ASSESSING THE INFLUENCE OF A WARMING CLIMATE ON THE BIOTA OF ARCTIC FRESHWATER SYSTEMS

by

Connor Nishikawa

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ABSTRACT

Understanding the effects of climate change is particularly important for northern regions, where temperatures have increased by considerably more than that of the global average since the start of the 20th century. As a result, freshwater lakes and ponds found throughout the subarctic and Arctic are now under a great degree of ecological stress. The way these systems have responded to this stress has not been uniform and it is unclear how aquatic biodiversity has been affected by climate related stress as well as anthropogenic development. As such, this thesis attempts to broaden our understanding of how aquatic biodiversity of small freshwater ponds and lakes in the eastern Canadian subarctic and Arctic have changed as warming has intensified during the Anthropocene. A chironomid-based paleolimnological analysis of lakes in northern Manitoba and Baffin Island, Nunavut, showed that the divergent prevailing ecological conditions in different northern regions have had significant effects on freshwater biodiversity. In the Arctic warming has led to an increase in species richness as the warm conditions increased the amount of habitat available to warm water taxa. In particular, lakes within the urban boundary of Iqaluit, Nunavut, primarily responded to regional warming, and had very little change associated with anthropogenic development during the last century. In the subarctic warming has increased evaporative pressure on the shallow lakes and ponds characteristic of the region. As such, warming has had less of an influence than landscape-mediated changes to aquatic environments. Understanding how these sensitive ecosystems have been affected by climate change is critical to understanding future responses as warming becomes more intense in the future

LIST OF ABBREVIATIONS USED

^{210}Pb	Lead-210
^{137}Cs	Cesium 137
CE	Common era
DOC	Dissolved organic carbon
KOH	Potassium hydroxide
LOI	Loss on ignition
$\%C_{org}$	Total organic carbon
%N	Total nitrogen content
C/N	Carbon/nitrogen ratio
$\delta\ ^{13}C_{org}$	Carbon isotope ratio
$\delta\ ^{15}N$	Stable isotope value of nitrogen
$\delta\ ^{18}O_{pw}$	Pond sediment cellulose-inferred pond water oxygen isotope composition

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CHAPTER 1: INTRODUCTION

1.1 Introduction

Understanding the effects of global anthropogenic climate change has become one of the great challenges of our time. It is well understood that climate change has had increasingly deleterious effects on Arctic regions (Serreze et al., 2008). Over the last few decades surface temperatures in the Arctic have increased by more than twice that of the global average (Hansen et al., 2010). As a result of this warming, freshwater lakes and ponds found throughout the Arctic are now under a great degree of ecological stress (Schindler & Smol, 2006). Climate has been shown to play an especially dominant role in moderating biodiversity in the Arctic (Michelutti et al., 2007). Arctic ecosystems are also considered to generally be more susceptible to the influence of any kind of environmental stressor than other ecological regions (Schindler & Smol, 2006). For example, warming has reduced the ice-cover season, which has led to changes in predation, habitat loss, fish migration patterns, and population declines (Post et al., 2009). Arctic lakes are also vital for the health and sustainability of northern communities, providing drinking water and a source of food as part of the subsistence economy (Vincent et al., 2012; Medeiros et al., 2017).

Understanding how Arctic environments have responded to climate warming over the Anthropocene is a good way to determine how these systems will respond in the future (Porinchu et al., 2009). A lack of long-term meteorological data often makes understanding long-term climate patterns difficult (Luoto et al., 2019). Using the science of paleolimnology, researchers can use bioindicator species preserved in lake sediment cores to reconstruct past environmental conditions (Luoto et al., 2019). Chironomids (Diptera: Chironomidae) are a family of insects that are known to be responsive to temperature (Walker, 2001).

These insects are commonly found in freshwater ecosystems in the Arctic and are very sensitive to a variety of environmental changes, especially temperature (Brooks et al, 2007). Changes in chironomid species assemblages in a downcore stratigraphic record can be used to infer how lake temperatures and by extension the climate have changed over time (Brooks and Birks, 2004). This inference is what allows for the creation of historical temperature reconstructions.

1.1.1 Overview of the problem to be addressed

While it is clear that the freshwater systems of the Arctic and subarctic have been put under a great degree of stress by climate change, the way these systems have responded to this stress has not been uniform. Moreover, there is also the potential issue of the increasing effect that local development has had as populations of communities increase. The human population across northern Canada has been increasing steadily over the last few decades, and that trend is expected to continue (Heleniak, 2020). With an increasing population comes with it an increased influence on the local environment which includes the nearby freshwater systems.

The combination of the Arctic's increased sensitivity to environmental change and the stress imposed by climate change means that the impacts of local peoples could be even more deleterious to the nearby freshwater systems than they might be in other regions (Smol, 2016). Of further concern is the fact that it is not entirely clear how the stress of climate change and an increased local human influence will affect the biodiversity of the freshwater systems in the Arctic. It is important to have solid understanding of how the biodiversity of these system will respond to these stressors because the biodiversity dynamics of an ecosystem are often a good proxy for the health of that ecosystem (Mayfield et al., 2020).

1.1.2 Research purpose

The purpose of this research is to help better understand how the biodiversity of the different freshwater systems in the Arctic and subarctic have changed as the effects of climate change and local human influence have become more intensive. To this end, a paleolimnological analysis of the sediment cores of 3 sets of lakes was performed. The first is a lake in the High Arctic that has little to no direct human influence or catchment inputs, therefore the ecology of this lake should be primarily governed by the regional climate. The second set includes 2 shallow ponds located in the subarctic Hudson Bay Lowlands. These shallow ponds are common in the region and are affected by a variety of terrestrial and climate related inputs. The final set includes two lakes located close to the city of Iqaluit, a city whose population has been increasing steadily and whose influence on the local freshwater systems has become increasingly noticeable.

1.2 Literature review

1.2.1 Paleolimnological Theory

Paleolimnology is a science that works by using the geological, chemical, and biological information stored in the sediment of a lake to reconstruct the environmental history of that lake (Smol, 1992). These reconstructions can potentially go back thousands of years and allows for the inference of historical lake alterations such as water level, water chemistry and climate related changes (Sullivan and Charles, 1994). To make inferences of environmental history paleolimnology often makes use of a variety of fossil bioindicator species stored in the lake sediment to act as proxies for past environmental conditions. Because bioindicator species have specific environmental preferences, changes in the species assemblages of these proxies over time combined with radioisotope dating of the individual layers of a sediment core can be used to reconstruct how a lake has changed over time (Millet et al., 2014).

Chironomids, also known as non-biting midges are a commonly used bioindicator species that have been shown to be an effective indicator for reconstructing Arctic climate changes (Luoto et al., 2019). This effectiveness is due to a few reasons, most chironomid species have an aquatic larval stage that leaves behind chitinous head capsules that preserve well in sediment (Axford et al., 2009).

Chironomids are also found commonly all over the world, even in the coldest Arctic lakes and are known to be highly sensitive to climatic temperature shifts because they depend on temperature for reaching their various life stages (Smol et al., 1992). Since climate and by extension temperature play such a dominant role in facilitating environmental change in the Arctic, chironomids have been shown to be an especially effective bioindicator for interpreting climatic shifts that have occurred in Arctic freshwater systems since the early Holocene (Brooks et al., 2012). Aside from climate and temperature, chironomids have also been shown to be effective indicators for environmental variables such as aquatic plant production (Rühland et al., 2014), water depth (Gajewski et al., 2005), habitat availability (Brooks et al., 2007), and heavy metal deposition (Zubova et al., 2020). Being a good indicator for this variety of environmental variables means that chironomids can be quite useful for assessing not only the impacts of climate change but other anthropogenic inputs as well.

Common Paleolimnological Methodology

The sediment cores used for the analysis of paleolimnological indicators are collected using a gravity coring device such as a Glew or Uwitec corer in the mid basin of the water body that is being analyzed. Cores can then be sectioned into 0.5 cm intervals in lab and stored at 4°C temperatures (Walker, 2001).

The individual sediment intervals for biostratigraphic analysis of sediments from the anthropocene are often chronologically dated using either ^{210}Pb (Belle et al., 2017) or ^{137}Cs (Self et al., 2011) isotope dating. The age of the various sediment layers can vary from study to study based on factors such as local sedimentation rate. In Macdonald et al. (2009), a 25 cm core went back 2000 years whereas in Thomas et al. (2008), a 12cm core went back 2400 years.

Samples from the individual sediment layers are chemically cooked so that it is easier to extract the chitinous chironomid head capsules (Walker, 2001). These head capsules are identified to the genus or species level if possible, using an identification guide. These guides tend to be region specific and mostly focus on commonly found species (Brooks et al., 2007). Most studies need at least 50 head capsules per layer to be deemed statistically significant (Quinlan and Smol, 2001) although some studies like Porinchu et al. (2009) or Millet et al. (2014) either required less than 50 heads or did not even require a minimum headcount. The chironomid species assemblage data is then run through a region-specific inference model to produce a historical climate reconstruction (Self et al., 2011). These models need to be region specific because ecological and climate related differences between regions creates differences in regional species assemblages. Using a model developed from a different region can produce inaccurate reconstructions (Self et al., 2011).

Limitations of Paleolimnology

While they can produce accurate long-term climate reconstructions, there are some potential complications that should be considered when performing a chironomid based paleolimnological study. Firstly, paleolimnological studies often need a pre-established inference model for proper climate reconstructions. These inference models are difficult to create and often require very intensive sampling.

