conditions (as presented in section 3.3) through the analysis of algal community composition in pond sediment cores.

3.2.1 Sampling methods

Surface sediment samples (0-1cm) were collected in June 2010 from the 27 study ponds and analyzed to determine the community composition of diatom assemblages in the study ponds and to determine the community composition and abundance of pigments. A lucite tube fitted with rubber stoppers was used to recover these samples in order to minimize disturbance to unconsolidated sediments at the surface-water interface. Samples were stored in Whirl-pak® bags at 4°C until returned to the Waterloo Environmental Change Research Lab (WATER Lab) at the University of Waterloo for processing.

3.2.2 Laboratory procedures

Diatom slides were prepared for each of the study ponds using standard techniques outlined by Rühland and Smol (2002). Sub-samples of 0.3 ± 0.05 g of wet sediment were treated with 10% HCl to remove carbonates, rinsed until neutral with de-ionized water and then treated with a mixture of concentrated sulfuric and nitric acids (50:50, by volume) at 80°C for 6 hours to digest the remaining organic material. Samples were then rinsed repeatedly with de-ionized water until neutral and then dispensed onto coverslips, air-dried and mounted onto microscope slides using Naphrax mounting medium. A minimum of 300 valves were identified and enumerated from each sample along transects using a Zeiss Axioskop II plus compound light microscope fitted with differential interference contrast optics (numerical aperture = 1.30). Diatom taxonomy followed designations by Krammer and Lange-Bertalot (1986-1991) and Lavoie et al. (2008).

Sedimentary pigments were extracted and analyzed using standard methods described by Reuss et al. (2010). Following freeze drying of sediments, pigments were extracted from a 300 mg sub-sample in 5 ml of extraction solution consisting of HPLC-grade acetone, methanol and de-ionized water (80:15:5 by volume). Samples were stored in a freezer (-20°C) for 24 hours for the extraction process to reach completion. Then, the solution was filtered through a 0.22μL PTFE filter before being dried under nitrogen gas. Calibration solution was added to the dried samples and then transferred to HPLC injection vials for analysis. Sudan II solution was used as an internal reference standard. Pigment concentrations were quantified at the Waterloo Environmental Change Research Laboratory using a WATERS 2695 HPLC following the reverse phase procedure outlined by Mantoura and Lleywellyn (1983) as modified by Leavitt et al. (1989).

3.2.3 Numerical analysis

Numerical analyses of the diatom and pigment data were conducted following methods modified from Sokal et al. (2008) and Reuss et al. (2010) to assess relationships between hydrolimnological conditions and the recently deposited sedimentary assemblages. Diatom taxa that were not represented in samples from atleast two study ponds or that did not have maximum abundances greater than 1% were excluded from the numerical analysis. Since Birch Lake, Imposter Lake, Lauren Lake and Puddle Pond were excluded from the analysis of hydrolimnological relationships due to outliers and many zero values in the dataset, diatom abundances and pigment concentrations from surficial sediments from these four ponds were also excluded from the numerical analysis. Photosynthetic pigments that were not found in five or more of the study ponds were excluded from the ordinations. Chlorophyll c_2 (chl-c_2), aphanizophyll, diadinoxanthin, myxoxanthophyll, alloxanthin, lutein, zeaxanthin, canthaxanthin, chlorophyll b (chl-*b*), chl-*a*, echinenone and β-carotene were included in the ordinations as active 'species' representative of the major freshwater algal groups. Fucoxanthin, which was the most labile of the quantified pigments, was included passively so that differences in degradation among the ponds did not exert a confounding signal that may influence the results of the ordination. East Twin Lake, West Twin Lake and Coast Lake were removed from the analysis of photosynthetic pigments due to anomalously high pigment concentrations relative to the other study ponds. The high concentrations (expressed per g OM) were due to the low organic content of the sediments collected from these ponds. The low organic content was due mainly to sampling methods because the ponds were ice-covered at the time of sampling and samples could only be collected from the shallow, wave-affected shoreline zone.

Multivariate ordination techniques were used to explore patterns in the surface sediment community composition of diatoms and pigments among the study ponds and to evaluate species-environment relationships. All biological data were square-root transformed and rare taxa were down-weighted prior to analysis. CANOCO version 4.5 was used for all ordinations (ter Braak and Smilauer, 2002).

Prior to assessing the diatom data, Detrended Correspondence Analysis (DCA) was performed to determine whether to utilize linear- or unimodal-based multivariate ordination methods on the diatom and pigment data. For the surface sediment diatom data, the gradient length of first axis in the DCA was greater than 2 standard deviation units. Therefore, unimodal-based methods were employed

(Correspondence Analysis (CA) or DCA, Canonical Correspondence Analysis (CCA)). Correspondence Analysis was used to assess the distribution of surface sediment diatom assemblages among the study ponds, but the resulting ordination diagrams identified a possible arch-effect, so DCA was used instead (ter Braak and Šmilauer, 2002). Ordination by CCA was used to identify the limnological variables that best accounted for independent and significant amounts of the variation in the surface sediment diatom community composition. CCA was performed using manual forward selection and Monte Carlo tests with 999 permutations under the full model.

For the pigment concentration data, the gradient length of the first DCA axis was less than 2 standard deviation units. Therefore, linear methods were employed (PCA and Redundancy Analysis (RDA)). PCA was used to assess the distribution of surface sediment pigments among the 20 study ponds. RDA was then used to explore the relationship between surface sediment pigment concentrations and limnological conditions in the study ponds. Redundancy Analysis of the species-environmental relationships using manual forward selection identified SiO_2 as only variable that accounted for significant amounts of variation between the 20 study ponds in the analysis and therefore was not very informative. Instead, limnological variables that are known to exert strong influence on algal communities in lakes and ponds were selected to assess pigment-environment relations in the 20 study ponds. Conductivity was selected to represent the ionic composition of the study ponds as it was positively correlated with most major ions (SiO_2, $lnCl^-$, $lnNa^+$, K^+, $lnSO_4^{2-}$, Mg^{2+}) in the PCA ordination of the environmental variables. TSS was the variable selected to represent suspended solids as all three measures of suspended solids were highly correlated (lnISS, lnAFDW, lnTSS). Both alkalinity and pH were both selected due to the large influence that these two variables have on algal community composition. lnTP and lnTKN were used in the ordination in order to assess the relationship between the phototrophic community and nitrogen and phosphorus levels, which are normally the limiting nutrient in high-latitude lakes (Reuss et al., 2010). $lnNO_3^-+NO_2^-$ was included in the analysis to assess the effect that biologically-available nutrients have on the distribution of photosynthetic pigments in the study ponds. SRP was excluded from the ordination due to numerous values that were below minimum detection limits in a large number of the study ponds, as previously mentioned.

3.3 Reconstructing past hydroecological conditions

This section describes the methods that were used to assess past hydroecological responses to climatic changes through the analysis of fossil diatoms and pigments within the pond sediment record. Geochemical components from the pond sediment cores investigated by Light (2011) were also incorporated in this section to further inform the paleolimnological reconstruction.

3.3.1 Sample collection

Two sediment cores were collected by Light (2011) from each of the seven study ponds in August 2009. The cores were collected from near the centre of each pond using a Glew gravity corer (Glew, 1989) fitted with a Lucite tube (7.6-cm inside diameter). The cores were sectioned into 0.5-cm intervals at the field base (CNSC). Sediment samples were stored at 4^{o}C until laboratory analysis.

3.3.2 Loss-on-ignition

LOI was conducted on both gravity cores (GC-1, GC-2) from each of the two ponds selected for paleolimnological reconstruction and on surface sediment samples from the 27 study ponds using methods modified from Dean (1974). A subsample of 0.5 ± 0.05 g of well-mixed sediment was weighed into a pre-weighed, clean dry crucible and the final weight was recorded along with the crucible number for each sample. Sediments and crucibles were then placed into a drying oven at 90^{o}C for 24 hours. After 24 hours, the crucibles were removed from the drying oven, allowed to cool to room temperature in a desiccator and then re-weighed. All of the crucibles were then combusted in a muffle furnace at 550^{o}C for one hour. The samples were allowed to cool, following the same procedure and then re-weighed. The crucibles were then ashed in the muffle furnace at 950^{o}C for one hour, allowed to cool and re-weighed once more.

The percent water content was calculated as:

$$\text{Moisture content} = ((WW-DW_{90}) / WW - CW) * 100$$

The percent organic matter was calculated as:

$$OM (\%) = ((DW_{90}-DW_{550}) / DW_{90} - CW) * 100$$

The percent mineral matter was calculated as:

$$MM (\%) = ((DW_{950}-CW) / DW_{90} - CW) * 100$$

The percent $CaCO_3$ was calculated as:

$$CaCO_3 = [((DW_{550}-DW_{950}) / DW_{90} - CW) * 100] /0.44$$

where, CW is the crucible weight, WW is the weight of the wet sediments and crucible and DW is the weight of the dried sediments and crucible.

3.3.3 Sediment core chronologies

To establish the chronologies of the cores, sediment samples were analyzed at the Waterloo Environmental Change Research Laboratory, Waterloo, Ontario for the radioactive isotopes ^{210}Pb, ^{137}Cs and ^{214}Bi using methods described by Oldfield and Appleby (1984). Approximately 1-cm^3 of epoxy (Devcon ®, No 14310) was used to seal > 3 g sample of freeze-dried sediments packed to a height of 35 mm in 8 ml polypropylene (SARSTEDT; product No. 55.523) tubes. Samples were then allowed to sit undisturbed for 2-3 weeks to trap any ^{222}Rn gas and establish equilibrium between ^{226}Ra and daughter isotopes (^{210}Pb, ^{137}Cs and ^{214}Bi). ^{210}Pb, ^{214}Pb, ^{137}Cs and ^{214}Bi activity were counted on a high-purity Ge co-axial Digital Gamma Ray Spectrometer. Both the Linear and CRS models were used to explore the activity profiles of the sediment cores. However, the CRS model was applied to the data to establish the sediment chronology. ^{137}Cs activity profiles were compared to the ^{210}Pb-based chronology.

3.3.4 Diatoms and fossil pigment analysis

Diatoms and photosynthetic pigment were analyzed for every second 0.5-cm interval from core GC-1 for Left Lake and Erin Lake using sample preparation methods described for surface sediment analysis described previously.

3.3.5 Numerical analysis

Sedimentary diatom community composition was assessed by plotting diatom assemblage data from the sediment cores for Left Lake and Erin Lake passively in the DCA ordinations of the surface sediment diatoms (as developed in section 3.2.3) using methods described by Sokal et al. (2008). This method provided a visual representation of the downcore variation in diatom community composition in the context of the spatial variation observed among the ponds in the surface sediment assemblages.

Chapter 4

Characterizing contemporary hydrolimnological conditions in shallow tundra ponds of the HBL

The purpose of Chapter 4 is to explore relations between the hydrological and limnological conditions in the study ponds. This analysis can be used to identify the hydrological factors that most strongly influence current limnological conditions in the study ponds. If relationships between hydrological and limnological conditions are sufficiently strong, methods can be developed and applied to obtain long-term records of past changes in hydrological and ecological conditions, as is required to anticipate future pond responses to climate warming and other factors.

4.1 Physical and chemical limnological conditions

The 23 ponds included in the final data set are typically less than 1.5 m deep (Macrae, 1996), fishless and freeze to the bottom during the winter months. Limnological analysis of water samples collected in July 2010 (Table 4.1) indicated that the ponds are alkaline (mean pH = 8.75; range 8.00 – 9.47), nutrient-rich (mean TKN = 1410.0 µg L^{-1}; range: 270 – 5170 µg L^{-1}: mean TP = 934.4 µg L^{-1}; range: 14.0 – 1991 µg L^{-1}), and high in concentrations of most major ions except Ca^{2+} and SiO_2 (Kalff, 2002; Table 4.1). Relatively low chl-*a* concentrations (mean chl-*a* = 1.821 µg L^{-1}; range: 0.34 – 8.84 µg L^{-1}) and relatively high AFDW concentrations (mean = 5.53 mg L^{-1}; range: 1.4 – 27.9 mg L^{-1}) indicated that most of the phosphorus (P) and nitrogen (N) in the water column was likely in the form of particulate organic matter rather than phytoplankton or other photosynthetic organisms (Sokal et al., 2008). Relatively low ISS (mean = 2.33 mg L^{-1}; range 0.0 – 18.5 mg L^{-1}) in comparison to AFDW further supports this interpretation as very little of the P or N likely originated from the inorganic fraction. DOC concentrations were high (mean = 13.82 mg L^{-1}; range: 4.8 – 33.9 mg L^{-1}) in comparison to values typical in tundra vegetation zones (Kalff, 2002).

There are a variety of seasonal trends in the limnological variables measured during the thaw-season in 2010. A slight decrease in pH was observed during the thaw-season in 2010 (Table 4.2), while other limnological variables progressively increased in concentration (conductivity, alkalinity, Ca^{2+}, Cl^-, SO_4^- ; Table 4.2). Large decreases in the concentration of biologically-available nutrients (NO_3^- + NO_2^-, NH_4^+) were observed early in the thaw-season, likely in response to increased uptake for primary production in the ponds. Concentrations of most ions, TKN, TP and TSS increased initially, likely in response to evaporative concentration during the early thaw-season (June to July). However, these variables decreased late in the thaw-season (July to September), likely due to dilution from a large precipitation event.

Principal Components Analysis (PCA) of the limnological and hydrological variables measured in June 2010 was used to explore relations between hydrological and limnological conditions among the study ponds from the three ecozones (Figure 4.1). PCA axis 1 captured 38.5% of the variation in the 17 active limnological variables and mainly captured gradients in concentrations of ions and nutrients. Ponds with high conductivity, alkalinity, and concentrations of nutrients and ions were positioned to the right along axis 1. Vectors for these variables were orthogonal to those for water isotope composition, indicating they were not correlated with pond water balance. PCA axis 2 captured 17.8% of the variance in limnological conditions and separated ponds mainly due to differences in concentrations of suspended solids (positively correlated with axis 2) and water isotope composition (negatively correlated with axis 2). Chl-*a* and pH explained equal amounts of variation in both axis 1 and axis 2. Ponds with higher pH and chl-*a* concentrations plotted in the upper left quadrant of the ordination diagram. There was a slight tendency for ponds in the open-forest ecozone to plot to the left along axis 1 associated with lower ionic concentrations. However, no distinct differences in water chemistry among ecozones are observed in early thaw-season.

The PCA ordination diagram of the 20 active environmental variables measured in July 2010 was used to assess the variation in limnological conditions among ponds from the different ecozones (Figure 4.2). The first ordination axis accounted for 42.6% of the variation among ponds and mainly captured a gradient of alkalinity and DIC concentrations. Ponds with high alkalinity and DIC concentrations were positioned to the right along axis 1. Vectors for these variables were negatively correlated with those for pH. Axis 2 of the PCA accounted for 18.6% of the variation in which limnological conditions separated the ponds mainly due to differences in concentrations of suspended solids and chl-*a* and water isotope composition (negatively correlated with axis 2) and concentrations of $NO_3^- + NO_2^-$ (positively correlated with axis 2). Ponds with lower concentrations of suspended solids and chl-*a* were positioned high along axis 2. Conductivity and concentrations of DOC, TKN, TP and most major ions (Ca^{2+}, Na^+, K^+, Mg^{2+}, Cl^-, SO_4^-) contributed equally to the variation among ponds along the first and second axis. Ponds with higher conductivity and concentrations of most major ions (Ca^{2+}, Na^{+}, K^{+}, Mg^{2+}, Cl^{-}, SO_4^{-}) and TP were located in the top right quadrant of the ordination diagram, while ponds with higher concentrations of DOC and TKN were located towards the lower right portion of the PCA space. Vectors for water isotope composition were orthogonal to vectors for conductivity and the concentration of most major ions, indicating that pond water balance was uncorrelated to ionic concentration. Interestingly, the three ponds in the open-forest ecozone clustered relatively close together on the left side of the ordination diagram, associated with lower ionic composition and pH – a feature which may be reflective of their large surface area and volume. Interestingly, some of the larger ponds from the other ecozones (Erin Lake, Swept Lake) also plot nearby. There was also a tendency for tundra ponds to possess the highest concentrations of ions and nutrients, as identified by high values along PCA axis 1.

PCA was also used to examine the influence that vegetation type had on the water chemistry of the 23 study ponds in September 2010 (Figure 4.3). The first ordination axis separated the ponds along a gradient of increasing ionic composition and nutrient concentrations and accounted for 39.8% of the variation between the ponds. Ponds that had higher conductivity, alkalinity, pH and ion concentrations were positioned to the right along axis 1. The vector for water isotope composition was positioned opposite these variables and, therefore, negatively correlated to them. Axis 2 captured 20.9% of the variation in the study ponds and captured mainly differences in the concentrations of nutrients and suspended solids among the study ponds. Ponds that had higher concentrations of NO_3^{-} + NO_2^{-} were located in the upper half of the ordination diagram, while ponds with higher concentrations of other nutrients (TKN, TP, NH_4) and suspended solids were positioned in the bottom half of the ordination diagram. Interestingly, Erin Lake, which is hydrologically connected to a larger adjacent basin, plotted in the cluster with the ponds from the open-forest ecozone to the left along axis 1 associated with lower alkalinity, pH and conductivity levels.

Boxplots were constructed and ANOVA tests were conducted on the July limnological variables to assess differences in limnological conditions among ponds in the different ecozones. July limnological variables were used as this sampling period was hypothesized to be most representative of limnological conditions during the thaw-season because the limnology of the ponds was less affected by the influence of snowmelt than June sampling and less affected by large rainfall events that occurred later in the thaw-season than September. Since evaporation most strongly influenced the pond water balances in July (Light, 2011), this sampling interval was also used to explore the role of this hydrological process on pond water chemistry. Boxplots (Figure 4.4), ANOVA tests and *post-hoc* tests of the limnological variables confirmed the observations made previously in regard to the ordination diagram. Results identified that alkalinity (F = 4.667, $p = 0.022$, d.f. = 2, 20), lnNa^+ (F = 10.288, $p = 0.001$, d.f. = 2, 20), lnCl^- (F = 9.784, $p = 0.001$, d.f. = 2, 20), lnSO_4^- (F = 8.130, $p = 0.003$, d.f. = 2, 20), and lnDIC (F = 5.822, $p = 0.015$, d.f. = 2, 20) differed significantly between the open-forest ponds and the ponds in the other two ecozones, but ponds in the tundra and forest-tundra ecozones did not differ significantly.

The limnological data from all three sampling periods were combined into one data set to examine changes in pond water chemistry throughout the thaw-season to determine if the hydrological characteristics of the ponds resulted in different limnological responses to evaporative drawdown. PCA of the 17 limnological variables that were measured during all three sampling periods (June, July, September) in 2010 indicated that the ponds differed mainly along a gradient of alkalinity, conductivity and concentrations of most major ions (Figure 4.5). The eigenvalue for the first and second ordination axes were 0.387 and 0.196, respectively. Alkalinity, conductivity and concentrations of most ions were positively correlated with the first ordination axis, while the concentration of NO_3^-+NO_2^- was negatively correlated. The concentration of chl-*a* was strongly and negatively correlated with the second ordination axis. Concentrations of ISS, TSS, pH, TKN and TP contributed almost equal amounts to the variation along the first and second ordination axes.

Closer inspection of the PCA ordination revealed that the study ponds followed one of four different seasonal water chemistry trajectories (SWCT) during the thaw-season . One identifiable group of ponds (hereafter called SWCT 1) had relatively high initial pH and NO_3^-+NO_2^- concentrations in June (Figure 4.6a). However, as the thaw-season progressed, pH and NO_3^-+NO_2^- concentrations declined, while conductivity, alkalinity and the concentration of most ions increased. Concentrations of suspended solids, TKN and TP remained relatively constant throughout the thaw-season. A second identifiable group (hereafter called SWCT 2) followed a similar pathway as SWCT 1 ponds in the early thaw-season (Figure 4.6b). However, between July and September a shift towards increased TSS, ISS, AFDW, TKN and TP accompanied the continued increasing concentrations of most of the limnological variables (excluding pH, NO_3^-+NO_2^-). A third group identified in the PCA (hereafter called SWCT 3) also increase in alkalinity, conductivity and the concentrations of most ions during the summer, while pH and NO_3^-+NO_2^- concentrations decreased (Figure 4.6c). However, these ponds also experienced increased concentrations of suspended solids, TKN and TP early in the thaw-season. After July, SWCT 3 ponds experienced a large decline in the concentrations of suspended solids, TKN and TP, while they continued to decline in pH and NO_3^-+NO_2^- concentrations and increase in alkalinity, conductivity and ion concentrations. The fourth and final group of ponds identified in the PCA (hereafter called SWCT 4) increased in all limnological variables (alkalinity, conductivity, concentrations of ions, TKN, TP and suspended solids) throughout the thaw-season, except for pH and NO_3^-+NO_2^- concentrations (Figure 4.6d).

In order to assess whether the ponds in each of the four SWCT catergories shared similar limnological properties in any of the sampling periods in 2010, PCA analysis of the environmental variables was reassessed with the ponds coded by SWCT (Figure 4.7). In the PCA of the environmental variables from June 2010, ponds categorized as SWCT3 and SWCT 4 generally plot towards the right lower quadrant in the ordination diagram associated with higher concentrations of suspended solids, TKN and TP (Figure 4.7; upper left panel). In the ordination of the 20 environmental variables measured in July 2010, ponds categorized as SWCT 3 and SWCT 4 became separated, forming distinct clusters in the ordination diagram (Figure 4.7; upper right panel). SWCT 3 ponds continue to plot towards the right side of the ordination diagram associated with higher alkalinity, conductivity, and concentrations of ions, TKN and DOC. Ponds categorized as SWCT 4 plot in the left lower quadrant associated with higher pH and concentrations of suspended solids. Ponds in the other two categorizes are clustered together in the upper left quadrant of the ordination diagram associated with lower alkalinity, conductivity and concentrations of suspended solids, DOC and TKN. In the PCA ordination diagram of the 17 limnological variables measured in September 2010, SWCT 1 and SWCT 3 ponds plot in the upper half of the ordination associated with lower concentrations of suspended solids, whereas ponds in SWCT 2 and SWCT 4 categories cluster together towards the lower right in the ordination diagram associated with higher concentrations of suspended solids, TP and chl-*a* .

Water isotope data (Light, 2011) were investigated to determine if the ponds differed in their hydrological response in relation to the SWCT categories that were identified. The ponds were coded on the isotopic framework based on SWCT and revealed no discernible trends that would suggest that the ponds from the different SWCT categories were isotopically dissimilar during any of the sampling periods (Figure 4.8). This was as expected, as Light (2011) was unable to identify any distinct differences among the study ponds. It was observed that all the ponds followed a similar pattern in which ponds were isotopically depleted in the spring, became more enriched throughout the thaw-season due to evaporation and then became depleted again in the fall due to dilution from increased late summer/early fall rain events.

4.2 Discussion

The climatic conditions during the thaw-season in 2010 provided an excellent opportunity to examine limnological conditions and to explore relations between hydrological processes and limnological conditions. This was because the extremely dry conditions observed in the early thaw-season followed by the pronounced precipitation in the late thaw-season mimicked the expected changes to future climate conditions (Macrae et al., 2010). Seasonal variations in concentrations of TP and TKN followed a similar pattern as was observed by Light (2011) for water isotope composition, in which concentrations increased between June and July due to evaporation and then decreased from July to September due to dilution by late thaw-season precipitation. TP and TKN concentrations observed during 2010 indicated that the study ponds on average were mesotrophic in nature during the June and September sampling periods and eutrophic when samples were collected in July (Table 4.3). The eutrophic status observed in July was similar to conditions previously observed by both Gray (1987, as cited by Macrae, 1998) and Macrae (1998) who reported the ponds' status as ultra-oligotrophic during wet years and eutrophic during dry years. Although nitrogen and phosphorus concentrations were elevated, phytoplankton chl-*a* concentrations remained typical of lakes that are oligotrophic (Table 4.1), which agrees with observations previously reported by Gray (1987, as cited by Macrae, 1998) and Macrae (1998). The phenomenon in which nutrient levels and chl-*a* concentrations are not correlated has also been observed in other Arctic lakes and is due to benthic photosynthesis dominating these systems whereas chl-*a* measurements are reflective of primary production in the water column (Michelutti, 2007). Because the ponds are fishless, they support a dense herbivorous zooplankton population which can also reduce the phytoplankton standing crop (Danger et al., 2009).

Pond water pH in the study ponds was alkaline averaging ~8.75, which was slightly higher than the pH values reported previously by Macrae (1998). The subset of ponds in this study was slightly different than the ponds in the Macrae (1998) study which may account for the majority of the difference observed between the two studies. Some of the difference in pH between studies may also be the result of evaporative concentration as 2010 was an extremely dry year. A minor amount of the difference in reported pH values between studies could also be due to the continued build-up of cations that are flushed into the ponds during precipitation events due to the presence of dolomites and limestone in the drainage basin (Macrae, 1998). The slight decrease in pH over the course of the 2010 field season was opposite to the seasonal trend reported by Macrae (1996). Macrae (1996) attributed the seasonal increase in pH to evaporative concentration of cations and the uptake of CO_2 by photosynthetic algae in the ponds. Macrae (1998) attributed lower pH values at the cessation of the melt period to an influx of melt water from the basin that was supersaturated in CO_2. However, analysis of water isotope composition at the onset of the 2010 thaw-season indicated that ponds consisted largely of rainfall due to late-season rain events prior to freeze up the previous year (Light, 2011). Because the ponds were full prior to freeze-up, they did not receive appreciable amounts of snowmelt that had been observed in the previous study (Light, 2011). Since the ponds did not receive a pulse of melt water supersaturated with CO_2, they had a relatively high pH at the start of the open water season compared to the study conducted in 1997. No statistically significant decline in pH was observed between the June and July 2010 sampling periods, but pH had declined significantly by the September sampling period. The decreased pH in September was likely in response to an influx of runoff from the catchment that contained higher concentrations of dissolved CO_2 due to late thaw-season rainfall events prior to sampling in September (Macrae, 1998). The decrease in pH may also be partially attributed to the tendency of rainfall to be slightly acidic (Charlson and Rodhe, 1982). Similar shifts in pH were observed throughout the 1997 field season when dramatic declines in pH and increases in DIC were observed following rainfall events (Macrae, 1998).

Hydrolimnological relationships among the ponds were explored across vegetation gradients from tundra to forest-tundra transition to open-forest ecozones based on findings by Turner et. al. (2010) in another thermokarst landscape which showed that vegetation can exert a strong influence over water balances. The hypothesis that a similar trend may be observed in the study ponds was based on the results of a previous study by Macrae (1998), which identified spring melt water as an important component of the ponds' hydrological budget. However, as previously mentioned, the study by Light (2011) indicated that snowmelt was not the main hydrological input into the study ponds of the HBL in the spring of 2010. Rather, it was an increase in late thaw-season rainfall that filled the ponds prior to freeze-up that was the main component of the water budget of the ponds at the onset of the thaw-season the following year. Even with the weaker influence of snowmelt on pond water balance, the analysis of limnological conditions based on ecozone revealed that the ponds located in the open-forest ecozone differed from the ponds located in the other two ecozones. Ponds in the open-forest ecozone were associated with lower alkalinity and lower concentrations of Na^+, Cl^-, SO_4^- and DIC. However, these results may be spurious as two of the three ponds in the open-forest ecozone are much larger and of a different origin (kettle lakes) than the rest of ponds included in the study (Light, 2011). The fact that two other large ponds (Erin Lake – transition zone and Swept Lake – tundra zone) have similar limnological conditions as the open-forest ponds further supports this interpretation. Therefore, the differences observed in limnological conditions among ecozones may be the result of pond volume and not catchment vegetation characteristics. Size, depth and volume of pond were determined to be important factors influencing the variability in water balances observed among the ponds through the analysis of water stable isotope composition (Light, 2011). These factors also appear to account for at least some the variability among ponds in limnological conditions.

Review of the seasonal trends in pond water chemistry over the thaw-season in 2010 identified four distinct patterns of limnological evolution. The different trajectories in limnological conditions among the study ponds closely align with the water isotope results reported by Wolfe et al. (2011) in which temporal and spatial patterns were attributed to pond catchment characteristics and hydrologic connectivity (timing and degree) with other basins.

Ponds characterized as Seasonal Water Chemistry Trajectory 1 (SWCT 1) undergo a seasonal pattern of increasing concentrations for almost all limnological variables (except pH and concentrations of suspended solids and NO_3^- + NO_2^-), likely associated with evaporative drawdown (Figure 4.6a). Left Lake, an example of a pond in SWCT 1, also experienced the greatest evaporative isotopic enrichment during the thaw-season of the four ponds presented in Wolfe et al. (2011). Seasonal trends in limnology and isotope hydrology along with no surface expression of hydrologic connectivity in aerial photos (Figure 4.9) indicated that Left Lake has a small catchment and is hydrologically isolated, leading to the strong role of evaporation on the pond water balance and chemistry. Further investigation of ponds categorized as SWCT 1 is required to determine if they share similar catchment characteristics as Left Lake that may be responsible for the seasonal trends observed in water chemistry.

Ponds identified as SWCT 2 initially (June to July) follow a similar trajectory as ponds in the SWCT 1 classification, but between July and September, following late thaw-season precipitation events, there was a large increase in the concentration of suspended solids and a decrease in the concentration of most ions (Figure 4.6b). These ponds like SWCT 1ponds are hydrologically isolated during drier periods in early thaw-season. “West Twin Lake” and “East Twin Lake” are much larger than “Box Lake” and are positioned in a different area in the ordination diagram due to differences in basin volume that make Box Lake more susceptible to evaporation than the other two ponds (Figure 4.6b). Between July and September all three ponds in this category increase markedly in the concentration of suspended solids in response to precipitation events that wash allochthonous material into the ponds. Alkalinity, conductivity and the concentrations of ions remain relatively constant during this period and it is believed that these ponds have sufficiently large catchments that allow for sufficient inflows following precipitation events that offset the earlier effects of evaporation. Interestingly, ponds in this category plot in similar ordination space in June and July as ponds in SWCT 1 when they are expected to be hydrologically isolated (Figure 4.7). However, in September after large late season precipitation events these ponds cluster in the ordination diagram with ponds from SWCT 3 that are known to have hydrological connections with other basins (Figure 4.7).

SWCT 3 ponds are characterized by early thaw-season increases in suspended solids and ion concentrations, but a decrease in pH and NO_3^-+NO_2^- concentration. This is followed by a marked decline in the concentration of suspended solids and a slight increase in almost all the limnological variables (except pH and NO_3^-+NO_2^-) throughout the remainder of the thaw-season. Larch Lake was categorized as SWCT 3 based on the seasonal trends that were observed in its limnology (Figure 4.6c). Larch Lake experienced less isotopic enrichment in comparison to Left Lake over the thaw-season in 2010 (Light, 2011; Wolfe et al., 2011). The weaker influence of evaporation on the water balance of this pond was attributed to lateral flow into Larch Lake. This interpretation was supported by the surface expression of hydrologic connectivity between adjacent basins in aerial photos (Wolfe et al., 2011; Figure 4.9). Seasonal trends in isotope hydrology and limnology are consistent with the interpretation that in early thaw-season, Larch Lake receives an inflow of water via lateral flow through the active layer, which acts to slow the evaporative concentration of ions and nutrients and transports suspended solids into the basin. Following the sampling period in July, Larch Lake becomes hydrologically isolated as water levels in the connected pond decline, which results in a decline in the concentration of suspended solids and the evaporative concentration of most ions and nutrients. PCAs of the individual sampling period supports this interpretation as SWCT 3 ponds cluster with ponds in SWCT 4 in the June and July ordination diagrams, while both groups of ponds are receiving inflows through hydrologic connections (Figure 4.7). However, in the PCA of environmental variables for September 2010, SWCT 3 ponds cluster with ponds from the SWCT category 1 that are known to be hydrologically isolated. Larch Lake is the only pond in the SWCT 3 category that has a detailed investigation into the hydrologic processes influencing its water balance. It is suspected that the other SWCT 3 ponds have similar catchment features as Larch Lake which are responsible for similar seasonal trends in pond water chemistry, but this requires a more in-depth investigation to make firm conclusions.

Ponds in SWCT 4 have very similar early thaw-season limnological conditions as ponds in SWCT 3. However, these ponds remain hydrologically connected to other basins through channels and subsurface flows over the entire thaw-season and differ markedly from ponds in SWCT 3 by the time sampling was conducted in September. Because these ponds receive inflows throughout the entire thaw-season, they plot in ordination space associated with higher concentrations of suspended solids and lower alkalinity, conductivity and ion concentrations during all sampling periods in 2010 (Figure 4.7). Erin Lake was included in the detailed hydrologic study by Wolfe et al. (2011) and was categorized as SWCT 4 based on this seasonal trend in water chemistry. Erin Lake experienced far less evaporative enrichment in comparison to Left Lake, which was attributed to its' large size and catchment area and a hydrological connection between the basin and the surrounding peatland (Wolfe et al., 2011; Figure 4.9).Trends in the isotope hydrology and limnology from Erin Lake are consistent with the interpretation that this pond is less susceptible to the effects of evaporative drawdown during drier periods of the thaw-season because Erin Lake is hydrologically connected to other basins and has a sufficiently large catchment to provide inflows to offset the effects of evaporation. These two factors provide season long inflows that act to wash allochthonous material into the basin and dilute ion concentrations and decrease alkalinity and conductivity. Based on similar seasonal trends in water chemistry of the other SWCT 4 ponds, it is hypothesized that they share similar basin and catchment characteristics as Erin Lake.

In summary, relationships exist between pond hydrology and seasonal patterns of limnological variation in ponds of the HBL near Churchill, Manitoba. More specifically, it is hypothesized that catchment features which allow for greater hydrologic inputs into the ponds, result in increased concentrations of suspended solids as allochthonous material from the catchment are washed into the ponds. Also, decreases in ion concentrations, alkalinity and conductivity occur as inflows offset the effects of evaporation on pond water chemistry. The opposite effect in these limnological parameters is observed in ponds during periods in which hydrological inputs decline. These associations are based on patterns in the seasonal trends in the water chemistry of the ponds that align closely with the basin and catchment characteristics of three ponds that were examined in-depth by Wolfe et al. (2011). However, further investigation into the basin and catchment characteristics of all the ponds that are grouped into each SWCT category is required to substantiate the interpretation that the ponds within each category share similar basin characteristics.

The hydrologic factors that appear to be influencing limnological conditions may also be reflected in variations in aquatic biota among the study ponds. The next chapter explores the relationships between community composition of recently deposited remains of diatom algae and photosynthetic pigments in surficial sediments of the study ponds and hydrolimnological conditions, which is an important step to help inform reconstructions of past hydrolimnological changes from information contained in pond sediment cores.

Table 4.1 Summary statistics for the 20 limnological variables measured in July 2010 from the 23 study ponds in the Churchill Wildlife Management Area.

Variable	Units	Mean	Std. Dev.	Maximum	Minimum
Depth	m	< 1.0	N/A	N/A	N/A
pH		8.76	0.33	9.47	8.00
Cond	$\mu S\ cm^{-1}$	353.9	214.83	997.0	108.0
Alkalinity	$mg\ L^{-1}$	125.87	52.25	44.61	229.05
Na	$mg\ L^{-1}$	10.219	6.143	26.81	2.17
K	$mg\ L^{-1}$	2.371	1.413	5.64	0.49
Mg	$mg\ L^{-1}$	8.024	2.532	13.47	4.20
Ca	$mg\ L^{-1}$	15.247	3.235	23.41	10.76
Cl	$mg\ L^{-1}$	45.392	50.427	3.00	225.25
SO_4	$mg\ L^{-1}$	9.781	8.586	33.53	0.27
SiO_2	$mg\ L^{-1}$	3.179	3.005	11.90	0.14
DIC	$mg\ L^{-1}$	25.46	10.67	47.8	10.7
DOC	$mg\ L^{-1}$	13.82	5.83	33.9	4.8
TKN	$\mu g\ L^{-1}$	1410.1	974.0	5170.0	270.0
$NO_3 + NO_2$	$\mu g\ L^{-1}$	17.20	2.35	25.00	15.00
NH_4	$\mu g\ L^{-1}$	98.20	71.67	409.0	47.0
TP	$\mu g\ L^{-1}$	261.2	403.5	1155.0	14.0
Chl-*a*	$\mu g\ L^{-1}$	1.821	1.751	8.84	0.34
TSS	$mg\ L^{-1}$	7.92	10.21	46.4	1.4
ISS	$mg\ L^{-1}$	2.39	4.63	18.5	0.0
AFDW	$mg\ L^{-1}$	5.53	6.34	27.9	1.4

Table 4.2 Mean values for the 17 limnological variables measured during the three sampling periods in the 2010 field season from the 23 ponds located in the Churchill Wildlife Management Area.