In creating their inference model, Porinchu et al. (2009) had to take multiple samples from 88 lakes and identify all the chironomid head capsules inside. A shortage of inference models means that researchers are sometimes limited on which lakes they wish to reconstruct (Self et al., 2011). These inference models also need to be very accurate, if these models do not include appropriate analogues for past chironomid assemblages and past climate conditions, they can produce false reconstructions (Axford et al., 2007). Another issue stems from the fact that chironomids will respond to changes in environmental factors aside from temperature that are often related to increased terrestrial catchment input. Changes in trophic status (Self et al., 2011), oxygen levels (Macdonald et al., 2008) and nutrient levels (Millet et al., 2013) have all been shown to have the ability to mask the true effect of climate change on chironomid assemblages. Even bird droppings have the potential to significantly mask the true influence of climate change on chironomids (Luoto et al., 2015). In short time scales with low amplitude temperature changes, the influence of temperature can be outmatched by the influence of these other environmental factors (Millet et al., 2013). This is an issue that tends to disappear once reconstructions start involving larger time scales as many of the previously mentioned environmental factors are themselves influenced by temperature changes (Luoto et al., 2015). This is also less of an issue in the Arctic where the smaller human population and lower catchment inputs reduce the impact of local environmental factors on climate reconstructions (Smol, 2016).

1.2.2 Local human influence in the Arctic

The influence of humans on the environment extends to freshwater lakes, ponds, and rivers found near human settlements. Farming practices have been shown to increase the nutrient load of nearby freshwater systems, which in turn can lead to eutrophication and the creation of harmful algal blooms (Garcıa-Rodrıguez et al., 2002).

Improperly managed sewage discharge has been shown to have a similar impact as farming but to a potentially even worse extent (Jarvie et al., 2006). Urban developments have been shown to increase the level of pollutants in nearby rivers (Mederios et al., 2011). Industrial manufacturing has been shown to lead to heavy metal contamination and acidification of nearby lakes (Greenway et al., 2012). The development of mineral industries has led to increased salt and ion influxes into nearby freshwater systems which has altered their pH and salinity levels (Rowell et al., 2016). All of these negative influences have been shown to not only be damaging to the local environment but also generally lead to a loss in freshwater biodiversity.

In the Arctic there is a commonly held belief that the population is too small to have any kind of notable impact on the freshwater systems. This is partially why many paleolimnological studies are done in the Arctic as there are no local influences on the environment so its easier to attribute environmental changes to shifts in climate (Smol, 2016). However, a growing population (Statistics Canada, 2022), outdated and inefficient infrastructure (Das and Canizares, 2019) and a growing oil and gas industry (National Petroleum Council, 2019) mean that this commonly held belief may no longer be true. Paleolimnological studies have shown that there are regions in the Arctic where the local populace has started to have a significant impact on nearby freshwater systems although these kinds of studies are limited (Antoniades et al., 2011; Belle et al., 2017). The Arctic is considered to be more environmentally sensitive than most other regions so monitoring the effects of the local population is crucially important because local anthropogenic impacts could be all the more damaging to Arctic freshwater environments and biodiversity. On top of the ecological damages are the social ones, freshwater sources are critical for the survival of Arctic communities.

If the freshwater sources in the Arctic become seriously contaminated by local pollutants it would be more damaging for Arctic communities than most other regions in North America (Medeiros et al., 2011).

1.2.3 Effects of climate change on the subarctic

Ecotones are environmental transition zones where two neighbouring ecosystems are compressed along a spatial gradient (Smith and Goetz, 2021). These transition zones are dynamic ecosystems that are known to be susceptible to environmental stressors, especially climate change (Zhu et al., 2011). The subarctic Hudson Bay Lowlands are one such ecotone, acting as a transition from the temperate boreal forest to the Arctic tundra (Hadley et al., 2019). The climate in Hudson Bay Lowlands is heavily moderated by Hudson Bay, which was able to mostly buffer the region from the effects of climate change throughout the mid twentieth century (Gough et al., 2004). However, melting sea in Hudson Bay has removed this buffer and as a result the Lowlands have experienced some of the most intense warming in the entire circumpolar Arctic over the past few decades (Kaufman et al., 2009). This warming has had significant impact on Hudson Bay Lowlands, especially the shallow ponds which dominate the region's landscape (Macrae et al., 2004). These ponds are usually frozen over during the coldest periods of the year but because of the intense warming that the lowlands have received the winter season has been noticeably shortened (Prowse et al., 2009). This shortened season limits the time when these ponds are frozen and makes them more susceptible to evaporative stress (Macrae et al., 2004). A shorter winter also limits snow buildup which limits the amount of water these ponds receive during the spring thaw that helps offset the evaporative pressure and reduced precipitation seen in the region's summer months (Bouchard et al., 2013).

The combination of increased evaporative pressure and reduced snowmelt mean that many of these ponds are at risk of drying up entirely (Prowse et al., 2006). Another concern is the fact that the Hudson Bay Lowlands also reside in a region of continuous permafrost which often acts as a barrier that prevents water from leeching into the ground, instead keeping it at the surface where local biota can make use of it (Dredge and Dyke, 2020). A further effect of regional warming is that this permafrost is starting to melt which is causing many of these ponds to drain into the surrounding landscape (Schindler and Smol, 2006). There have been a variety of studies that have looked at the long-term hydrological changes that shallow ponds like those in the Hudson Bay Lowlands have undergone (Wolfe et al., 2011), but few have looked at how those hydrological changes have affected the biological aspects of these ponds (Hargan et al., 2020). These shallow ponds are considered to be vital for the ecological functioning of the Lowlands (Meerhoff et al., 2012), are important for Northern communities and act as important components in the region's carbon cycle (Abnizova et al., 2011). Developing a full understanding of the impact that climate change has had on the ponds is important to determine how they may be impacted in the future and because shallow ponds often act as early previews for the changes that much larger lakes may undergo with continued ecological stress (Smol, 2016).

1.2.4 Intermediate Disturbance Hypothesis

Due to the nature of this study in that the lakes being studied can be scaled along a gradient of ecological disturbance; this research provides an excellent opportunity to test the intermediate disturbance hypothesis and how it could be used as a model for how the taxonomic richness of various Arctic and subarctic lakes could change as climate change worsens. Climate change has been shown to have a deleterious effect on species richness as a warming climate tends to homogenize aquatic environments, thereby reducing the habitats for specialist taxa and leaving only those generalist taxa that can survive the ecosystem changes (Mayfield et al., 2020).

The intermediate disturbance hypothesis is a theory about the relationship between the species richness of an ecosystem and the level of ecological disturbance (Dial and Roughgarden, 1998). The theory states that species richness is at its highest at intermediate levels of disturbance. If the levels of disturbance are too low a few species will dominant the ecosystem and exclude others due to the low selection pressure. If the levels of disturbance are too high only a few species will be able survive the harsh environment (Dial and Roughgarden, 1998). If disturbance levels are somewhere in between there is enough variability in the environmental inputs and selection pressure for an environment to support a variety of species without any being able to be especially dominant (Roxburgh et al., 2004). There is an ongoing debate about the validity of the intermediate disturbance hypothesis. Some studies have argued that disturbance is not a dominant factor in determining species richness (Mackey and Currie, 2001) or that the hypothesis fails to adequately address the complex ecological relationships between species richness and ecological disturbance (Fox, 2013).

There have also been studies that support the intermediate disturbance hypothesis such as England et al. (2008), who looked at the affect of wave disturbance on macroalgal communities or Rogers (1993) who looked at the affect of hurricanes on shallow water coral reefs. There has been limited research looking at applying this theory to the Canadian Arctic or to chironomids, although there is evidence that some species in the Arctic and Antarctic may follow the intermediate disturbance hypothesis (Gutt et al., 2004). This research represents a good opportunity to test this theory because each set of lakes in this thesis represent one of the three major points on the intermediate disturbance curve (Figure 1.1). The High Arctic Lake has very little terrestrial input or human presence so the disturbance levels should be quite low. The subarctic ponds of the Hudson Bay Lowlands represent the intermediate stage on the curve. Being located in the subarctic ecotone means that these ponds receive inputs and are affected by ecological influences from both the boreal forest and tundra biomes, being in such a heterogeneous environment should allow for greater species richness.

Finally, the lakes that may have been altered by the urbanization of Iqaluit represent the most disturbed lakes as local anthropogenic pollutants have been shown to lower the biodiversity of nearby freshwater environments (Medeiros et al., 2011; Zubova et al., 2020). One potential concern with testing this hypothesis, and something that Fox (2013) points to as being common, is that the most disturbed lake might not be disturbed enough to reach the species richness peak let alone see the point when species richness might start to drop off. This could be a potential issue in the Arctic where the ecosystems are considered to be some of the least impacted by local pollutants in the world (Weckstrom et al., 2010).