Variable	Units	June	July	September
pH		8.97	8.76	8.25
Cond	µS cm^{-1}	214.1	353.9	685.5
Alkalinity	mg L^{-1}	83.64	125.87	134.68
Na	mg L^{-1}	6.474	10.219	8.189
K	mg L^{-1}	1.686	2.371	1.654
Mg	mg L^{-1}	4.577	8.024	6.976
Ca	mg L^{-1}	11.606	15.247	17.532
Cl	mg L^{-1}	15.247	45.39	48.61
SO_4	mg L^{-1}	4.646	9.781	14.593
TKN	µg L^{-1}	752.4	1410.1	710.6
$NO_3 + NO_2$	µg L^{-1}	62.2	17.2	17.9
NH_4	µg L^{-1}	113.1	98.2	72.8
TP	µg L^{-1}	466.5	261.2	86.6
Chl-*a*	µg L^{-1}	1.375	1.821	1.493
TSS	mg L^{-1}	4.13	7.92	3.90
ISS	mg L^{-1}	1.40	2.39	0.74
AFDW	mg L^{-1}	2.73	5.53	3.16

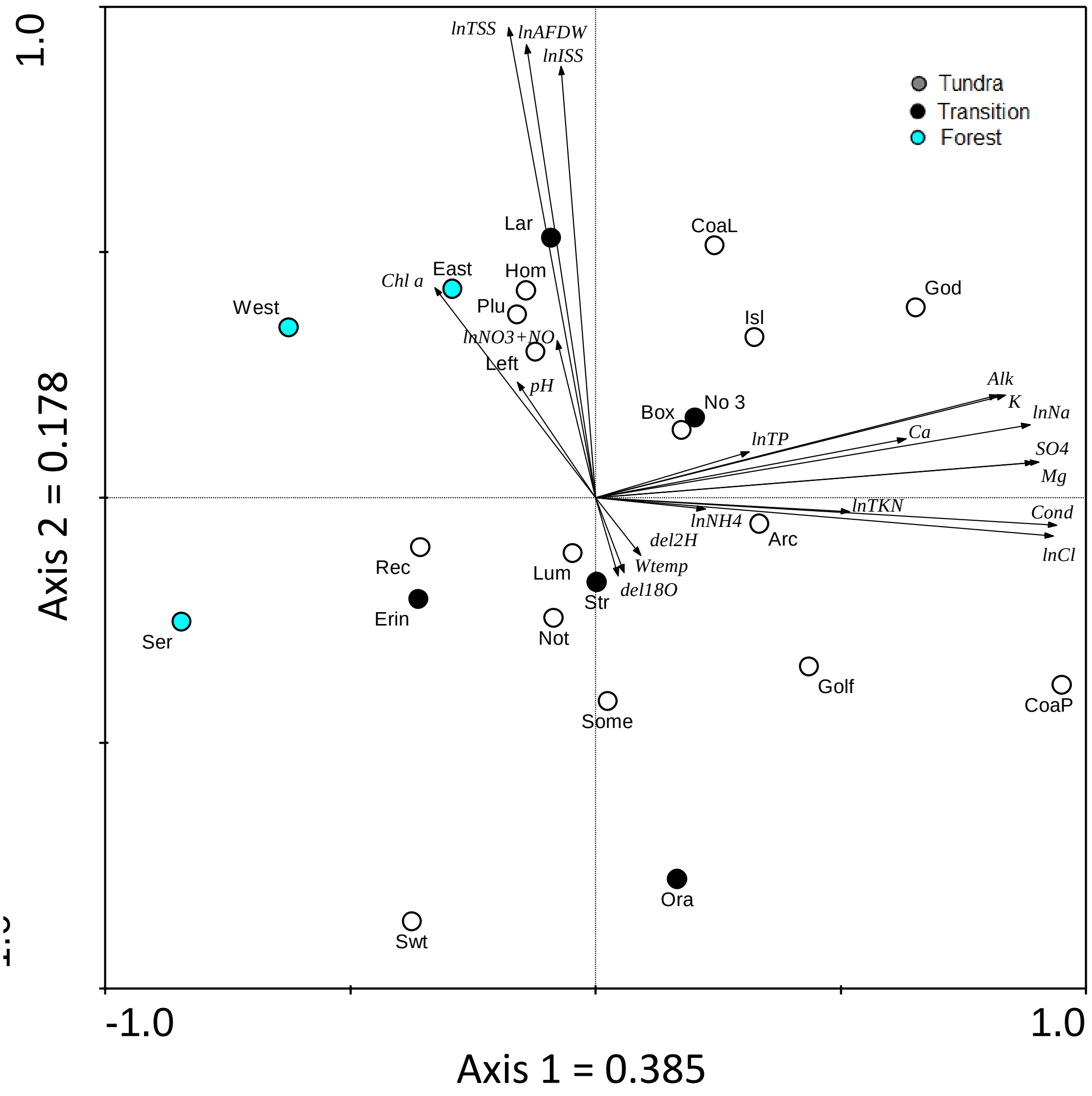

Figure 4-1 Principal Component Analysis (PCA) ordination diagram of the 23 study ponds from the CMWA with respect to the 17 limnological variables measured in June 2010. Ponds are coded based on the vegetation zone in which they are located.

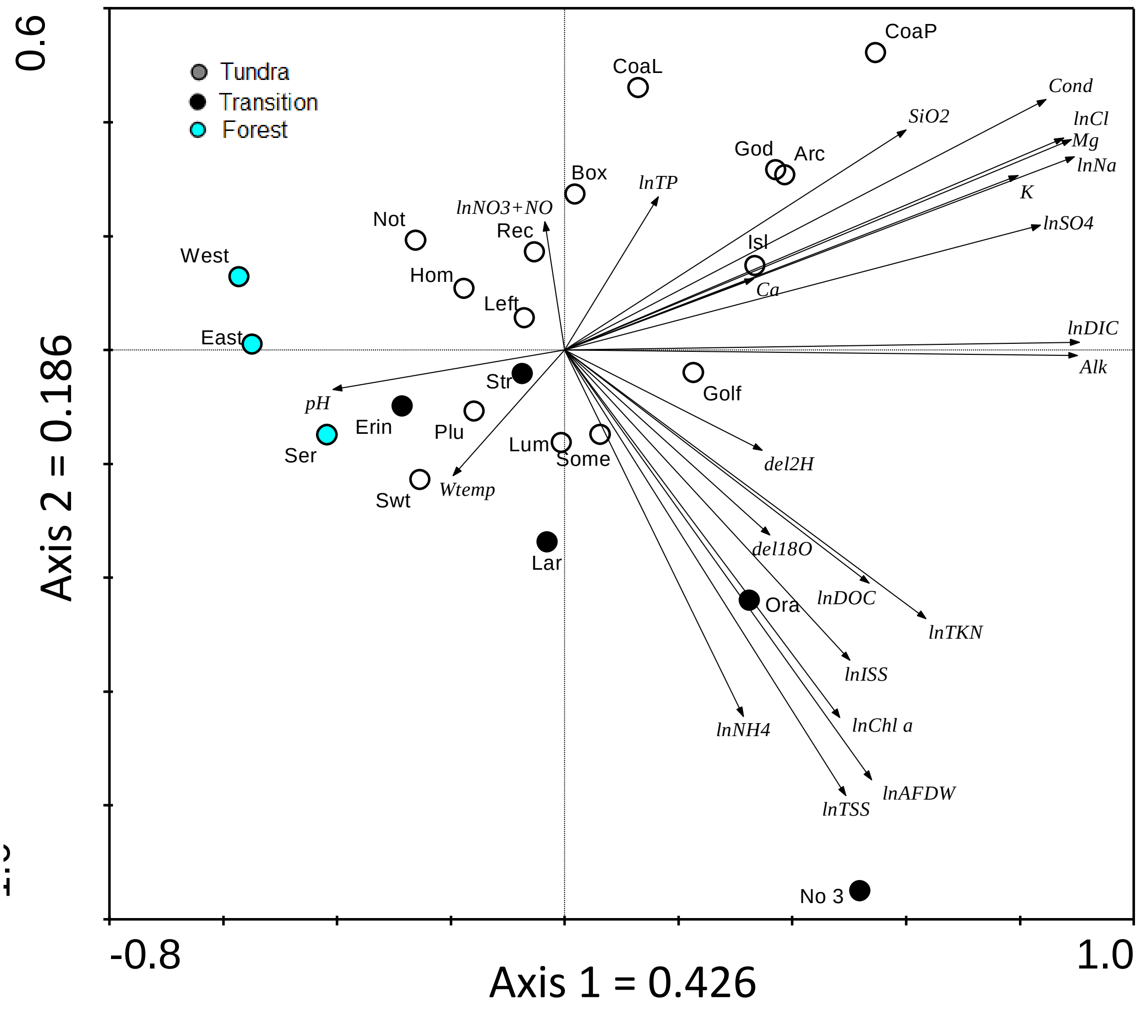

Figure 4-2 Principal Component Analysis (PCA) ordination diagram of the 23 study ponds from the CMWA with respect to the 20 limnological variables measured in July 2010. Ponds are coded based on the vegetation zone in which they are located.

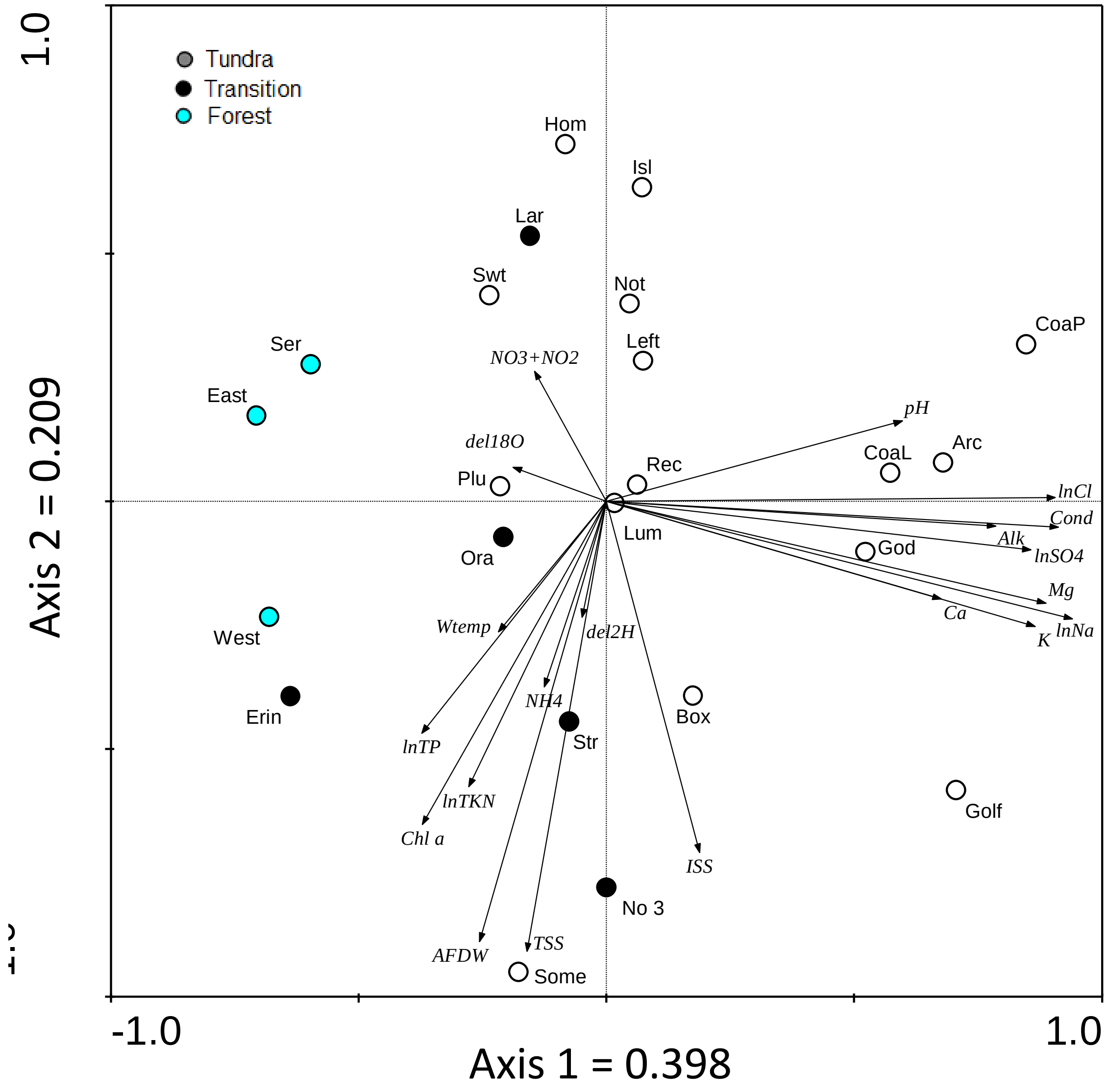

Figure 4-3 Principal Component Analysis (PCA) ordination diagram of the 23 study ponds from the CMWA with respect to the 17 limnological variables measured in September 2010. Ponds are coded based on the vegetation zone in which they are located.

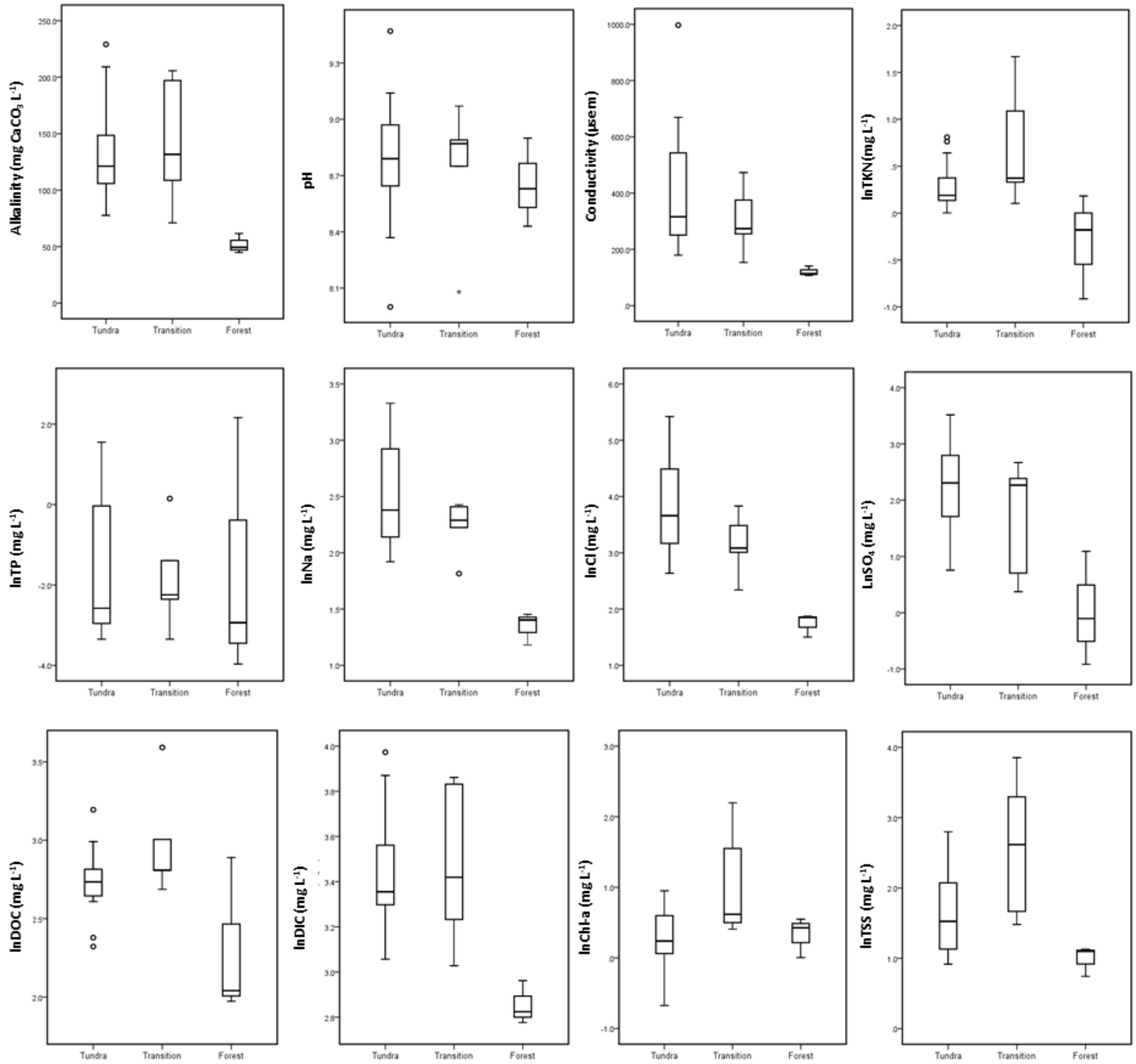

Figure 4-4 Boxplots of 12 selected variables for the 23 study ponds sampled in July 2010. Ponds are categorized based on the vegetation zone (Tundra; n=15, Transition; n=5, Forest; n=3) in which they are located.

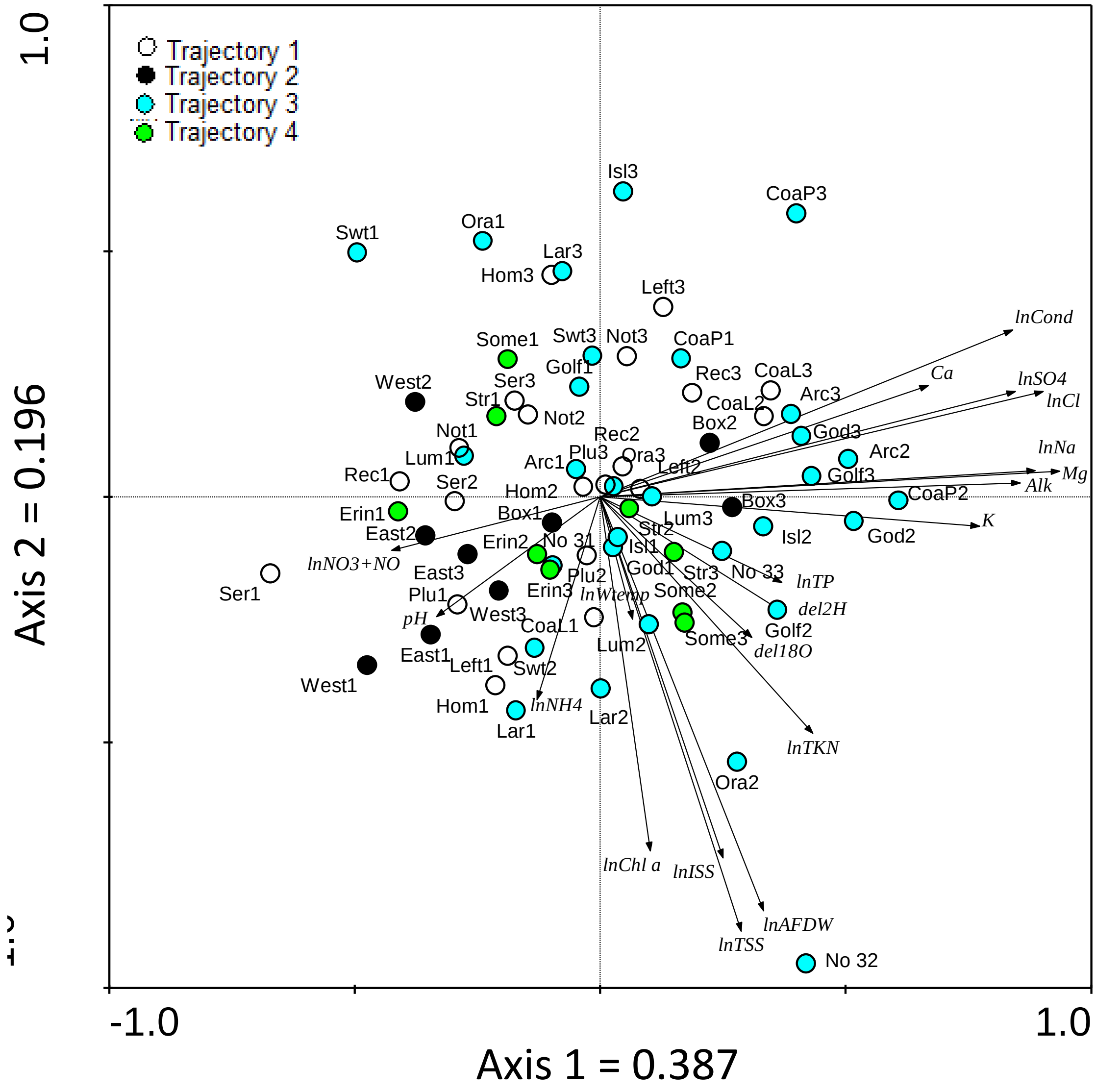

Figure 4-5 Principal component analysis (PCA) ordination diagram illustrating the evolution of the 23 study ponds during the 2010 field seasons with respect to the 17 limnological variables measured during June, July and September. The sample scores (n=69) for all three sampling periods with respect to17 limnological variables are presented.

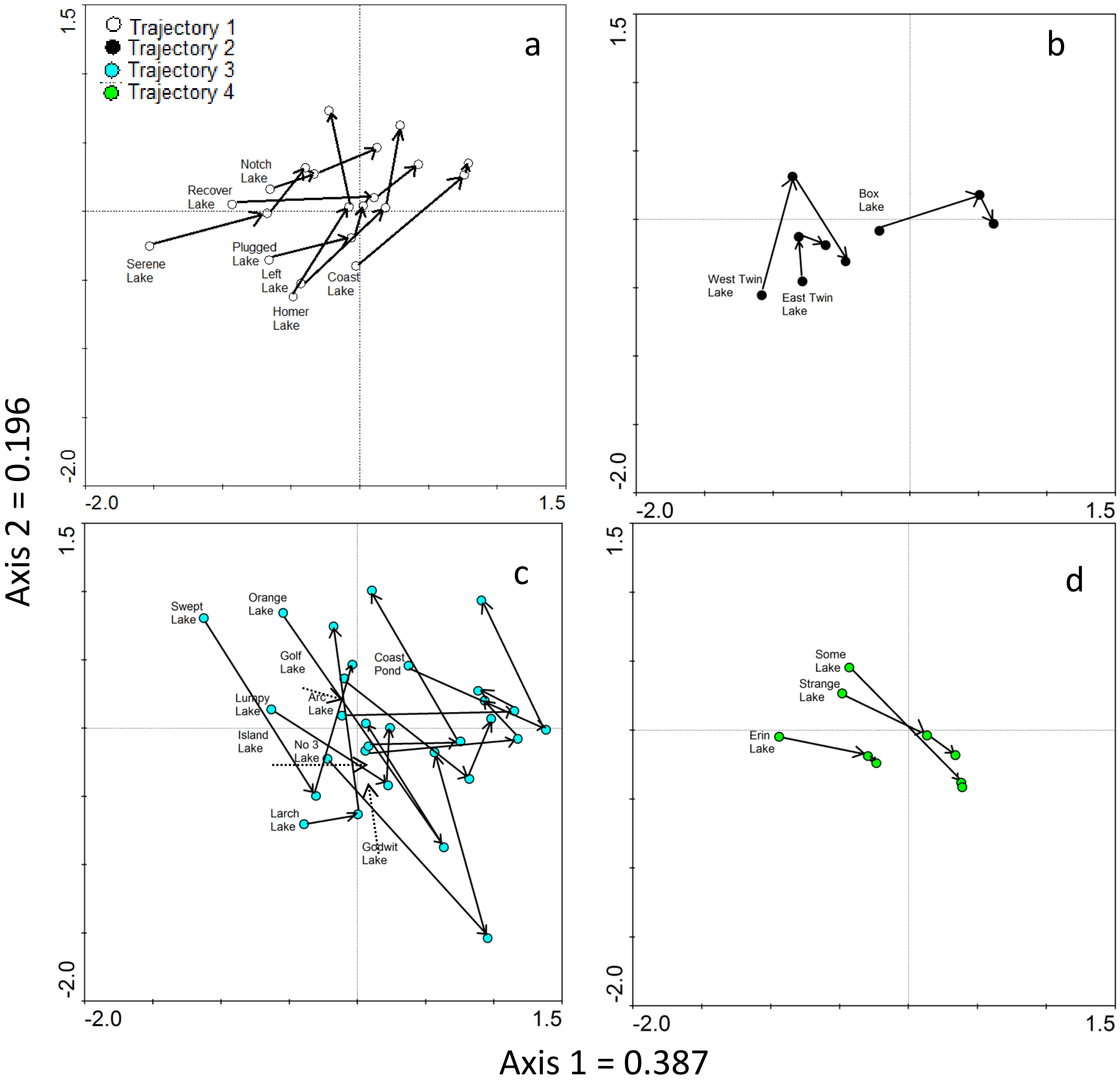

Figure 4-6 Principal component analysis (PCA) ordination diagram illustrating the evolution of the 23 study ponds during the 2010 field seasons with respect to the 17 limnological variables measured during June, July and September. The SWCT of the ponds defined as trajectory 1 (n=7) ponds and trajectory 2 (n=3) ponds are presented in the top left and right panels, respectively. Trajectory 3 (n=10) ponds and trajectory 4 (n=3) ponds are presented in the bottom left and right panels, respectively.

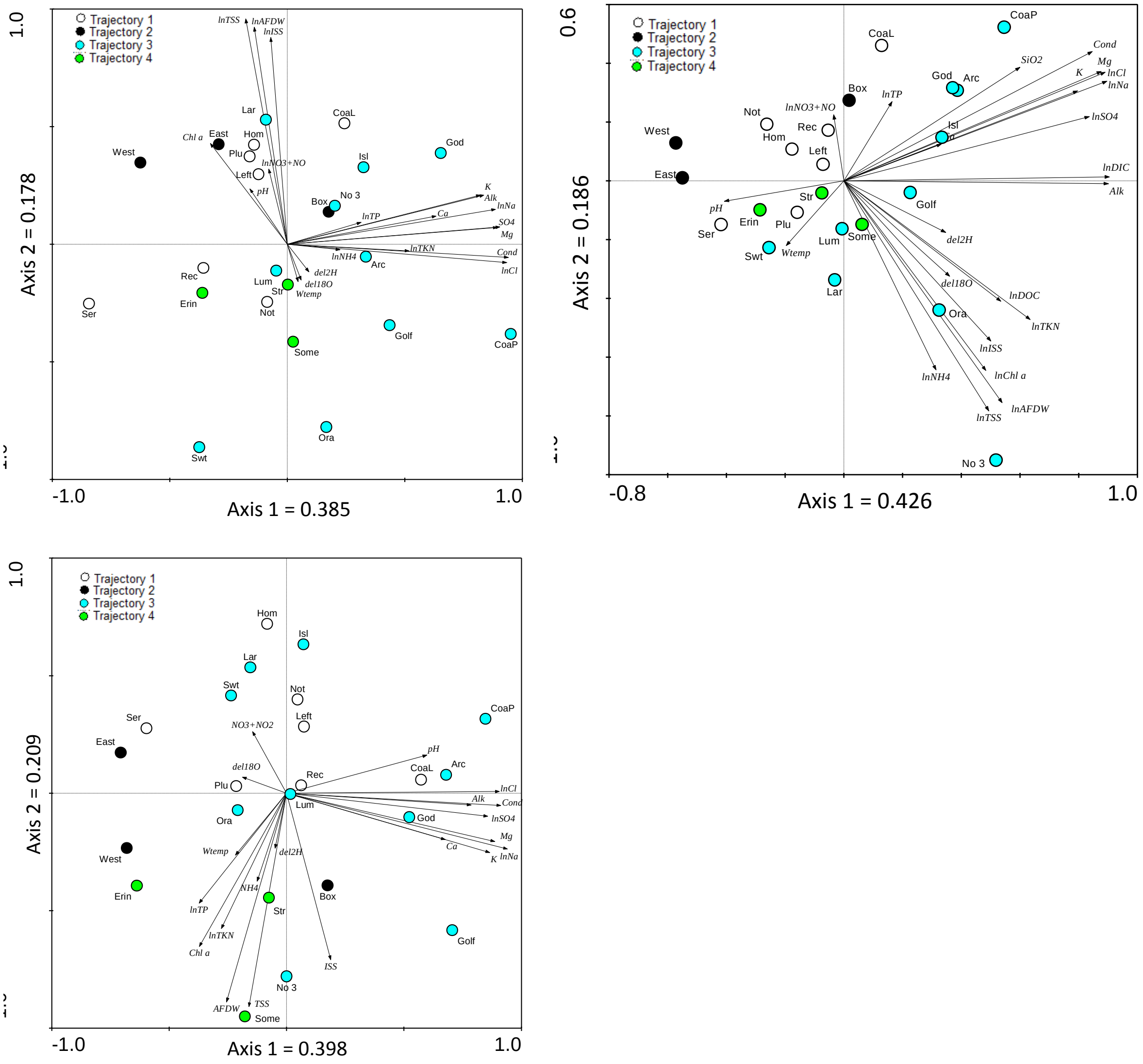

Figure 4-7 Principal Components Analysis of the limnological variables measured in June (top left panel), July (top right panel) and September (lower left panel) from the 23 study ponds coded based on the SWCT identified during the thaw-season in 2010.

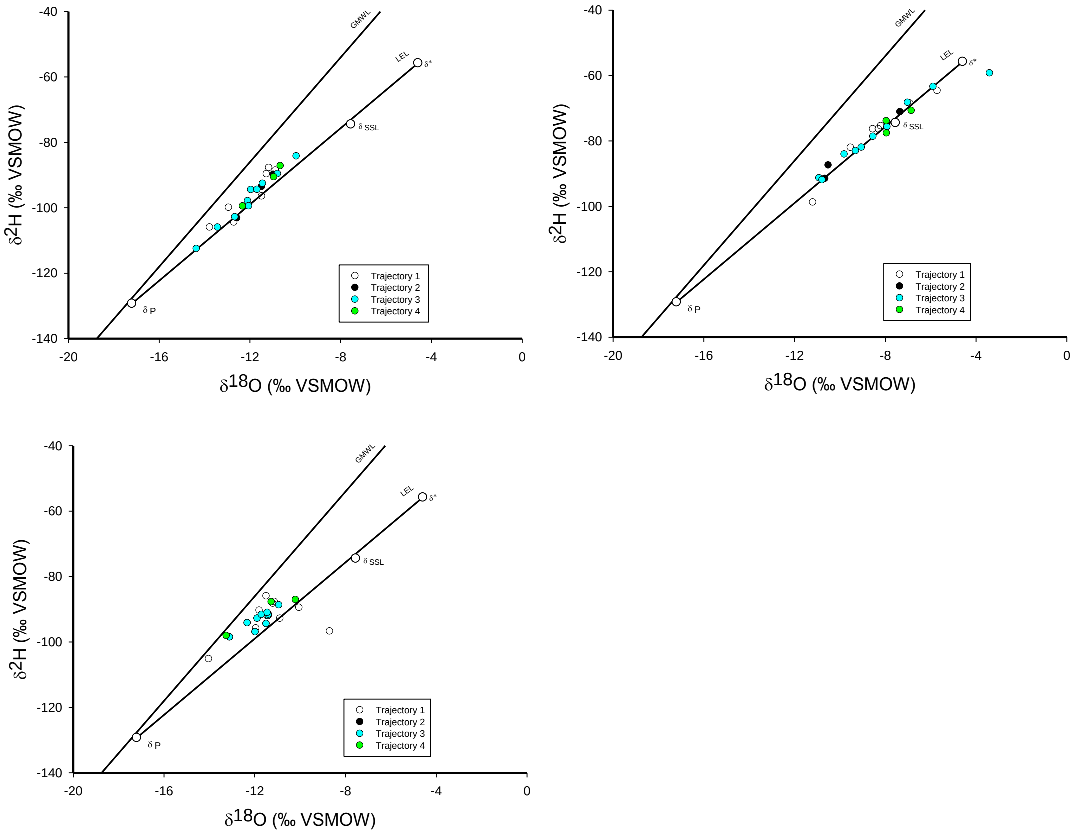

Figure 4-8 Water isotope data from June (top left panel), July (top right panel) and September (lower left panel) plotted on the isotopic framework developed for Churchill, MB with the 23 study ponds coded based on SWCT identified from the limnological analysis.

Table 4.3 Total phosphorus, total nitrogen and chl-*a* concentrations used in the general trophic classification of lakes and reservoirs. From Wetzel (2001).

Parameter		Oligotrophic	Mesotrophic	Eutrophic	Hypereutrophic
Total phosphorus	Mean	8.0	26.7	84.4	
(ug/L)	Range	3.0 - 17.7	10.9 – 95.6	16.0 – 386	750 - 1200
Total nitrogen	Mean	661	753	1875	
(ug/L)	Range	307 – 1630	361 - 1387	393 - 6100	
Chl-*a* (phytoplankton)	Mean	1.7	4.7	14.3	
(ug/L)	Range	0.3 – 4.5	3.0 – 11.0	3.0 – 78.0	100.0 – 150.0

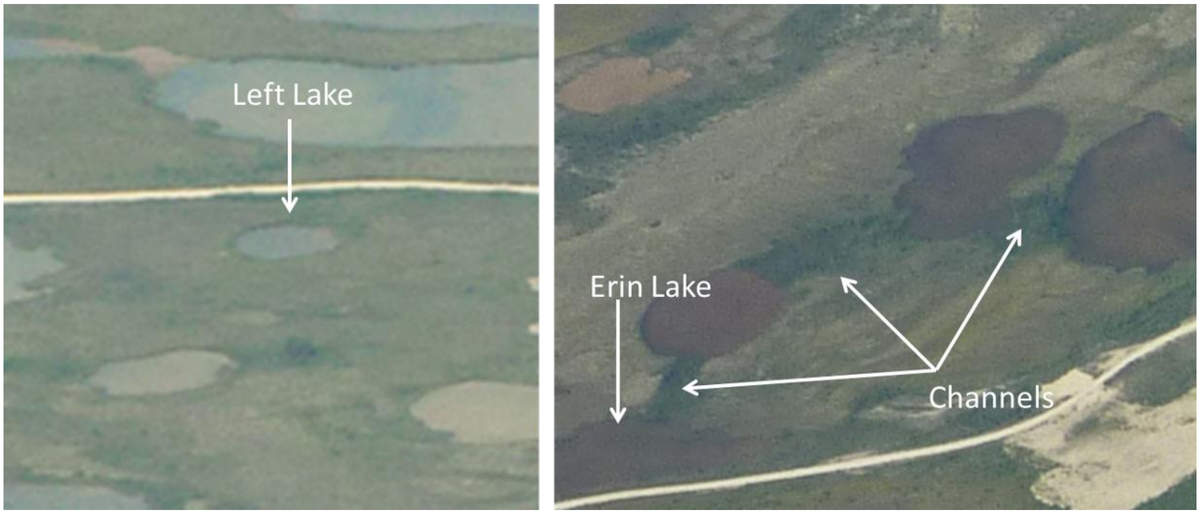

Figure 4-9 Aerial photo of the study ponds in July 2010. The left panel illustrates Left Lake which is hydrologically isolated throughout the thaw-season as illustrated by no apparent inflow or outflow. The right panel shows three ponds which are hydrologically connected as indicated by the dark green vegetation areas between the ponds. The timing of these connections was suspected to differentiate between ponds in SWCT 2 and 3.

Chapter 5

Characterizing the relationship between hydrolimnological conditions in shallow tundra ponds in the HBL and sedimentary diatom and pigment assemblages

The purpose of this chapter is to assess relations between biological communities (diatom algae, photosynthetic pigments) in recently-deposited surficial sediments (upper 1 cm) and hydrolimnological conditions of the 23 study ponds, which is an important step to guide interpretations of past hydro-ecological changes contained in pond sediment cores.

5.1 Relationships between surface-sediment diatoms and limnological conditions

Enumeration of diatoms found in the surface sediments of the 23 study ponds identified 67 different taxa from 31 different genera that met the data screening requirements defined in section 3.2.3 (Table 5.1). Epiphytic diatom taxa, *Denticula kuetzingii* (Kützing) and *Nitzschia amphibia* (Grunow), dominated in most of the study ponds and are closely associated with cyanobacteria-dominated algal mats that provide abundant benthic habitat in many of the ponds. Because these two species are morphologically similar and almost indistinguishable in girdle-band view, they were grouped as a ‘complex’ to ensure consistent taxonomic identification. Ponds not dominated by *D. kuetzingii* complex were found to have higher abundances of *Staurosirella pinnata* ((Ehrenberg) Williams & Round) or *Achnanthidium rivulare* (Potapova & Ponader). Planktonic diatom taxa were almost entirely absent from surface sediments of the 23 study ponds.

Detrended Correspondence Analysis (DCA) was used to examine the variability in diatom community composition among the study ponds (Figure 5.1). Eigenvalues of the first and second axis were 0.217 and 0.115 respectively, and explained 33.2% of the variation in diatom community composition among ponds. DCA identified three groups among the 23 study ponds differing in surface-sediment diatom community composition. The majority of ponds (identified with a red ellipse in Fig. 5.1; Group 1) were characterized by high relative abundance of *D. kuetzingii* complex along with lower abundances of epiphytic taxa indicative of mesotrophic conditions (e.g., *Encyonopsis microcephala* ((Grunow) Krammer) (Lim et al., 2007). Two ponds comprise a second group (green ellipse in Figure 5.1; Group 2), characterized by high relative abundances of *A. rivulare* and *Nitzschia fonticola* (Grunow). *N. fonticola* has been associated with small ponds that have high conductivity and high pH values (Oppenheim and Greenwood, 1990). It has been reported that *A. rivulare* has an affinity for low calcium and high chloride concentrations (Ponader and Potapova, 2007). However, a manipulative experiment indicated that *A. rivulare* could thrive in waters of a wide range of ionic composition indicating that it may be indicative of high nutrient levels (Cohen, 2010). Diatom assemblages in a third group of three ponds (identified by a blue ellipse in Fig. 5.1; Group 3) were characterized by higher relative abundance of *S. pinnata.* This taxon has been reported in surveys of other northern Canadian lakes as an alkalphilic benthic species associated with fine mineral grain substrates and low-light conditions (Rühland et al., 2003a, 2003b). No 3 Lake was left unclassified as it plotted alone in the ordination diagram characterized by higher relative abundance of *Achnanthidium minutissima* var. *minutissima* (Kützing), *Amphora veneta* (Kützing), *Hantzschia amphioxys* ((Ehrenberg) Grunow) and *Pinnularia microstauron* ((Ehrenberg) Cleve). *A. minutissima* var. *minutissima* is considered a cosmopolitan species and is found in a variety of ecological conditions in other high-latitude lake studies (Lim et al., 2007). Both *A. veneta* and *H. amphioxys* have been associated with lakes that have high proportions of sediment habitat and high Na^+ concentrations (Douglas and Smol, 1995). The ecological preferences of *P. microstauron* could not be determined from any North American high-latitude diatom training sets. No 3 Lake was one of the study ponds that was ephemeral as it was completely desiccated by mid-summer 2010 (e.g., July 2010; Figure 2.7), and its unique surface-sediment diatom assemblage may reflect community shifts that will be observed in ponds sensitive to drying if climate warming increases evaporative water losses. Interestingly, ponds of the three distinctive surface sediment diatom assemblage types did not correspond with either the ecozone the pond was situated in or the SWCT type. Diatom group 1 and 3 ponds were characterized by surficial diatom assemblages dominated by *D. kuetzingii* complex and differed from each other in the composition of less abundant taxa. Diatom group 2 ponds and No 3 Lake were very distinct from the other two groups as no *D. kuetzingii* complex diatoms were detected in the surficial sediment diatom assemblages using the methods incorporated in the study (Fig. 4-8).

Canonical Correspondence Analysis (CCA) was used to examine the relations between the diatom community composition and the environmental variables during July 2010. The July sampling period was selected for this analysis as conditions during this time were hypothesized to be most representative of limnological conditions during the thaw-season as this sampling period was less affected by the influence of snowmelt than June's sampling and less affected by large rainfall events that occurred later in the thaw-season than September's sampling. July's limnological conditions also allowed for the assessment of the effect that varying degrees of evaporative drawdown among the ponds had on the diatom community composition. Using manual forward selection in CCA, five limnological variables were identified that accounted for independent and significant amounts of the variation in diatom community composition among the ponds (Figure 5.2). The eigenvalue of the first ordination axis was 0.184 and explained 31.3 % of the variation in the species-environment relationship. lnChl-*a* and lnTKN were strongly and positively correlated to the first ordination axis (intraset correlations = 0.6173 and 0.7930, respectively). The eigenvalue for the second ordination axis was 0.167 and explained 28.5% of the variation in the species-environment relationship. pH was strongly and negatively correlated to axis 2 (intraset correlations = -0.6828). Alkalinity and $\ln NH_4^+$ contributed nearly equal amounts to both ordination axes (intraset correlations ax_1 = 0.7140; 0.5382, ax_2 = 0.7042; -0.4506, respectively). The ordination diagram was coded based on the diatom community groups identified in the DCA of the diatom data (Figure 5.1) and did not result in a similar pattern of sample scores as the DCA of the diatom data alone, suggesting the limnological variables selected did not characterize the species-environment relationships well. Interestingly, the first ordination axis separated the ponds distinguished by higher abundances of *A. rivulare* and *N. fonticola* in the previous DCA (Group 2; Figure 5.1) from the rest of the ponds based on lower lnChl-*a*, TKN, NH_4^+ and alkalinity. However, the sample scores from ponds in group 1 and group 3 are intermixed, which indicated that the variables that best account for the variation in surficial sediment diatom assemblages between these groups had not been included in the CCA ordination.

5.2 Relationships between surface-sediment pigments and limnological conditions

Principal Components Analysis was used to assess the distribution of surface sediment pigments among the study ponds (Figure 5.3). PCA axis 1 captured 64.1% of the variation in the 12 active pigment variables and indicated that the ponds differed along a gradient of pigment concentration. Almost all pigments, with the exception of chl-b, chl-*a*, chl-c_2 and diadinoxanthin, were strongly and negatively correlated with PCA axis 1. Ponds with high pigment concentrations were positioned to the left along axis 1. Vectors for these pigments were orthogonal to the vector for chl-b indicating that they were uncorrelated. PCA axis 2 captured 15.8% of the variation and separated the ponds mainly due to differences in chl-b concentration, which was strongly and negatively correlated to axis 2. Concentrations of chl-*a*, chl-c_2 and diadinoxanthin explained equal amounts of the variation along axis 1 and axis 2. PCA of surface sediment pigment assemblages indicated that the composition of pigment assemblages did not differ systematically among the study ponds in the three ecozones, as ponds from each ecozone did not form distinct clusters in the ordination diagram (top left panel; Figure 5.3). However, there was a tendency for ponds in the tundra ecozone to plot to the left along axis 1 associated with higher concentrations of sedimentary pigments than ponds in the other ecozones. Relationships between the SWCT pond categories and the variation in pigment community composition among ponds were also explored (bottom left panel; Figure 5.3). No trends were observed in the ordination diagram when the ponds were coded based on SWCT as sample scores for the ponds from all four SWCT categories overlapped considerably.

Redundancy Analysis (RDA) of the pigment-environmental relationships using seven representative limnological variables (conductivity, alkalinity, lnTKN, lnTP, $\ln NO_3^-+NO_2^-$, pH, lnTSS) indicated that the study ponds spanned a gradient along the first axis of increasing pigment concentration (Figure 4.11). The first RDA axis was significant ($p \leq 0.05$) and captured 76.6% of the variation. The variable lnTSS was weakly and positively correlated with RDA axis, while $\ln NO_3^- +NO_2^-$ was strongly and negatively correlated with axis 1 (intraset correlation = 0.2101 and -0.4393, respectively). The second axis separated the ponds mainly along a gradient of chl-b concentration, which captured 12.6% of the species-environment relationships. Pond water pH was strongly and negatively correlated with RDA axis 2 (intraset correlations = -0.5527), while conductivity, alkalinity, lnTP and lnTKN explained almost equal amounts of the variation observed in both axes. Indeed, the relative orientation of the pigment vectors is comparable between the PCA and RDA ordinations, suggesting the 7 limnological variables included in the RDA captured most of the variance in pigment abundance and composition among the ponds. Coding of the ponds based on ecozone did not reveal distinct separation or grouping of sample scores according to the dominant vegetation.

5.3 Discussion

Diatom assemblages in most of the ponds in the CWMA were dominated by epiphytic taxa. This was reflective of the large amount of epiphytic habitat available on the thick cyanobacterial mats that line the bottom of these shallow ponds. Planktonic taxa were in very low relative abundance in the surficial pond sediments, which is similar to findings in other northern ponds and a consequence of shallow depths and a short thaw-season (Michelutti et al., 2003). *Denticula kuetzingii*, the most abundant epiphytic taxon which was found in high abundances (>30%) in all but five ponds, has been closely associated with ponds that have lush catchment areas consisting of mosses and grasses (Lim et al., 2008). However, the large amount of habitat provided by the cyanobacteria mats on many of the pond bottoms and the low abundance of macrophytes suggests that the abundant *D. kuetzingii* in these ponds may be reflective of the colonization of this rich habitat. The main difference between the ponds categorized as group 1(red ellipse in figure 5.1) and group 2 (green ellipse in Figure 5.1) was the absence of *D. kuetzingii* in the surficial sediments of ponds in group 2. Most other taxa were present but at higher abundances in group 2, which was as expected given that diatom relative abundance is a closed data set. As per communication with LeeAnn Fishback (Science Coordinator at the Churchill Northern Studies Centre (CNSC)), it was determined that the difference between group 1 and group 2 ponds may be entirely related to sampling locations. Both West Twin Lake and Serene Lake (group 2) were sampled on the eastern shore of the ponds, which due to wind direction, has thicker ice cover and an increased scouring of the pond bottom that effectively removes the algal mat. East Twin Lake, which is just across the road from West Twin Lake and shares similar physical and limnological characteristics, was sampled on the western shore where the impact of ice is less severe. Here, the algal mats were intact and *D. kuetzingii* was present in the surface sediment samples. Resampling of both West Twin Lake and Serene Lake in a different location would be required to confirm whether the diatom community composition in these two ponds was in fact different than group 1 ponds or if the difference was an artifact of sampling techniques.

Ponds characterized as group 3 differed from group 1 mainly by the presence of *S. pinnata* in the surface sediments. *D. kuetzingii* were still present in the surficial pond sediments, which suggests that algal mat habitat is present in these ponds. However, the presence of *S. pinnata* indicates that these ponds also provide suitable microhabitat, consisting of fine mineral grains, for this taxon. As per communication with LeeAnn Fishback (Science Coordinator, CNSC), all three ponds in group 3 (Godwit, Golf and Orange) have areas of mineral sediment in the ponds that would provide suitable habitat for this taxon. As these species are still present in the surface sediments along with more complex epiphytic diatom communities, this may be an indication that these ponds are going through a period of transition. Often floristic changes in sedimentary diatoms from small adnate taxa (e.g., benthic *Fragilaria* taxa) to more complex epiphytic taxa (e.g., *Cymbella* taxa*)* are interpreted as a climate warming signal as an increased length of growing season results in additional substrates and more complex diatom communities (Douglas et al., 2004).

The diatom community composition of No 3 Lake was substantially different from the other groups of ponds. This is likely due to the ephemeral nature of the pond during the thaw-season. Under conditions of complete desiccation, it would be expected that more aerophilic and terrestrial taxa would be observed in the sediment record (Douglas et al., 2004). Interestingly, *H. amphioxys* and *P. microstauron* were found in the surface sediments of No 3 Lake which have been described as aerophilic and terrestrial in other regions. *H. amphioxys* has been found in samples of North African wind-blown dust collected in southern France and in soil samples analyzed in northeastern Greenland (Harper, 1999; van Kerekvoorde, 2000). *P. microstauron* was also found in the soil samples collected in Greenland but at a much lower abundance than was observed for *H. amphioxys.* Also of interest, *H. amphioxys* was found in the surficial sediments analyzed from Birch Lake, which was another pond originally included in the study that became ephemeral in the 2010 field season but was removed from the numerical analyses due to an incomplete set of environmental data. For this reason, *H. amphioxys* may be a reliable indicator species to identify time periods in a pond's sediment record that indicate desiccation.

Unfortunately, the environmental variables that were included in the CCA did not sufficiently explain the distribution of the diatoms within the study ponds. However, given the information provided concerning habitat specificity in diatom training sets, there appears to be a strong relationship between the diatom variation observed in the study ponds and habitat availability. Macrae (1998) identified diatom assemblages similar to groups 1 and 2 in a previous survey of the ponds in the CWMA. Numerical analysis in that study revealed that the environmental variables measured only explained a small fraction of the variation in diatom species distributions among the ponds. Macrae (1998) also suggested that microhabitat was influencing the ponds diatom assemblages and recommended a more detailed analysis of the ponds' substrates to test this hypothesis.

Analysis of photosynthetic pigment concentrations from the 20 study ponds revealed that there was a trend for the ponds located in the tundra ecozone to have the highest concentrations of most pigments. These ponds plotted to the left in the RDA ordination diagram associated with higher alkalinity, conductivity and NO_3^-+NO_2^- concentrations (Figure 5.4). However, none of these variables proved to be statistically significant in the forward selection of environmental variables. In the geochemical analysis of the pond sediment record from three of the study ponds, Light (2011) reported that at the beginning of the 20th century the ponds were severely nitrogen limited as $\delta^{15}N$ values were very close to zero which suggested that nitrogen-fixation by cyanobacteria was the main source of nitrogen stored in the pond's sediment. In the latter half of the 20th century, $\delta^{15}N$ values in the three ponds deviated away from zero, which was interpreted as a signal that the nutrient status in the ponds was changing and dissolved nitrogen was becoming more available. Although the geochemical analysis indicated that nitrogen availability may be changing in more recent times, the dominance of nitrogen-fixing cyanobacteria throughout the fossil pigment record of two of these ponds (discussed in Chapter 6) supports the interpretation that nitrogen is still a limiting nutrient in the ponds. However, the severity of the limitation has changed since the beginning of the 20th century. This interpretation is further supported by the fact that aphanizophyll is the dominant photosynthetic pigment in the surface sediments in the majority of the study ponds (Table B2, Appendix B). In the RDA, pigment concentrations were positively correlated with NO_3^-+NO_2^- concentrations, which is consistent with fossil pigment data that suggests that productivity within the ponds is tightly coupled to nitrogen availability. Increasing NO_3^-+NO_2^- concentration presents a plausible explanation for the gradient in pigment concentrations observed in the RDA among the ponds given the fact that the ponds are still N-limited even with recent changes in nitrogen availability observed in the geochemical stratigraphic record. However, the gradient in NO_3^-+NO_2^- concentrations among the study ponds in July 2010 was very small (0.015 mg/L – 0.025 mg/L) and the higher productivity inferred by high pigment concentrations in tundra ponds may also reflect differences in duration of the growing season among ponds, which can be attributed to the location and the size of these ponds. The study ponds in the tundra ecozone tend to be smaller and shallower than the ponds in the other ecozones. Shallow ponds heat faster in the spring due to smaller volume and experience ice-off conditions much earlier in the spring (Duguay et al., 2003). Ponds in the tundra ecozone also typically have little or no snow cover due to exposure to severe winds while ponds located in the forest-tundra transition zone typically are snow covered with a few bare ice patches and ponds in the open-forest zone are completely snow covered (Duguay et al., 2003). Increased snow cover translates into higher surface albedos, which results in a decreased amount of absorbed solar radiation at the onset of the spring thaw (Rouse, 2000). In the open-forest and forest-tundra transition zones where snow accumulation is greater, this could result in a slightly delayed thaw-season. Also in areas where snow accumulates over ice, snow-ice forms due to bottom accretion, which has different optical and reflective properties than clear ice (Duguay et al., 2003). As clear ice allows for greater light penetration, photosynthesis can begin before ice-off, which may also partially explain the difference in primary production inferred by pigment concentrations between the ponds in the tundra ecozone and the other two ecozones.

The analysis of diatom and pigment community composition from the study ponds has provided a valuable tool that can be used to interpret paleoecological conditions archived in the sediment record in the following chapter. Shifts in diatom community composition from small benthic taxa (e.g. *Fragilaria* spp.) to more complex diatom communities (e.g. *Denticula* spp., *Cymbella* spp.) in the pond sediments can be used to infer changes in microhabitat from episammic substrates to cyanobacterial mats that likely reflects an increased length in the growing season. Increased concentrations of the photosynthetic pigment, aphanizophyll, can be used to support this interpretation as a larger proportion of cyanobacterial mat habitat will result in higher aphanizophyll concentrations. The presence of aerophilic and terrestrial diatom taxa (e.g. *H. amphioxys; P. microstauron*) in the sediment record can be used to detect environmental conditions that result in ponds becoming ephemeral. Increased pigment concentrations in pond sediments can be used to infer changes in productivity that are the result of a change in nutrient status (nitrogen availability) as long as the changes in productivity coincide with shifts in the δ^{15}N values in the geochemical record. If pigment concentrations increase with no coinciding change in the δ^{15}N values in the geochemical record then this may better reflect an increased productivity in response to a longer growing season.

Table 5.1 Diatom taxa (n=67) found in more than 1 study pond (n=23) at ≥1% abundance and whether the taxon was present in the Left Lake and Erin Lake sediment cores.

#	Taxon Name	Left Lake	Erin Lake
1	*Achnanthes petersenii* Hustedt	N	N
2	*Achnanthidium deflexum* Reimer	N	N
3	*Achnanthidium minutissima* var. *jackii* Rabhenhorst	N	N
4	*Achnanthidium minutissima* var. *minutissima* Kutzing	Y	Y
5	*Achnanthidium minutissima* var. *saprophila* Kobaysi & Mayama	N	N
6	*Achnanthidium rivulare* Potapova & Ponader	N	Y
7	*Amphora libyca* Ehrenberg	N	N
8	*Amphora veneta* Kützing	N	N
9	*Aneumastus tusculus* (Ehrenberg) Mann & Stickle	Y	Y
10	*Anomoeoneis brachysira* (Brebisson) Grunow	N	N
11	*Brachysira microcephala* (Grunow) Compere	N	N
12	*Caloneis silicula* (Ehrenberg) Cleve	N	N
13	*Cocconeis placentula* var. *euglypta* Ehrenberg	N	N
14	*Cyclotella antigua* Smith	N	N
15	*Cyclotella distinguenda* Hustedt	N	Y
16	*Cymbella affinis* Kützing	N	N
17	*Cymbella alpina* Grunow	N	N
18	*Cymbella angustata* Smith	Y	Y
19	*Cymbella cymbiformis* Agardh	N	N
20	*Cymbella descripta* (Hustedt) Krammer & Lange-Bertalot	N	N
21	*Cymbella delicatula* Kützing	N	N
22	*Cymbella incerta* (Grunow) Cleve	N	N
23	*Cymbella neocistula* Krammer	N	N
24	*Cymbella norvegica* var. *norvegica* Grunow	N	N
25	*Cymbella naviculacea* Grunow	N	N
26	*Cymbella subaequalis* Grunow	Y	Y
27	*Cymbopleura lapponica* Grunow ex Cleve	N	N

#	Taxon name	Left Lake	Erin Lake
28	*Diatoma tenuis* Agardh	N	N
29	*Encyonema minutum* (Hilse) Mann	Y	Y
30	*Encyonema silesiacum* (Bleish) Mann	Y	N
31	*Encyonopsis cesatii* (Rabenhorst) Krammer	Y	Y
32	*Encyonopsis microcephala* (Grunow) Krammer	Y	Y
33	*Eunotia praerupta* Ehrenberg	N	N
34	*Epithemia adnata* (Kützing) Brebisson	N	N
35	*Eucocconeis flexella* (Kützing) Meister	N	N
36	*Eucocconeis laevis* (Ostrup) Lange-Bertalot	N	N
37	*Fragilaria capucina* var. *vaucheriae* (Kutzing) Lange-Bertalot	Y	Y
38	*Fragilaria nanana* Lange-Bertalot	N	N
39	*Fragilaria tenera* (Smith) Lange-Bertalot	Y	N
40	*Gomphonema angustum* Agardh	N	N
41	*Hantzschia amphioxys* (Ehrenberg) Grunow	N	N
42	*Kobayasiella jaagii* (Meister) Lange-Bertalot	N	N
43	*Denticula kuetzingii* complex Grunow	Y	Y
44	*Mastogloia elliptica* (Agardh) Cleve	N	N
45	*Mastogloia grevillei* Smith	N	N
46	*Mastogloia smithii* Smith	N	N
47	*Navicula cryptocephala* Kützing	Y	Y
48	*Navicula cryptotenella* Lange-Bertalot	Y	N
49	*Navicula halophila* (Grunow) Cleve	N	N
50	*Navicula radiosa* Kützing	N	Y
51	*Navicula vulpina* Kützing	Y	Y
52	*Navicula wildii* Lange-Bertalot	Y	Y
53	*Neidium ampliatum* (Ehrenberg) Krammer	N	N
54	*Nitzschia fonticola* Grunow	N	N
55	*Nitzschia frustulum* (Kützing) Grunow	Y	Y
56	*Nitzschia inconspicua* Grunow	N	N

#	Taxon name	Left Lake	Erin Lake
57	*Nitzschia intermedia* Hantzsch	N	N
58	*Nitzschia palea* (Kützing) W. Smith	Y	Y
59	*Nitzschia perminuta* (Grunow) M. Peragallo	N	Y
60	*Nitzschia pura* Hustedt	Y	Y
61	*Nitzschia radicula* Hustedt	N	N
62	*Pinnularia microstauron* (Ehrenberg) Cleve	N	Y
63	*Pinnularia viridis* (Nitzsch) Ehrenberg	N	N
64	*Psammothidium bioretii* (Germain) Bukhtiyarova & Round	N	N
65	*Rhopalodia gibba* (Ehrenberg) O. Muller	Y	N
66	*Sellaphora pupula* (Kützing) Mereschkovsky	Y	Y
67	*Staurosirella pinnata* (Ehrenberg) Williams & Round	Y	Y

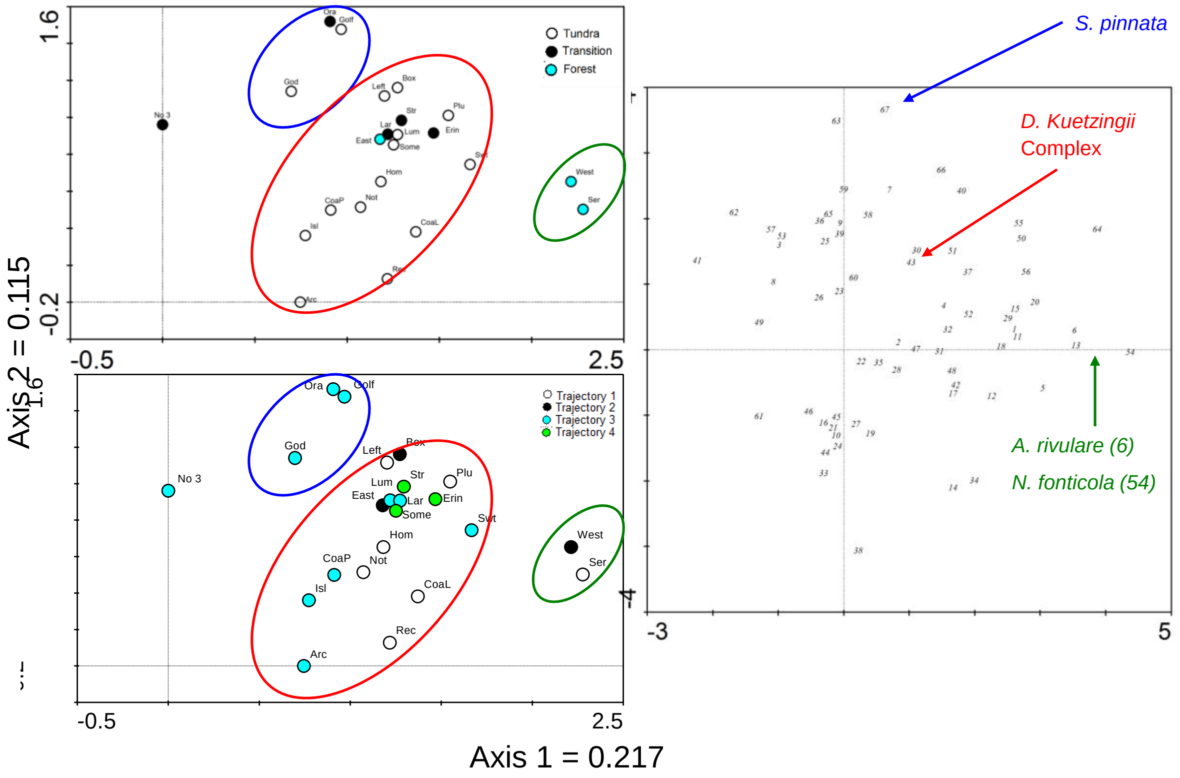

Figure 5-1 Ordination diagram displaying the results of the Detrended Correspondence Analysis (DCA) of surface sediment diatom community composition from the 23 study ponds in the Churchill Wildlife Management Area. Sample scores are located in the left panels (coded by vegetation zone top panel, SWCT bottom panel) while species scores are located in the right panel (note, corresponding taxon names for the numbers presented in the figure are provided in Table 5.1).

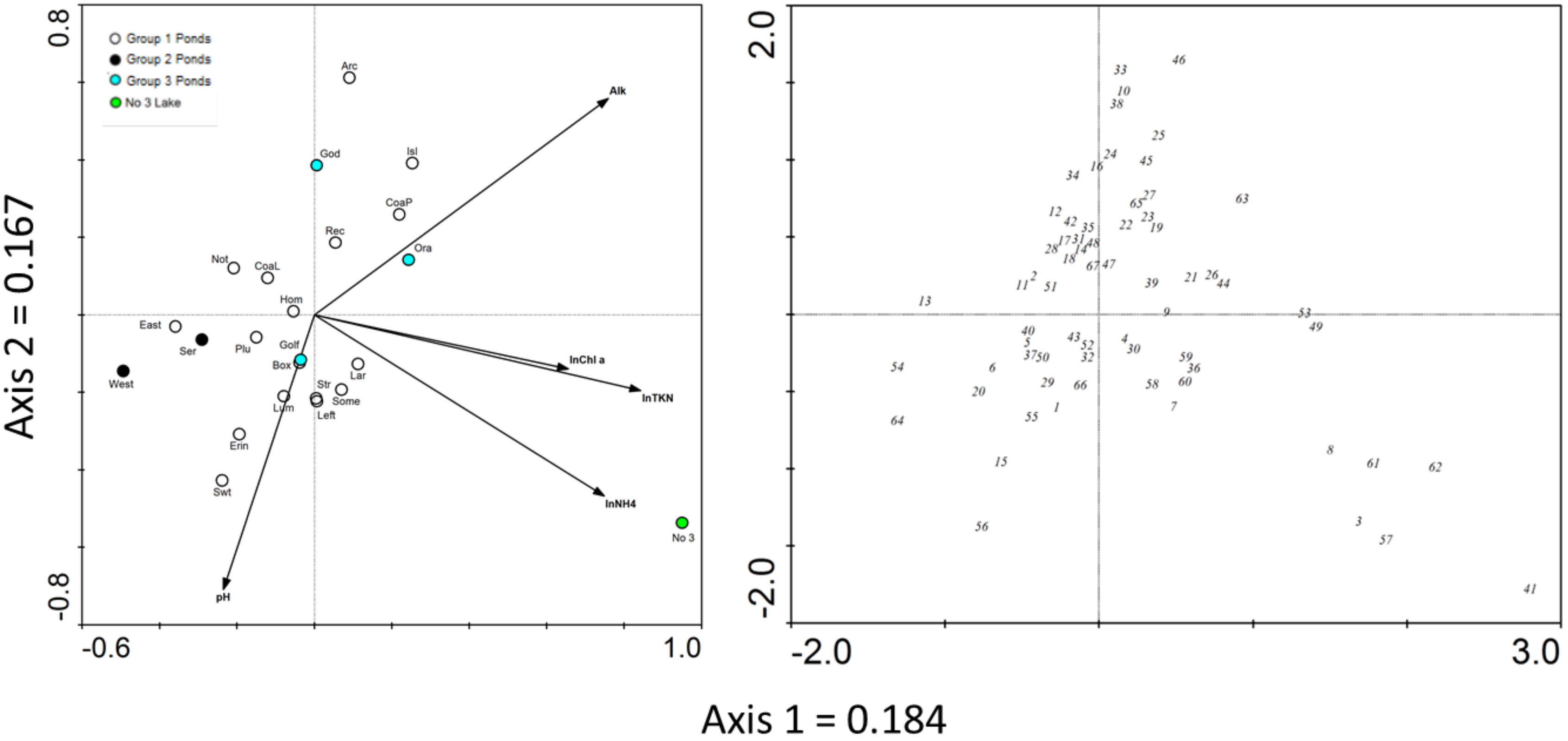

Figure 5-2 Canonical Correspondence Analysis (CCA) of the forward-selected environmental variables measure in July 2010 and sedimentary diatom taxa (n=67) from the 23 study ponds located in the Churchill Wildlife Management Area. Sample scores are displayed in the left ordination panel and species scores are displayed in the right panel. Numbers presented for the species scores correspond to taxon names in Table 5.1. Ponds are coded based on the groups identified in the DCA ordination.

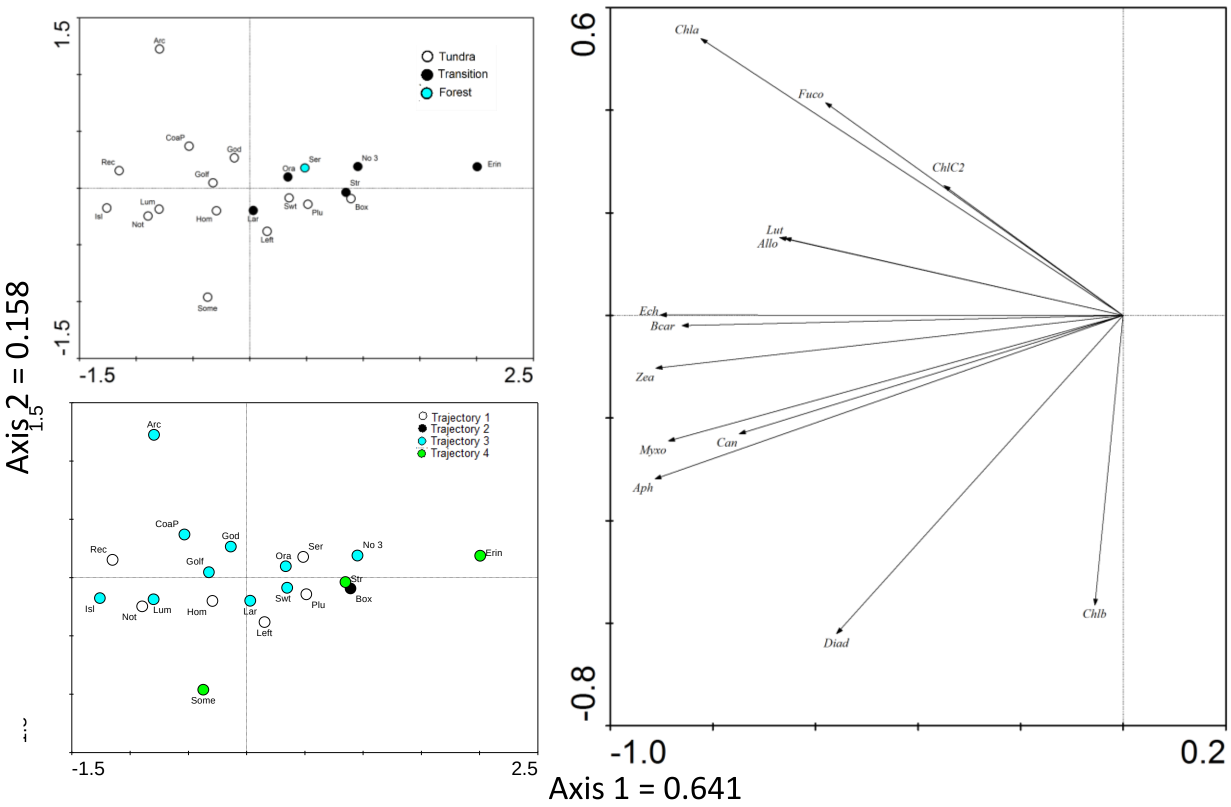

Figure 5-3 Principal components analysis (PCA) of surface sediment photosynthetic pigment community composition from the 20 study ponds located in the Churchill Wildlife Management Area. Sample scores are located in the left panels (coded by vegetation zone top panel, SWCT bottom panel) while species scores are located in the right panel. (Allo-alloxanthin, Aph-aphanizophyll, Bcar-β-carotene, Can-canthaxanthin, Chla-chlorophyll-*a*, Chlb-chlorophyll-*b*, ChlC2-chlorophyll-c_2, Diad-diadinoxanthin, Ech-echinenone, Fuco-fucoxanthin, Lut-lutein, Myxo-myxoxanthophyll, Zea-zeaxanthin).

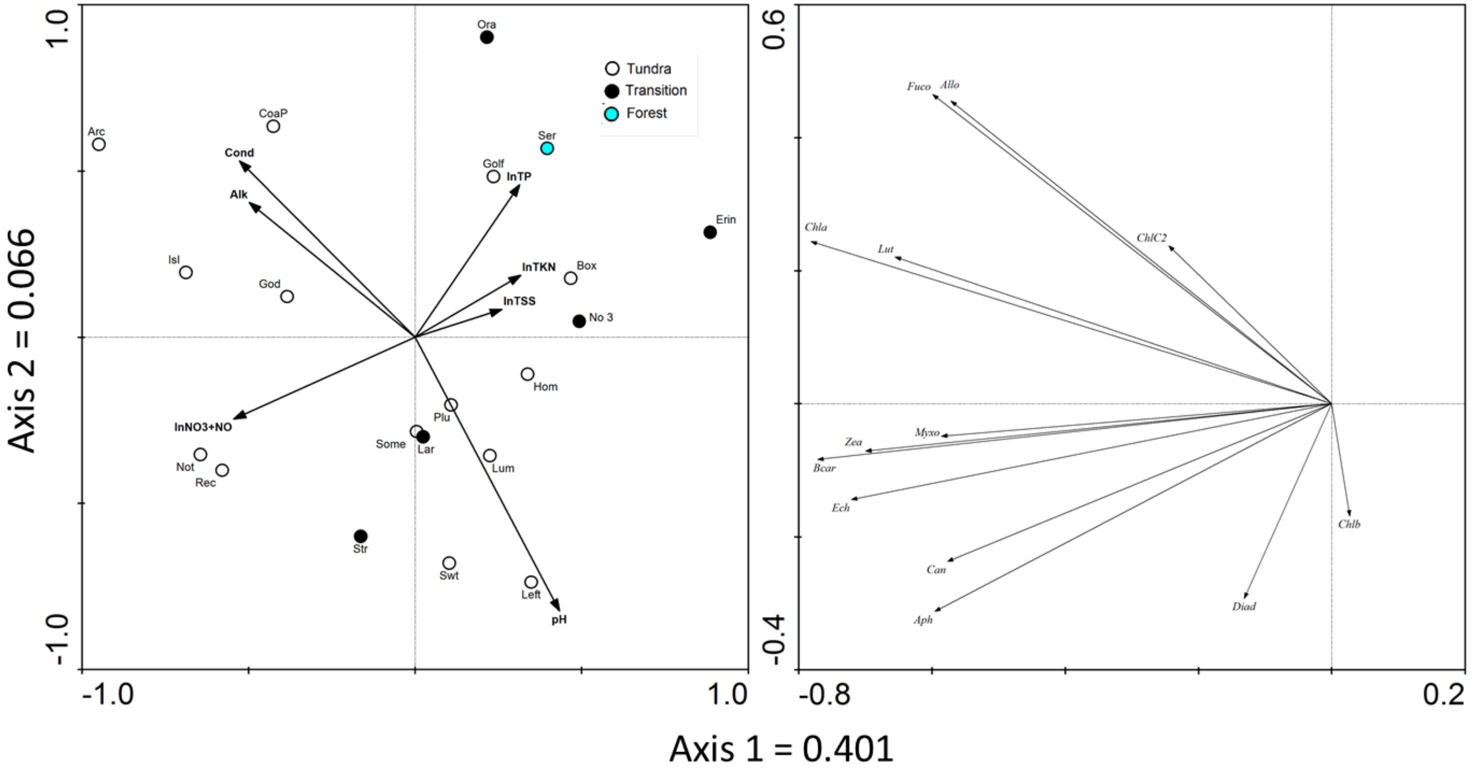

Figure 5-4 Redundancy Analysis (RDA) ordination diagram of the photosynthetic pigments and the environmental variables from July 2010 selected to represent limnological conditions in the 20 study ponds from the Churchill Wildlife Management Area. Samples scores are presented in the left panel and species scores are in the right panel. The ponds are coded based on ecozone location. (Allo-alloxanthin, Aph-aphanizophyll, Bcar-β-carotene, Can-canthaxanthin, Chla-chlorophyll-*a*, Chlb-chlorophyll-*b*, ChlC2-chlorophyll-c_2, Diad-diadinoxanthin, Ech-echinenone, Fuco-fucoxanthin, Lut-lutein, Myxo-myxoxanthophyll, Zea-zeaxanthin).