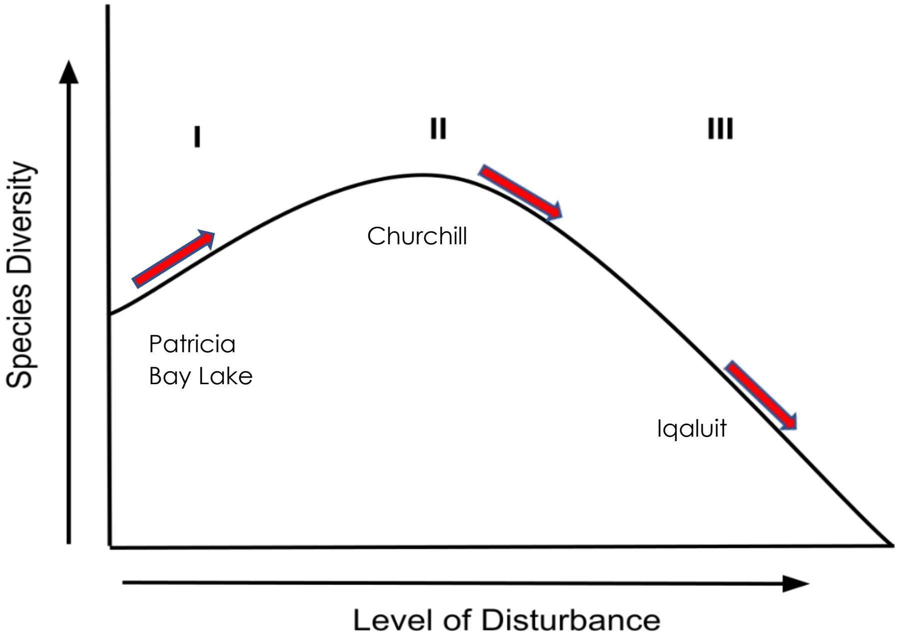

Figure 1.1: Intermediate disturbance hypothesis curve with the three sets of study lakes and their proposed position on the curve. Red arrows indicate the directional shift along the curve for each set of lakes under continued stress from climate change.

Another portion of this theory that has not been largely explored is how environments on the intermediate disturbance curve will respond as the effects of climate change become more severe. The responses to climate change in Arctic freshwater environments has not been uniform but that could be due to the different environmental influences eliciting variation in ecosystem responses (Wolfe et al., 2011). The structure of the intermediate disturbance hypothesis allows for the observation of different lakes with their various inputs aggregated together as influences on biodiversity. Therefore, these lakes and their response to climate change can be studied holistically without having to account for outside influences because they are already included. We believe that each of the study lakes will shift along the disturbance curve as the historical effects of climate become worse. Patricia Bay Lake has no other primary influences except climate and a warming climate has been shown to increase species richness (Engels et al., 2020). Climate change has also been shown to reduce richness in Arctic lakes as cold-water taxa and other specialist taxa are lost as temperatures rise beyond a threshold that those taxa can not tolerate (Mayfield et al., 2020). Therefore, we believe that species richness in Hudson Bay Lowland ponds and the lakes near the city of Iqaluit will decrease over time as the disturbance caused by climate change becomes worse. Part of the response for the Iqaluit lakes depends on the severity of the local anthropogenic impacts. If these impacts are so severe that it means that local influences are now the primary driver of environmental change than a warming climate may not have an effect on species richness after the time when impacts have become severe enough. In short, we expect every lake to shift to the right from its original position on the curve as climate change gets worse over time.

1.3 Research locations

1.3.1 Patricia Bay Lake

Patricia Bay is located along the eastern coast of Baffin Island, Nunavut near the Inuit hamlet of Clyde River. Patricia Bay Lake is located above treeline in the zone of continuous permafrost with a geology consisting primarily of Precambrian bedrock (Miller et al., 2005) and experiences a tundra climate. There is little human activity in the area and while the community of Clyde River is nearby, there are no indications of any local human activities around the catchment area. Patricia Bay Lake is also located not far from one of the glaciers that are not uncommon on Baffin Island (Miller et al. 2005). Therefore, climate should be the only real factor influencing the environmental change on this lake.

1.3.2 IQ01 and IQ04

Lakes IQ01 and IQ04 are located near the city of Iqaluit, Nunavut. Iqaluit is located in the southern end of Baffin Island in Frobisher Bay. Iqaluit is also located above treeline and also experiences a tundra climate although it is in a warmer region than Patricia Bay Lake. Iqaluit is one of the most populated urban centres in the Canadian Arctic and has grown in population to around 7429 people (Statistics Canada, 2022). Iqaluit was founded in the 1940s as the United States airbase "Crystal Two". Over time this base transitioned into a major settlement and then into the capital of Nunavut, receiving city status in 2001(Shields and Weber, 2013). There is a noticeable human presence near these lakes with a newly built road near the catchment of one and a bag of asbestos was pulled from the catchment of one lake during data collection. There is also a distinct lack of vegetation around the catchment area of these lakes. Therefore, the only factors that should be having an influence on these lakes are the local human population and/or the climate.

1.3.3 Left Lake and Larch Lake

Left Lake and Larch Lake are located near the town of Churchill, Manitoba in the subarctic Hudson Bay Lowlands. The Lowlands are located in an environmental transition zone between the boreal forest and Arctic tundra. Being in this transition zone means that lakes in this region are exposed to a variety of boreal and tundra inputs and vegetation. Churchill has a population of less than 900 people and has shown no indications of having a substantial impact on the nearby lakes (Statistics Canada, 2022). This lack of local human disturbance means that the primary influences effecting these lakes are the terrestrial inputs and warming climate.

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CHAPTER 2: RESPONSES OF SHALLOW SUBARCTIC PONDS TO A WARMING CLIMATE IN THE ANTHROPOCENE: A PERSPECTIVE FROM THE HUDSON BAY LOWLANDS, CANADA

2.1 Statement of Student Contribution

Study design by Connor Nishikawa and Andrew Medeiros, limnological data collection by Connor Nishikawa, Erin Light, Roland Hall, Brent Wolfe, Merrin Macrae, LeeAnn Fishback and Johan Wiklund, chapter writing by Connor Nishikawa with editorial contributions by Andrew Medeiros, Jennifer Eamer and Brent Wolfe.

2.2 Abstract

Arctic ecotones contain dynamic freshwater ecosystems where aquatic biota vary across a small spatial transition and as such can be especially susceptible to environmental change. Here, we examine the paleohistory of two lakes in the ecotonal Hudson Bay Lowlands, subarctic Canada, to understand how aquatic biota have responded in an increasingly climate-stressed Anthropocene, and to better anticipate future changes. Using a multi-proxy paleolimnological analysis, we reconstruct past environmental conditions through the examination of subfossil chironomids (Diptera: Chironomidae) and compare these records to organic carbon and nitrogen elemental and isotope composition, and prior cellulose-inferred lake water d^{18}O records. Despite their close proximity, the two lakes had vastly different hydrological responses to climate change. Larch Lake, which was once a small shallow pond, became connected to other ponds due to the formation of permafrost degradation channels.

The increased connectivity of the lake to adjacent ponds increased water levels, created new aquatic habitats, and lead to a community shift from an environment that was once dominated by *Limnophyes* and *Psectrocladius* to one dominated by various species of Tanytarsini. In contrast, Left Lake was once a deeper isolated basin, but increased evaporation has significantly decreased its surface water volume. A shift in chironomids from a community dominated by *Cricotopus* and Tanytarsini to one with higher abundances of *Dicrotendipes* and *Procladius* suggests that this system has increasingly transitioned from an isolated deeper-water system to one which has a larger, much shallower, littoral environment. Despite their divergent hydrological responses, both lakes had similar ecological responses to climate change as declining C/N ratios and a loss of chironomid taxa associated with macrophytes indicate a potential shift from a macrophyte dominated ecosystem to one dominated by phytoplankton. Likewise, even though large catchment-mediated changes resulted in a shift in some chironomids of both lakes, a chironomid-based paleo-temperature reconstruction successfully demonstrated regional warming with a similar trend in both systems. Shifts in the ecology of subarctic lakes and ponds are expected to increase as the effects of climate change, which could include continued pond desiccation and enhanced algal proliferation, become more severe.

2.3 Introduction

The transition between the northern boreal forest and subarctic tundra is an especially dynamic region and susceptible to environmental change; yet, our knowledge of how temperature is influencing these landscapes is not well understood. Ecotones are transition zones between ecosystems where the overlap of the neighboring systems creates an intermediate area where different ecological communities are spatially compressed (Smith and Goetz, 2021).

The transition from the boreal forest to tundra in the subarctic is especially environmentally sensitive (Zhu et al., 2011), where temperatures have been shown to have increased by a considerably greater rate than the global average (Kaufman et al., 2009; Ballinger et al., 2021). Of particular concern are shallow freshwater ponds characteristic of northern regions, which are vital for maintaining ecological stability (Meerhoff et al., 2012), act as important carbon sinks (Abnizova et al., 2012), and are important sources of freshwater for northern communities (White et al., 2007; Medeiros et al., 2017). Shallow ponds (typically less than 2 m deep) with a large surface area are the most common freshwater basins in the Arctic (Smol, 2016). These ponds have been strongly affected by the warming climate, in part because of their shallow depth; temperature will have a larger effect on systems with a large surface area:volume ratio (Michelutti et al., 2007; Smol, 2016). In recent decades, ponds in Nunavut (Abnizova and Young, 2009), the Northwest Territories (Campbell et al., 2018), Greenland (Higgens et al., 2019), and Alaska (Koch et al., 2014), have all experienced some degree of desiccation during drier summer months (Medeiros and Bakaic, 2016). Smol and Douglas (2007) noted several high Arctic ponds that dried up for the first time in thousands of years due to increases in evaporative water loss and a longer ice-free season. Since shallow ponds are considered to act as early indicators for environmental change, their continued desiccation could be a preview of what awaits lakes with deeper basins (Smol, 2016; Webb et al., 2022).