Chapter 6

Reconstructing past hydrolimnological changes from fossil diatom and pigment sedimentary assemblages

The purpose of this chapter is to use the relations between surficial sediment assemblages of diatoms and photosynthetic pigments and hydrolimnological conditions in the study ponds to interpret past changes in environmental conditions of two of the study ponds (Left Lake and Erin Lake) based on analyses of diatom assemblages and fossil pigments archived in the sediment records.

6.1 Paleolimnological analyses for Left Lake

6.1.1 Sediment core visual stratigraphy

Visual observations of the sediment cores were provided by field notes reported in Light (2011). Left Lake core GC-1, used for geochemical, diatom and pigment analyses, was 39.5 cm long. Core GC-2, used for ^{210}Pb dating, was 39.0 cm long (Figure 6.1). Both cores were collected on the same date and in close proximity to each other, and therefore share similar stratigraphic sequences. Sediments from the base of the cores (LL GC-1 = 39.5 cm; LL GC-2 = 39.0) to 34 cm consisted of black, crumbly sediment. At 34 cm, there was a 1-cm interval of dark brown sediment. From 33 cm to 26.5 cm, the pond sediments were a homogenous brownish-orange colour. Sediments from ~26.5 cm to 6.5 cm were lighter coloured and had distinct black-brown bands at ~26.5 cm and ~13.5 cm. The dark black-brown band at 26.5 cm was ~10 degrees from horizontal in GC-1. Sediments between ~6.5 cm to 0.0 cm consisted of a rusty-orange organic material.

6.1.2 Loss-on-ignition

Loss-on-ignition results illustrated that the two sediment cores from Left Lake shared similar profiles, which were highly organic (Figure 6.2). Average organic matter (OM) content was ~87% and was relatively stable, but with an interval of lower OM content (low 80s-%) in both cores between ~34-38 cm. Calcium carbonate ($CaCO_3$) content was low and variable in both sediment cores averaging ~3.5% (±1.7%, 1 std. dev.) over the entire profile.

6.1.3 Core chronology and sedimentation rate

The total ^{210}Pb activity in core GC-2 remained relatively stable in the the top ~3 cm of the core profile after which the activity declined exponentially until reaching background levels at ~ 21.5 cm (Figure 6.3). Near-constant total ^{210}Pb activity between 0 and 3 cm depth could be the result of sediment mixing or an increased sedimentation rate beginning ~1998. ^{210}Pb activity ranged from its highest value of 0.320 Bq g^{-1} at ~ 3.0 cm to 0.007 Bq g^{1}, which was the average ^{210}Pb activity measured below 20.0 cm. ^{214}Bi activity, which is normally used to infer background ^{210}Pb activity, proved unreliable due to values below minimum detection limits as a result of insufficient sample mass. Consequently, mean ^{214}Pb activity was used to estimate supported ^{210}Pb activity (0.005 Bq g^{-1}). Close agreement between mean ^{214}Pb activity (0.005 Bq g^{-1}) and average ^{210}Pb values below 20 cm (0.007 Bq g^{-1}) validated the use of this method of determining background from ^{214}Pb activity. Background depth was estimated to have been reached at ~21.5 cm when ^{210}Pb activity decreased and remained relatively constant with values below the mean ^{214}Pb activity used to infer background activity. The CRS model was used to estimate the chronology and sedimentation rates from the unsupported ^{210}Pb activity profile and then a linear extrapolation was used to estimate older ages, which provided a basal date of 1625. Radiocesium activity increased markedly above 11 cm. However, the lack of a well-defined peak suggested that ^{137}Cs may have been mobile in the sediments, as has been observed in sediments with high concentrations of organic matter (Brenner et al., 2004).

The dry mass sedimentation rate averaged 0.013 g cm^{-2} yr^{-1} from ~1860 – 1960 (Figure 6.4). The sedimentation rate increased during ~1966 to 1990 and then remained relatively constant at 0.017 g cm^{-2} yr^{-1} for a decade (1990 – 2000). Subsequently, it has been on an increasing trend since ~ 2001. An increased sedimentation rate in most recent years may represent an increase in productivity and/or may be the result of incomplete decomposition of the most recently deposited sediments.

Profiles of organic matter and $CaCO_3$ content in the sediment cores GC-1 and GC-2 from Left Lake were very similar (Figure 6.2) and therefore, the assumption was made that the chronology developed from core GC-2 could be directly applied to core GC-1.

6.1.4 Diatom stratigraphy

A total of 63 diatom taxa were identified in sediment samples from core GC-1 of Left Lake. Twenty-one of these taxa were also present at abundances ≥1% in at least two of the study ponds in which surface sediment samples were collected (Table 5.1). Indicator species and patterns identified in the analysis of the surface sediment diatoms in Chapter 5 were used to interpret the fossil diatom record. Training sets developed for lakes in the Northwest Territories by Lim et al. (2007) and Rühland and Smol (2002) were used to further assist the interpretation of environmental conditions based on the diatom assemblages identified in the sediment intervals, since direct gradient ordination of the surface sediment diatom assemblages and environmental variables identified that assemblages were most strongly influenced by microhabitat availability in the ponds.

Denticula kuetzingii complex dominated diatom assemblages throughout the core (Figure 6.5). This feature likely reflects the consistent and dominant epiphytic habitat provided for diatoms on and within cyanobacterial mats that were present throughout the record since the early 1600s.Two distinct zones were identified based on changes in percent abundance of other less common taxa in the sediment core. Sediments deposited prior to ~1820 (39.0 cm to 24.0 cm) were characterized by relatively higher abundances of *Staurosirella pinnata* and *Nitzschia frustulum. S. pinnata* declined below detection limits after ~1850. Assemblages deposited since ~1820 (~24.0 cm to 0 cm) were more diverse with modest abundances of five taxa (*Aneumastus tusculus* ((Ehrenberg) D.G. Mann & A.J. Stickle), *Cymbella angustata* ((Smith) Cleve), *Encyonopsis cesatii* ((Rabenhorst) Krammer), *E. microcephala and Navicula vulpina* (Kützing)). The sedimentary diatom assemblages indicated that prior to ~1820, Left Lake consisted of a very simple and species-poor environment that was characterized by dominance of taxa living in association with cyanobacterial mats or on mineral grains. After ~ 1820, richness increases slightly and includes some epiphytic taxa suggesting modest expansion of macrophyte and cyanobacterial mat habitats. These epiphytic taxa are also associated with mesotrophic conditions, which may be reflective of a slight change in nutrient status.

Diatom concentrations were relatively constant until ~1925, after which concentrations increased and remained generally higher for the remainder of the sediment record (Figure 6.5). Diatom fluxes, which could only be calculated for the time period above background ^{210}Pb activity, correspond with the pattern observed for the concentration data, which indicates that diatom numbers are fluctuating as a result of a change in productivity and not due to a change in sedimentation rates alone.

In order to further explore the nature of the changes in the diatom assemblages in the sediment core, the sample scores for the sediment samples were added passively to the DCA ordination of the surface sediment diatom assemblages in the 23-pond data set (Figure 6.6). The ordination diagram illustrated that fossil diatom assemblages in the sediment core were initially (~ 39.0 cm - 24.0 cm; ~1630 – 1820) more similar to the contemporary diatom assemblages found in the study ponds (group 3; blue ellipse) that were distinguished by the presence of the small, adnate benthic taxon, *S. pinnata*. However, after ~1820 until present (23.0 – 0.0 cm), diatom community composition evolved and was more similar to the diatom assemblages found in the study ponds (group 1; red ellipse) that were dominated by *D. kuetzingii* and had no *S. pinnata* in their surface sediments.

6.1.5 Fossil pigment stratigraphy

Analysis of the sedimentary photosynthetic pigment concentrations identified two distinct zones that align with the zones observed in the diatom stratigraphy (Figure 6.7). Concentrations of all pigments were markedly higher in zone 1 (below 19 cm; pre-~1870) than in zone 2 (19.0 cm to 0 cm; ~1870 to present). Chl-*a* (which indicates total algal biomass) averaged 178.0 nMol g^{-1} OM in zone 1, but only 100.0 nMol g^{-1} OM in zone 2. Similar magnitudes of decline were observed for β-carotene (also estimates total algal biomass; 32.3 to 14.1 nMol g^{-1} OM), lutein (green algae; 212.0 to 70.1 nMol g^{-1} OM), aphanizophyll (N_2-fixing cyanobacteria; 632.3 to 220.1 nMol g^{-1} OM) and diadinoxanthin (diatoms; 72.7 to 29.3 nMol g^{-1} OM). Within both zones, aphanizophyll was the most abundant pigment, which indicated the importance of N_2-fixing cyanobacteria to primary production throughout the record from Left Lake. There was a slight trend in the most recent sediments (~1980 - present) towards increased concentrations of chl-*a*, β-carotene and aphanizophyll.

Pigment data, when expressed as fluxes, indicated that the large decrease in productivity that was observed in pigment concentrations at ~1870 was not due to a change in productivity (Figure 6.7). Chl-*a* flux rates remain relatively unchanged until ~1920 when an increase in productivity is first observed in the sedimentarypigment record.

6.1.6 Multi-proxy analysis

Combing the knowledge gained from this study along with the hydrological and geochemical analysis of the Left Lake core conducted by Light (2011) provides a useful approach for interpreting past changes in hydro-ecological conditions in Left Lake.

D. kuetzingii complex, which occurred at high relative abundance throughout the core, were interpreted to be associated with the cyanobacteria mats that currently line the bottom of the pond. Analysis of fossil pigments supported this interpretation as pigments representative of nitrogen-fixing cyanobacteria (aphanizophyll) dominated the entire sedimentary record. The $\delta^{15}N$ record from the geochemical analysis further supported the dominance of N_2-fixing cyanobacteria as values were very close to that of atmospheric N (~0‰), indicating that atmospheric N was a dominant source of N for primary production (Meyers and Teranes, 2001). Although N_2-fixation remained important to the ecology of the pond, there was a decline in the concentration of aphanizophyll which coincided with a rising trend in N content and a declining trend in $\delta^{15}N$ values in the early 20th century which was suggestive of weakening N-limitation in the ponds (Light, 2011). LOI analysis of the pond sediments (Figure 6.2) did not provide any additional information related to the loss of *S. pinnata* between zone 1 and zone 2 in the sediment record (Figure 6.5), as no observable change in sediment composition was detected that would indicate a change in benthic habitat from fine mineral grain substrate to thick algal mats. The assumption that a marked decline in aquatic productivity occurred around 1870, as indicated by decreased sedimentary pigment concentrations, was not supported in the $\delta^{13}C$ or N record from the geochemical analysis or chl-*a* flux rates. Chl-*a* flux rates remain relatively unchanged from 1850 to ~1920 when a trend towards increasing values is observed. This increase in productivity as inferred by higher chl-*a* flux rates may be in response to an increase in nitrogen availability or a lengthened growing season. There was a slight decrease in $\delta^{13}C$ values around ~1950 but this difference was likely in response to the Suess effect in which $\delta^{13}C$ values of atmospheric CO_2 have declined due to the burning of fossil fuels depleted in ^{13}C (Verberg, 2007). This interpretation of the geochemical record is supported by the pigment data as chl-*a* flux rates have actually increased since ~1920 which would infer an increase in productivity, not a decline.

The cellulose-inferred pond water $\delta^{18}O$ values from the geochemical analysis indicated that evaporation has been exerting a larger influence on pond water balance since around ~ 1920. The increase in evaporation may be attributed to temperature increases of 0.3 to 0.8 °C per decade during 1958-2000 in the HBL and a lengthened thaw-season (Schindler and Smol, 2006; Kaufman et al., 2009; Wolfe et al., 2011). Increased precipitation over adjacent lands has also been observed due to climate warming influences on regional climate (Macrae et al., 2010). This is expected to offset the effects of increased evaporation (Macrae et al., 2010). However, these increases are being observed in the late thaw-season (August-September) and it is actually drier in early thaw-season (May-July;

Macrae et al., 2010) when primary productivity is highest and most of the cellulose that is incorporated into the lake sediment record is likely produced. As previously discussed, small basin size, small catchment area and hydrologic isolation from other basins has made Left Lake highly susceptible to the effects that evaporation is having on water balance due to changing climatic conditions (Wolfe et al., 2011). While there were no apparent shifts in the diatom or pigment community composition related to 20th century changes in water balance, there was an increase in diatom concentrations and algal productivity possibly in response to an increased length of the thaw-season.

6.2 Erin Lake paleo-reconstruction

6.2.1 Sediment cores visual stratigraphy

Visual observations of the sediment cores from Erin Lake were reported by Light (2011). Sediment core EL GC-1 was 32.0 cm in length (Figure 6.9). Sediments at the base of the core (32.0 cm) to 26.0 cm were dark brown and appeared to be high in organic matter content. Sediments between 26.0 cm and 3.5 cm were described as orange-brown. Above 3.5 cm, sediments were light grey and abundant chironomid tubes were visible.

The second core EL GC-2 was shorter (28.5 cm; Figure 6.9) and although the stratigraphic sequences are in the same order, the changes occurred at slightly different depths. The change from dark brown to orange-brown was noted at ~23.0 cm and the subsequent transition to light grey sediment occurred at 3.5 cm.

6.2.2 Loss-on-Ignition

Pond sediments from Erin Lake were highly organic, averaging ~85 % over the entire profile of both sediment cores (Figure 6.10). Organic matter content was lower at the base of both cores (~70-80%) and increased until ~24.0 cm when concentrations stabilized and remained near-constant for the remainder of the core profile. $CaCO_3$ content was low in both sediment cores (< 5%). $CaCO_3$ content at the base of core GC-1 was relatively high and declined gradually in the lower half of the core. However, around ~15.0 cm, $CaCO_3$ content stabilized and values were comparable to those observed in core GC-2.

6.2.3 Core Chronology and Sedimentation Rate

In core EL GC-2, total ^{210}Pb activity initially increased to its highest value of ~0.478 Bq $g^{-1}$from 0 cm until 3-cm core-depth after which the activity decreased exponentially to background levels of 0.006 Bq g^{-1} by approximately 11.25 cm (Figure 6.11). The near-constant total ^{210}Pb activity from the surface down to the 3-cm interval may be the result of sediment mixing or an increase in sedimentation rate. Background activity was determined as the average of ^{214}Pb and ^{214}Bi and was in close agreement with the mean ^{210}Pb values (0.006 Bq g^{-1}) measured below 20 cm depth. The CRS model was used to estimate the chronology and sedimentation rates from the unsupported ^{210}Pb activity profile in the sediment core and then a linear extrapolation was used to estimate older ages, which provided a basal date of ~1398 C.E. for the core. The activity profile for ^{137}Cs demonstrated a reasonably well-defined peak at ~5.5 cm, corresponding to a CRS date ~1959 – close to peak above-ground nuclear weapon testing that occurred in 1963.

The dry mass sedimentation rate for EL GC-2 was considerably slower than the sedimentation rate observed in the Left Lake sediment core (Figure 6.12). The dry mass sedimentation rate averaged ~0.00786 g cm^{-2} yr^{-1} from ~1876 to 1993. From 1993 until the core was collected in 2009 the sedimentation rate increased to 0.0137 g cm^{-2} yr^{-1}, which may be the result of in an increase in productivity and/or the result of incomplete decomposition of the most recently deposited sediments.

Profiles of the OM and the $CaCO_3$ content in sediment cores GC-1 and GC-2 from Erin Lake were very similar and provided very few stratigraphic markers with which to cross-date GC-2 with GC-1 (Figure 6.10). However, a few peaks and troughs in the OM profile at the base of both cores could be used to align the cores, which indicated that the cores may be offset by approximately 0.5 cm. Given that the sampling resolution was greater than 10 years at this depth, the small difference in alignment should be negligible and the chronology developed from GC-2 could be directly applied to GC-1. Because GC-1 was 3.5 cm longer than GC-2, the same linear regression used to extrapolate the age-depth profile for GC-2 was used to extend the chronology for the entire core, which provided a basal date ~1255 C.E for GC-1.

6.2.4 Diatom Stratigraphy

A total of 48 diatom taxa were identified in the samples from core GC-1 of Erin Lake. Twenty-two of these taxa were also present at abundances ≥1% in at least two of the study ponds in which surface sediment samples were collected (Table 5.1). Three distinct zones were identified based on diatom community composition in the sediment core (Figure 6.13). Zone 1, from ~1280-1550, was characterized by high relative abundance (60-80%) of the alkalphilic benthic taxon *S. pinnata*. This trend suggested that cyanobacterial mat habitat was not extensive, nor were macrophyte or planktonic habitats available. *S. pinnata* is considered a colonizing species that dominates in substrates consisting of rocks and fine mineral grain substrates. These habitats are typically located in environments that have extended ice cover and short growing seasons with limited primary production, which does not allow more complex diatom communities to evolve (Douglas et al., 2004). In Zone 2, from ~1550-1850, diatom numbers declined to almost zero, which may have been the result of a shift in ecological conditions or a shift to poor preservation due to silica limitation, increased alkalinity or other factors which can reduce preservation (Battarbee et al., 2001). Consistent concentrations of the pigment diadinoxanthin (Fig. 6.15,; however, identifies that diatom biomass was unaltered during the period of low diatom valve abundance, suggesting poor preservation of the siliceous valves is the cause of the reduced concentration of diatom valves in zone 2. In zone 3, from ~1850-2009, *S. pinnata* was below detection limits and *D. kuetzingii* complex appeared in the sedimentary record at relatively high abundances (>50%), likely in response to the availability of extensive cyanobacterial mat habitat. This interpretation is consistent with the modern-day association between *D. kuetzingii* complex and the dense cyanobacteria mats that line the pond bottoms, as was identified from the surface sediment survey of the study ponds. These mats have likely been a feature in the pond since ~1850, as indicated by the high abundance of *D. kuetzingii* complex in the sediment record. The poor preservation of diatom microfossils from ~1550 to 1850 prevents interpreting the exact time in which the cyanobacterial mats became a feature in the pond. The emergence of epiphytic taxa that prefer more nutrient-rich waters (*Encyonopsis microcephala*, *Navicula vulpina*, *Cymbella angustata)* was observed in this zone.

Diatom concentration from ~1850-1900 was low in comparison to the concentration of diatom valves in the most recent sediments, especially after ~ 1920. Diatom absolute abundance declined substantially from ~1500 until 1850, which appeared to be related to the dissolution of diatom valves in the sedimentary record. From ~1850 to present there was a trend towards increased values in the stratigraphic record. Diatom flux over the last ~150 years mimic similar trends that were observed in the diatom absolute abundance, which indicated that the changes in the number of diatom valves enumerated were the result of increased diatom productivity and not due to changes in sedimentation rates.

In order to further explore the nature of the changes in the diatom assemblages in the sediment core, the sample scores for the sediment samples were added passively to the DCA ordination of the surface sediment diatom assemblages in the 23-pond data set (Figure 6.14). Early diatom assemblages deposited in Erin Lake (31.0 cm - 25.0 cm; ~1280 – 1550) were positioned high on DCA axis 2, well above assemblages in surface sediments of the 23-pond'training set', indicating the assemblages at Erin Lake were different from the contemporary diatom assemblages found in any of the study ponds. However, they are located closest to group 3 ponds (Group 3; blue ellipse) in the ordination diagram because of the presence of *S. pinnata*, but at much higher abundances than observed in the surface sediments of any of the study ponds. After ~1850 until present, when diatoms reappeared in the sedimentary record, community composition changed and was more similar to the diatom assemblages found in the first group of ponds (Group 1; red ellipse) characterized by high abundances of *D. kuetzingii* complex and epiphytic taxa which prefer higher nitrogen concentrations. Another important feature illustrated in the ordination diagram is that diatom community composition since ~ 1850 is comparable to the contemporary assemblage composition, as indicated by the post ~ 1850 samples plotting close to the surface sediment sample from Erin Lake.

6.2.5 Fossil pigment stratigraphy

Analysis of the sedimentary photosynthetic pigment concentrations indicated that pigment composition and concentrations have remained relatively unchanged during the entire period contained in the sediment record (Figure 6.15). Chl-*a* and β-carotene, which represents total algal productivity, averaged 126.4 nMol g^{-1} OM and 24.9 nMol g^{-1} OM, respectively. Aphanizophyll, which is produced mainly by N_2-fixing cyanobacteria, was the most abundant of all pigments (mean = 328.0 nMol g^{-1} OM), a finding that highlights the importance of N_2-fixation to the past ecology of the pond. Green algae were represented by the pigment lutein, which averaged 223.1 nMol g^{-1} OM with a trend towards slightly decreased values in the most recently deposited sediments. The pigment diadinoxanthin, which represented diatoms, averaged 59.3 nMol g^{-1} OM with relatively consistent concentrations throughout the core.

Chl-*a* flux rates trend towards increased values in the Erin Lake sediment record indicating total algal production has increased over the last ~100 years. Flux of aphanizophyll was variable in the early part of the twentieth century. However, the most recent flux rates were similar to the rates observed prior to ~1900 in the sediment record. Diadinoxanthin flux rates indicated that diatom productivity has increased slightly over the last ~100 years.

6.2.6 Multi-proxy analysis

Integrating the results from the geochemical analysis conducted by Light (2011) with knowledge gained from the analysis of fossil diatoms and pigments in the sediment core, allows for a more complete understanding of the past ecological conditions in Erin Lake.

D. kuetzingii complex, which includes taxa that are associated with the algal mats within the pond, were observed throughout the sedimentary record, which is similar to Left Lake but in lower relative abundances. Aphanizophyll, the photosynthetic pigment associated with nitrogen-fixing cyanobacteria that produce the algal mats, was a dominant pigment in the sediment record but it was found in lower concentration than was observed in Left Lake. The $\delta^{15}N$ values from the geochemical analysis were very close to 0 ‰, consistent with the dominance of cyanobacteria and the importance of atmospheric nitrogen as a source for primary production. However, a trend towards higher $\delta^{15}N$ values and a slight decrease in aphanizophyll concentration and flux rates after ~1950 suggest that other sources of dissolved nitrogen are becoming increasingly available.

From ~1250 – 1550, *S. pinnata* were in the highest abundance and were hypothesized to be associated with benthic habitat consisting of fine mineral grains. The decrease in diatom numbers to almost zero in from ~1550-1850 can be attributed to preservation issues as the photosynthetic pigment (diadinoxanthin) associated with diatoms remained constant in the sediment record. From ~1850 – 2009, *S. pinnata* are no longer present and epiphytic diatom taxa that prefer mesotrophic conditions appear in the sediment record. However, the timing of the change cannot be determined with any degree of accuracy due to the dissolution of diatoms prior to the stratigraphic change. Similar stratigraphic changes in the diatom record occurred in Left Lake likely in response to an increase in epiphytic habitat. Although chl-*a* concentrations have remained relatively constant over the entire core profile, a trend towards higher chl-*a* flux rates over the last ~100 years indicates that productivity is increasing. Increased nitrogen availability around ~1950 coincides with an increase in flux rates of chl-*a* and higher $\delta^{13}C$ values in the geochemical stratigraphy, consistent with the interpretation of increased algal productivity.

Cellulose-inferred pond water $\delta^{18}O$ values from Erin Lake have remained relatively unchanged over the entire core profile even in response to the temperature increases that have been observed in the HBL over the later part of the 20th century (Kaufman et al., 2009). Erin Lake has had a much different hydrological response to changing climatic conditions than Left Lake, probably due to catchment and basin characteristics, which allow Erin Lake to capture sufficient precipitation to offset the effects of increased evaporation (Wolfe et al., 2011). Analysis of diatom and pigment community composition also supports this interpretation as no changes in either the diatom or pigment stratigraphies indicate that any hydrological change has occurred over the past century.

6.3 Discussion

The chronologies of both the Left Lake and Erin Lake sediment cores display similar trends in the activity profiles. Both cores demonstrate near-constant ^{210}Pb activity from 0 cm – 3 cm before the typical exponential decline in activity is observed in the remainder profile. The near-constant activity in the top 3 cm of the core can be interpreted in a number of ways, as previously mentioned. The anomaly in the upper intervals of both profiles could be due to the mixing of sediments. This explanation would be highly plausible as the ponds in the study are exposed to high winds which could create sufficient wave action to mix the upper sediments. However, this scenario is unlikely as the thick cyanobacterial mats that line the bottom of the ponds protect the uppermost sediments from this process (Wolfe et al., 2011). Non-homogeneous values in the stratigraphies of other proxies measured in this study (LOI, diatoms, pigments) also suggest that the near-constant values of ^{210}Pb at the top of the core are not the result of sediment mixing. Instead, increased sedimentation rates or the incomplete decomposition and compaction of the algal mats appear to be more likely than sediment mixing.

Analyses of the biological records from the sediment cores from Left Lake and Erin Lake displayed temporal patterns of change that provided important information concerning past hydroecological responses to environmental change. Sediment cores from both Left and Erin Lake revealed a marked change in diatom assemblages characterized by the complete disappearance of small, adnate, benthic taxa from the most recent sediments. Similar floristic changes have also been observed in the sediment cores of shallow ponds on Ellesmere Island in which benthic *Fragilareaceae* spp. were replaced post-18th century by aquatic moss epiphytes (Douglas et al., 1994). The Ellesmere Island ponds underwent a state change due to changes in ice-cover duration and extent. *Fragilareaceae* spp. dominated under colder climatic conditions when only a narrow moat of melted ice developed around the near-shore zone of the ponds. When the climate warmed, ice-cover was lost over a larger area of the pond for much longer each year, allowing aquatic mosses to colonize the ponds and associated epiphytic taxa to proliferate. In the CWMA, climate warming may have also played a role in the floristic changes that occurred in the mid-19th century in the sediment record. A longer thaw-season would have allowed for the expansion of the cyanobacterial mats that cover the pond bottoms. The increased length of the growing season may have also resulted in an increase in plant cover in the pond's catchment which would have reduced the input of mineral matter supply from the adjacent landscape. One or both of these processes may be responsible for the loss of mineral grain sediment substrates that *S. pinnata* grow on and the disappearance of *S. pinnata* from the sedimentary diatom record.

Very high C/N ratios and $\delta^{15}N$ signatures close to 0‰ in the geochemical analysis of the sediment cores indicated that algal growth in both Left Lake and Erin Lake was nitrogen-limited (Light, 2011). This evidence, along with high concentrations of aphanizophyll, supports the interpretation that nitrogen-limitation in the CWMA has promoted the proliferation of cyanobacterial mats, which typically dominate under such conditions (Vreca and Muri, 2010). This has resulted in the complete dominance of diatom assemblages by *D. kuetzingii* in the sediment record due to its association with the cyanobacterial mats. Increased concentrations of nitrogen in recent sediments (post -~1940 in Left Lake and post-~1880 in Erin Lake) have corresponded with lower C/N ratios. Also, $\delta^{15}N$ values have deviated away from zero, which indicated an increase in nitrogen availability in the study ponds. Consistent with this interpretation was a slight decline in the concentrations of aphanizophyll as an increased supply of dissolved nitrogen allows other algal groups to proliferate (Vreca and Muri, 2010). Higher total algal production, as indicated by increasing pigment flux rates, was also observed in the sediment cores of both ponds that coincided with the increased concentrations of nitrogen in pond sediments. The change in the availability of nitrogen in the ponds may be the result of a number of factors. Increased atmospheric deposition of anthropogenic nitrogen (Wolfe et al., 2006; Light 2011), increased deposition of bird guano from populations of the Lesser Snow Goose and Canada Goose in the area (Michelutti et al., 2006; Light, 2011) or the increased mobilization of nitrogen in the active layer in response to increased precipitation (Light, 2011) are all plausible explanations for the change in nitrogen availability. Changes in diatom assemblages from oligotrophic taxa to mesotrophic taxa were observed in a lake sediment core collected in a remote location of the Rocky Mountains in response to increased amounts of fixed nitrogen from anthropogenic sources transported to the lake (Wolfe et al., 2001). In the CWMA, epiphytic taxa that prefer mesotrophic conditions are present in very small relative abundances in the more recent sediments. They probably do not represent a greater percentage of the diatom assemblages because the majority of epiphytic habitat in the ponds still consists largely of cyanobacterial mats as aquatic macrophytes are still sparsely distributed in the ponds even with the changes in nitrogen availability.

The paleohydrological record from Left Lake and Erin Lake showed very different responses to changing climatic conditions at the beginning of the 20th century. The $\delta^{18}O_{PW}$ record from the two ponds indicated that Left Lake was much more influenced by increased evaporation than Erin Lake (Light, 2011; Wolfe et al., 2011). Although the water balance in the two ponds differs, the diatom communities in both ponds are very similar because microhabitat availability is the factor that is most strongly influencing community composition at this time. However, there is an increase in productivity (chl-*a* concentration) and diatom concentrations in both ponds that occurred at ~1920 when the increased influence of evaporation is first observed in Left Lake. This indicates that the longer growing season and warmer temperatures that are driving changes in the water balances of the ponds are also influencing ecological conditions.

Figure 6-1 Photos of gravity core #1 (LL GC-1) and Left Lake gravity core #2 (LL GC-2) collected in July 2009 from Left Lake.

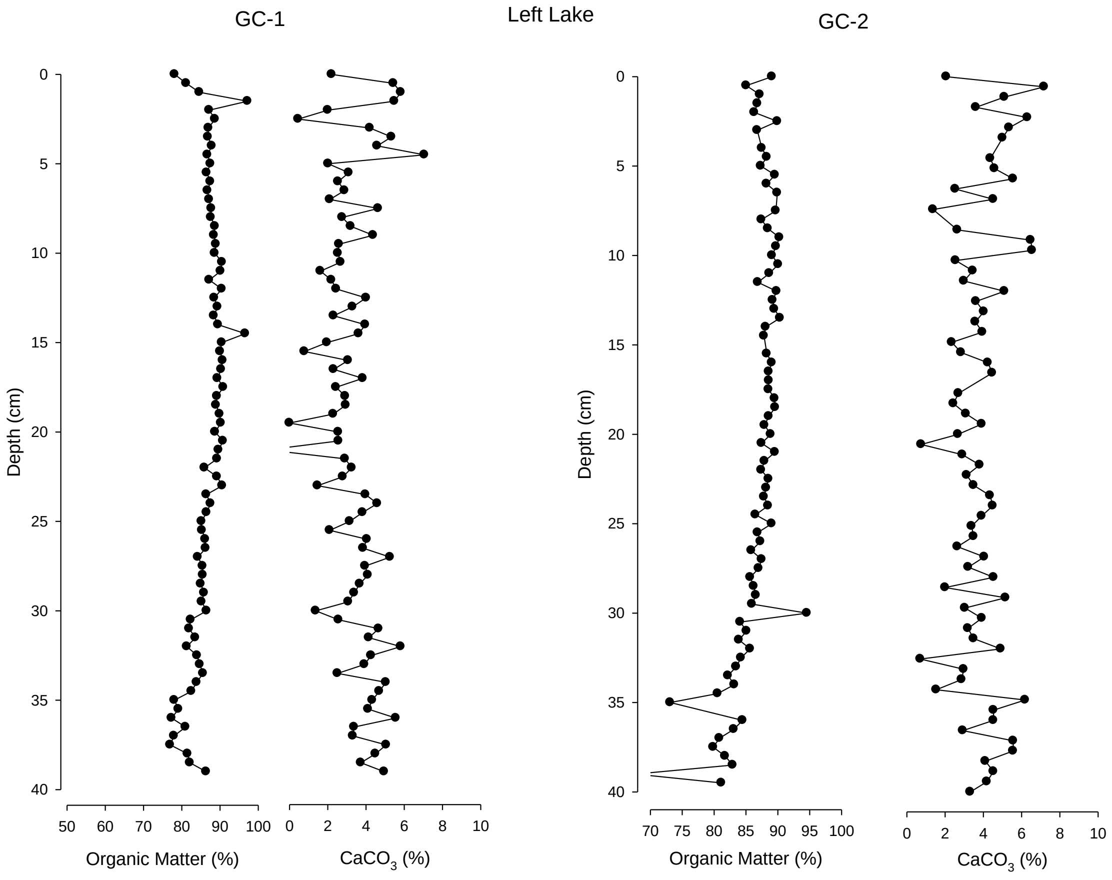

Figure 6-2 Stratigraphic plots showing patterns of change in the organic matter and calcium carbonate content (expressed as percent of dry sediment mass) with depth in sediment cores GC-1 and GC-2 from Left Lake (Churchill Wildlife Management Area, northern Manitoba).

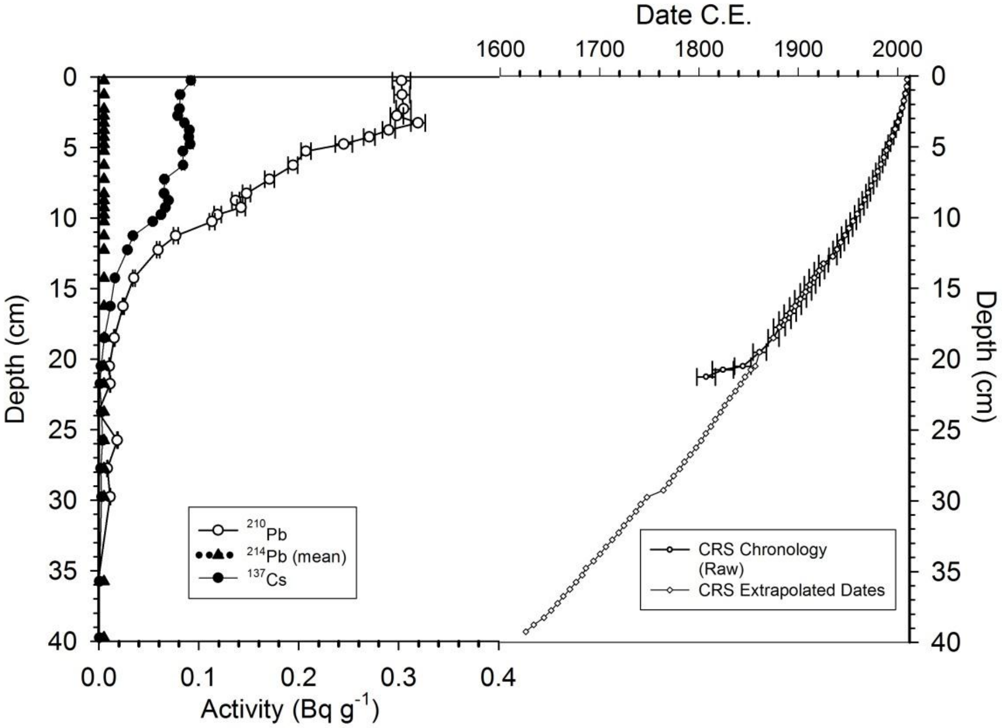

Figure 6-3 Activity profiles of ^{210}Pb, ^{214}Pb and ^{137}Cs and the age-depth relationship based on use of the constant rate of sedimentation (CRS) model for sediment core GC-2 from Left Lake, located in the Churchill Wildlife Management Area, northern Manitoba.

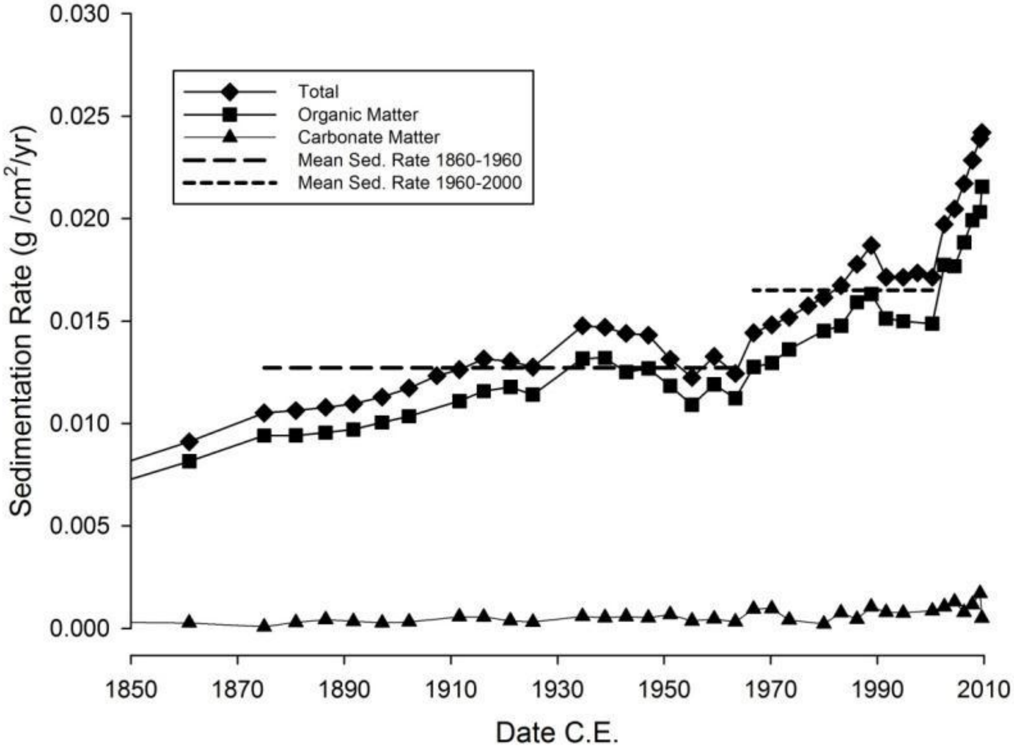

Figure 6-4 Total and average (1860-1960; 1960-2000) dry mass sedimentation rate, organic matter sedimentation rate and carbonate matter sedimentation rate for sediment core GC-2 from Left Lake, located in the Churchill Wildlife Management Area, northern Manitoba.

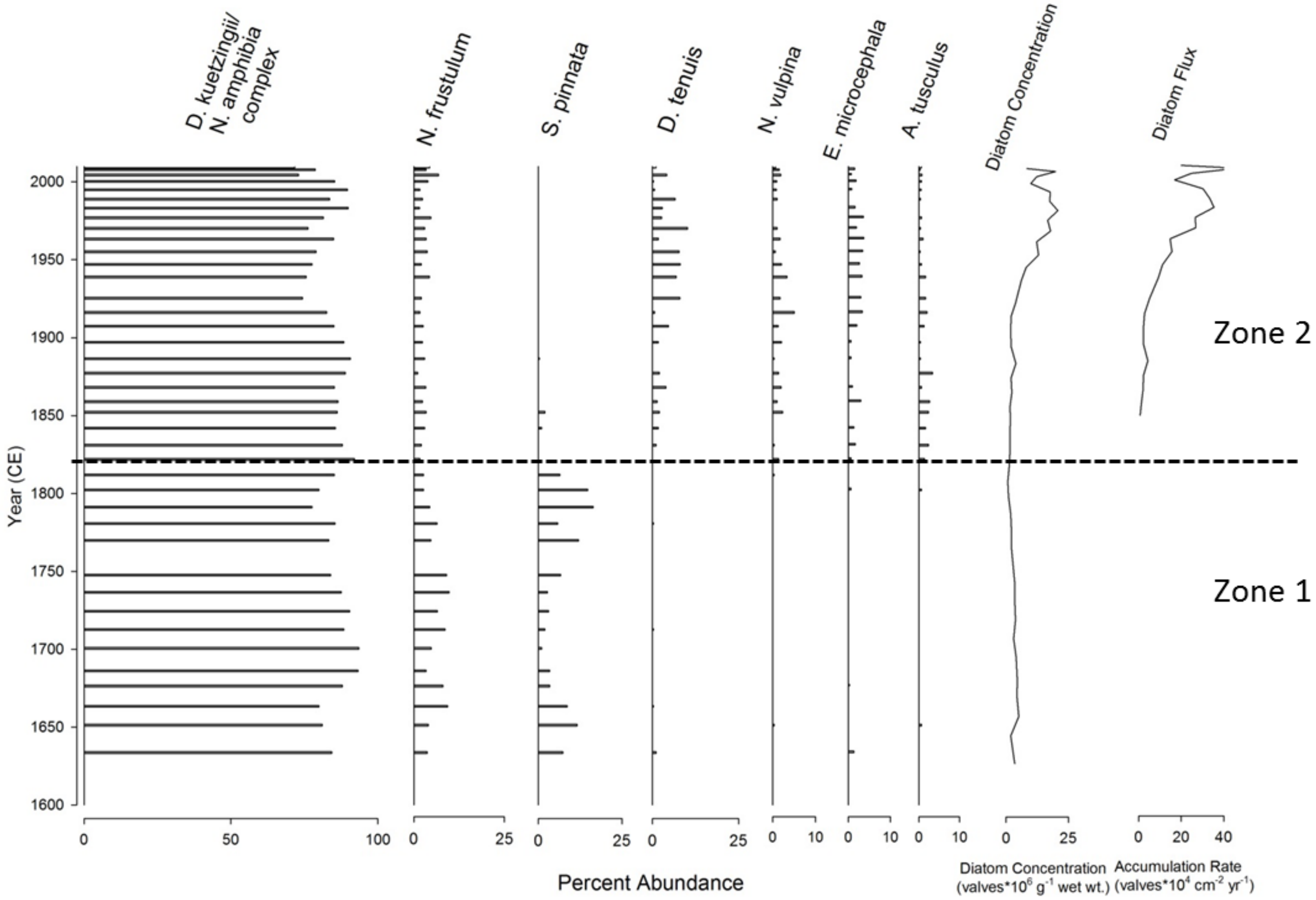

Figure 6-5 Stratigraphic profiles (by estimated age) showing diatom assemblage composition (percent abundance), and total diatom concentration and flux in sediment core GC-1 from Left Lake, located in the Churchill Wildlife Management Area, northern Manitoba.

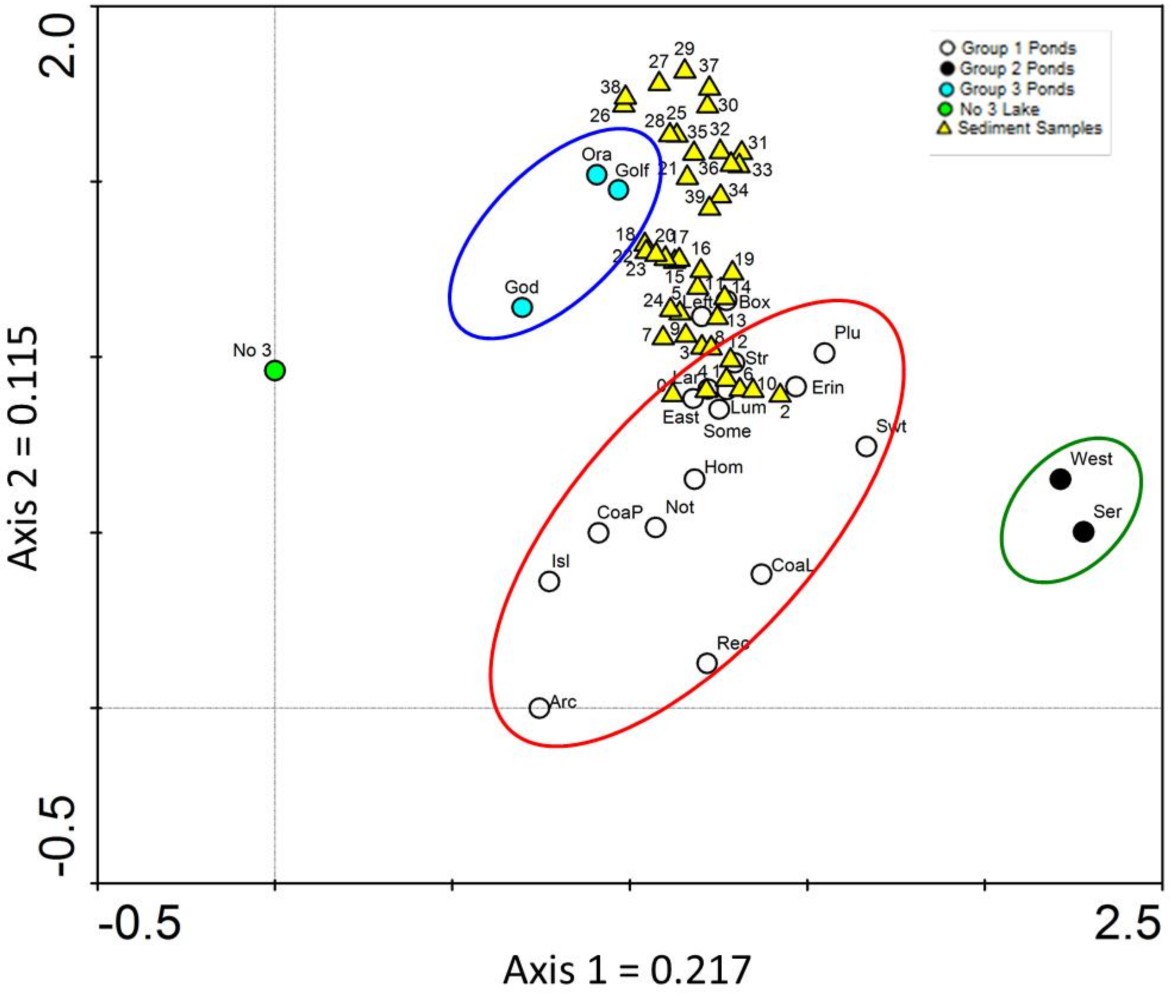

Figure 6-6 Detrended correspondence analysis (DCA) of surface sediment diatom community composition from the 23 study ponds with Left Lake sediment core diatom assemblages (n=39) plotted passively. The numbers associated with sediment samples represent the depth interval in cm which the sample corresponds. 0 cm represents the most recent interval enumerated while 39 cm is the oldest.

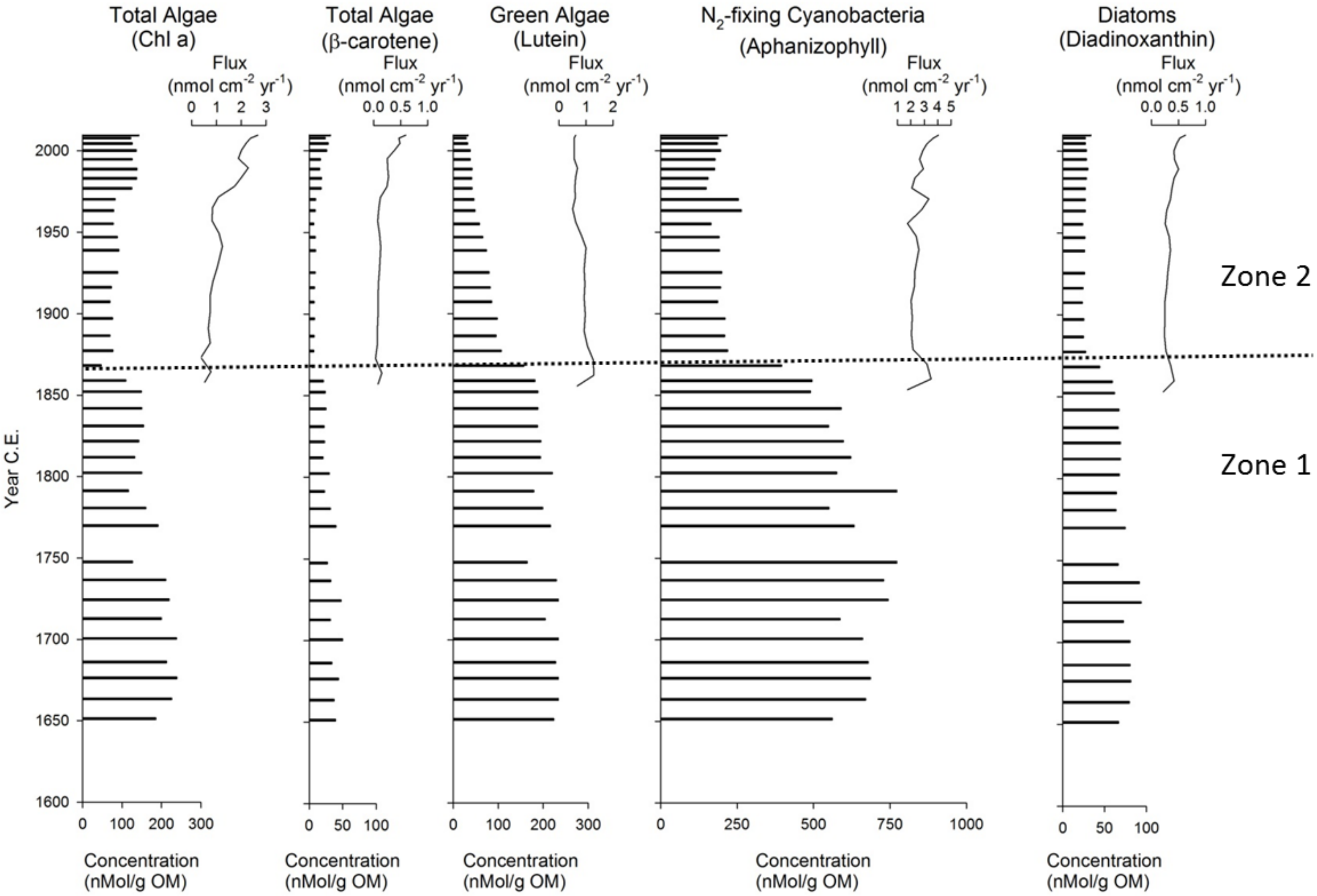

Figure 6-7 Stratigraphic profiles (by estimated age) showing the concentration (horizontal bars; nMol/g OM) and flux (lines; nMol/cm/yr) of photosynthetic pigments in sediment core GC-1 from Left Lake, located in the Churchill Wildlife Management Area, northern Manitoba.

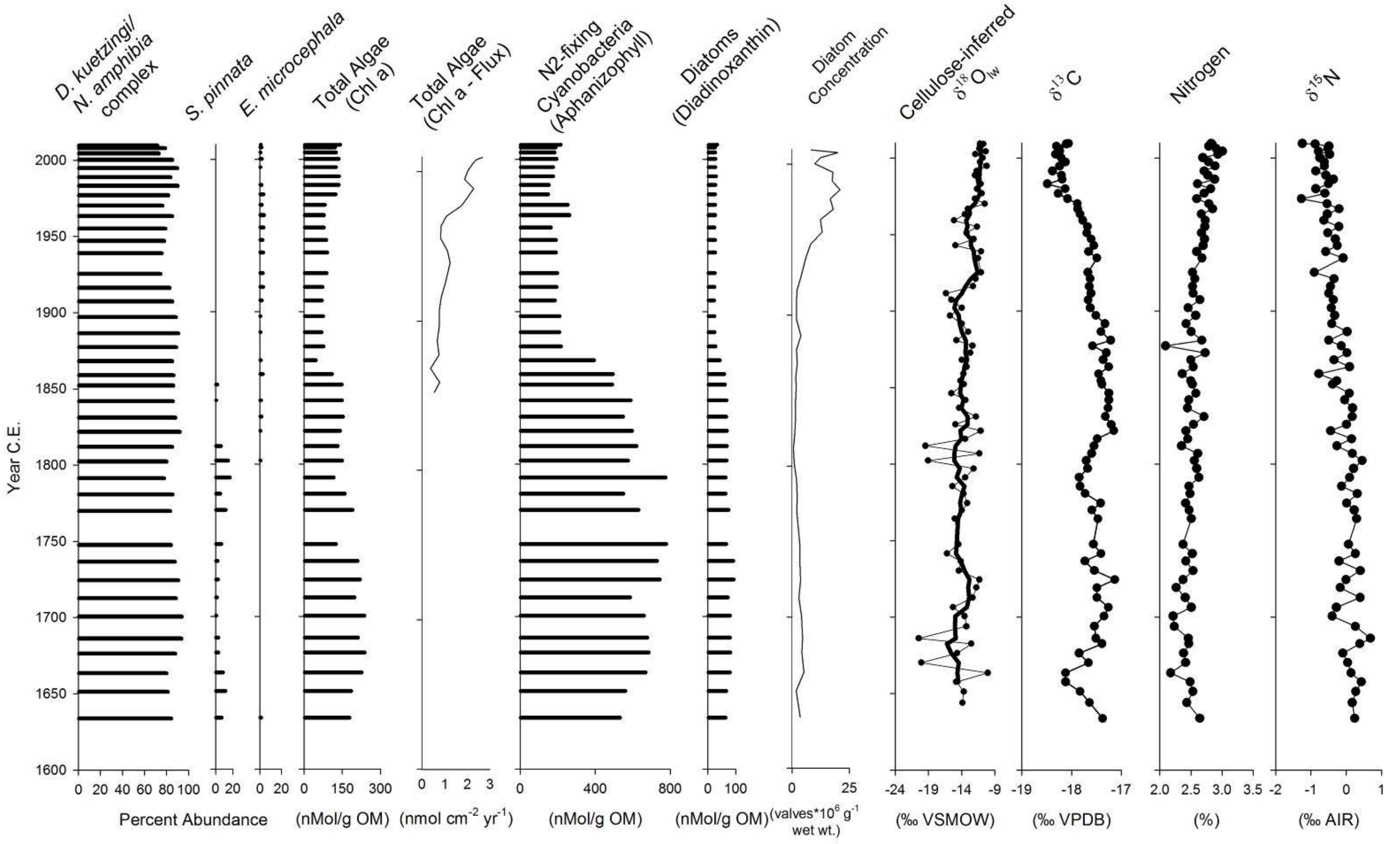

Figure 6-8 Summary diagram showing stratigraphic profiles of selected biological (diatoms, pigments) and geochemical (Light, 2011) variables used in the multi-proxy analysis of past hydroecological conditions from sediment cores from Left Lake, located in the Churchill Wildlife Management Area, northern Manitoba.

EL GC-1 EL GC-2

Figure 6-9 Photos of gravity core #1 (EL GC-1) and gravity core #2 (EL GC-2) collected in July 2009 from Erin Lake.

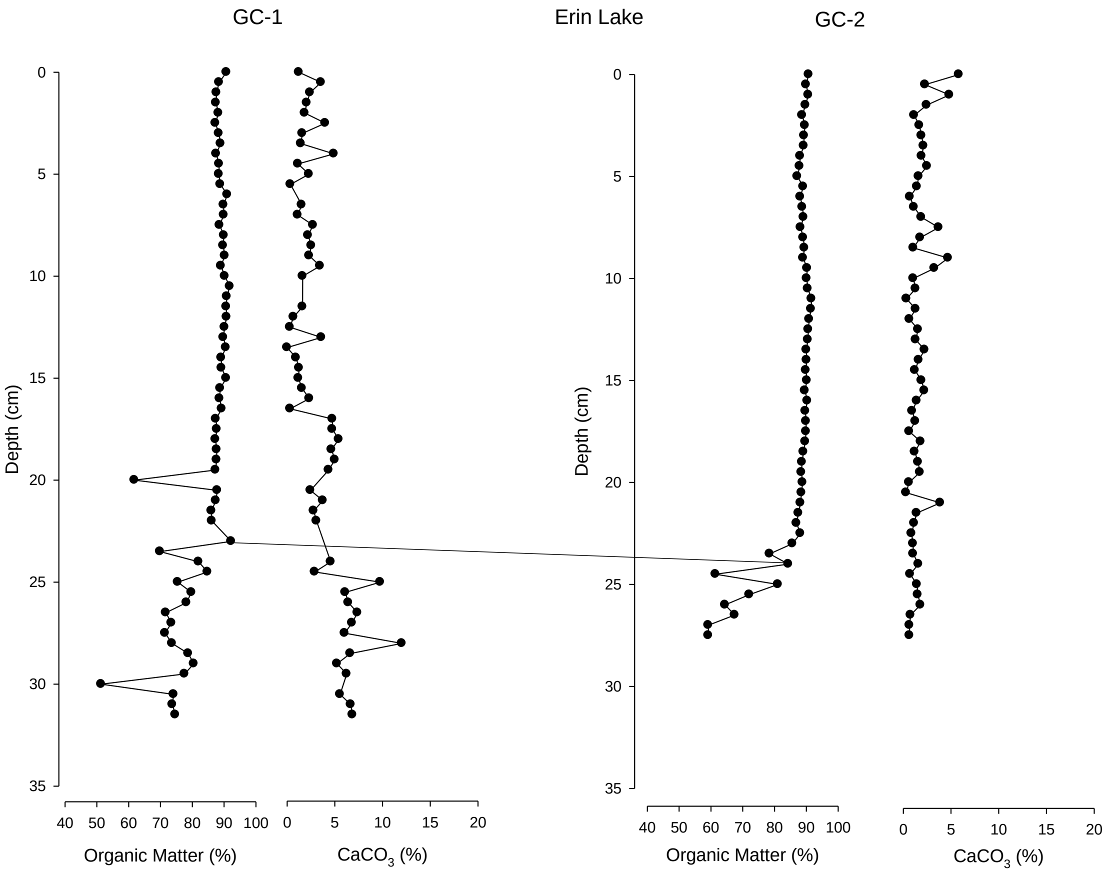

Figure 6-10 LOI results displaying the organic matter and calcium carbonate content of Erin Lake sediment cores GC-1 and GC-2. Line connecting organic profiles of both cores represents the stratigraphic offset between cores (~0.5 cm).

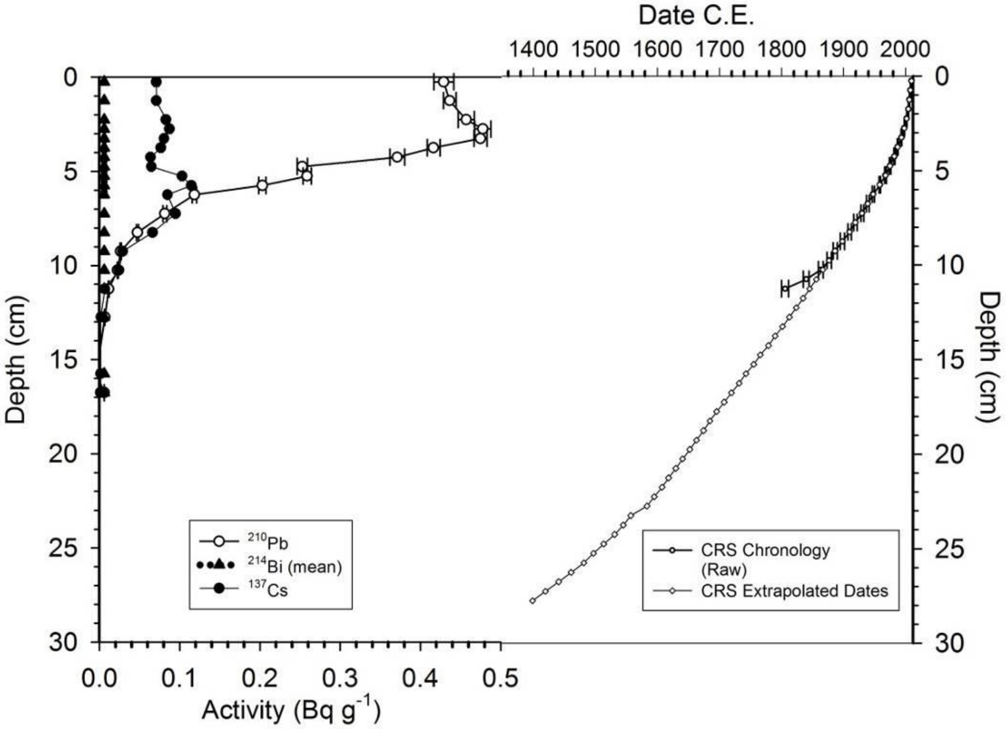

Figure 6-11 Activity profile of ^{210}Pb and ^{137}Cs and CRS dating profiles from Erin Lake GC-2 collected from the CWMA in August 2009 analyzed at the WATER Lab at the University of Waterloo.

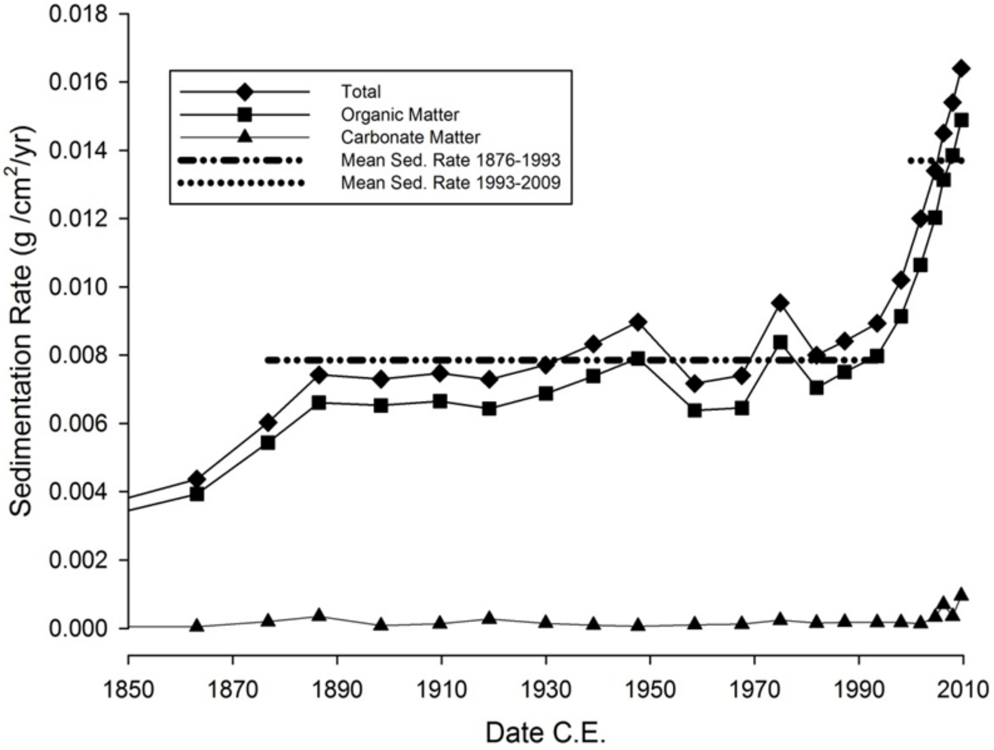

Figure 6-12 Total and average dry mass sedimentation rate, organic matter sedimentation rate and carbonate matter sedimentation rate for Erin Lake sediment core GC-2.

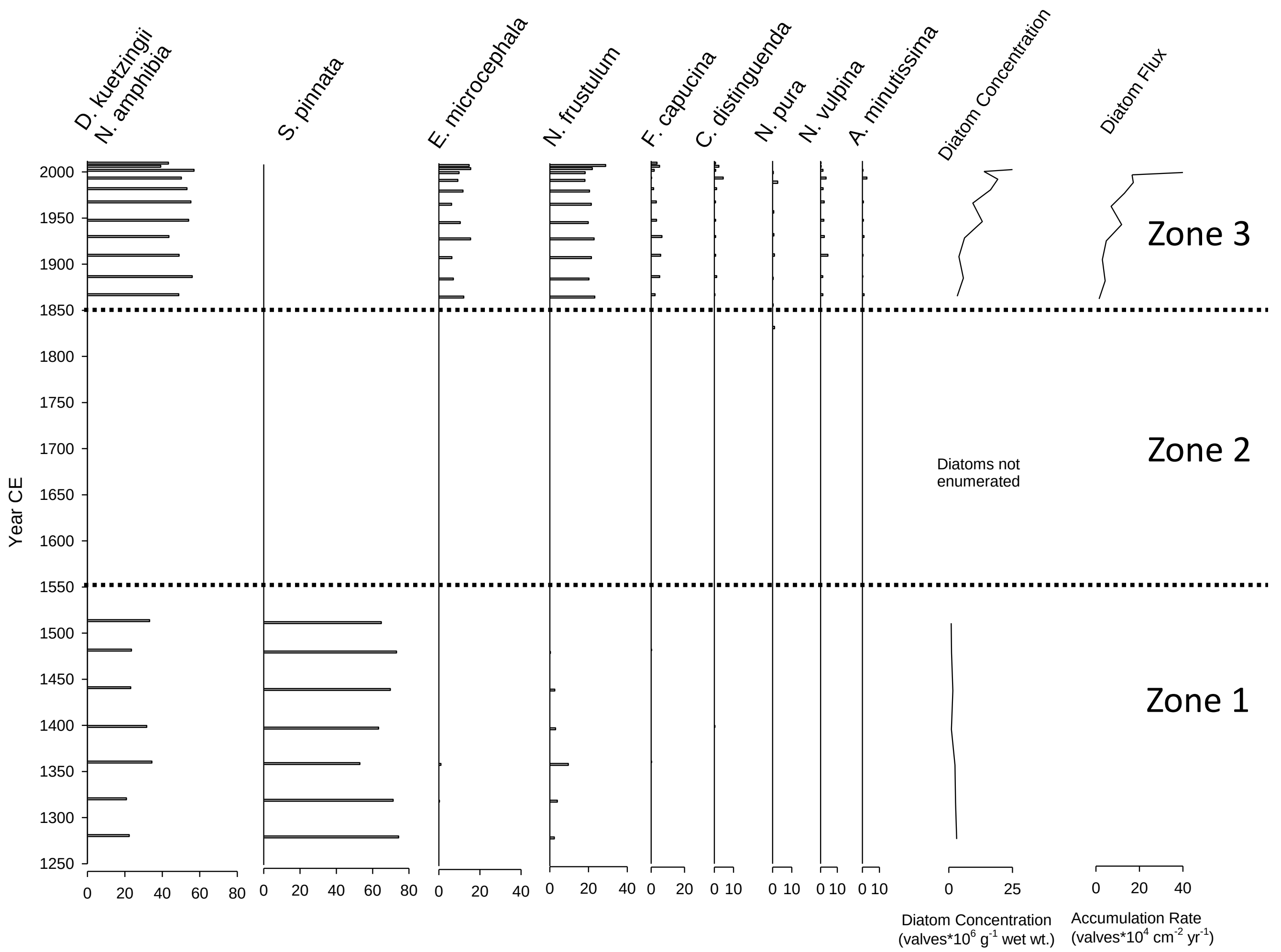

Figure 6-13 Stratigraphic profiles (by estimated age) showing diatom assemblage composition (percent abundance), and total diatom concentration and flux in sediment core GC-1 from Erin Lake, located in the Churchill Wildlife Management Area, northern Manitoba.

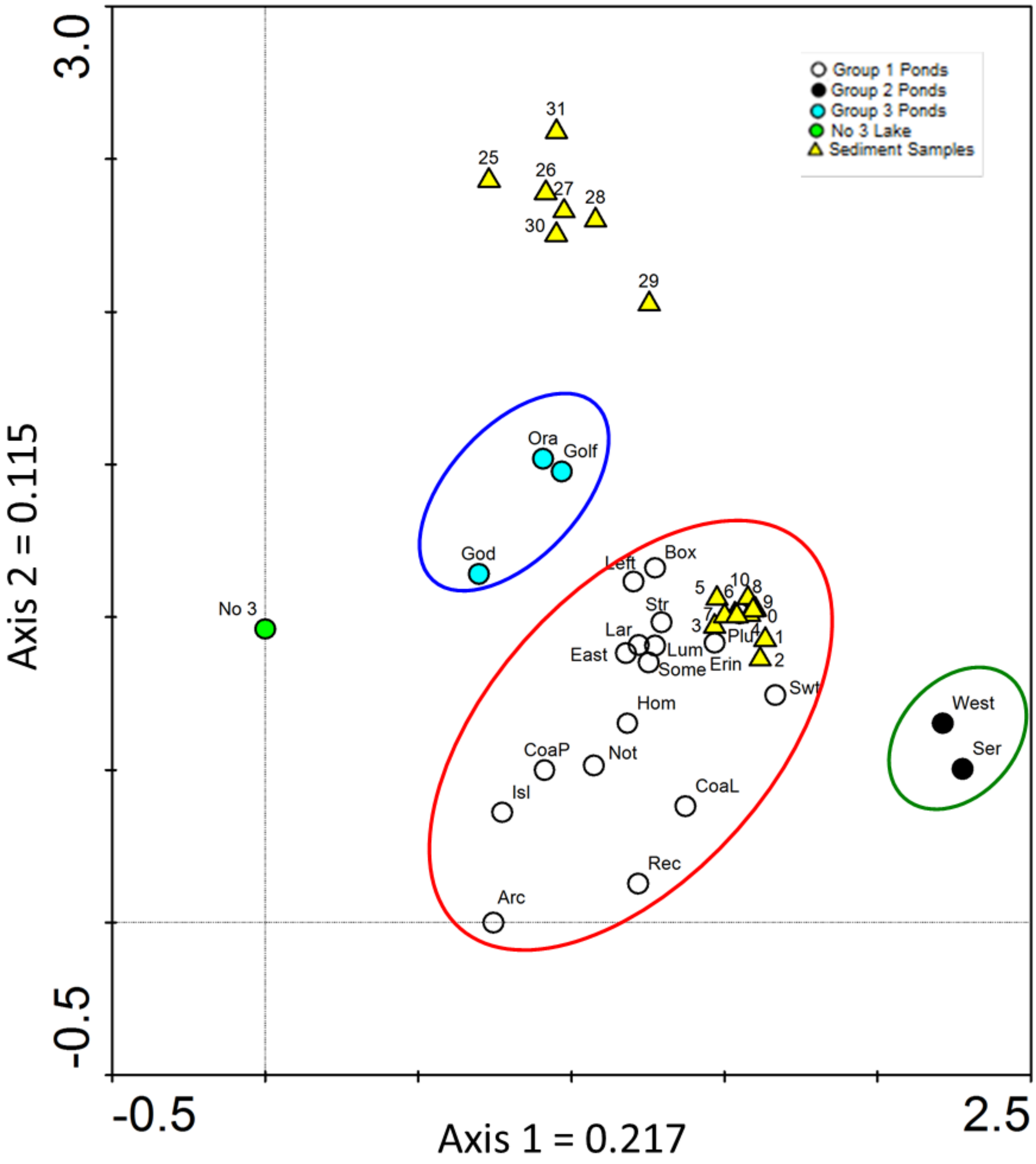

Figure 6-14 Detrended Correspondence analysis of surface sediment diatom community composition from the 23 study ponds in the CWMA with Erin Lake sediment core diatom assemblages (n=18) plotted passively. The numbers associated with sediment samples represent the depth interval in cm which the sample corresponds. 0 cm represents the most recent interval enumerated while 31 cm is the oldest.

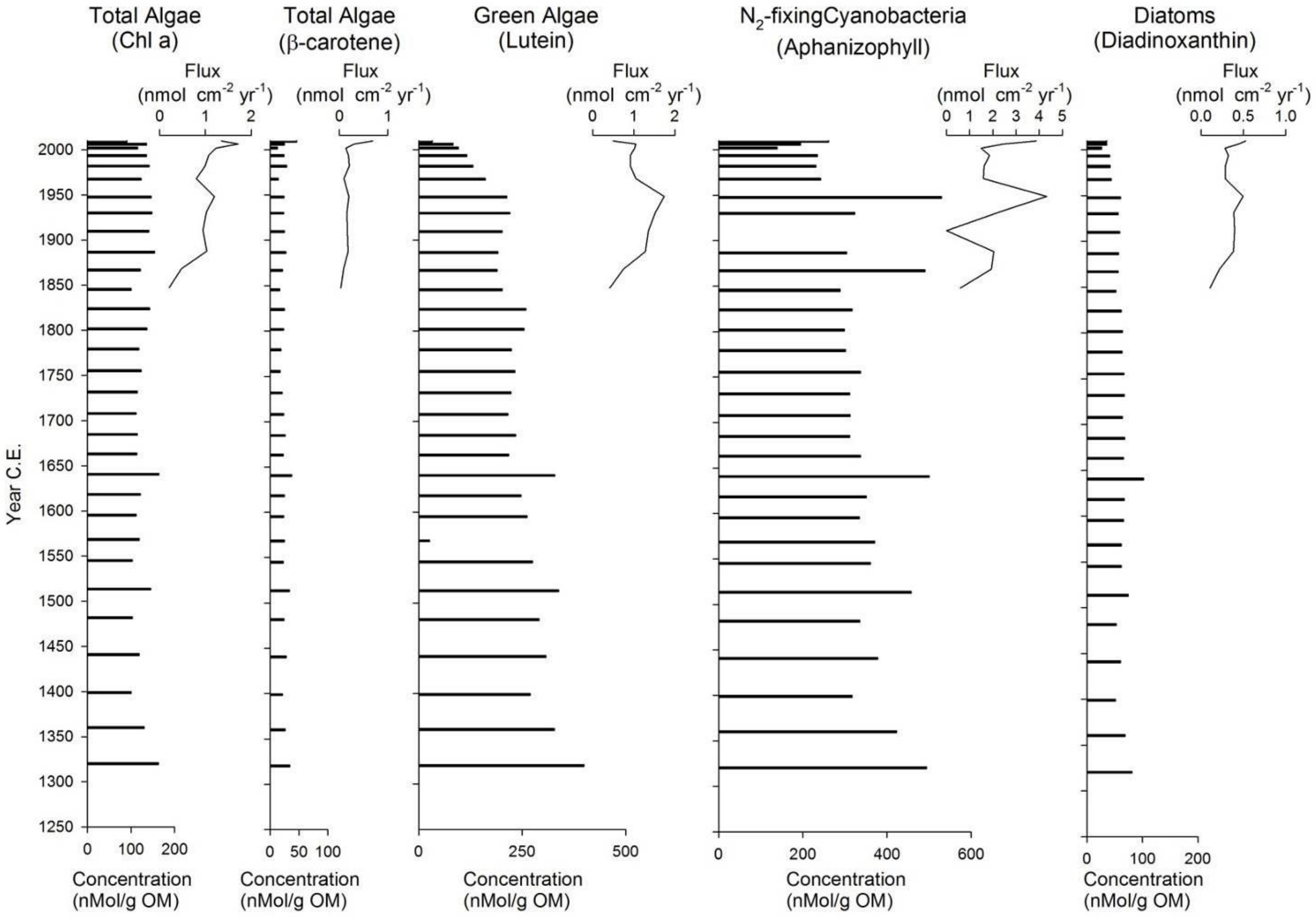

Figure 6-15 Stratigraphic profiles (by estimated age) showing the concentration (horizontal bars; nMol/g OM) and flux (lines; nMol/cm/yr) of photosynthetic pigments in sediment core GC-1 from Erin Lake, located in the Churchill Wildlife Management Area, northern Manitoba.