Hudson Bay Lowlands are an extensive subarctic wetland region within the forest-tundra ecotone (Hadley et al., 2019). The lowlands are dominated by shallow ponds, with some areas having 25 % of their total landscape consisting of shallow freshwater wetlands (Macrae et al., 2004). The region is heavily influenced by its proximity to Hudson Bay to the north, with a coastal-mediated climate (Martini, 2006).

Throughout the mid-twentieth century, Hudson Bay was able buffer the region from the effects of climate change as the region generally experienced minimal warming (Gough et al., 2004); however, during the past few decades it has experienced some of the greatest warming in the entire circumpolar Arctic (Kaufman et al., 2009). Increased summer temperature and reduced winter precipitation have had a noticeable effect, with shortened winter seasons (Prowse et al., 2009), increased evaporative water loss (Macrae et al., 2004), and reduced spring freshet (Bouchard et al., 2013). While there has been an increase in the amount of rain the region receives as a result of a shorter winter, the increased evaporative water loss and reduced spring snowmelt input means many of these ponds are at risk of drying up (Prowse et al., 2006; Bouchard et al., 2013). This risk is exacerbated by the fact that much of the lowlands are located in the zone of discontinuous to continuous permafrost that has started to degrade with the increasing temperatures (Dredge and Dyke, 2020). As a result, many of the ponds in the lowlands have started to drain into surrounding peatlands (Schindler and Smol, 2006).

Wolfe et al. (2011) analyzed the recent impact of climate change on the hydrology of four lowland ponds near Churchill, Manitoba, using cellulose-inferred pond water oxygen isotope records extracted from sediment cores. Two of these lakes, Left Lake and Larch Lake, had profoundly different hydrologic responses to anthropogenic warming despite being located only a few kilometres from one another (Wolfe et al., 2011). In the early decades of the 20th century, similar signals of increased evaporative water loss caused by regional warming were observed in both systems, but after ~1960 their hydrological responses to warming started to diverge (Wolfe et al., 2011). Left Lake continued to display evidence of evaporative water loss, whereas Larch Lake started to show indications of decreasing stress.

Wolfe et al. (2011) postulated that the different hydrological responses was due to the development of a channel fen that increased hydrological connectivity and caused the water levels of Larch Lake to increase. Here, we aim to understand the influence of warming and associated variation in hydrologic change on the biological communities of Left Lake and Larch Lake. As such, we have undertaken additional paleolimnological assessment of these lakes using chironomid-based inferences as well as organic carbon and nitrogen elemental and isotope composition to reconstruct the ecological history of these lakes. Previous studies (Perren et al., 2003; Keatley et al., 2006; Kaufman et al., 2009) have pinpointed the beginning of the 20th century as the period when anthropogenic climate change began having noticeable effects on Arctic ecosystems; therefore, our reconstructions focus on ecological changes that have occurred since that time. To date, there have been limited paleo-environmental reconstructions in the Hudson Bay Lowlands that have focused on changes since the start of the 20th century (Camill et al., 2012; Rühland et al., 2013; Dickson et al., 2014; Hargan et al., 2020), and fewer that examined changes in chironomid assemblages. However, these reconstructions have been shown to be a useful tool for understanding the historical biological impacts of climate change on other Arctic freshwater bodies (Medeiros et al., 2012). As freshwater ponds provide unique habitat to wildlife and are sentinels of environmental change (Smol, 2016), it is crucial that we have a solid understanding of how climate change has historically altered these ponds if we are to determine how they may evolve in the future as the landscape continues to change.

2.4 Methods

2.4.1 Site description

Both Left Lake and Larch Lake are located in the western Hudson Bay Lowlands region near the town of Churchill in northern Manitoba, subarctic Canada (Figure 2.1).

The region experiences an average annual temperature of -6.5°C, an average July temperature of 12.7°C, and receives 452 mm of precipitation annually (Station 5060600, Environment Canada). Larch Lake is located in the forest-tundra transition zone and is in close proximity to a network of other shallow ponds and is most likely fed by a channel fen (Wolfe et al., 2011). Vegetation around Larch Lake consists of various trees such as dwarf birch and spruce as well as sedges and mosses. Left Lake is located north of treeline in the tundra zone and is geographically isolated from any other nearby water bodies; therefore, Left Lake is considered a closed drainage pond. Catchment vegetation around Left Lake consists of mosses, sedges, shrubs and tundra grasses.

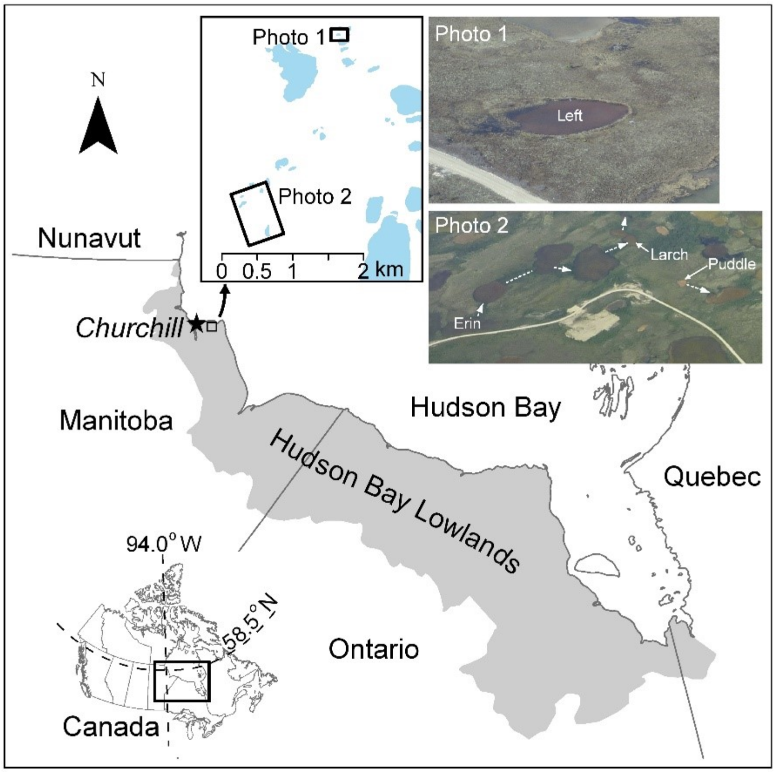

Figure 2.1: Location of the Left Lake and Larch Lake coring sites in the northwestern Hudson Bay Lowlands region near Churchill, Manitoba (from Wolfe et al., 2011).

2.4.2 Field sampling

Duplicate sediment cores were collected from the mid-basin of Larch Lake and Left Lake in July 2009, with the use of a Glew gravity corer (10 cm diameter) (Glew, 1989). Cores were sectioned at 0.5 cm intervals for the entire lengths collected: 21 cm from Larch Lake and 39 cm from Left Lake. Each interval was placed in labelled Whirl-Pak® bags, transported in a cooler, and stored at 4 °C for processing and analysis. Environmental data was collected from each of the study ponds in September 2010 using a YSI multi-parameter probe (Table 2.1)

Table 2.1 Environmental characteristics of sampled ponds Left Lake and Larch Lake collected from Churchill, Manitoba, Canada in September 2010 (adapted from White, 2012).

Parameter	Units	Left	Larch
Lat	DD°	58.75	58.72
Long	DD°	- 93.82	-93.84
Area	m^2	181.0	471.5
Alkalinity	mg/L	122.91	124.30
Depth	cm	23.7	20.0
Temp	°C	16.0	17.8
pH		9.5	8.9
COND	µs/cm	630	546
TP	mg/L	0.053	0.056

(Lat = latitude, Long = longitude, Area = lake surface area, Alkalinity= lake buffering capacity, Depth = maximum depth, Temp = surface water temperature, COND = conductivity at 25 C, TP = Total phosphorus)

2.4.3 Laboratory analysis

The upper 15 cm and 10 cm of the sediment cores for Left Lake and Larch Lake respectively were analyzed for subfossil chironomids at Dalhousie University following standard methods (Walker, 2001), where each sample was treated with potassium hydroxide (KOH) and heated at 75 °C for 30 minutes, with stirring after 15 minutes to deflocculate the sediment. The treated sediment was then poured through nested sieves of 212 and 106 µm and rinsed with 95 % ethanol. Residues were then sifted with the aid of a Borogov counting chamber using a stereomicroscope at 200-400X magnification. Specimens were extracted with fine tipped forceps from each subsample and permanently mounted on glass slides using Entellen®. Each individual head capsule was identified using a compound microscope at 400X magnification to the most specific taxonomic resolution possible following Brooks et al. (2007) and Medeiros and Quinlan (2011). A minimum of 50 head capsules were enumerated from each sample (Quinlan and Smol, 2001); samples below 50 head capsules were resampled until the minimum head capsule count was surpassed.

Measurement of organic carbon and nitrogen elemental and isotope composition was performed on every 0.5 cm section of the sediment cores from Larch Lake and Left Lake following methods described in Wolfe et al. (2001). Samples were prepared for analysis by washing ~15 mg of sediment with 10% HCl at 60 °C to remove carbonate, followed by rinsing with deionised water until samples were pH neutral. Samples were then freeze-dried and sieved through a 500-µm sieve to remove potential terrestrial coarse debris. Subsamples were sent to UW-EIL for organic carbon and nitrogen elemental and stable isotope analysis, which was performed using an elemental analyzer coupled to a continuous-flow isotope-ratio mass spectrometer (EA-IRMS).