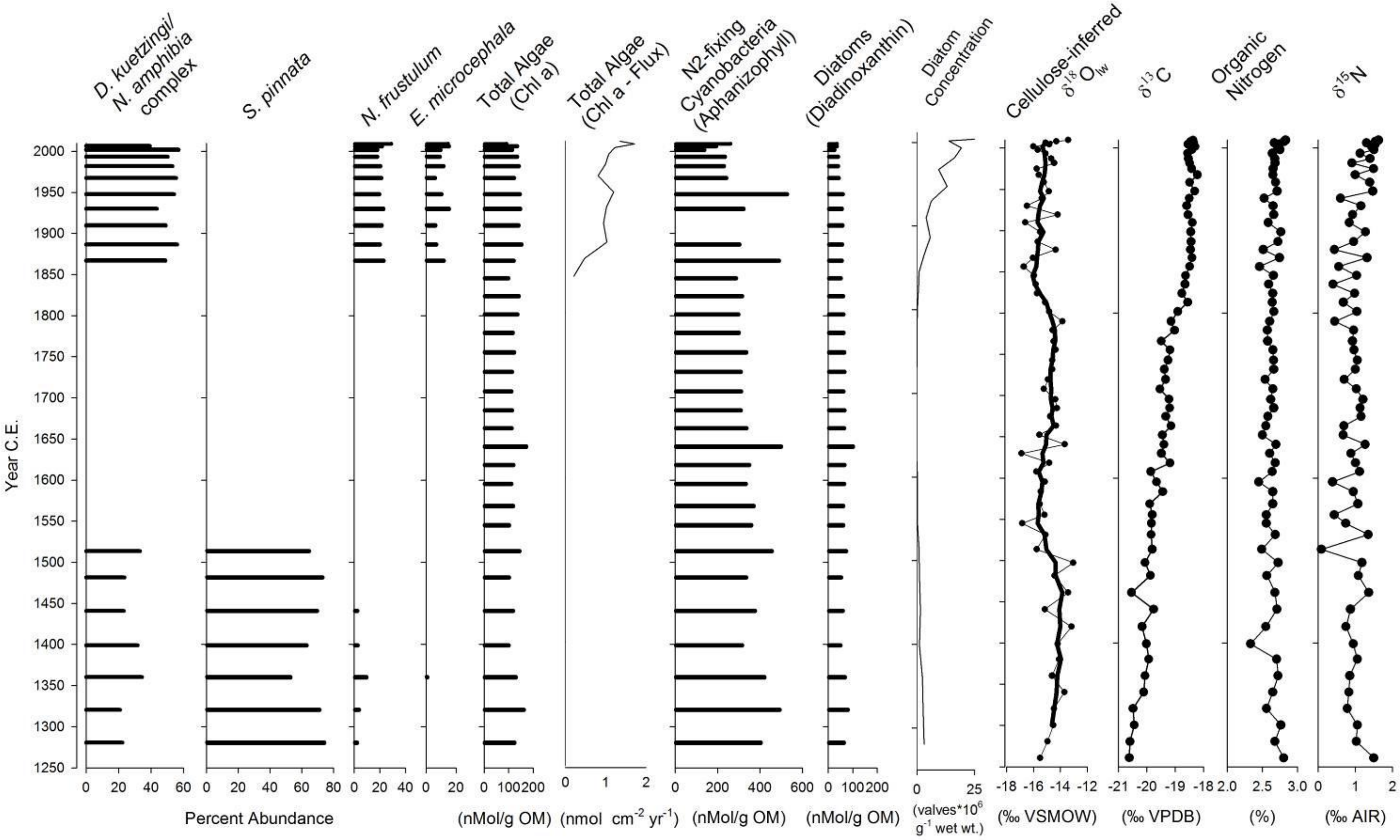

Figure 6-16 Summary diagram showing stratigraphic profiles of selected biological (diatoms, pigments) and geochemical (Light, 2011) variables used in the multi-proxy analysis of past hydroecological conditions from sediment cores from Erin Lake, located in the Churchill Wildlife Management Area, northern Manitoba.

Chapter 7

Synthesis & Recommendations

7.1 Synthesis

The combination of spatial surveys of contemporary hydrolimnological relations with multiproxy paleolimnological studies of past variations in hydrolimnological conditions is particularly informative in developing a better understanding of the factors that drive ecological conditions in the HBL. This more complete understanding will provide the tools to better predict how the shallow tundra ponds that dominate the HBL landscape will respond to continued climate warming. The purpose of this section is to synthesize the results of the individual chapters and address the stated objectives of the thesis.

1. What are the relations between the hydrological processes that control the pond water balances and the limnological conditions that are observed in these shallow tundra ponds?

Light (2011) identified a seasonal trend in water isotope composition in which all the ponds experienced systematic evaporative enrichment from spring to summer followed by isotopic depletion due to dilution in response to late thaw-season rainfall. Although the overall pattern in water isotope composition among the study ponds indicated a seasonal wet-dry-wet pattern, there were differences in the degree of evaporative enrichment the ponds experienced over the thaw-season that were attributed to basin and catchment characteristics and hydrologic connectivity (Light, 2011; Wolfe et al., 2011).

The seasonal evolution of water chemistry among the study ponds was also variable, and the ponds followed one of four distinctive patterns. The four seasonal limnological patterns appeared to be related to hydrologic factors that influence pond water balance, and they differed mainly in the evolution of ion concentrations, alkalinity, conductivity and suspended solids. The first group of ponds (SWCT 1) is influenced strongly by evaporation and includes basins that have small catchments and are hydrologically isolated from other ponds and wetlands throughout the thaw-season. These features result in progressively decreasing concentrations of suspended solids over the course of the thaw-season due to an absence of inflows into the ponds along with corresponding increases in ion concentration, conductivity and alkalinity due to the effects of evaporation. The second group of ponds (SWCT 2) experience varying degrees of influence by evaporation early in the thaw-season, depending on basin size. These ponds are hydrologically isolated following the snow-melt period but then receive inflows in late thaw-season in response to large precipitation events. This acts to offset the effects of evaporation on pond water balance. These ponds initially experience decreasing concentrations of suspended solids in response to the lack of inflows and increasing ion concentrations, alkalinity and conductivity as the ponds experience evaporative drawdown, similar to patterns of the first group. But, later in the thaw-season, inflows from the catchment wash allochthonous material into the ponds and the concentrations of suspended solids increase. These inflows also act to offset the effects of evaporation and ion concentration, alkalinity and conductivity decline. The third group of ponds (SWCT 3) initially is less influenced by evaporation as they receive inputs in early thaw-season (through lateral subsurface or surface flow) which offsets the effects of evaporation, but then become more susceptible to evaporation as they become isolated later in the thaw-season. This pattern is mirrored in the water chemistry of these ponds as concentrations of suspended solids increase as allochthonous inputs are initially washed into the ponds from the catchment. Ion concentration, alkalinity and conductivity remain relatively constant as hydrological inputs offset the effects of evaporation. Later in the thaw-season concentrations of suspended solids decline as inflows cease and ion concentration, alkalinity and conductivity increase as evaporation plays a larger role in the water balance of the ponds. The fourth group of ponds is hydrologically connected to their basins throughout the thaw-season and has consistently high concentrations of suspended solids as allochthonous material is washed into the ponds. These ponds also have lower alkalinity, conductivity and ion concentrations as inflows act to offset the effects of evaporative concentration. As previously noted, the generalizations made between pond categories and basin and catchment characteristics are based on three ponds, one from each category except SWCT 2 (SWCT 1- Left Lake,, SWCT 3- Larch Lake, SWCT 4 – Erin Lake), that were intensively studied during the 2010 thaw-season. A more detailed study of basin and catchment characteristics of the other ponds within each group needs to be conducted to further validate these interpretations.

The size, depth and volume of water of the ponds were also found to influence pond water balance as these variables have a large effect on the amount of potential evaporation (Light, 2011; Wolfe et al., 2011). These physical characteristics were also found to have a direct effect on limnological conditions as the larger ponds shared similar limnological conditions during all the sampling periods in 2010.

These relationships are highly informative as they provide valuable insight into the hydrological factors that not only influence pond water balance but also influence the limnological conditions in the ponds.

2. How are the hydrolimnological conditions reflected in recently deposited sedimentary assemblages of diatoms and pigments?

Hydrolimnological variations within and among the ponds identified in Chapter 4 are not well captured by surface sediment diatom assemblages because microhabitat availability appears to be the dominant driver of community composition. Although there are differences among the ponds in hydrological and limnological conditions due to basin characteristics and hydrological connectivity, this has not been reflected in diatom and pigment community composition as the variations in hydrolimnological conditions among the ponds has not resulted in differences in microhabitat availability.

Nitrogen-limitation appears to be the factor that exerts the strongest influence on diatom and pigment composition and abundance. Nitrogen limitation favours the proliferation of the benthic mats consisting of nitrogen-fixing cyanobacteria. This has resulted in the relatively high concentrations of aphanizophyll in sediments of most ponds. The abundance of benthic habitat provided on the cyanobacterial mats under these conditions has resulted in the domination of diatom assemblages by *D. kuetzingii* in the sediments of most ponds due to their close association with these substrates. Ponds that varied from this pattern lacked *D. kuetzingii* or had *D. kuetzingii* in lower abundances along with small, benthic, adnate taxa that are associated with fine mineral grain substrates. A continued increase in nitrogen availability, due to the factors previously discussed, is expected to result in increased nitrogen availability in the ponds. This may drive a change in diatom and pigment community composition as reduced nitrogen limitation is likely to place nitrogen-fixing benthic cyanobacteria at a disadvantage and favour the growth of other algae and aquatic plants, which will alter microhabitat availability.

Spatial analysis of surface sediment diatom assemblages revealed that a pond which desiccated during the 2010 field season had taxa (*H. amphioxys*, *P. microstauron*) that have been associated with aerial or terrestrial habitats. In particular, increases in percent abundance of the desiccation-indicator species *H. amphioxys* can be used in future paleolimnological investigations in the HBL to identify periods when ponds became ephemeral due to changing environmental conditions.

Ponds located in the tundra ecozone were found to have higher concentrations of most photosynthetic pigments, likely due to physical and hydrological characteristics such as size, depth and volume of water and location which promote earlier ice-off, lengthen the growing season and increase water temperature.

In summary, the analysis of surface sediment diatom assemblages and pigments has permitted a better understanding of the factors that are driving the distribution of diatoms and pigments among the study ponds. This approach has helped to identify that habitat specificity is the main factor that drives variations in diatom and pigment community composition among the ponds. Many of the differences in habitat availability are mediated by hydrological and geochemical variations within the study ponds, especially changes in the severity of nitrogen limitation that regulates the degree of benthic cover provided by the nitrogen-fixing cyanobacterial mats. It was also found that surface sediment diatom assemblages reflected extreme hydrolimnological conditions (desiccation) in which aerophilic and terrestrial taxa were detected in pond sediments and that pigment concentrations were found to respond strongly to the physical (size, volume, location) characteristics of the ponds that resulted in a longer growing season and stimulated growth.

3. How have hydroecological conditions changed over the period represented within the sediment record. In particular, how have conditions changed over the past 50 years in response to climate warming? Can these changes be used to anticipate the future hydroecological conditions in these ponds under current climate change scenarios?

The divergent hydrological responses of the study ponds to 20th century climate warming, identified through the analysis of the oxygen isotope composition of aquatic cellulose (Light, 2011; Wolfe et al. 2011), are not reflected by similar timing of changes in diatom community composition in the sediment cores. The cellulose-inferred $\delta^{18}O_{PW}$ values from the Left Lake sediment record trend towards more enriched values since the beginning of the 20th century as Left Lake experienced greater evaporation due to increased temperatures and longer thaw-seasons. Left Lake is highly sensitive to these changing climatic conditions, as no increases in hydrological inputs have been observed to offset the increases in evaporation. This is due to Left Lake’s small catchment size and its isolation from other pond basins. Unlike Left Lake, cellulose-inferred $\delta^{18}O_{PW}$ values at Erin Lake have remained relatively unchanged over the 20th century due to a larger basin size and catchment area which allowed for sufficient rainfall inputs to offset evaporation. Erin Lake is also connected to a much larger upstream pond which provides additional inflows and a more stable water balance. The differences between the two ponds in hydrological conditions are not reflected in differences in diatom community assemblages because hydrological conditions are not driving sufficient changes in microhabitat availability. Microhabitat availability has been identified to be the main driver of diatom community composition in the HBL, which appears to be more strongly related to changes in nitrogen availability and the length of the growing season in the study ponds.

The biggest change in ecological conditions apparent in the sediment records from Left Lake and Erin Lake was the complete disappearance of the small, benthic, adnate taxon *S. pinnata* during the past ~150 years. This shift in diatom community structure occurred in Left Lake in the mid-19th century, which is congruent with similar changes in other Arctic lakes and coincides with the end of the "Little Ice-Age". The timing of the ecological shift in Erin Lake cannot be determined with any degree of accuracy due to diatom dissolution issues during this time period. These changes could be interpreted as an indicator of the climate warming signal as warmer temperatures result in a longer growing season, which allows for greater primary production leading to increased nutrient limitation. Increasing nitrogen limitation promoted proliferation of cyanobacterial mats on the pond bottoms, which effectively reduced the mineral grain and sediment habitat required for the growth of *S. pinnata*.

Increases in pigment accumulation rates coincided with increased concentrations of organic nitrogen in sediment cores from both ponds. The changes in nitrogen supply were postulated to be the result of three possible factors; increased atmospheric deposition of anthropogenic nitrogen, increased deposition of bird guano, and increased inflows from the catchment due to increased precipitation. The relative importance of these three sources of nitrogen to the ponds remains unknown and requires further investigation (Light, 2011). An increased supply of nitrogen to nitrogen-limited ecosystems is a plausible mechanism that accounts for the rise in aquatic productivity inferred by increased pigment fluxes and increased sedimentation rates at both ponds. Even with an increased nitrogen supply, the algal communities continue to be dominated by N_2-fixing cyanobacteria and associated biota which indicates that the study ponds remain nitrogen-limited and have not yet crossed thresholds to a different ecological state.

This study has identified that hydrological process not only influence pond water balance but also limnological conditions of the study ponds in the HBL. However, variations in hydrological conditions among the ponds do not appear to strongly determine surficial sediment diatom and pigment assemblages. The variations observed in the biological indicators among the ponds are influenced mainly by differences in microhabitat availability, which appears to be driven by nitrogen availability and the length of the growing season. For this reason, the divergent hydrological responses of Left Lake and Erin Lake to climate warming over the last century are not captured by floristic changes in diatoms or pigments in the sediment record. Instead, the paleoecological reconstruction of Left Lake and Erin Lake detected changes in diatom and pigment community composition that occurred ~1850 in relation to a change in microhabitat availability.

It is predicted that as climate warming continues in the HBL, limnological conditions will continue to change in response to changing water balances in the ponds. This will lead to very different conditions that may begin to exert an influence on the diatom and pigment assemblages in the ponds, as was observed in No 3 Lake in this study. Because the water balances of the study ponds have demonstrated an array of responses to climate warming due to different basin characteristics and hydrologic connectivity (Wolfe et al., 2011), a more complete understanding of how these hydrological factors influence the apparent drivers of ecological conditions (microhabitat availability, nitrogen availability) is required to inform decision makers in the CWMA. Although our understanding of these relationships has greatly improved since the onset of the studies of the Churchill ponds, there are still knowledge gaps that need to be addressed before we can predict what the final ecological fate of the shallow tundra ponds in the HBL in the wake of changing climate conditions. These knowledge gaps are presented in the next section (Recommendations).

7.2 Recommendations

The main findings from the study of the ponds in the Churchill Wildlife Management Area indicate that: 1) climatic and hydrological changes in the HBL lead to microhabitat changes that exert a strong influence on primary producer communities, and 2) ponds vary in the degree of nitrogen-limitation which markedly influences the composition of microhabitats (especially cyanobacterial mats). As these two factors appear to be the main drivers of ecological conditions in the ponds, a better understanding of how climatic and hydrological fluctuations affect N-limitation and growth of the cyanobacterial mats is required. Once this relationship is better understood, future ecological responses to changing climate and N-supply can be more accurately predicted which can then be used to guide management decisions in the HBL.

The relation between hydrological connectivity in the ponds and seasonal patterns of limnological conditions was another important finding from the study. In order to make more effective use of the information provided by the seasonal patterns of limnological conditions among the study ponds that were uncovered in Chapter 4, a more detailed analysis of hydrological connections between ponds and their contributing catchments needs to be conducted. In particular, better knowledge of lateral surface and subsurface flow is required. In order to better detect the hydrological connections between ponds, a parameter that only reflects allochthonous inputs is required. One possibility is the analysis of inorganic suspended solids (ISS) following treatment by dilute hydrochloric acid to remove carbonates, which may be reflective of in-pond processes, is one possibility. Tracking seasonal changes in ISS throughout the thaw-season may be an effective method of determining the timing and degree of hydrological connections between ponds and their basins. A detailed landscape survey may also better identify the connections which exist between basins. This information can then be used by decision makers to guide management decisions in the HBL, as ponds with different basin and catchment characteristics have been found to respond differently to climatic changes.

The study also identified a difference in limnological conditions between the ponds in the open-forest ecozone and the other two ecozones. However, because two of the three ponds in the open-forest category were physically different from the other ponds in the study these conclusions could not be validated. The addition of ponds in the open-forest ecozone that are more similar in physical characteristics of the other study ponds would allow for better assessment of hydrological, limnological and ecological conclusions across ecozones to be made. The addition of ponds to both the open-forest and forest-tundra transition zones, so that equal numbers of ponds are included in each category, would also improve the statistical power of the study. However, these changes may still not be sufficient to detect differences in hydroecological conditions across ecological zones as the spatial gradient would still be very small in comparison to other studies which have been able to detect these differences.

The analyses of living diatom communities collected from a range of microhabitats in ponds may be beneficial in order to further substantiate the associations that exist between diatom taxa and habitat specificity described in the study. This would provide a better understanding of the microhabitat requirements of taxa and improve paleolimnological interpretations from the ponds. If these investigations improve our knowledge of the relationships between diatom indicator species and habitat specificity, then sediment cores could be collected from a number of ponds to better assess the hydroecological shifts that are occurring in the Churchill ponds.

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Appendix A

Raw Limnology Data

Table A1 Raw limnology data from the 23 study ponds collected in June 2010

Pond	pH	Cond µsem	Alkalinity (mg/L)	TKN (mg/L)	NO_3^-+NO_2^- (mg/L)	NH_4^+ (mg/L)	TP (mg/L)	Na^+ (mg/L)	K^+ (mg/L)	Mg^{2+} (mg/L)	Ca^{2+} (mg/L)
Arc	8.88	273	101.66	0.577	0.020	0.158	0.009	8.58	2.73	6.34	12.96
Box	9.30	293	96.79	0.773	0.027	0.098	0.004	7.09	1.67	5.69	13.12
CoaL	9.03	241	66.78	0.449	0.022	0.109	0.006	10.59	2.99	5.64	9.72
CoaP	8.91	554	132.46	2.326	0.029	0.104	4.432	13.71	2.65	8.01	13.13
East	9.40	80	38.11	0.436	0.230	0.093	0.334	8.27	2.15	3.21	8.05
Erin	9.78	126	38.48	0.603	0.033	0.118	0.047	4.04	1.02	3.24	10.08
God	9.98	492	164.37	0.543	0.140	0.163	0.022	10.87	3.20	6.39	13.64
Golf	8.84	307	85.16	1.438	0.054	0.106	4.839	10.26	2.07	5.16	12.99
Hom	8.83	90	94.97	0.909	0.036	0.106	0.807	4.42	1.43	4.60	12.71
Isl	8.70	326	154.49	0.599	0.029	0.061	0.051	7.77	1.83	7.82	12.10
Lar	9.10	119	108.77	0.811	0.040	0.138	0.040	5.51	1.37	4.98	12.17
Left	7.94	177	67.98	0.906	0.026	0.164	0.025	4.56	1.38	4.33	13.48
Lum	8.88	197	86.63	0.642	0.170	0.098	0.005	7.18	1.87	3.63	9.91
No 3	8.80	261	107.08	0.950	0.043	0.164	0.028	6.02	1.95	5.61	13.45
Not	8.82	192	61.63	0.630	0.037	0.165	0.006	4.69	1.08	4.14	11.33
Ora	8.83	239	91.52	0.612	0.023	0.141	0.001	6.82	2.01	4.90	12.12
Plu	9.28	162	65.30	0.782	0.204	0.083	0.000	7.55	1.59	2.65	11.84
Rec	8.69	132	70.63	0.463	0.120	0.102	0.028	2.91	0.58	1.81	11.35
Ser	9.85	25	11.43	0.745	0.027	0.098	0.005	1.74	0.31	1.47	6.67
Some	7.24	213	90.00	0.654	0.020	0.091	0.000	5.06	1.25	5.25	13.60
Str	8.84	220	96.04	0.708	0.037	0.063	0.002	5.56	2.09	4.74	11.29
Swt	8.87	106	40.81	0.275	0.027	0.102	0.002	3.54	0.95	3.37	9.35
West	9.45	100	52.66	0.469	0.037	0.076	0.043	2.20	0.57	2.27	11.88

Pond	Cl^- (mg/L)	SO_4^- (mg/L)	Chl-a (mg/L)	TSS (mg/L)	ISS (mg/L)	AFDW (mg/L)	Water Temperature (°C)
Arc	21.96	6.46	1.54	1.5	0.3	1.2	11.93
Box	26.52	6.64	2.36	2.8	0.5	2.3	9.67
CoaL	31.74	9.56	0.78	15.0	7.2	7.8	3.50
CoaP	54.09	11.67	0.68	0.4	0.0	0.4	10.54
East	5.38	3.12	0.71	9.5	7.1	2.4	2.90
Erin	5.98	1.56	1.04	2.3	0.0	2.3	16.15
God	29.94	10.72	1.50	3.1	0.9	2.2	9.04
Golf	34.83	6.20	0.60	0.5	0.1	0.4	13.86
Hom	6.50	1.19	1.46	9.4	3.1	6.3	12.50
Isl	17.96	6.19	1.37	5.6	1.6	4.0	5.96
Lar	9.86	2.46	2.19	12.9	1.2	11.7	18.11
Left	7.34	1.72	2.21	7.8	1.3	6.5	15.78
Lum	10.80	4.99	1.31	1.1	0.2	0.9	12.58
No 3	12.02	6.63	2.45	2.5	0.7	1.8	19.50
Not	13.01	4.06	0.57	1.4	0.7	0.7	12.50
Ora	16.36	4.18	0.51	0.0	0.0	0.0	12.30
Plu	8.94	4.68	1.62	5.6	1.6	4.0	16.74
Rec	8.30	3.30	0.74	2.0	0.7	1.3	5.65
Ser	2.05	0.85	3.07	1.5	0.0	1.5	11.92
Some	9.51	3.63	1.02	1.2	0.0	1.2	8.70
Str	7.88	4.42	0.88	1.2	0.4	0.8	11.30
Swt	7.92	2.13	0.73	0.0	0.0	0.0	11.60
West	1.78	0.48	2.27	7.8	4.6	3.2	2.10

Table A2 Raw limnology data from the 23 study ponds collected in July 2010

Pond	pH	Cond µsem	Alkalinity (mg/L)	TKN (mg/L)	NO_3^-+NO_2^- (mg/L)	NH_4^+ (mg/L)	TP (mg/L)	Na^+ (mg/L)	K^+ (mg/L)	Mg^{2+} (mg/L)	Ca^{2+} (mg/L)
Arc	8.00	603	208.99	0.866	0.017	0.070	0.030	17.28	4.32	11.32	23.41
Box	8.96	371	121.60	1.766	0.016	0.067	0.002	12.12	3.18	9.48	15.99
CoaL	8.67	539	137.96	0.980	0.018	0.059	0.878	18.30	4.50	9.62	13.39
CoaP	8.59	997	159.01	2.009	0.020	0.113	0.005	26.81	4.51	13.47	18.14
East	8.63	108	49.35	1.064	0.017	0.062	0.009	2.17	0.49	4.29	14.19
Erin	8.89	154	70.93	1.257	0.015	0.078	1.155	5.06	2.03	5.11	15.23
God	8.70	670	197.62	1.056	0.015	0.047	0.954	20.07	5.64	12.05	12.46
Golf	8.62	520	121.24	1.143	0.016	0.088	1.041	17.76	2.55	8.67	21.59
Hom	9.00	217	118.89	1.073	0.016	0.087	0.971	6.32	1.16	7.58	17.77
Isl	8.37	547	229.05	1.393	0.016	0.075	0.071	12.08	2.23	11.19	16.51
Lar	8.87	274	131.70	1.315	0.018	0.121	0.090	8.17	1.54	7.52	11.32
Left	9.47	285	94.80	2.111	0.015	0.092	0.068	9.72	2.03	8.12	14.75
Lum	8.98	195	106.75	1.220	0.016	0.082	0.048	8.97	2.36	8.38	13.05
No 3	8.75	375	205.79	5.173	0.017	0.409	0.243	10.05	3.76	9.07	17.94
Not	8.73	298	104.34	0.982	0.025	0.062	0.041	5.78	0.82	4.72	12.39
Ora	8.08	473	197.05	2.833	0.016	0.068	0.101	10.23	1.62	7.15	11.28
Plu	8.87	217	105.08	1.039	0.017	0.072	0.081	5.95	1.59	7.03	15.78
Rec	8.79	316	125.84	1.038	0.022	0.124	0.046	8.82	2.13	8.84	10.76
Ser	8.43	114	44.61	0.701	0.016	0.106	0.048	3.19	0.56	4.20	15.81
Some	8.90	291	116.50	1.249	0.017	0.124	0.050	8.68	3.17	8.66	15.81
Str	9.07	255	108.69	0.972	0.017	0.121	0.030	8.78	2.44	7.70	17.09
Swt	9.14	180	77.57	0.922	0.018	0.070	0.030	5.74	1.20	5.74	11.01
West	8.90	141	61.72	0.266	0.016	0.062	0.014	2.98	0.71	4.65	15.01

Pond	Cl^- (mg/L)	SO_4^- (mg/L)	Chl-a (mg/L)	TSS (mg/L)	ISS (mg/L)	AFDW (mg/L)	Water Temperature (°C)	DOC (mg/L)	DIC (mg/L)	SiO_2 (mg/L)
Arc	88.71	23.70	2.36	3.8	0.7	3.1	11.2	42.6	4.97	6.34
Box	50.56	9.90	1.10	2.1	0.0	2.1	13.5	23.3	1.23	5.69
CoaL	86.52	22.96	1.46	1.8	0.1	1.7	8.4	28.4	1.98	5.64
CoaP	225.25	33.53	0.97	4.0	1.1	2.9	15.0	31.4	11.90	8.01
East	3.00	0.27	0.84	2.4	1.0	1.4	4.8	11.5	0.14	3.21
Erin	8.91	1.89	1.48	3.7	0.1	3.6	12.3	15.3	0.56	3.24
God	106.87	21.00	1.04	7.0	2.4	4.6	11.2	40.4	3.49	6.39
Golf	106.45	10.63	2.41	10.9	2.2	8.7	12.7	25.0	2.32	5.16
Hom	14.97	1.99	0.79	2.4	0.3	2.1	12.5	20.6	4.82	4.60
Isl	53.37	7.23	2.42	3.9	1.3	2.6	17.5	47.8	10.40	7.82
Lar	18.77	1.32	1.68	13.0	1.3	11.7	14.2	25.2	3.29	4.98
Left	37.38	9.74	0.34	3.2	0.4	2.8	22.0	16.2	4.47	4.33
Lum	29.28	10.59	1.39	9.7	2.5	7.2	12.2	21.5	3.38	3.63
No 3	31.10	14.27	8.84	46.4	18.5	27.9	33.9	40.8	0.49	5.61
Not	16.79	4.01	0.82	1.9	0.0	1.9	14.3	21.9	3.17	4.14
Ora	44.56	9.51	4.54	26.3	7.3	19.0	14.2	42.2	5.44	4.90
Plu	13.78	3.75	1.10	5.8	0.6	5.2	13.0	22.1	0.81	2.65
Rec	33.13	8.50	0.74	2.4	0.1	2.3	14.3	25.6	3.96	1.81
Ser	4.86	2.84	1.56	2.3	0.1	2.2	15.6	10.7	0.28	1.47
Some	31.94	12.56	1.68	7.5	1.5	6.0	14.1	23.1	2.39	5.25
Str	20.32	10.73	1.34	4.6	0.0	4.6	17.8	20.0	1.65	4.74
Swt	12.49	3.27	1.62	15.7	13.5	2.2	7.8	15.9	1.74	3.37
West	4.99	0.77	1.36	1.4	0.0	1.4	5.3	14.0	0.23	2.27

Table A3 Raw limnological data collected from the 23 study ponds in September 2010

Pond	pH	Cond µsem	Alkalinity (mg/L)	TKN (mg/L)	NO_3^-+NO_2^- (mg/L)	NH_4^+ (mg/L)	TP (mg/L)	Na^+ (mg/L)	K^+ (mg/L)	Mg^{2+} (mg/L)	Ca^{2+} (mg/L)
Arc	8.52	1110	189.80	0.437	0.018	0.062	0.023	14.29	3.52	9.51	16.49
Box	8.35	689	138.33	0.969	0.016	0.105	0.147	9.83	2.45	7.60	19.27
CoaL	8.45	978	152.54	0.639	0.017	0.055	0.025	14.08	3.31	8.62	20.24
CoaP	8.50	1219	187.20	0.434	0.017	0.093	0.018	16.97	2.96	9.55	22.49
East	8.10	272	47.52	0.446	0.019	0.074	0.029	2.36	0.28	4.09	10.55
Erin	7.93	323	66.16	0.931	0.017	0.102	0.038	4.44	0.72	4.35	13.89
God	8.25	659	201.85	0.828	0.017	0.048	0.042	13.81	3.75	9.63	16.83
Golf	8.51	1402	149.52	0.498	0.017	0.083	0.039	16.03	2.25	8.46	25.44
Hom	8.51	704	181.49	0.341	0.017	0.058	0.018	4.14	0.69	5.77	14.98
Isl	7.95	855	88.42	0.412	0.024	0.067	0.025	7.55	1.59	7.78	19.22
Lar	8.26	546	124.30	0.530	0.016	0.058	0.056	6.93	1.01	6.27	18.53
Left	8.17	630	122.91	0.922	0.016	0.057	0.053	7.66	1.52	7.32	18.01
Lum	8.54	607	152.41	0.664	0.020	0.069	0.089	7.52	1.67	7.81	17.34
No 3	7.83	797	162.32	0.654	0.016	0.048	0.139	8.91	2.20	8.26	19.37
Not	8.50	794	179.06	0.534	0.021	0.081	0.057	6.38	0.86	5.67	17.63
Ora	8.16	618	130.17	0.753	0.016	0.064	0.102	6.33	1.06	5.13	15.22
Plu	8.50	539	130.97	0.995	0.017	0.081	0.154	5.85	0.89	6.55	15.85
Rec	8.11	664	143.17	0.900	0.016	0.056	0.159	7.35	1.78	7.95	17.31
Ser	7.99	312	66.90	0.900	0.022	0.084	0.296	3.81	0.56	4.92	18.45
Some	8.21	687	140.62	1.288	0.020	0.102	0.229	7.12	1.52	7.32	19.60
Str	8.27	543	165.64	0.955	0.018	0.094	0.140	9.50	2.13	8.23	15.15
Swt	8.03	541	113.35	0.658	0.018	0.063	0.061	4.21	0.69	5.01	15.61
West	8.03	278	63.07	0.651	0.016	0.070	0.052	3.29	0.67	4.65	15.78

Pond	Cl^- (mg/L)	SO_4^- (mg/L)	Chl-a (mg/L)	TSS (mg/L)	ISS (mg/L)	AFDW (mg/L)	Water Temperature (°C)
Arc	127.42	39.27	1.68	3.3	0.9	2.4	7.11
Box	61.36	25.22	2.18	4.0	0.7	3.3	6.94
CoaL	80.76	28.85	0.94	3.1	1.0	2.1	7.31
CoaP	220.38	45.96	0.35	0.9	0.0	0.9	6.81
East	10.69	2.75	1.85	5.6	1.2	4.4	7.92
Erin	17.10	6.85	5.17	6.0	0.0	6.0	7.40
God	109.23	33.53	1.46	3.6	1.0	2.6	6.73
Golf	177.77	29.87	1.58	7.9	2.6	5.3	7.04
Hom	15.39	3.41	1.07	0.5	0.0	0.5	5.27
Isl	29.87	13.63	0.25	0.5	0.0	0.5	6.67
Lar	9.69	1.98	0.35	0.6	0.1	0.5	8.11
Left	50.65	16.31	0.53	1.4	0.0	1.4	6.65
Lum	15.22	6.78	1.48	4.3	1.0	3.3	5.58
No 3	16.72	8.64	2.60	10.0	1.6	8.4	7.41
Not	30.82	6.95	0.28	2.1	0.8	1.3	5.30
Ora	18.18	5.95	0.93	4.8	1.7	3.1	8.55
Plu	15.62	5.86	2.29	2.1	0.4	1.7	5.42
Rec	35.86	19.48	0.58	2.9	0.3	2.6	6.62
Ser	7.43	2.57	1.14	1.3	0.0	1.3	8.42
Some	16.24	7.55	3.20	8.6	2.0	6.6	6.73
Str	11.80	6.20	1.13	7.0	0.8	6.2	7.32
Swt	35.95	16.90	1.37	1.5	0.0	1.5	5.31
West	3.98	1.15	1.92	7.8	1.0	6.8	7.76

Appendix B

Surface Sediment Diatom Percent Abundance and Pigment Concentration Data

Table B1 Surface sediment diatom percent abundances from a spatial survey of Churchill Wildlife Management Area in 2010.

Taxon name	**Arc**	**Box**	**CoaL**	**CoaP**	**East**	**Erin**	**God**	**Golf**	**Hom**	**Isl**	**Lar**	**Left**
Achnanthes delicatula (Kützing) Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthes lanceolata (Brébisson ex Kützing) Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthes petersenii Hustedt	0.00	0.00	2.26	0.00	0.00	1.03	0.00	0.00	0.00	0.00	3.75	0.00
Achnanthidium deflexum Reimer	0.00	0.00	0.00	0.00	0.00	0.26	0.00	0.00	0.25	0.00	0.00	0.00
Achnanthidium minutissima var. *jackii* Rabhenhorst	0.00	0.00	0.00	0.00	0.00	0.26	0.00	0.00	0.00	0.00	1.07	0.00
Achnanthidium minutissima var. *minutissima* Kützing	7.93	1.64	6.77	7.24	5.30	5.43	3.63	2.59	9.41	9.77	6.97	1.07
Achnanthidium minutissima var. *macrocephala* Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthidium minutissima var. *saprophila* Kobaysi & Mayama	0.00	0.33	0.65	0.33	0.00	0.00	0.00	0.00	0.00	0.00	2.41	0.00
Achnanthidium minutissima var. *scotia* (J.R. Carter) Lange-Bertalot	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.80
Achnathidium reimeri (Camburn) Ponader & Potapova	0.00	0.00	0.32	0.00	0.00	0.00	0.99	0.00	0.00	0.00	0.00	0.00
Achnanthidium rivulare Potapova & Ponader	0.00	0.00	0.97	0.33	0.00	0.00	0.00	0.00	0.50	0.98	0.00	0.27
Amphora libyca Ehrenberg	0.00	0.00	0.32	0.00	0.00	0.00	0.00	0.29	0.00	0.00	1.61	0.00
Amphora veneta Kützing	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.53

Taxon name	**Arc**	**Box**	**CoaL**	**CoaP**	**East**	**Erin**	**God**	**Golf**	**Hom**	**Isl**	**Lar**	**Left**
Aneumastus tusculus (Ehrenberg) Mann & Stickle	0.00	0.99	0.32	1.64	0.00	0.00	1.65	0.86	0.25	0.98	0.80	0.53
Anomoeoneis brachysira (Brebisson) Grunow	8.28	0.00	0.97	0.00	0.00	0.00	0.33	0.00	0.25	0.65	0.00	0.00
Brachysira microcephala (Grunow) Compere	0.34	0.33	1.94	0.66	0.62	0.00	0.99	0.00	0.99	0.00	0.00	0.00
Caloneis alpestris (Grunow) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.00
Caloneis silicula (Ehrenberg) Cleve	0.69	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.65	0.54	0.00
Caloneis thermalis (Grunow) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cavinula cocconeiformis (Gregory in Greville) Mann & Stickle in Round, Crawford & Mann	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cocconeis placentula var. *euglypta* Ehrenberg	0.69	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.00
Craticula cuspidata (Kützing) Mann in Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.00
Cyclotella antigua Smith	0.00	0.00	0.65	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Cyclotella distinguenda Hustedt	0.00	0.00	0.32	0.00	0.31	1.55	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella affinis Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.98	0.00	0.00
Cymbella alpina Grunow	0.69	0.00	1.29	0.66	0.62	0.00	0.66	0.00	0.25	0.98	0.00	0.00
Cymbella amphicephala Nägeli	0.34	0.00	0.00	0.66	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**Arc**	**Box**	**CoaL**	**CoaP**	**East**	**Erin**	**God**	**Golf**	**Hom**	**Isl**	**Lar**	**Left**
Cymbella angustata Smith	3.79	0.00	1.29	4.28	0.93	0.52	0.99	0.29	2.23	5.54	1.34	0.27
Cymbella austriaca Grunow	0.00	0.00	0.32	0.00	0.00	0.00	0.99	0.00	0.00	0.98	0.54	0.00
Cymbella cymbiformis Agardh	0.69	0.00	0.00	0.66	0.00	0.00	0.00	0.00	0.00	0.00	1.61	0.00
Cymbella descripta (Hustedt) Krammer & Lange-Bertalot	0.00	0.00	1.29	0.00	0.00	0.52	0.00	0.29	0.00	0.00	1.07	0.00
Cymbella designata Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Cymbella delicatula Kützing	0.00	0.00	0.32	0.00	0.00	0.00	0.00	0.00	1.24	1.30	0.00	0.00
Cymbella falaisensis (Grunow) Krammer & Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella incerta (Grunow) Cleve	0.34	0.00	0.00	1.97	0.00	0.00	1.32	0.00	1.24	0.98	0.54	0.27
Cymbella latens Krasske	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella neocistula Krammer	0.69	0.00	0.00	0.66	0.00	0.00	0.99	0.00	0.00	0.33	1.07	0.00
Cymbella norvegica var. *norvegica* Grunow	3.45	0.00	0.00	0.66	0.93	0.26	0.00	0.00	0.25	1.95	0.00	0.00
Cymbella naviculacea Grunow	1.03	0.66	0.00	0.33	0.00	0.00	0.66	0.00	0.00	0.33	0.00	0.00
Cymbella paucistriata Cleve-Euler	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella sinuata Gregory	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella subaequalis Grunow	1.38	0.00	0.00	0.66	0.00	0.00	0.66	1.15	0.74	2.93	1.88	0.00

Taxon name	**Arc**	**Box**	**CoaL**	**CoaP**	**East**	**Erin**	**God**	**Golf**	**Hom**	**Isl**	**Lar**	**Left**
Cymbopleura lapponica Grunow ex Cleve	1.03	0.00	0.32	1.64	0.00	0.00	0.00	0.29	0.25	0.98	0.54	0.00
Cymbopleura naviculiformis (Auerswald in Heiberg) Krammer	0.34	0.00	0.00	0.00	0.31	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbopleura subcuspidata (Krammer) Krammer	0.00	0.00	0.00	0.00	0.00	0.52	0.00	0.00	0.00	0.00	0.54	0.00
Cymbopleura turgidula Grunow	0.69	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Denticula kuetzingii Complex	15.86	71.05	57.42	51.97	47.35	50.90	58.09	28.16	52.48	34.20	38.61	86.93
Denticula tenuis Kützing	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.98	0.00	0.00
Diatoma tenuis Agardh	5.17	0.00	3.23	0.66	9.03	0.00	0.00	1.72	1.49	0.00	0.00	0.00
Encyonema minutum (Hilse) Mann	0.00	0.33	0.32	1.32	0.93	0.78	0.00	0.00	1.73	0.00	1.34	0.00
Encyonema silesiacum (Bleish) Mann	0.34	0.00	0.32	0.00	0.00	0.26	0.99	0.00	0.25	0.00	0.80	0.00
Encyonema triangulum (Ehrenberg) Kützing	0.00	0.00	0.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonopsis cesatii (Rabenhorst) Krammer	0.34	0.00	0.00	0.99	0.93	0.26	0.99	0.00	0.25	3.91	0.00	0.00
Encyonopsis microcephala (Grunow) Krammer	3.79	2.96	1.94	2.30	1.56	9.56	1.65	0.29	7.67	1.95	1.07	2.13
Eunotia praerupta (Ehrenberg)	6.55	0.00	0.00	0.00	0.62	0.00	0.00	0.00	0.00	0.65	0.00	0.00
Epithemia adnata (Kützing) Brebisson	4.48	0.00	1.29	0.66	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**Arc**	**Box**	**CoaL**	**CoaP**	**East**	**Erin**	**God**	**Golf**	**Hom**	**Isl**	**Lar**	**Left**
Eucocconeis flexella (Kützing) Meister	0.34	0.00	1.61	0.33	2.49	0.52	0.00	0.86	0.00	3.58	0.00	0.00
Eucocconeis laevis (Ostrup) Lange-Bertalot	0.00	0.33	0.00	0.00	1.87	0.00	0.66	0.00	0.00	1.30	0.00	0.00
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	1.72	0.00	3.87	0.66	9.35	1.55	0.33	11.21	2.97	0.98	6.17	0.00
Fragilaria crotonensis Kitton	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria nanana Lange-Bertalot	1.38	0.00	0.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria tenera (Smith) Lange-Bertalot	1.72	2.30	0.00	2.63	1.25	0.52	0.66	2.01	1.73	0.33	2.41	0.00
Geissleria acceptata (Hustedt) Lange-Bertalot & Metzeltin	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema angustum Agardh	0.00	0.00	0.00	0.00	2.80	0.00	3.63	0.00	0.00	0.00	1.61	0.53
Gomphonema clavatum Ehrenberg	0.00	0.00	0.00	0.66	0.31	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema parvulum (Kützing) Kützing	0.00	0.00	0.97	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Hantzschia amphioxys (Ehrenberg) Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Kobayasiella jaagii (Meister) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.99	1.63	0.00	0.00
Kobayasiella subtilissima (Cleve) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**Arc**	**Box**	**CoaL**	**CoaP**	**East**	**Erin**	**God**	**Golf**	**Hom**	**Isl**	**Lar**	**Left**
Mastogloia elliptica (Agardh) Cleve	0.00	0.00	2.58	0.00	0.31	0.00	0.00	0.00	0.00	1.63	0.00	0.00
Mastogloia grevillei Smith	1.38	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.98	0.00	0.00
Mastogloia smithii Smith	6.90	0.00	0.00	3.29	0.00	0.00	1.98	0.00	0.00	3.58	0.00	0.00
Navicula cryptocephala Kützing	1.72	0.99	0.32	0.00	0.00	0.00	0.00	0.00	0.74	0.00	0.80	0.00
Navicula cryptotenella Lange-Bertalot	5.86	0.33	0.32	2.96	0.31	0.00	0.00	0.29	1.73	1.63	0.80	0.27
Navicula difficillima Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.00
Navicula halophila (Grunow) Cleve	0.69	0.00	0.00	0.99	0.00	0.00	0.33	0.29	0.00	0.00	0.00	0.00
Navicula halophilioides Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.00
Navicula lanceolata Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula libonensis Schoeman	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula radiosa Kützing	0.00	0.66	0.00	0.66	0.00	0.00	0.00	0.57	0.25	0.00	0.80	0.27
Navicula rhynchocephala Kützing	0.00	0.66	0.00	0.00	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.00
Navicula salinarum Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00
Navicula viridula var. *linearis* Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.65	0.00	0.00
Navicula vitiosa Schimanski	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.00
Navicula vulpina Kützing	1.03	0.33	0.32	0.33	0.93	0.26	1.65	1.44	0.99	0.65	0.00	0.53
Navicula wildii Lange-Bertalot	2.76	0.00	0.65	0.66	1.87	2.33	0.00	1.15	0.00	2.28	1.61	0.27

Taxon name	**Arc**	**Box**	**CoaL**	**CoaP**	**East**	**Erin**	**God**	**Golf**	**Hom**	**Isl**	**Lar**	**Left**
Neidium ampliatum (Ehrenberg) Krammer	0.00	0.00	0.00	0.99	0.00	0.00	0.66	0.57	0.00	1.30	0.54	0.00
Nitzschia bacillum Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia fonticola Grunow	0.00	0.00	0.97	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia frustulum (Kützing) Grunow	0.00	14.14	0.00	0.66	2.49	18.35	0.00	4.89	3.22	0.00	5.09	2.40
Nitzschia gracilis Hantzsch	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.54	0.00
Nitzschia inconspicua Grunow	0.00	0.00	0.00	0.00	0.00	1.81	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia intermedia Hantzsch	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia linearis Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia palea (Kützing) W. Smith	0.69	1.32	0.65	0.33	0.31	0.78	0.66	2.87	0.74	0.65	1.88	1.33
Nitzschia perminuta (Grunow) Peragallo	0.34	0.00	0.00	0.00	0.00	0.26	0.00	0.57	0.50	0.00	0.54	0.00
Nitzschia pura Hustedt	2.41	0.00	0.00	0.66	0.00	1.29	0.00	0.86	1.98	0.33	3.22	0.53
Nitzschia radicula Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nupela lapidosa (Krasske) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia gigas Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.74	0.00	0.80	0.80
Pinnularia maior (Kützing) Cleve	0.34	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.00	0.00
Pinnularia microstauron (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	2.97	0.00	0.00	0.00	0.00	0.00

Taxon name	**Arc**	**Box**	**CoaL**	**CoaP**	**East**	**Erin**	**God**	**Golf**	**Hom**	**Isl**	**Lar**	**Left**
Pinnularia sbrostrata (Cleve) Cleve-Euler	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Pinnularia viridis (Nitzsch) Ehrenberg	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Placoneis explanata (Hustedt) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Psammothidium bioretii (Germain) Bukhtiyarova & Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pseudostaurosira brevistriata (Grunow) Williams & Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rhoicospenia abbreviate (Agardh) Lange-Bertalot	0.69	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rhopalodia gibba (Ehrenberg) O. Muller	0.00	0.00	0.00	1.64	0.00	0.00	2.97	0.00	0.00	0.00	0.00	0.00
Sellaphora pupula (Kützing) Mereschkovsky	0.00	0.33	0.00	0.00	0.62	0.26	0.00	0.57	0.50	0.00	2.41	0.27
Stauroneis amphicephala Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**Arc**	**Box**	**CoaL**	**CoaP**	**East**	**Erin**	**God**	**Golf**	**Hom**	**Isl**	**Lar**	**Left**
Stauroneis phoenicenteron (Nitzsch) Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.54	0.00
Stauroneis subgracilis Lange-Bertalot & Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.57	0.00	0.98	0.80	0.00
Staurosira construens var. *venter* (Ehrenberg) Hamilton	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	0.00	0.00	2.26	0.00	3.74	0.00	5.61	31.32	0.00	2.93	0.00	0.00
Staurosirella pinnata var. *intercedens* (Grunow) Hamilton	0.00	0.00	0.00	0.00	0.93	0.00	0.33	2.87	0.00	0.98	0.00	0.00
Tabellaria flocculosa (Roth) Kützing	0.69	0.00	0.00	0.33	0.93	0.00	0.66	0.00	0.74	0.33	0.54	0.00
Tryblionella angustata Smith	0.00	0.00	0.00	0.66	0.00	0.00	0.33	0.00	0.50	0.33	0.00	0.00

Taxon name	**Lum**	**No 3**	**Not**	**Ora**	**Plu**	**Rec**	**Ser**	**Some**	**Str**	**Swt**	**West**
Achnanthes delicatula (Kützing) Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.88
Achnanthes lanceolata (Brébisson ex Kützing) Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.24
Achnanthes petersenii Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.20	0.00	0.32	0.36
Achnanthidium deflexum Reimer	0.00	0.00	2.43	0.29	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthidium minutissima var. *jackii* Rabhenhorst	0.00	3.63	0.00	0.00	0.00	0.00	0.00	0.48	0.00	0.00	0.00
Achnanthidium minutissima var. *minutissima* Kützing	5.92	20.13	3.65	7.23	5.97	7.62	21.05	4.34	5.67	1.28	5.76
Achnanthidium minutissima var. *macrocephala* Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	2.63	0.00	0.00	0.00	0.00
Achnanthidium minutissima var. *saprophila* Kobaysi & Mayama	0.31	0.00	0.00	0.00	0.00	1.99	1.50	0.00	0.00	0.00	2.52
Achnanthidium minutissima var. *scotia* (J.R. Carter) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.00
Achnathidium reimeri (Camburn) Ponader & Potapova	0.00	0.33	0.00	0.58	0.00	0.33	0.75	0.00	0.00	0.00	0.00
Achnanthidium rivulare Potapova & Ponader	0.00	0.33	0.00	0.29	1.57	0.66	45.11	0.00	2.27	0.96	19.42
Amphora libyca Ehrenberg	0.00	0.66	0.00	0.58	0.00	0.00	0.00	0.00	0.57	0.00	0.36
Amphora veneta Kützing	0.00	9.90	0.61	0.29	0.00	0.00	0.00	0.24	0.00	0.00	0.36

Taxon name	**Lum**	**No 3**	**Not**	**Ora**	**Plu**	**Rec**	**Ser**	**Some**	**Str**	**Swt**	**West**
Aneumastus tusculus (Ehrenberg) Mann & Stickle	0.00	1.32	1.52	0.29	0.31	0.00	0.00	0.24	1.98	0.00	0.00
Anomoeoneis brachysira (Brebisson) Grunow	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00	0.00	0.00	0.00
Brachysira microcephala (Grunow) Compere	0.62	0.00	0.91	0.58	0.63	0.33	2.26	0.48	0.00	0.64	1.44
Caloneis alpestris (Grunow) Cleve	0.00	0.00	0.00	0.00	0.63	0.00	0.00	0.00	0.00	0.00	0.00
Caloneis silicula (Ehrenberg) Cleve	0.00	0.00	0.91	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.08
Caloneis thermalis (Grunow) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.57	0.00	0.00
Cavinula cocconeiformis (Gregory in Greville) Mann & Stickle in Round, Crawford & Mann	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00	0.00	0.00	0.00
Cocconeis placentula var. *euglypta* Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.83
Craticula cuspidata (Kützing) Mann in Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cyclotella antigua Smith	0.00	0.00	0.30	0.00	0.00	1.32	0.00	0.00	0.00	0.32	0.00
Cyclotella distinguenda Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.48	0.00	1.92	0.00
Cymbella affinis Kützing	0.00	0.00	1.52	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella alpina Grunow	0.31	0.00	0.30	0.00	0.63	0.99	0.00	0.24	0.00	0.32	0.72
Cymbella amphicephala Nägeli	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**Lum**	**No 3**	**Not**	**Ora**	**Plu**	**Rec**	**Ser**	**Some**	**Str**	**Swt**	**West**
Cymbella angustata Smith	1.25	0.00	0.91	0.87	0.31	1.32	6.02	0.48	0.28	0.00	4.32
Cymbella austriaca Grunow	0.62	0.00	0.00	0.00	0.00	0.00	0.00	0.48	0.00	0.00	0.00
Cymbella cymbiformis Agardh	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.28	0.00	0.00
Cymbella descripta (Hustedt) Krammer & Lange-Bertalot	0.00	0.00	1.22	0.00	0.94	0.00	0.00	0.00	0.57	0.64	4.32
Cymbella designata Krammer	0.93	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella delicatula Kützing	0.31	0.66	0.30	0.00	0.00	1.32	0.00	0.00	0.00	0.00	0.00
Cymbella falaisensis (Grunow) Krammer & Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.00	0.00
Cymbella incerta (Grunow) Cleve	0.31	0.00	0.61	0.29	0.00	0.66	0.00	0.00	0.00	0.32	0.00
Cymbella latens Krasske	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella neocistula Krammer	0.31	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.28	0.00	0.00
Cymbella norvegica var. *norvegica* Grunow	0.00	0.00	0.30	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00
Cymbella naviculacea Grunow	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella paucistriata Cleve-Euler	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.48	0.00	0.00	0.00
Cymbella sinuata Gregory	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.04
Cymbella subaequalis Grunow	0.62	3.96	1.82	1.16	0.00	0.66	0.00	0.00	0.85	0.00	0.00
Cymbopleura lapponica Grunow ex Cleve	0.00	0.00	0.00	0.00	0.00	0.99	0.00	0.00	0.28	0.00	0.00
*Cymbopleura naviculiformis (*Auerswald in Heiberg) Krammer	0.00	0.00	0.61	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**Lum**	**No 3**	**Not**	**Ora**	**Plu**	**Rec**	**Ser**	**Some**	**Str**	**Swt**	**West**
Cymbopleura subcuspidata (Krammer) Krammer	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.00	0.00
Cymbopleura turgidula Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Denticula kuetzingii Complex	59.50	1.32	71.12	39.31	48.11	57.62	0.00	78.07	45.04	53.99	0.00
Denticula tenuis Kützing	0.31	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Diatoma tenuis Agardh	0.93	0.00	0.61	0.58	0.00	0.33	0.00	0.00	0.00	0.64	0.00
Encyonema minutum (Hilse) Mann	0.62	0.00	0.00	0.00	0.63	0.33	0.00	0.48	0.85	1.28	1.08
Encyonema silesiacum (Bleish) Mann	0.62	0.66	0.61	0.29	0.00	0.00	1.13	0.00	1.13	0.00	0.00
Encyonema triangulum (Ehrenberg) Kützing	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonopsis cesatii (Rabenhorst) Krammer	0.62	0.00	0.61	0.00	0.31	0.99	0.38	0.24	0.57	0.00	0.36
Encyonopsis microcephala (Grunow) Krammer	5.61	2.31	0.91	0.00	3.77	7.62	0.38	4.34	2.83	8.31	1.08
Eunotia praerupta (Ehrenberg)	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00
Epithemia adnata (Kützing) Brebisson	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.32	0.72
Eucocconeis flexella (Kützing) Meister	0.00	0.00	0.91	0.29	0.00	0.33	0.00	0.72	0.00	0.00	0.00
Eucocconeis laevis (Ostrup) Lange-Bertalot	0.00	3.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.08

Taxon name	**Lum**	**No 3**	**Not**	**Ora**	**Plu**	**Rec**	**Ser**	**Some**	**Str**	**Swt**	**West**
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	3.43	0.00	0.30	0.00	9.12	1.32	0.00	1.45	1.42	8.63	0.36
Fragilaria crotonensis Kitton	0.00	0.00	0.00	1.45	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria nanana Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.00	0.00
Fragilaria tenera (Smith) Lange-Bertalot	0.31	0.99	0.61	1.16	0.00	0.00	0.00	0.24	0.00	0.00	0.00
Geissleria acceptata (Hustedt) Lange-Bertalot & Metzeltin	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.72
Gomphonema angustum Agardh	1.56	0.00	0.30	0.00	1.26	0.00	0.75	0.00	0.85	0.00	0.00
Gomphonema clavatum Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema parvulum (Kützing) Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Hantzschia amphioxys (Ehrenberg) Grunow	0.00	4.29	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.00	0.00
Kobayasiella jaagii (Meister) Lange-Bertalot	0.00	0.00	0.61	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.36
Kobayasiella subtilissima (Cleve) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	2.26	0.00	0.00	0.00	0.00
Mastogloia elliptica (Agardh) Cleve	0.00	1.65	0.00	0.00	0.00	2.98	0.00	0.00	0.00	0.00	0.00
Mastogloia grevillei Smith	0.00	0.00	0.30	0.00	0.00	0.00	0.00	0.00	1.13	0.00	0.00
Mastogloia smithii Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula cryptocephala Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.57	0.00	0.00

Taxon name	**Lum**	**No 3**	**Not**	**Ora**	**Plu**	**Rec**	**Ser**	**Some**	**Str**	**Swt**	**West**
Navicula cryptotenella Lange-Bertalot	0.62	0.00	1.82	0.29	0.31	0.00	1.13	0.24	0.00	0.32	0.36
Navicula difficillima Hustedt	0.00	0.00	0.61	0.00	0.00	0.00	0.00	0.00	0.85	0.00	0.00
Navicula halophila (Grunow) Cleve	0.31	3.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula halophilioides Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.00
Navicula lanceolata Ehrenberg	0.31	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula libonensis Schoeman	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.00	0.00
Navicula radiosa Kützing	0.31	0.00	0.30	1.16	0.63	0.66	1.13	0.00	0.00	0.64	1.44
Navicula rhynchocephala Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula salinarum Grunow	0.00	1.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.36
Navicula viridula var. *linearis* Hustedt	0.00	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.57	0.00	0.00
Navicula vitiosa Schimanski	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula vulpina Kützing	0.31	0.00	0.61	0.58	0.00	0.66	0.00	0.72	1.13	0.64	2.16
Navicula wildii Lange-Bertalot	1.87	1.32	1.52	0.87	1.89	1.66	1.13	0.24	2.83	1.28	2.16
Neidium ampliatum (Ehrenberg) Krammer	0.00	2.64	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia bacillum Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	1.88	0.00	0.00	0.00	0.00
Nitzschia fonticola Grunow	0.00	0.00	0.00	0.00	0.00	0.00	7.52	0.00	0.00	0.00	5.40

Taxon name	**Lum**	**No 3**	**Not**	**Ora**	**Plu**	**Rec**	**Ser**	**Some**	**Str**	**Swt**	**West**
Nitzschia frustulum (Kützing) Grunow	6.23	0.00	0.00	0.00	15.09	1.66	0.75	2.17	20.11	13.42	15.83
Nitzschia gracilis Hantzsch	0.62	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia inconspicua Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.96	0.00
Nitzschia intermedia Hantzsch	0.00	1.32	0.00	0.00	0.00	0.00	0.00	0.00	1.13	0.00	0.00
Nitzschia linearis Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.80
Nitzschia palea (Kützing) W. Smith	1.87	8.25	0.00	1.16	3.46	0.00	1.13	1.20	1.98	0.32	0.00
Nitzschia perminuta (Grunow) Peragallo	0.00	0.66	0.00	1.16	1.26	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia pura Hustedt	0.00	5.28	0.00	0.29	0.63	0.00	0.38	0.00	0.85	1.92	0.00
Nitzschia radicula Hustedt	0.31	2.31	0.00	0.00	0.00	0.99	0.00	0.00	0.00	0.00	0.00
Nupela lapidosa (Krasske) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.00	0.00
Pinnularia gigas Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia maior (Kützing) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia microstauron (Ehrenberg) Cleve	0.00	17.82	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.00	0.00
Pinnularia sbrostrata (Cleve) Cleve-Euler	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia viridis (Nitzsch) Ehrenberg	0.00	0.00	0.00	1.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**Lum**	**No 3**	**Not**	**Ora**	**Plu**	**Rec**	**Ser**	**Some**	**Str**	**Swt**	**West**
Placoneis explanata (Hustedt) Lange-Bertalot	0.00	0.00	0.00	0.00	0.63	0.00	0.00	0.00	0.00	0.00	0.00
Psammothidium bioretii (Germain) Bukhtiyarova & Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.00	1.44
Pseudostaurosira brevistriata (Grunow) Williams & Round	0.00	0.00	0.00	1.73	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rhoicospenia abbreviate (Agardh) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rhopalodia gibba (Ehrenberg) O. Muller	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.57	0.00	0.00
Sellaphora pupula (Kützing) Mereschkovsky	0.00	0.00	0.00	0.29	0.31	0.00	0.00	0.00	0.28	0.00	0.00
Stauroneis amphicephala Kützing	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis phoenicenteron (Nitzsch) Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.57	0.00	0.00
Stauroneis subgracilis Lange-Bertalot & Krammer	0.00	0.00	0.00	0.00	0.63	0.00	0.00	0.00	0.57	0.00	0.00
Staurosira construens var. *venter* (Ehrenberg) Hamilton	0.00	0.00	0.00	0.87	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	1.87	0.00	0.00	34.10	0.00	0.00	0.00	0.00	0.00	0.00	3.60

Taxon name	**Lum**	**No 3**	**Not**	**Ora**	**Plu**	**Rec**	**Ser**	**Some**	**Str**	**Swt**	**West**
Staurosirella pinnata var. *intercedens* (Grunow) Frenguelli	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tabellaria flocculosa (Roth) Kützing	0.31	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00
Tryblionella angustata Smith	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00	0.00	0.00	0.00

Table B2 Pigment concentration data from the surface sediments collected from the study ponds in 2010. Pigment concentrations are expressed in nMol per gram organic matter.

	ChlC2	**Fuco**	**Aph**	**Diad**	**Myxo**	**Allo**	**Lut**	**Zea**	**Can**	**Chlb**	**Chla**	**Ech**	**Bcar**
Arc	20.56	285.18	254.13	0	27.54	45.63	164.19	89.56	31.58	0	736.4	84.19	66.19
Box	0	13.35	122.75	42.73	21.64	26.42	43.68	36.14	15.4	0	74.25	0	2.69
CoaP	0	160.45	197.41	99.97	76.39	74.13	77.4	65.24	28.25	0	429.54	73.86	49.13
Erin	0	4.17	6.54	0	0	1.5	6.32	3.19	2.42	1.32	12.47	1.6	1.17
God	0	6.75	368.71	0	40.97	29.65	45.11	46.4	11.25	0	216.2	76.87	49.05
Golf	10.19	99.11	264.34	65.69	59.13	25.05	62.58	55.1	41.76	22.51	312.31	40.12	49.88
Hom	7.18	64.15	382.86	102.8	61.29	23.35	79.29	74.34	20.44	0	216.22	40.83	7.05
Isl	0	51.5	526.05	93.7	64.08	51.02	154.13	73.6	54.88	42.25	392.52	162.98	146.71
Lar	0	30.5	261.41	65	40.03	8.41	90.09	47.71	25.53	39.25	204.32	19.55	24.95
Left	0	7.2	282.74	58.83	34.91	0	90.56	58.37	32.1	30.17	132.17	30.04	19.59
Lum	0	6.78	488.97	102.88	47.08	50.84	25.45	56.68	33.43	0	286.34	145.75	77.71
No 3	0	12.97	69.29	12.79	12.09	3.63	34.77	11.98	11.55	6.73	114.11	13	20.54
Not	0	57.79	482.81	106.51	81.68	44.31	77.9	85.36	39.12	15.12	320.45	91.48	74.75
Ora	0	49.39	133.78	46.28	30.13	39.69	85.14	39.85	19.65	17.59	191.9	16.84	14.47
Plu	0	16.59	158.39	42.11	27.7	7.83	57.78	36.86	19.08	10.16	110.06	16.68	26.98
Rec	1.59	57.31	491.77	111.95	66.38	20.83	64.67	89.05	35.78	0	526	124.3	86.87
Ser	0.58	12.94	130.21	33.51	28.36	4.88	65.01	34.23	0	7.8	185.72	21.78	25.13
Some	8.27	83.38	488.71	111.72	61	35.52	77.49	91.61	38.45	41.54	54.2	76.38	73.62
Str	0	5.33	88.76	28.28	15.05	5.81	41.66	26.28	19.1	9.17	86.24	14.98	20.52
Swt	0	5.55	219.58	35.88	18.56	0	27.32	25.5	14.06	4.91	149.01	53.48	31.77

Appendix C

Left Lake and Erin Lake Fossil Diatom Percent Abundance and Pigment Concentration Raw Data

Table C1 Sedimentary diatom percent abundance data from the Left Lake sediment core.

Taxon name	**0-0.5**	**1.0-1.5**	**2.0-2.5**	**3.0-3.5**	**4.0-4.5**	**5.0-5.5**	**6.0-6.5**	**7.0-7.5**	**8.0-8.5**	**9.0-9.5**
Achnanthidium deflexum Reimer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthidium minutissima var. *minutissima* Kützing	1.68	0.91	1.50	0.00	1.11	0.99	1.57	0.66	0.33	0.00
Achnanthidium rivulare Potapova & Ponader	0.67	0.91	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Amphora libyca Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Amphora veneta Kützing	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Aneumastus tusculus (Ehrenberg) Mann & Stickle	0.67	0.30	0.75	0.64	0.56	0.33	0.00	0.66	0.33	1.00
Caloneis silicula (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cocconeis placentula var. *euglypta* Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.00
Craticula cuspidata (Kützing) Mann in Round	0.34	0.00	0.00	0.32	0.00	0.00	0.00	0.00	0.00	0.00
Cyclotella distinguenda Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella alpina Grunow	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella angustata Smith	1.35	0.91	2.63	0.64	0.28	0.00	0.31	0.33	0.66	0.33
Cymbella aspera (Ehrenberg) Cleve	0.00	0.00	0.75	0.64	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella cymbiformis Agardh	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella neocistula Krammer	0.00	0.00	0.00	0.64	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**0-0.5**	**1.0-1.5**	**2.0-2.5**	**3.0-3.5**	**4.0-4.5**	**5.0-5.5**	**6.0-6.5**	**7.0-7.5**	**8.0-8.5**	**9.0-9.5**
Cymbella subaequalis Grunow	1.01	0.30	0.00	0.64	0.28	0.33	0.00	0.66	0.33	1.00
Cymbopleura lapponica Grunow ex Cleve	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbopleura subcuspidata (Krammer) Krammer	0.00	0.61	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Denticula kuetzingii Complex	71.72	78.79	72.93	85.26	89.72	83.55	89.94	81.46	76.24	84.95
Denticula tenuis Kützing	1.01	0.00	4.14	0.32	0.56	6.58	2.83	2.65	10.23	1.67
Encyonema minutum (Hilse) Mann	0.00	0.00	0.38	0.32	0.00	0.00	0.31	0.00	1.32	0.00
Encyonema silesiacum (Bleish) Mann	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonopsis cesatii (Rabenhorst) Krammer	2.36	1.21	1.13	0.32	1.11	0.00	0.31	0.33	0.33	0.00
Encyonopsis microcephala (Grunow) Krammer	1.01	1.52	0.75	1.92	0.83	0.00	1.57	3.64	1.98	3.68
Eunotia pectinalis (Dyllwyn) Rabenhorst	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eunotia praerupta (Ehrenberg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Epithemia adnata (Kützing) Brebisson	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Epithemia turgida (Ehrenberg) Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	0.00	0.00	0.00	0.00	0.56	0.66	0.00	0.00	0.00	0.00

Taxon name	**0-0.5**	**1.0-1.5**	**2.0-2.5**	**3.0-3.5**	**4.0-4.5**	**5.0-5.5**	**6.0-6.5**	**7.0-7.5**	**8.0-8.5**	**9.0-9.5**
Fragilaria crotonensis Kitton	0.00	0.00	0.75	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria exigua Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria tenera (Smith) Lange-Bertalot	1.01	0.00	0.00	0.00	0.00	0.99	0.00	1.32	0.00	0.00
Fragilaria ulna (Nitzsch) Lange-Bertalot	1.68	1.52	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema acuminatum Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema angustum Agardh	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema clavatum Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Navicula cryptocephala Kützing	1.01	0.61	1.88	0.64	0.83	0.00	0.00	0.00	0.00	0.00
Navicula cryptotenella Lange-Bertalot	2.02	1.52	2.26	0.64	1.39	0.99	0.31	0.66	0.66	0.33
Navicula radiosa Kützing	0.00	0.30	0.00	0.00	0.28	0.00	0.00	0.00	0.00	0.00
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Navicula viridula var. *linearis* Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.00
Navicula vulpina Kützing	0.67	1.52	1.88	0.96	0.83	0.99	0.00	0.00	0.99	1.67
Navicula wildii Lange-Bertalot	2.36	2.42	0.75	0.32	0.00	1.64	0.31	0.66	0.00	0.67
Neidium ampliatum (Ehrenberg) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**0-0.5**	**1.0-1.5**	**2.0-2.5**	**3.0-3.5**	**4.0-4.5**	**5.0-5.5**	**6.0-6.5**	**7.0-7.5**	**8.0-8.5**	**9.0-9.5**
Nitzschia bacillum Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia filiformis (Smith) Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00
Nitzschia frustulum (Kützing) Grunow	4.38	3.33	6.77	3.85	1.67	2.30	1.57	4.64	2.97	3.34
Nitzschia microcephala Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Nitzschia palea (Kützing) W. Smith	1.01	0.30	0.00	0.00	0.00	0.66	0.00	0.66	1.32	0.00
Nitzschia pura Hustedt	3.37	0.30	0.00	0.32	0.00	0.00	0.31	0.00	0.00	0.00
Nitzschia pusilla Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00
Pinnularia gigas Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia maior (Kützing) Cleve	0.00	0.00	0.00	0.32	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia microstauron (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia subgibba Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rhopalodia gibba (Ehrenberg) O. Muller	0.00	0.00	0.00	0.64	0.00	0.00	0.00	0.99	0.66	0.67
Sellaphora pupula (Kützing) Mereschkovsky	0.00	0.61	0.38	0.64	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis anceps Ehrenberg	0.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis phoenicenteron (Nitzsch) Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**0-0.5**	**1.0-1.5**	**2.0-2.5**	**3.0-3.5**	**4.0-4.5**	**5.0-5.5**	**6.0-6.5**	**7.0-7.5**	**8.0-8.5**	**9.0-9.5**
Stauroneis subgracilis Lange-Bertalot & Krammer	0.00	0.00	0.38	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tryblionella angustata Smith	0.00	0.61	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Achnanthidium deflexum Reimer	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthidium minutissima var. *minutissima* Kützing	0.00	0.00	1.32	0.67	0.67	0.00	0.00	0.00	1.35	0.33
Achnanthidium rivulare Potapova & Ponader	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Amphora libyca Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.68	0.33	0.00	0.00
Amphora veneta Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Aneumastus tusculus (Ehrenberg) Mann & Stickle	0.31	0.67	1.64	1.67	2.01	1.28	0.34	0.33	3.37	0.66
Caloneis silicula (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Cocconeis placentula var. *euglypta* Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Craticula cuspidata (Kützing) Mann in Round	0.00	0.00	0.00	0.33	0.00	0.51	0.34	0.00	0.00	0.00
Cyclotella distinguenda Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella alpina Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella angustata Smith	1.55	0.00	0.99	0.33	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella aspera (Ehrenberg) Cleve	0.31	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.67	0.33
Cymbella cymbiformis Agardh	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Cymbella neocistula Krammer	0.00	0.00	0.00	0.33	0.00	0.26	0.68	0.00	0.00	0.00
Cymbella subaequalis Grunow	0.62	0.67	0.99	0.00	0.00	0.26	0.34	2.01	0.00	0.00
Cymbopleura lapponica Grunow ex Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbopleura subcuspidata (Krammer) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Denticula kuetzingii Complex	78.95	77.59	75.66	74.33	82.61	85.13	88.44	90.64	88.89	85.25
Denticula tenuis Kützing	7.74	8.03	6.91	8.00	0.67	4.62	1.70	0.00	2.02	3.93
Encyonema minutum (Hilse) Mann	0.62	0.00	0.66	0.33	0.67	0.00	0.00	0.67	0.00	0.00
Encyonema silesiacum (Bleish) Mann	0.00	0.33	0.00	5.67	0.00	1.03	0.00	0.00	0.00	0.00