Organic carbon and nitrogen stable isotope results are expressed in δ-notation in per mil (‰) relative to the Vienna PeeDee Belemnite (VPDB) and atmospheric (AIR) standards, respectively. Radiometric analysis of sediments using gamma spectrometry for analysis of ^{210}Pb and ^{137}Cs isotopes for both cores was conducted at the University of Waterloo as described in Wolfe et al. (2011); dating profiles can be found in the associated auxiliary material.

2.4.4 Statistical Analysis and Climate Reconstructions

For each core, taxa were represented by their relative abundances, calculated as the percentage of identifiable chironomids in each sample for each interval, and plotted stratigraphically over depth and chronological time. Zonation was established using Constrained Cluster Analysis (CONISS) with the number of significant zones determined using a broken-stick model (Bennett, 1996). The taxonomic diversity of each interval was determined using Hills N2 analysis, where diversity is E(Tn) or the number of each taxon present in each sample of n = 50 (Birks and Line, 1992; Birks and Birks, 2008). Zonation for organic carbon and nitrogen elemental and isotope composition was based on chironomid CONISS zonation. A principal components analysis (PCA) was used to reduce the dimensionality of the assemblage dataset for each core and examine the degree of variation between the chironomid assemblages of each sediment interval. To aid in interpretation of dominant trends, rare taxa (i.e., those whose relative abundance never exceeded 1% of the total identifiable chironomids in any interval, were removed from the stratigraphic diagrams. Data analysis was completed using R statistical software v.4.1.2 (R Core Team, 2021).

A chironomid-based paleotemperature model (Fortin et al., 2015) was applied to the relative abundances of chironomid taxa from each interval of both cores to estimate mean-July air temperature. A weighted-averaging partial-least squares (WA-PLS) transfer function was applied to the chironomid assemblages of each core interval using taxa-environmental relationships from 403 lakes from across northern North America (Fortin et al., 2015). The chironomid assemblages from each core were compared through analogue matching to the calibration dataset and inferred reconstructions were tested with a goodness-of-fit approach using the "palaeoSig" package v.2.0-3 (Telford, 2019). The core trajectories of both lakes were passively plotted across calibration space using the "analogue" package (Simpson, 2007).

2.5 Results

2.5.1 Chironomid and geochemical stratigraphy

2.5.1.1 Left Lake

A total of 1229 subfossil chironomid head capsules were extracted from the upper 15 cm of the sediment core from Left Lake comprising 42 taxa. Based on the compositional similarity of samples, 2 zones were identified as statistically significant (Figure 2.2). **Zone 1** (15.25-5.25 cm; 1907-1989 CE) was the larger of these zones and was mostly dominated by various taxa of the Tribe Tanytarsini, such as *Tanytarsus lactescens*-type*, Tanytarsus glabrecens*-type*, Tanytarsus pallidicornis*-type and *Cladotanytarsus mancus*-type.

From 15.25-10.25 cm (1907-1955 CE) the abundances of the various Tanytarsini taxa varied a notable amount between intervals with little directional trend in any taxa. *Tanytarsus lactescens*-type had an average abundance of ~ 7% with intervals ranging from 12% abundance (15.25 cm; 1907 CE) to 3% abundance 13.25 cm (1921 CE). In the same period *Cladotanytarsus mancus*-type had an average abundance of 8% with a high of 11% occurring at 15.25 cm (1907 CE) and a low of 5 % occurring at 12.5 cm (1937 CE), all other intervals during this period were closer to the zone average with no directional trend. Tanytarsus pallidicornis-type had an average abundance of 6% during this period with an abundance high of 12% occurring at 14.5 cm (1914 CE) and a low of 2 % occurring at 10.25 cm (1955 CE). *Tanytarsus glabrecens*-type had an average abundance of 9% during this period with high of 21% occurring at 14.5 cm (1914 CE) and a low of 2 % % occurring at 10.25 cm (1955 CE). *Cricotopus intersectus*-type and *Psectrocladius sordidellus*-type were also present during this period with each an average abundance ~6% and also both experiencing a general decrease in abundance over time. The average diversity value during this period was 12.9 with a decreasing trend over time.

From 9.5-5.25 cm; 1961-1989 CE. The abundance of *Tanytarsus glabrecens*-type increased in almost every successive interval from 5% at 9.5 cm (1961 CE) to 11% at 6.25 cm (1983 CE), after which the abundance of this taxon declined to 0% at 5.25 cm (1989 CE). *Cladotanytarsus mancus-type* had an average abundance of 11%. Aside from the Tanytarsini, an increasing number of *Dicrotendipes nervosus*-type occurred, with an average abundance of 5% and a maximum of 10% that was reached at 6.25 cm (1983 CE). This increase is notable because prior to this zone the abundance of this taxa was often between 1-2%. Taxonomic diversity was notably lower during this period than during the previously mentioned intervals as the average diversity value for this period was 7.3

Zonation of the geochemical analysis of Left Lake was based on the statistical separation of the chironomid assemblages (Figure 2.3). From 15.25-10.25 cm; (1907-1955 CE) The average %C_{org} was 49.4% with no interval deviating far from that average. The average %N value was 2.6% with an increasing trend over time and no notable deviations from that average. The average C/N value in this period was 18.7 with a decreasing trend over time and no deviations from the average. The average $\delta^{13}C_{org}$ value in this period was -17.6 ‰ with no notable trends or deviations from the zone average. The average $\delta^{15}N$ value in this zone was -0.4‰ with a decreasing trend over time.

Between 9.5-5.25 cm (1961-1989 CE) The average $\%C_{org}$ was 48.1%, the only deviation from this average occurred at 6.25 cm (1983 CE) where $\%C_{org}$ was 45.6%. The average %N value during this period was 2.7% with no notable deviations from that average. The average C/N value in this period was 17.7 with no deviations from that average. The average $\delta^{13}C_{org}$ value in this period was -18.1‰ with a trend of increasing values over time. The average $\delta^{15}N$ value in this period was -0.6‰ with no trend or deviation from that average.

Zone 2 (4.75-0.25 cm; 1989-2010 CE) was also mostly dominated by Tanytarsini taxa. A trend of decreasing abundance for *Tanytarsus glabrecens*-type characterized this zone from other period, where abundances declined from 17% for the next five intervals (4.25-1.75 cm; 1994-2006 CE) to 1% at (1.25 cm; 2008 CE). *Tanytarsus pallidicornis*-type had an average abundance of 10% in this zone with a maximum occurring at 1.25 cm (2008 CE), where abundance was 17% of the assemblage. These results are notable because prior to this zone, the average abundance for this taxon was 5% with few notable deviations. There was a trend of increasing abundance for *Cladotanytarsus mancus*-type in this zone. For the first four intervals of this zone (4.75-3.25 cm; 1989-2000 CE), the average abundance for this taxon was 8%; in the later 5 intervals (2.5-0.25 cm; 2003-2009 CE), average abundance increased to 20% with the last two intervals (0.75-0.25 cm; 2009-2010 CE) having abundances around 26%. *Dicrotendipes nervosus*-type was once again present in this zone with an average abundance of 7% and a maximum of 12.5% that was attained at 4.25 cm (1994 CE). The average diversity value in this Zone 2 was 7.0.

The average %C_{org} in Zone 2 was 48.3%, similar to values seen in Zone 1, with no discernable trend or deviation from the average. The average %N value in this zone was 2.8%, higher than the previous zone with no notable deviations from that average or directional trend. The average C/N value in this zone was 17.1, slightly lower than the previous zone with one notable deviation from the average occurring at 2.5 cm (2004 CE) where this value was 16.5. The average $\delta^{13}C_{org}$ value in Zone 2 was -18.2‰ no notable trend or deviation from the zone average. The average $\delta^{15}N$ value in this zone was -0.7‰ with no trend and one deviation from the average occurring at 0.25 cm (2010 CE) where this value was -1.24‰, easily the lowest of the entire core.

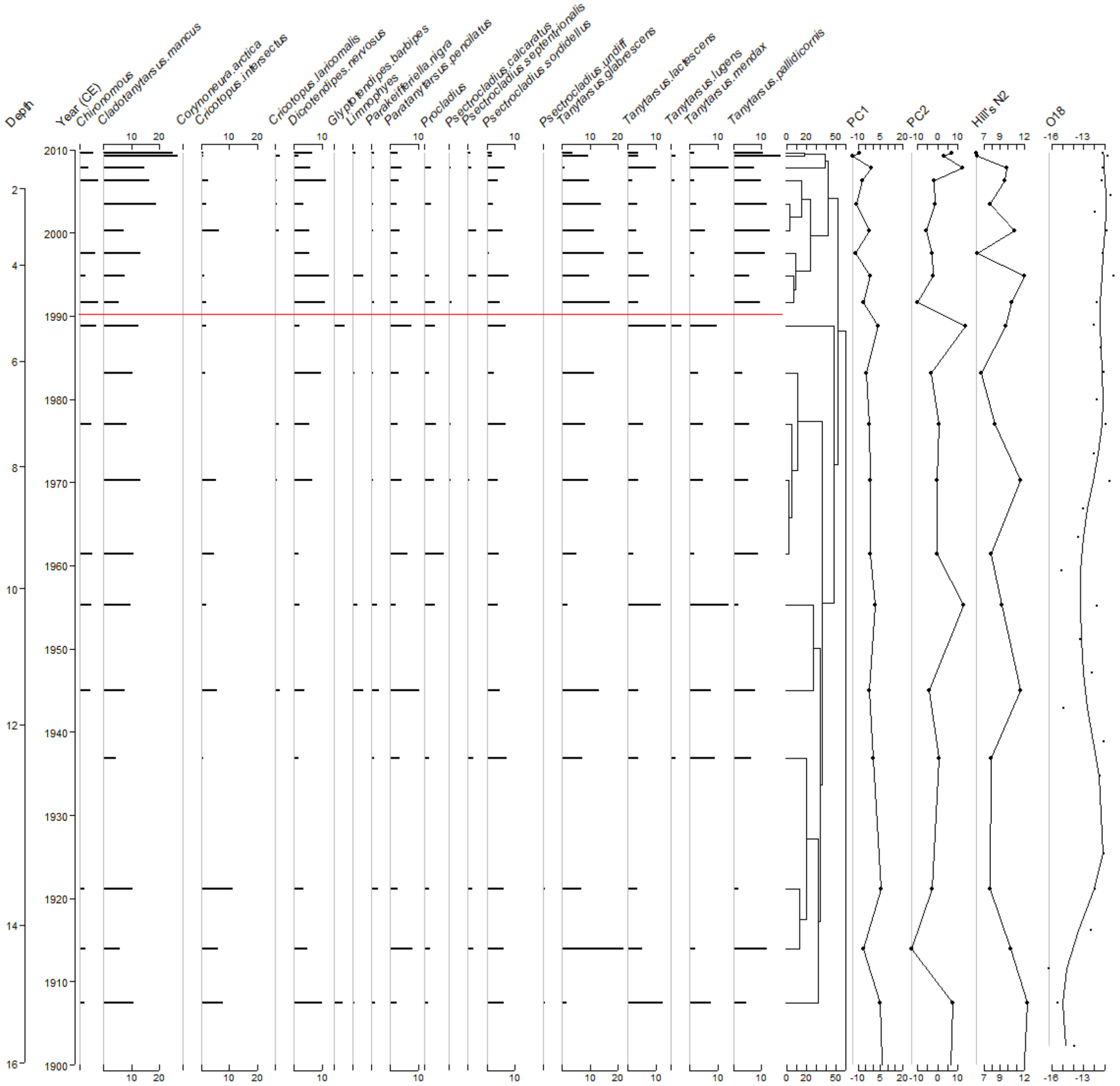

Figure 2.2: Stratigraphy of chironomid taxa and PCA sample scores as well as a cellulose-inferred pond water δ^{18}O (‰ VSMOW) record for the core of Left Lake, with only taxa present at a minimum of 1 % in 2 lakes shown. Species diversity calculated through Hills N2 analysis using all taxa from the respective depth interval. Horizontal red lines denote the time zones within the respective cores based on a constrained cluster analysis with significant zones identified through a broken stick model. Cellulose-inferred pond water δ^{18}O record extracted from Wolfe et al. 2011.

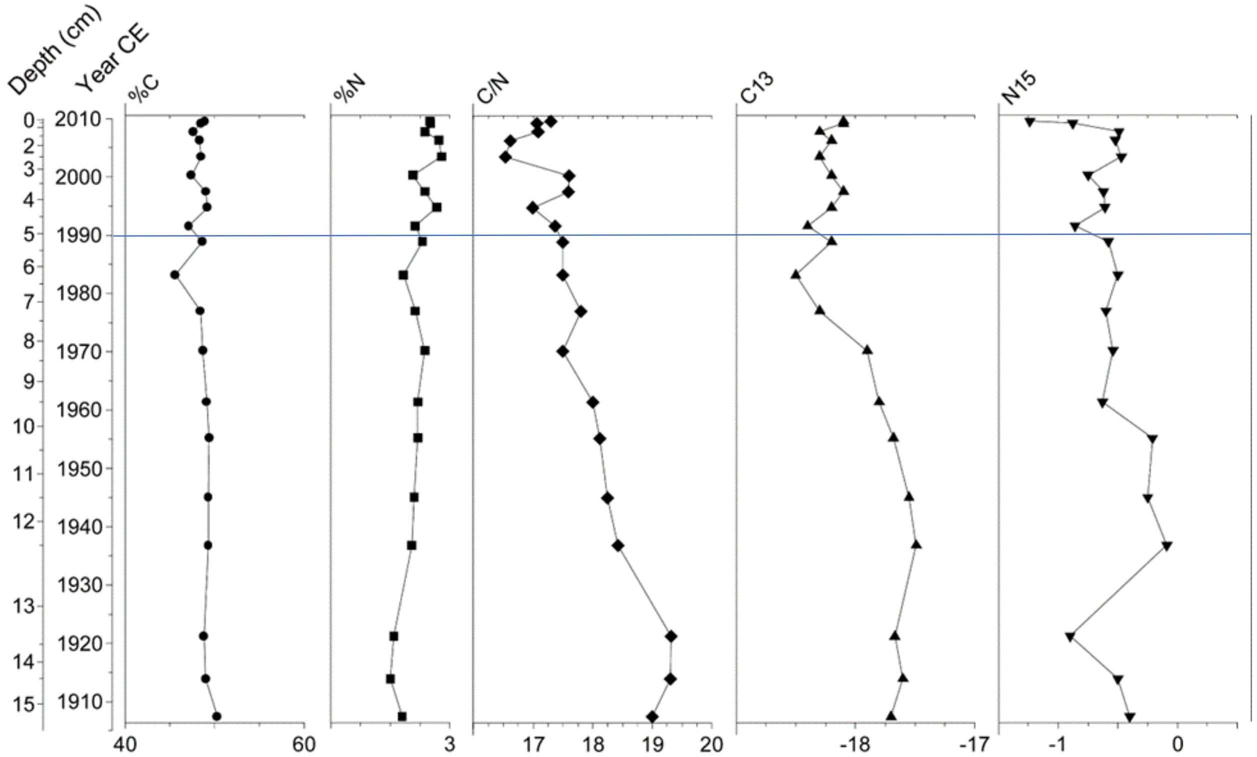

Figure 2.3: Geochemical stratigraphy for Left Lake including % organic carbon (%C_{org}), % total nitrogen (%N), carbon to nitrogen ratio (C/N), organic carbon isotope composition $\delta^{13}C_{org}$ (‰ VPDB) and nitrogen isotope composition $\delta^{15}N$ (‰ AIR). Blue line indicates zonation based on chironomid assemblage data

2.5.1.2 Larch Lake

A total of 1150 subfossil chironomid head capsules were extracted from the upper 10 cm of the sediment core from Larch Lake comprising 65 taxa that are described in 2 significant CONISS zones (Figure 2.4). **Zone 1** (9.25-8.25 cm; 1921-1938 CE) was dominated by *Limnophyes, Psectrocladius septentrionalis*-type*, Psectrocladius sordidellus*-type and *Cricotopus intersectus*-type. The average abundance of *Limnophyes* in this zone was 11% with a maximum of 14%

which occurred at 9.25 cm (1921 CE); this was the highest abundance of *Limnophyes* in this core. Both *Psectrocladius sordidellus*-type and *Psectrocladius septentrionalis*-type showed a trend of decreasing abundance in this zone. *Psectrocladius sordidellus*-type abundance decreased from 20% at 9.25 cm (1921 CE) to 19% at 8.75 cm (1926 CE) to 14% at 8.25 cm (1932 CE). *Psectrocladius septentrionalis*-type abundance decreased from 11% at 9.25 cm (1921 CE) to 9% at 8.75 cm (1926 CE) to 4% at 8.25 cm (1932 CE). *Cricotopus intersectus*-type had an average abundance of 4% in this zone with a low point of 2% occurring at 8.75 cm (1926 CE). Species richness increased from 21 at 9.25 cm (1921 CE) to 22 at 8.75 cm (1926 CE) to 26 at 8.25 cm (1932 CE).

The zonation for the geochemical stratigraphy for Larch Lake was again based on the statistical separation of the chironomid assemblages (Figure 2.5). The average %C_{org} in this zone was 58.3% with no directional trend and one notable deviation from the zone average occurring at 8.75 cm (1927 CE) when this value was 57.4%. The average %N value in this zone was 2.8% with no directional trends and no notable deviations from the zone average. The average C/N value in this zone was 20.9 with a decreasing trend over time and no notable deviations from the zone average. The average $\delta^{13}C_{org}$ value in this zone was -20.7‰ with no directional trend or deviations from the zone average value. The average $\delta^{15}N$ value in this zone was 0.1‰.