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Encyonopsis cesatii (Rabenhorst) Krammer	0.31	0.33	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonopsis microcephala (Grunow) Krammer	3.41	2.68	3.29	3.00	3.34	2.05	0.68	0.67	0.00	0.98
Eunotia pectinalis (Dyllwyn) Rabenhorst	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eunotia praerupta (Ehrenberg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Epithemia adnata (Kützing) Brebisson	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.00	0.00	0.00
Epithemia turgida (Ehrenberg) Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.00
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.64
Fragilaria crotonensis Kitton	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria exigua Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria tenera (Smith) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria ulna (Nitzsch) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema acuminatum Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema angustum Agardh	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Gomphonema clavatum Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula cryptocephala Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula cryptotenella Lange-Bertalot	0.62	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula radiosa Kützing	0.31	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.66
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula viridula var. *linearis* Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula vulpina Kützing	0.62	2.01	3.29	1.67	5.02	1.28	2.04	0.33	1.35	1.97
Navicula wildii Lange-Bertalot	0.62	0.67	0.33	0.00	0.33	0.00	0.34	0.00	0.00	0.00
Neidium ampliatum (Ehrenberg) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia bacillum Hustedt	0.00	0.00	0.00	0.33	0.00	0.00	0.34	0.00	0.34	0.00
Nitzschia filiformis (Smith) Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia frustulum (Kützing) Grunow	3.72	2.01	4.28	2.00	1.67	2.56	2.38	3.01	1.01	3.28
Nitzschia microcephala Grunow	0.00	1.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia palea (Kützing) W. Smith	0.00	0.00	0.00	0.00	0.00	0.26	0.00	0.00	0.00	0.00
Nitzschia pura Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia pusilla Grunow	0.31	0.00	0.00	0.33	0.00	0.26	0.00	0.00	0.00	0.00

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Pinnularia gigas Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia maior (Kützing) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia microstauron (Ehrenberg) Cleve	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia subgibba Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Rhopalodia gibba (Ehrenberg) O. Muller	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.00
Sellaphora pupula (Kützing) Mereschkovsky	0.00	2.34	0.00	0.00	1.00	0.51	0.68	0.67	0.34	0.33
Stauroneis anceps Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis phoenicenteron (Nitzsch) Ehrenberg	0.00	0.00	0.00	0.00	0.67	0.00	0.00	0.33	0.00	0.00
Stauroneis subgracilis Lange-Bertalot & Krammer	0.00	0.67	0.00	0.00	1.34	0.00	0.68	0.00	0.00	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Tryblionella angustata Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**20.0-20.5**	**21.0-21.5**	**22.0-22.5**	**23.0-23.5**	**24.0-24.5**	**25.0-25.5**	**26.0-26.5**	**27.0-27.5**	**28.0-28.5**	**29.0-29.5**
Achnanthidium deflexum Reimer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthidium minutissima var. *minutissima* Kützing	0.00	0.00	1.68	0.00	0.33	0.34	0.00	0.00	0.00	0.00
Achnanthidium rivulare Potapova & Ponader	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00
Amphora libyca Ehrenberg	0.00	0.00	0.00	0.00	0.33	0.34	0.00	0.00	0.00	0.00
Amphora veneta Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Aneumastus tusculus (Ehrenberg) Mann & Stickle	2.66	2.36	1.68	2.33	1.34	0.00	0.67	0.00	0.00	0.00
Caloneis silicula (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.67	0.00	0.00	0.00	0.00
Cocconeis placentula var. *euglypta* Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Craticula cuspidata (Kützing) Mann in Round	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00
Cyclotella distinguenda Hustedt	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella alpina Grunow	0.00	0.00	0.67	0.00	0.00	0.00	0.00	0.00	0.34	0.00
Cymbella angustata Smith	0.00	0.34	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00
Cymbella aspera (Ehrenberg) Cleve	0.00	0.00	0.34	0.33	0.00	1.01	0.00	0.33	0.00	0.00
Cymbella cymbiformis Agardh	0.00	0.34	0.34	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Cymbella neocistula Krammer	0.33	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.34	0.00

Taxon name	**20.0-20.5**	**21.0-21.5**	**22.0-22.5**	**23.0-23.5**	**24.0-24.5**	**25.0-25.5**	**26.0-26.5**	**27.0-27.5**	**28.0-28.5**	**29.0-29.5**
Cymbella subaequalis Grunow	0.66	0.00	0.67	0.67	0.67	0.00	0.67	0.33	0.68	0.00
Cymbopleura lapponica Grunow ex Cleve	0.00	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbopleura subcuspidata (Krammer) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Denticula kuetzingii Complex	86.38	86.20	85.57	88.00	91.97	85.19	79.93	77.59	85.47	83.28
Denticula tenuis Kützing	1.33	2.02	1.68	1.00	0.00	0.00	0.00	0.00	0.34	0.00
Encyonema minutum (Hilse) Mann	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonema silesiacum (Bleish) Mann	0.00	0.34	0.34	0.00	0.00	0.34	0.00	0.00	0.00	0.00
Encyonopsis cesatii (Rabenhorst) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Encyonopsis microcephala (Grunow) Krammer	2.99	0.00	1.34	1.67	0.67	0.00	0.67	0.00	0.00	0.00
Eunotia pectinalis (Dyllwyn) Rabenhorst	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eunotia praerupta (Ehrenberg)	0.00	0.00	0.00	0.00	0.00	0.67	0.00	0.00	0.00	0.00
Epithemia adnata (Kützing) Brebisson	0.33	0.00	0.34	0.00	0.00	0.00	0.33	0.00	0.00	0.00
Epithemia turgida (Ehrenberg) Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**20.0-20.5**	**21.0-21.5**	**22.0-22.5**	**23.0-23.5**	**24.0-24.5**	**25.0-25.5**	**26.0-26.5**	**27.0-27.5**	**28.0-28.5**	**29.0-29.5**
Fragilaria crotonensis Kitton	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria exigua Grunow	0.00	0.00	0.00	0.00	0.00	0.67	0.00	0.00	0.00	0.00
Fragilaria tenera (Smith) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria ulna (Nitzsch) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.67	0.00	0.00	0.00	0.00
Gomphonema acuminatum Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema angustum Agardh	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema clavatum Ehrenberg	0.00	0.00	0.34	0.33	0.00	0.34	0.00	0.00	0.00	0.00
Navicula cryptocephala Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula cryptotenella Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula radiosa Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula viridula var. *linearis* Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula vulpina Kützing	1.00	2.36	0.00	0.33	1.34	0.34	0.00	0.00	0.00	0.00
Navicula wildii Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Neidium ampliatum (Ehrenberg) Krammer	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00

Taxon name	**20.0-20.5**	**21.0-21.5**	**22.0-22.5**	**23.0-23.5**	**24.0-24.5**	**25.0-25.5**	**26.0-26.5**	**27.0-27.5**	**28.0-28.5**	**29.0-29.5**
Nitzschia bacillum Hustedt	0.00	0.00	0.00	1.00	0.33	0.00	0.00	0.00	0.00	0.00
Nitzschia filiformis (Smith) Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia frustulum (Kützing) Grunow	2.33	3.37	3.02	2.00	1.67	2.69	2.68	4.35	6.42	4.68
Nitzschia microcephala Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia palea (Kützing) W. Smith	0.00	0.00	0.34	0.00	0.00	0.34	0.00	0.00	0.34	0.00
Nitzschia pura Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia pusilla Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia gigas Ehrenberg	0.00	0.00	0.34	0.00	0.00	0.00	0.00	0.00	0.34	0.00
Pinnularia maior (Kützing) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia microstauron (Ehrenberg) Cleve	0.00	0.00	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia subgibba Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rhopalodia gibba (Ehrenberg) O. Muller	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Sellaphora pupula (Kützing) Mereschkovsky	1.99	0.34	0.00	1.67	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis anceps Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis phoenicenteron (Nitzsch) Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**20.0-20.5**	**21.0-21.5**	**22.0-22.5**	**23.0-23.5**	**24.0-24.5**	**25.0-25.5**	**26.0-26.5**	**27.0-27.5**	**28.0-28.5**	**29.0-29.5**
Stauroneis subgracilis Lange-Bertalot & Krammer	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	0.00	2.02	1.01	0.00	0.00	6.40	14.72	16.39	5.74	12.04
Tryblionella angustata Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00

Taxon name	30.0-30.5	31.0-31.5	32.0-32.5	33.0-33.5	34.0-34.5	35.0-35.5	36.0-36.5	37.0-37.5	38.0-38.5	39.0-39.5
Achnanthidium deflexum Reimer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthidium minutissima var. *minutissima* Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthidium rivulare Potapova & Ponader	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Amphora libyca Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.00
Amphora veneta Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Aneumastus tusculus (Ehrenberg) Mann & Stickle	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.00
Caloneis silicula (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.32	0.00	0.00	0.00	0.00	0.33

Taxon name	30.0-30.5	31.0-31.5	32.0-32.5	33.0-33.5	34.0-34.5	35.0-35.5	36.0-36.5	37.0-37.5	38.0-38.5	39.0-39.5
Cocconeis placentula var. *euglypta* Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Craticula cuspidata (Kützing) Mann in Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cyclotella distinguenda Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella alpina Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Cymbella angustata Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella aspera (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella cymbiformis Agardh	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00
Cymbella neocistula Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella subaequalis Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.00
Cymbopleura lapponica Grunow ex Cleve	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00
Cymbopleura subcuspidata (Krammer) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.67	0.00	0.00
Denticula kuetzingii Complex	83.95	87.54	90.46	88.45	93.57	93.31	88.00	80.00	81.13	84.33
Denticula tenuis Kützing	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.33	0.00	1.00
Encyonema minutum (Hilse) Mann	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonema silesiacum (Bleish) Mann	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	30.0-30.5	31.0-31.5	32.0-32.5	33.0-33.5	34.0-34.5	35.0-35.5	36.0-36.5	37.0-37.5	38.0-38.5	39.0-39.5
Encyonopsis cesatii (Rabenhorst) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Encyonopsis microcephala (Grunow) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	1.33
Eunotia pectinalis (Dyllwyn) Rabenhorst	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00
Eunotia praerupta (Ehrenberg)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Epithemia adnata (Kützing) Brebisson	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Epithemia turgida (Ehrenberg) Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria crotonensis Kitton	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria exigua Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria tenera (Smith) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria ulna (Nitzsch) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema acuminatum Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Gomphonema angustum Agardh	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	30.0-30.5	31.0-31.5	32.0-32.5	33.0-33.5	34.0-34.5	35.0-35.5	36.0-36.5	37.0-37.5	38.0-38.5	39.0-39.5
Gomphonema clavatum Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula cryptocephala Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula cryptotenella Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Navicula radiosa Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula viridula var. *linearis* Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula vulpina Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Navicula wildii Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.67
Neidium ampliatum (Ehrenberg) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia bacillum Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia filiformis (Smith) Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia frustulum (Kützing) Grunow	9.03	9.76	6.46	8.58	4.82	3.34	8.00	9.33	3.97	3.67
Nitzschia microcephala Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia palea (Kützing) W. Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00
Nitzschia pura Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia pusilla Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	30.0-30.5	31.0-31.5	32.0-32.5	33.0-33.5	34.0-34.5	35.0-35.5	36.0-36.5	37.0-37.5	38.0-38.5	39.0-39.5
Pinnularia gigas Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia maior (Kützing) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Pinnularia microstauron (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.32	0.00	0.00	0.33	0.00	0.00
Pinnularia subgibba Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rhopalodia gibba (Ehrenberg) O. Muller	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Sellaphora pupula (Kützing) Mereschkovsky	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis anceps Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis phoenicenteron (Nitzsch) Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis subgracilis Lange-Bertalot & Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	6.69	2.69	3.08	1.98	0.96	3.34	3.38	8.67	11.59	7.33
Tryblionella angustata Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table C2 Diatom percent abundance data from the Erin Lake sediment core.

Taxon name	**0-0.5**	**1.0-1.5**	**2.0-2.5**	**3.0-3.5**	**4.0-4.5**	**5.0-5.5**	**6.0-6.5**	**7.0-7.5**	**8.0-8.5**	**9.0-9.5**
Achnanthidium minutissima var. *minutissima* Kützing	1.92	6.08	2.19	5.94	2.42	2.48	1.63	0.97	2.35	1.93
Achnanthidium rivulare Potapova & Ponader	0.32	0.34	0.73	0.00	0.30	0.00	0.00	0.65	0.00	1.29
Amphora libyca Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Aneumastus tusculus (Ehrenberg) Mann & Stickle	0.64	0.00	0.36	0.66	0.00	0.62	0.65	0.32	0.00	0.96
Brachysira microcephala (Grunow) Compere	0.64	0.68	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00
Cocconeis placentula var. *euglypta* Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Craticula cuspidata (Kützing) Mann in Round	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00
Cyclotella antigua Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cyclotella distinguenda Hustedt	0.64	2.36	0.73	4.62	1.21	0.62	0.65	0.65	0.67	1.29
Cymbella angustata Smith	0.00	0.68	2.55	0.66	1.51	0.31	0.98	0.32	1.01	0.00
Cymbella aspera (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00	0.00
Cymbella neocistula Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella subaequalis Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**0-0.5**	**1.0-1.5**	**2.0-2.5**	**3.0-3.5**	**4.0-4.5**	**5.0-5.5**	**6.0-6.5**	**7.0-7.5**	**8.0-8.5**	**9.0-9.5**
Denticula kuetzingii Complex	43.27	39.19	56.93	50.17	53.17	55.28	54.07	43.55	48.99	55.95
Denticula tenuis Kützing	0.64	1.01	0.36	0.00	0.00	0.00	0.00	0.00	1.34	0.64
Encyonema minutum (Hilse) Mann	0.96	1.01	0.73	0.33	0.00	0.00	0.65	0.00	1.01	0.32
Encyonema silesiacum (Bleish) Mann	0.00	0.00	0.00	0.00	0.30	0.00	0.00	0.00	0.00	0.00
Encyonopsis cesatii (Rabenhorst) Krammer	0.00	0.68	1.09	0.66	0.91	1.86	0.00	0.00	0.00	0.00
Encyonopsis microcephala (Grunow) Krammer	14.74	15.54	9.85	9.24	11.78	6.21	10.42	15.48	6.38	7.07
Epithemia adnata (Kützing) Brebisson	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eucocconeis flexella (Kützing) Meister	0.32	0.00	0.00	0.00	0.00	0.62	0.00	0.32	0.00	0.00
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	3.53	5.07	1.82	0.33	1.51	3.11	3.26	6.45	5.70	5.14
Fragilaria ulna (Nitzsch) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.01	0.00
Gomphonema angustum Agardh	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema Spp. 1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Kobayasiella jaagii (Meister) Lange-Bertalot	0.00	0.00	0.36	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**0-0.5**	**1.0-1.5**	**2.0-2.5**	**3.0-3.5**	**4.0-4.5**	**5.0-5.5**	**6.0-6.5**	**7.0-7.5**	**8.0-8.5**	**9.0-9.5**
Navicula cryptocephala Kützing	0.32	1.01	0.73	0.66	0.60	0.62	0.98	1.29	0.67	0.64
Navicula cryptotenella Lange-Bertalot	0.32	0.00	0.36	0.00	0.30	0.00	0.65	0.65	0.67	0.32
Navicula radiosa Kützing	0.00	0.00	0.00	0.00	0.30	0.00	0.65	0.00	0.00	0.96
Navicula rostellata Kützing	0.00	0.68	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula vulpina Kützing	0.32	0.34	1.46	3.30	1.51	2.17	1.95	2.26	4.36	1.29
Navicula wildii Lange-Bertalot	1.60	1.35	1.09	0.66	1.51	1.24	1.95	1.29	1.34	0.96
Neidium ampliatum (Ehrenberg) Krammer	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia frustulum (Kützing) Grunow	28.85	21.96	18.25	18.15	20.54	21.43	19.87	22.90	21.48	20.26
Nitzschia palea (Kützing) W. Smith	0.00	0.68	0.00	0.99	0.30	0.31	0.00	0.00	0.00	0.00
Nitzschia perminuta (Grunow) Peragallo	0.96	0.34	0.00	0.00	1.21	0.00	0.00	0.97	0.67	0.00
Nitzschia pura Hustedt	0.00	0.00	0.36	2.64	0.00	0.62	0.65	0.97	0.34	0.32
Nitzschia pusilla Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia microstauron (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.32	0.00	0.64

Taxon name	**0-0.5**	**1.0-1.5**	**2.0-2.5**	**3.0-3.5**	**4.0-4.5**	**5.0-5.5**	**6.0-6.5**	**7.0-7.5**	**8.0-8.5**	**9.0-9.5**
Sellaphora pupula (Kützing) Mereschkovsky	0.00	0.68	0.00	0.33	0.60	1.55	0.33	0.00	1.01	0.00
Stauroneis anceps Peragello	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis gracilior Reichardt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.00	0.00
Stauroneis subgracilis Lange-Bertalot & Krammer	0.00	0.34	0.00	0.00	0.00	0.62	0.00	0.32	0.00	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Staurosirella pinnata var. *intercedens* (Grunow) Hamilton	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tabellaria flocculosa (Roth) Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Achnanthidium minutissima var. *minutissima* Kützing	2.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthidium rivulare Potapova & Ponader	1.35	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Amphora libyca Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Aneumastus tusculus (Ehrenberg) Mann & Stickle	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Brachysira microcephala (Grunow) Compere	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cocconeis placentula var. *euglypta* Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Craticula cuspidata (Kützing) Mann in Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cyclotella antigua Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cyclotella distinguenda Hustedt	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella angustata Smith	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella aspera (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella neocistula Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cymbella subaequalis Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Denticula kuetzingii Complex	48.82	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Denticula tenuis Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonema minutum (Hilse) Mann	1.35	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonema silesiacum (Bleish) Mann	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonopsis cesatii (Rabenhorst) Krammer	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonopsis microcephala (Grunow) Krammer	12.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Epithemia adnata (Kützing) Brebisson	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eucocconeis flexella (Kützing) Meister	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	2.36	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria ulna (Nitzsch) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema angustum Agardh	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gomphonema Spp. 1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Kobayasiella jaagii (Meister) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula cryptocephala Kützing	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula cryptotenella Lange-Bertalot	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula radiosa Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula rostellata Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula vulpina Kützing	1.35	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula wildii Lange-Bertalot	1.68	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Neidium ampliatum (Ehrenberg) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia frustulum (Kützing) Grunow	23.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia palea (Kützing) W. Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia perminuta (Grunow) Peragallo	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia pura Hustedt	1.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia pusilla Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pinnularia microstauron (Ehrenberg) Cleve	1.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Sellaphora pupula (Kützing) Mereschkovsky	0.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis anceps Peragello	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis gracilior Reichardt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis subgracilis Lange-Bertalot & Krammer	0.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**10.0-10.5**	**11.0-11.5**	**12.0-12.5**	**13.0-13.5**	**14.0-14.5**	**15.0-15.5**	**16.0-16.5**	**17.0-17.5**	**18.0-18.5**	**19.0-19.5**
Staurosirella pinnata var. *intercedens* (Grunow) Hamilton	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Tabellaria flocculosa (Roth) Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**20.0-20.5**	**21.0-21.5**	**22.0-22.5**	**23.0-23.5**	**24.0-24.5**	**25.0-25.5**	**26.0-26.5**	**27.0-27.5**	**28.0-28.5**	**29.0-29.5**
Achnanthidium minutissima var. *minutissima* Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Achnanthidium rivulare Potapova & Ponader	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Amphora libyca Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.00	0.33
Aneumastus tusculus (Ehrenberg) Mann & Stickle	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Brachysira microcephala (Grunow) Compere	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cocconeis placentula var. *euglypta* Ehrenberg	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Craticula cuspidata (Kützing) Mann in Round	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.00	0.00
Cyclotella antigua Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Cyclotella distinguenda Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00

Taxon name	**20.0-20.5**	**21.0-21.5**	**22.0-22.5**	**23.0-23.5**	**24.0-24.5**	**25.0-25.5**	**26.0-26.5**	**27.0-27.5**	**28.0-28.5**	**29.0-29.5**
Cymbella angustata Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.66
Cymbella aspera (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Cymbella neocistula Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.65	0.00	0.00	0.00
Cymbella subaequalis Grunow	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.00
Denticula kuetzingii Complex	0.00	0.00	0.00	0.00	0.00	33.22	23.53	23.15	31.67	34.44
Denticula tenuis Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonema minutum (Hilse) Mann	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Encyonema silesiacum (Bleish) Mann	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonopsis cesatii (Rabenhorst) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Encyonopsis microcephala (Grunow) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.99
Epithemia adnata (Kützing) Brebisson	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33
Eucocconeis flexella (Kützing) Meister	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.33
Fragilaria ulna (Nitzsch) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**20.0-20.5**	**21.0-21.5**	**22.0-22.5**	**23.0-23.5**	**24.0-24.5**	**25.0-25.5**	**26.0-26.5**	**27.0-27.5**	**28.0-28.5**	**29.0-29.5**
Gomphonema angustum Agardh	0.00	0.00	0.00	0.00	0.00	0.00	0.65	0.00	0.00	0.00
Gomphonema Spp. 1	0.00	0.00	0.00	0.00	0.00	0.00	0.65	0.00	0.00	0.00
Kobayasiella jaagii (Meister) Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula cryptocephala Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.64	0.00	0.00
Navicula cryptotenella Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula radiosa Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula rostellata Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.00
Navicula vulpina Kützing	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Navicula wildii Lange-Bertalot	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.64	0.00	0.00
Neidium ampliatum (Ehrenberg) Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia frustulum (Kützing) Grunow	0.00	0.00	0.00	0.00	0.00	0.00	0.33	2.57	3.00	9.60
Nitzschia palea (Kützing) W. Smith	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia perminuta (Grunow) Peragallo	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Taxon name	**20.0-20.5**	**21.0-21.5**	**22.0-22.5**	**23.0-23.5**	**24.0-24.5**	**25.0-25.5**	**26.0-26.5**	**27.0-27.5**	**28.0-28.5**	**29.0-29.5**
Nitzschia pura Hustedt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitzschia pusilla Grunow	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00
Pinnularia microstauron (Ehrenberg) Cleve	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.33	0.00
Sellaphora pupula (Kützing) Mereschkovsky	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.00	0.00
Stauroneis anceps Peragello	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00
Stauroneis gracilior Reichardt	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stauroneis subgracilis Lange-Bertalot & Krammer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.00	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	0.00	0.00	0.00	0.00	0.00	64.80	57.52	59.81	56.00	50.00
Staurosirella pinnata var. *intercedens* (Grunow) Hamilton	0.00	0.00	0.00	0.00	0.00	0.00	15.69	9.97	7.33	2.98
Tabellaria flocculosa (Roth) Kützing	0.00	0.00	0.00	0.00	0.00	0.33	0.00	1.61	0.00	0.00

Taxon name	30.0-30.5	31.0-31.5
Achnanthidium minutissima var. *minutissima* Kützing	0.00	0.00
Achnanthidium rivulare Potapova & Ponader	0.00	0.00
Amphora libyca Ehrenberg	0.00	0.33
Aneumastus tusculus (Ehrenberg) Mann & Stickle	0.00	0.00
Brachysira microcephala (Grunow) Compere	0.00	0.00
Cocconeis placentula var. *euglypta* Ehrenberg	0.33	0.00
Craticula cuspidata (Kützing) Mann in Round	0.00	0.00
Cyclotella antigua Smith	0.00	0.00
Cyclotella distinguenda Hustedt	0.00	0.00
Cymbella angustata Smith	0.00	0.00
Cymbella aspera (Ehrenberg) Cleve	0.00	0.00
Cymbella neocistula Krammer	0.00	0.33
Cymbella subaequalis Grunow	1.63	0.00
Denticula kuetzingii Complex	20.85	22.33

Taxon name	30.0-30.5	31.0-31.5
Denticula tenuis Kützing	0.00	0.00
Encyonema minutum (Hilse) Mann	0.00	0.00
Encyonema silesiacum (Bleish) Mann	0.00	0.00
Encyonopsis cesatii (Rabenhorst) Krammer	0.00	0.00
Encyonopsis microcephala (Grunow) Krammer	0.33	0.00
Epithemia adnata (Kützing) Brebisson	0.00	0.00
Eucocconeis flexella (Kützing) Meister	0.00	0.00
Fragilaria capucina var. *vaucheriae* (Kützing) Lange-Bertalot	0.00	0.00
Fragilaria ulna (Nitzsch) Lange-Bertalot	0.00	0.00
Gomphonema angustum Agardh	0.65	0.00
Gomphonema Spp. 1	0.00	0.00
Kobayasiella jaagii (Meister) Lange-Bertalot	0.00	0.00
Navicula cryptocephala Kützing	0.00	0.00

Taxon name	30.0-30.5	31.0-31.5
Navicula cryptotenella Lange-Bertalot	0.33	0.00
Navicula radiosa Kützing	0.00	0.00
Navicula rostellata Kützing	0.00	0.00
Navicula tripunctata (O.F. Müller) Bory de Saint-Vincent	0.00	0.00
Navicula vulpina Kützing	0.00	0.00
Navicula wildii Lange-Bertalot	0.65	0.00
Neidium ampliatum (Ehrenberg) Krammer	0.00	0.00
Nitzschia frustulum (Kützing) Grunow	3.91	2.33
Nitzschia palea (Kützing) W. Smith	0.00	0.00
Nitzschia perminuta (Grunow) Peragallo	0.00	0.00
Nitzschia pura Hustedt	0.00	0.00
Nitzschia pusilla Grunow	0.00	0.00
Pinnularia microstauron (Ehrenberg) Cleve	0.00	0.00

Taxon name	30.0-30.5	31.0-31.5
Sellaphora pupula (Kützing) Mereschkovsky	0.00	0.00
Stauroneis anceps Peragello	0.00	0.00
Stauroneis gracilior Reichardt	0.00	0.00
Stauroneis subgracilis Lange-Bertalot & Krammer	0.00	0.00
Staurosirella pinnata (Ehrenberg) Williams & Round	61.24	64.67
Staurosirella pinnata var. *intercedens* (Grunow) Hamilton	10.10	9.67
Tabellaria flocculosa (Roth) Kützing	0.00	0.33

Table C3 Pigment concentration data from the Left Lake sediment core collected in 2009. Pigment concentrations are expressed in nMol per gram organic matter.

Depth	ChlC2	Fuco	Aph	Diad	Myxo	Allo	Lut	Zea	Can	Chlb	Chla	Ech	Bcar
0	0.0	16.85	216.26	33.77	0.73	10.81	32.57	40.09	14.19	0	142.04	28.53	31.74
1	0.0	14.8	189.23	27.04	3.64	10.57	28.96	30.23	11.66	0	121.88	23.48	24.19
2	0.0	13.34	184.5	26.81	4.54	12.3	31.92	29.63	14.01	0	125.46	25.03	28.2
3	0.0	3.11	195.75	28.43	4.52	12.44	37.63	29.59	14.17	0	136.14	23.78	26.46
4	0.0	5.92	177.76	28.38	5.58	14.47	37.62	31.44	14.42	0	125.92	18.88	16.83
5	0.0	2.62	176.82	30.18	5.17	14.07	41.52	32.16	13.9	0	137.32	20.09	15.83
6	0.0	5.95	154.72	28.27	4.72	13.67	41.32	30.15	14.48	0	136.75	20.71	18.44
7	0.0	0.53	149.41	27.17	4.3	12.63	42.04	29.4	14.05	0	124.29	19.17	17.87
8	0.0	39.43	254.31	26.83	9.82	18.72	45.5	36.03	19.24	9.57	82.41	11.78	9.59
9	0.0	15.02	263.42	26.91	9.92	19.54	47.98	36.28	20.64	12.06	78.54	11.74	8.97
10	0.0	2.8	164.58	23.97	5.07	14.23	58.14	29.43	17.82	8.55	77.89	9.91	7.09
11	0.0	2.95	191.06	26.53	5.77	14.55	65.5	20.43	18.75	9.63	88.13	11.44	9.08
12	0.0	3.5	192.42	26.19	6.01	16.37	74.04	14.12	20.93	12.99	91.6	11.9	9.71
13	0.0	3.16	199.36	26.14	6.11	15.38	79.86	13.61	21.95	13.44	88.62	12.07	9.28
14	0.0	2.7	195.65	24.1	5.73	13.05	82.35	11.66	21.49	16.83	73.4	10.26	7.86
15	0.0	1.78	186.14	23.27	5.43	11.24	85.31	10.45	21.56	14.75	69.61	10.13	7.5
16	0.0	2.63	211.17	24.97	6.1	12.62	97.53	10.99	24.9	19.07	75.67	10.64	8.12
17	0.0	2.82	209.25	24.68	6.37	12.46	95.29	10.9	23.41	17.77	69.48	9.65	7.2
18	0.0	2.72	219.24	27.32	7.07	19.11	107.06	12.75	27.96	17.07	76.59	10.45	6.87
19	0.0	4.21	395.62	44.12	9.7	27.92	157.19	57.32	29.5	10.17	46.6	6.72	4.12

Depth	ChlC2	Fuco	Aph	Diad	Myxo	Allo	Lut	Zea	Can	Chlb	Chla	Ech	Bcar
20	0.0	0	494.59	59.26	15.89	23.49	181.75	63.41	37.12	15.27	109.55	33.26	21.36
21	0.0	2.56	490.7	62.12	16	23.64	188.03	25.61	40.53	26.51	148.62	38.6	24.08
22	0.0	37.4	589.87	67.34	18.24	26.42	187.89	30.21	38.38	34.06	149.9	23.1	25.25
23	0.0	7.02	549	66.45	15.51	26.74	187.31	31.34	36.98	33.38	154.46	22.67	22.21
24	0.0	24.27	597.22	69.1	18.01	28.22	194.75	72.83	37.62	27.25	142.15	21.46	22.69
25	0.0	49.99	621.83	68.85	20.19	30.53	193.69	75.6	38.16	22.75	132.35	19.54	20.56
26	0.0	3.2	575.35	67.6	16.6	31.42	220.7	76.04	41.95	41.55	150.26	22.35	30.27
27	0.0	26.17	774.99	64	27.35	41.92	179.23	87.93	47.64	53.76	116.15	17.95	22.79
28	0.0	2.35	550.9	63.81	14.1	31.25	198.61	76.56	32.83	34.2	160.63	20.34	31.8
29	0.0	2.8	632.88	75.01	15.86	38.27	216.09	92.09	37.57	26.04	191.28	24.49	39.99
30	0.0	27.57	779.05	66.39	24.85	45.6	164.36	91.86	42.19	36.64	126.41	18.94	27.3
31	0.0	3.6	730.2	91.87	19.2	39.02	229.68	106.39	36.39	27.4	210.45	22.76	32.3
32	0.0	12.64	744.34	93.84	16.1	39.85	241.7	106.9	32.2	24.53	219.43	27.71	47.83
33	0.0	1.98	587.18	72.71	15.01	39.44	204.97	90.34	37.17	25.53	199.46	22.25	31.49
34	0.0	0	659.94	80.6	12.35	40.67	243.4	95.34	34.98	31.02	237.78	26.9	50.01
35	0.0	3.06	679.08	80.45	16.14	40.61	227.78	98.99	36.67	24.79	212.41	23.9	33.98
36	0.0	0	686.5	81.52	11.63	38.45	262.1	93.89	36.97	23.69	238.66	24.92	44.06
37	0.0	1.23	670.4	79.79	13.94	41.28	257.87	96.56	39.31	27.72	226.16	23.79	37.05
38	0.0	0	561.23	66.8	11.52	34.04	223.35	77.01	37.48	29.5	186.12	22.13	39.7
39	0.0	3.85	532.1	63.9	14.66	35.66	207.43	75.54	42.01	30.85	179.36	22.53	29.6

Table C4 Pigment concentration data from the Erin Lake sediment core collected in 2009. Pigment concentrations are expressed in nMol per gram organic matter.

Depth	**ChlC2**	**Fuco**	**Aph**	**Diad**	**Myxo**	**Allo**	**Lut**	**Zea**	**Can**	**Chlb**	**Chla**	**Ech**	**Bcar**
0	0.00	26.58	261.67	35.45	4.10	8.78	33.31	31.75	12.58	5.50	91.06	55.45	46.31
1	0.00	20.65	194.92	35.37	4.29	9.52	83.03	36.86	20.72	6.29	135.56	21.99	24.50
2	0.00	4.45	139.47	26.57	3.51	7.44	96.45	13.22	22.11	12.14	115.16	17.96	12.95
3	0.00	19.90	235.24	40.91	4.84	11.26	116.13	21.38	25.02	11.77	135.59	23.64	24.21
4	0.00	19.33	231.62	41.33	4.20	10.73	131.82	20.78	26.89	12.39	142.12	23.53	29.03
5	0.00	16.95	242.54	44.09	5.44	12.30	160.81	20.97	29.49	21.19	123.37	15.79	14.09
6	0.00	0.00	529.55	60.52	0.00	14.54	213.29	24.49	35.65	26.92	146.49	16.59	24.42
7	0.00	0.00	322.93	55.90	0.00	13.74	220.19	25.51	36.88	19.51	147.77	17.79	23.28
8	0.00	0.00	0.00	59.02	0.00	12.43	202.06	25.46	33.61	28.79	141.24	17.27	25.00
9	0.00	1.74	304.54	57.18	5.66	12.56	191.87	26.85	31.10	25.62	153.87	21.37	27.31
10	0.00	0.00	490.24	56.09	0.00	12.03	189.97	24.30	33.23	18.56	121.71	15.43	21.44
11	0.00	0.00	288.61	51.89	7.32	15.00	201.72	23.92	35.47	21.95	100.03	10.93	17.10
12	0.00	0.00	316.99	61.48	0.00	13.83	259.36	29.22	42.22	38.57	142.54	6.44	24.96
13	0.00	0.00	298.99	63.93	0.00	14.18	254.33	30.08	42.92	22.70	136.48	15.93	23.53
14	0.00	0.00	302.00	63.48	6.61	14.37	224.10	28.88	36.68	27.46	118.09	12.56	18.65
15	0.00	0.00	336.94	66.36	0.00	14.11	232.83	17.20	36.62	20.04	123.62	14.25	17.73
16	0.00	1.95	311.09	67.25	7.50	15.54	223.57	32.45	36.50	27.38	114.62	13.49	20.49
17	0.00	0.00	312.80	63.97	1.69	13.96	215.52	16.05	35.50	25.51	111.76	13.41	23.59
18	0.00	1.34	311.26	67.89	97.93	19.09	234.52	34.69	38.23	25.65	114.68	13.75	26.18
19	0.00	4.11	337.62	65.80	11.37	19.86	217.40	34.31	38.18	29.37	113.24	16.41	23.02
20	0.00	4.22	500.27	102.09	12.53	28.42	329.15	50.33	56.48	57.39	173.45	26.93	37.59

Depth	**ChlC2**	**Fuco**	**Aph**	**Diad**	**Myxo**	**Allo**	**Lut**	**Zea**	**Can**	**Chlb**	**Chla**	**Ech**	**Bcar**
21	0.00	3.67	350.59	67.59	11.75	20.25	246.94	36.75	42.48	29.33	121.07	17.03	24.67
22	0.00	2.08	334.56	66.07	9.87	17.03	261.95	34.34	44.32	24.41	111.89	13.61	23.49
23	0.00	4.78	371.06	61.90	12.73	22.13	26.56	35.56	46.11	32.91	119.35	17.54	24.50
24	0.00	4.94	360.79	61.59	13.63	22.39	275.29	35.86	50.37	25.73	103.10	13.36	23.03
25	0.00	5.15	457.44	74.25	17.21	29.31	338.39	45.77	62.45	50.29	145.16	23.33	33.73
26	0.00	5.68	335.66	53.02	13.81	24.90	290.49	46.18	57.66	30.65	102.78	16.48	24.41
27	0.00	4.66	378.40	60.28	15.41	26.77	307.89	51.72	63.54	39.42	119.34	18.02	28.36
28	0.00	3.23	316.78	51.10	12.74	22.24	269.56	60.10	53.56	29.82	100.88	15.00	21.80
29	0.00	3.23	423.10	68.45	12.34	22.33	328.60	71.57	61.09	33.21	130.14	16.79	26.06
30	0.00	4.79	493.98	80.87	18.52	34.50	399.48	95.52	76.20	46.62	163.04	23.15	34.54
31	0.00	0.00	404.03	64.85	13.50	25.13	363.89	71.00	70.85	28.18	124.15	14.17	26.86