Zone 2 (7.75-0.25 cm; 1938-2010 CE) was dominated by *Psectrocladius sordidellus*-type, *Tanytarsus glabrecens*-type*, Tanytarsus mendax*-type*, Tanytarsus pallidicornis*-type and *Cladotanytarsus mancus*-type. In the early stages of Zone 2 (7.75-5.25 cm; 1938-1966 CE) *Psectrocladius sordidellus*-type had an average abundance of 10% with a high of 19% occurring at 7.75 cm (1938 CE) and a low of 6% occurring at 5.25 cm (1962 CE). The average abundance of *Tanytarsus glabrecens*-type during this same period was 6% with two low abundances occurring at 7.75 cm (1938 CE) and 5.25 cm (1966 CE), where abundance was 1% and 2%, respectively. *Tanytarsus mendax*-type had an average abundance of 8% in this period with the main deviations occurring at 5.75 cm (1957 CE), where abundance was 2% and at 5.25 cm (1962 CE), where this taxon's abundance was at its highest at 18%. *Tanytarsus pallidicornis*-type had an average abundance of 6% in this period with a high of 10% occurring at 7.25 cm (1943 CE) and a low of 1% occurring at 6.75 cm (1948 CE). *Cladotanytarsus mancus*-type had an average abundance of 7% in this period with the main deviations occurring at 6.75 cm (1948 CE) where abundance was 11% and at 6.25 cm (1953 CE) where abundance was 2%. During this period the average diversity value was 10.0

From 4.75-2.75 cm (1966-1990 CE) the chironomid assemblages were dominated by *Tanytarsus mendax*-type*, Cladotanytarsus mancus*-type and *Tanytarsus glabrecens*-type. *Tanytarsus mendax*-type had an average abundance of 11% during this period with the only deviation from that average occurring at 2.75 cm (1985 CE) where abundance was 7%. The average abundance of *Cladotanytarsus mancus*-type in this period was 8% with no directional shift in abundance and two notable deviations occurring at 4.75 cm (1971 CE) where abundance was 0% and at 3.25 cm (1980 CE) where abundance was 18%. *Tanytarsus glabrecens*-type had an average abundance of 6% during this period with one notable deviation from that average occurring at 3.75 cm (1976 CE) where the abundance of this taxon was 2%. *Psectrocladius sordidellus*-type was also present during this period with an average abundance of 7% although that average is slightly inflated by the 3.75 cm (1976 CE) interval where abundance was 12.5%; only one other interval was even above 6% (2.75 cm, 1990 CE). During this period the average diversity value was 9.9

From 2.25-0.25 cm; 1990-2010 CE the chironomid assemblages were dominated by *Tanytarsus mendax*-type*, Tanytarsus pallidicornis*-type*, Cladotanytarsus mancus*-type and *Psectrocladius sordidellus*-type. The abundance of *Tanytarsus mendax*-type fluctuated during this period. At 2.25 cm (1990 CE) abundance for this taxon was 13% and then decreased in the next interval to 8% before rising to 16% only to decrease to 8% at 0.25 cm (2010 CE). The abundance of *Psectrocladius sordidellus*-type followed a similar fluctuation trend as *Tanytarsus mendax*-type during this period. The abundance of *Psectrocladius sordidellus*-type was 11% at 2.25 cm (1990 CE), then decreased to 4% in the next interval before increasing to 10% and finally decreasing to 5% at 0.25 cm (2010 CE). The average abundance for *Tanytarsus pallidicornis*-type during this period was 8% with a high of 12% occurring at 1.75 cm (1994 CE) and a low of 5% occurring at 2.25 cm (1990 CE). The average abundance of *Cladotanytarsus mancus*-type during this period was 9% which was the highest average abundance of this taxa in this core; no intervals deviated far from that average. During this period the average diversity value was 9.8.

From 7.75-5.25 cm (1938-1966 CE) The average %C_{org} in Larch Lake was 57.2%. There was a general trend of decreasing %C_{org} over time during this period with a notable decrease from 58.5% to 56.5% that occurred between 7.25 cm (1943 CE) and 6.75 cm (1948 CE). The average %N value during this period was 3.3% with an increasing trend over time and a notable increase from 2.9 to 3.3% that occurred between 7.75 cm (1938 CE) and 7.25 cm (1943 CE). The average C/N value during this period was 17.3 with a decreasing trend over time with a noticeable decrease occurring between 7.75 cm (1938 CE) and 7.25 cm (1943 CE) where the C/N value fell from 20.1 to 18.0. The average $\delta^{13}C_{org}$ value in this period was -21.1‰ with most of the individual intervals having a value very close to that average with no directional trend.

The average $\delta^{15}N$ value during this period was 0.6‰, substantially higher than in Zone 1. There is a trend of increasing $\delta^{15}N$ over time during this period with a notable increase occurring between at 7.75 cm (1938 CE) and 7.25 cm (1943 CE), where $\delta^{15}N$ moved from 0.1 to 0.5‰.

From 4.75-2.75 cm (1966-1990 CE) The average $\%C_{org}$ was 56.2%. There was a general trend of decreasing $\%C_{org}$ over time during this period as almost every interval had a lower value than the previous. The average %N value in this period was 3.5% with no notable trends or deviations from that average. C/N values during this period were very consistent with all of the intervals having a value close to the average of 15.8. The average $\delta^{13}C_{org}$ value during this period was -21.0‰ with no directional trend or deviations from the average value. $\delta^{15}N$ values during this period were higher than they were between 7.75-5.25 cm (1938-1966 CE), with an average of 0.9‰, and more consistent with all of the intervals having a value close to the mentioned average.

From 2.25-0.25 cm; 1990-2010 CE the average $\%C_{org}$ was 55.6% with no intervals notably deviating from that average. The average %N value in this period was 3.5% with no noticeable trends or deviations from the previously mentioned average. The average C/N value in this period was 15.8, which is close to the average from the period between 4.75-2.75 cm (1966-1990 CE), though in this zone period the values were not quite as consistent with a notable deviation from the previously mentioned average occurring at 1.25 cm (1999 CE) where the C/N value was 15.3. The average $\delta^{13}C_{org}$ value in this period was -21.0‰ with no directional trends or deviations from that average. The average $\delta^{15}N$ value in this period was 1.0‰ with one notable deviation from that average occurring at 2.25 cm (1990 CE) where the $\delta^{15}N$ value was 1.1‰, which was the highest value in this core.

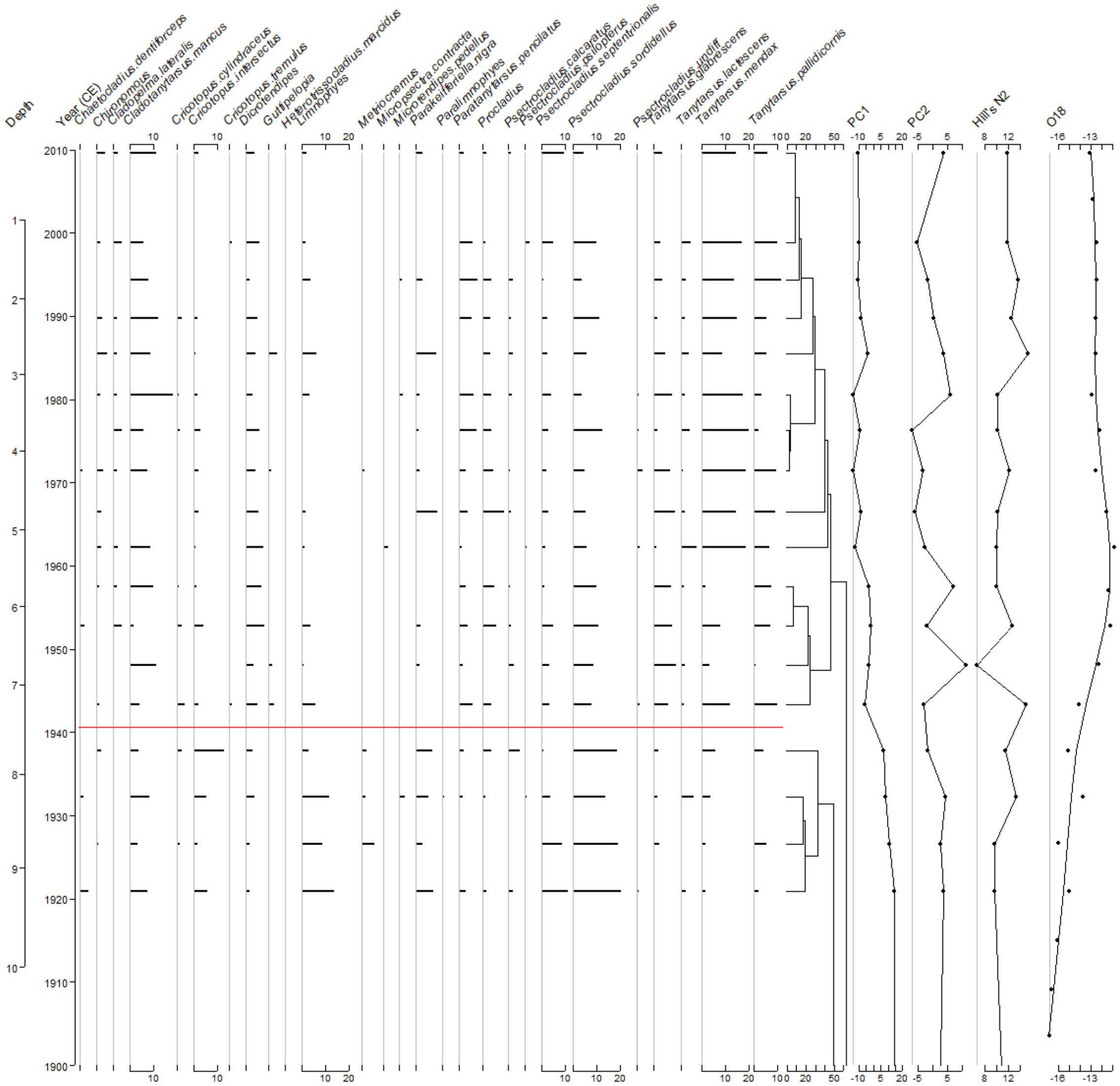

Figure 2.4: Stratigraphy of chironomid taxa and PCA sample scores as well as a cellulose-inferred pond water δ^{18}O (‰ VSMOW) record for the core of Larch Lake, with only taxa present at a minimum of 1% in 2 lakes shown. Species diversity calculated through Hills N2 analysis using all taxa from the respective depth interval. Horizontal red line denotes the time zones based on a constrained cluster analysis with significant zones identified through a broken stick model. The cellulose-inferred pond water δ^{18}O record was previously published in Wolfe et al. (2011).

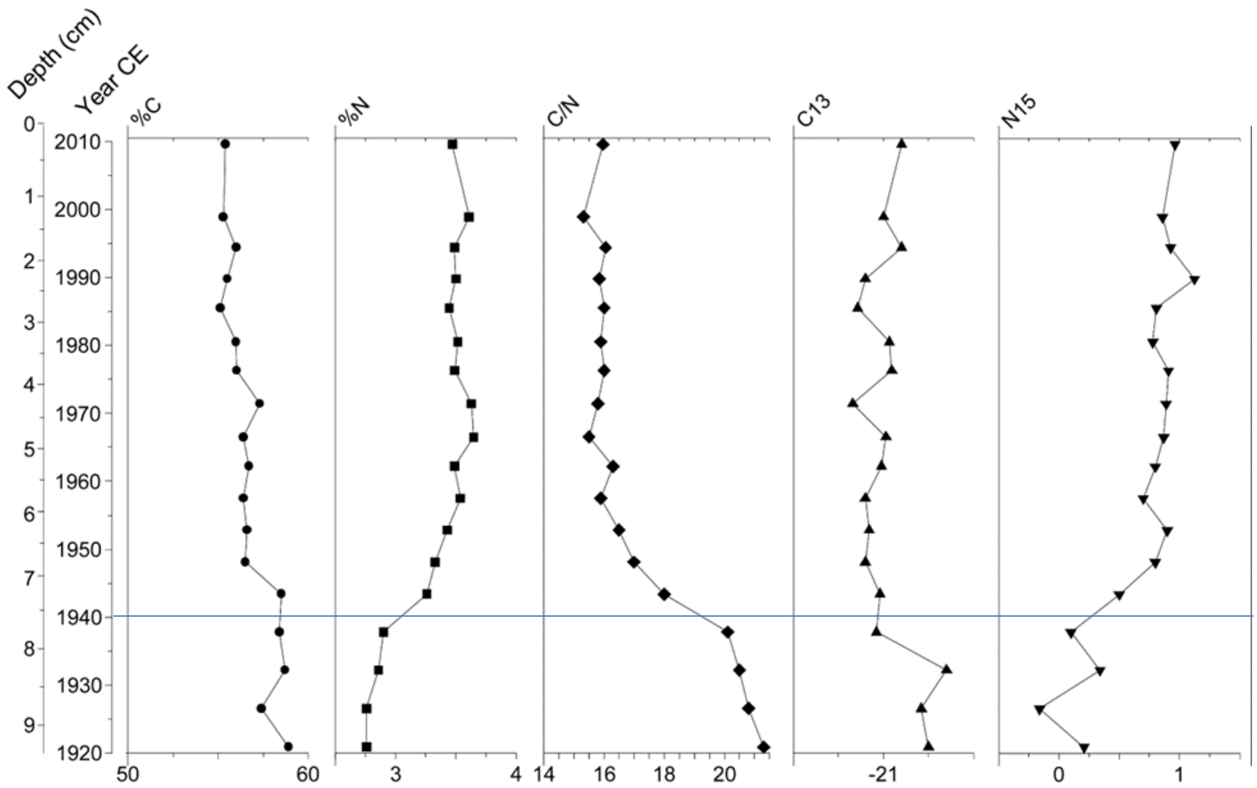

Figure 2.5: Geochemical stratigraphy for Larch Lake including % organic carbon (%C_{org}), % total nitrogen (%N), carbon to nitrogen ratio (C/N), organic carbon isotope composition $\delta^{13}C_{org}$ (‰ VPDB) and nitrogen isotope composition $\delta^{15}N$ (‰ AIR). Blue line indicates zonation based on chironomid assemblage data

2.5.2 Temperature reconstructions

Reconstructed July air temperatures from the core of Left Lake ranged from 10.8 to 12.4 °C with an increasing trend over time (Figure 2.6a). Zone 1 (15.25-14.5 cm; 1907-1913 CE) had an average inferred temperature of 11.7 °C. Zone 2 had an average inferred temperature of 11.6 °C. Zone 3 (9.5-5.25 cm; 1961-1989 CE) had an average inferred temperature of 11.9 °C and Zone 4 (4.75-0.25 cm; 1992-2010 CE) has an average temperature of 12.1 °C, the highest average temperature for any zone in this core. Reconstructed temperatures in Larch Lake ranged from

11.0 to 12.8 °C (Figure 2.6b.) Inferred temperatures in Larch Lake also showed an increasing trend over time and were generally less variable than those in Left Lake.

Zone 1 (9.25-8.25 cm; 1921-1932 CE) of Larch Lake had an average temperature of 11.6° C. Zone 2(7.75-5.25 cm; 1938-1962 CE) has an average temperature of 11.9 °C. Zone 3 (4.75-2.75 cm; 1971-1985 CE) has an average temperature of 12.0 °C and Zone 4 (2.25-0.25 cm; 1990-2010 CE) had an average temperature of 12.4 °C.

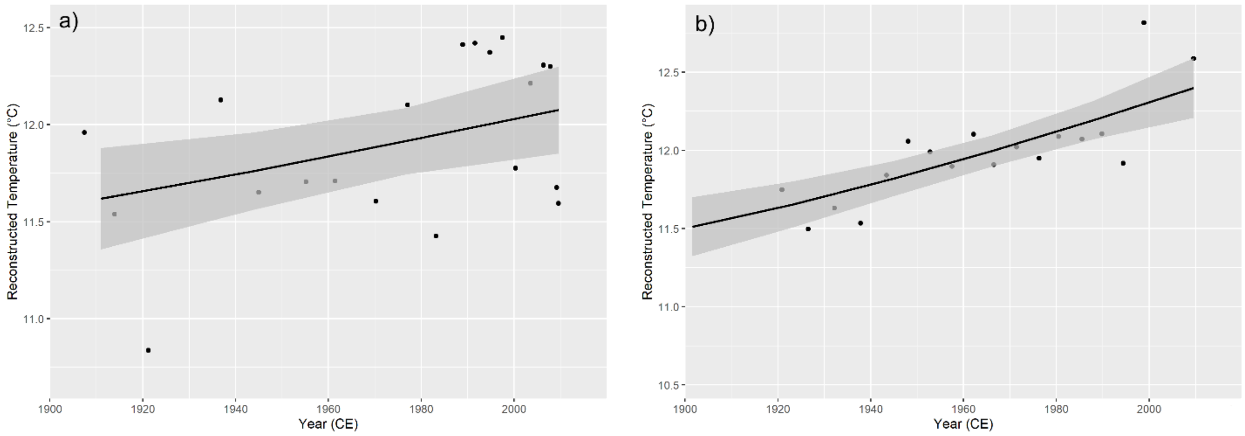

Figure 2.6: Reconstructed mean-July air temperatures for a) Left Lake, and b) Larch Lake using the Fortin et al. (2015) modern training set and the WA-PLS2 transfer function. A smoothing line and confidence intervals represent a generalized additive model (GAM) of the time series of inferred temperatures.

2.6 Discussion

While Hudson Bay Lowlands have experienced some of the most intensive warming in the entire circumpolar Arctic during the past century (Kaufman et al., 2009), our knowledge of how a combination of increasing temperatures and associated catchment-mediated changes may affect aquatic biota is limited.

Using cellulose-inferred pond water oxygen isotope records, Wolfe et al. (2011) found that the responses of ponds in western Hudson Bay Lowlands to anthropogenic warming was variable; Larch Lake and Left Lake had divergent hydrological responses to recent warming, with Left Lake showing clear signs of evaporative water loss throughout the 20th century and Larch Lake experiencing greater hydrological connectivity due to the establishment of a channel fen. Here, we show that the biological communities of these lakes experienced shifts consistent with inferred hydrological change; yet, we also show that both ponds also had a clear anthropogenic warming signal as demonstrated by similar results in a chironomid-based paleo-temperature reconstruction. Thus, the aquatic biota of these systems appear to have responded to an ecological shift that occurred as a result of warming that was independent of each pond's differing hydrology.

2.6.1 Inferred 20th century environmental changes

2.6.1.1 Left Lake

The most abundant chironomid taxa in Left Lake during the early stages of 20th century were *Cricotopus intersectus*-type*, Psectrocladius sordidellus*-type, and various Tanytarsini. These taxa are often indicators of a lake with a large littoral zone and macrophyte habitat (Brooks et al., 2007; Rühland et al., 2014). The high C/N ratios in this early period indicate that most of the organic matter input into Left Lake was from terrestrial sources (Meyers, 1997; Cummins et al., 2022). As $\delta^{15}N$ values during this period were close to 0‰, the system may have been nitrogen limited (Meyers, 2003; Wagner and Adrian, 2009). At around 1920 CE *Cricotopus intersectus*-

type and *Psectrocladius sordidellus*-type both began decreasing in abundance while various Tanytarsini taxa increased. C/N ratios also began declining at time indicating a possible shift to greater algal production, yet $\delta^{13}C_{org}$ and $\delta^{15}N$ values were mostly unchanged. The decrease in *Psectrocladius sordidellus*-type and *Cricotopus intersectus*-type could suggest a reduction of macrophytes, which would have also led to a decline in the C/N ratio (Meyers, 1997; Brooks et al., 2007; Rühland et al., 2014) and a transition to planktonic or benthic algal production more consistent with a shallow lake basin (Brodersen and Quinlan, 2006; Bornette and Puijalon, 2011; Qin et al., 2013; Moss et al., 2013). Cellulose-inferred pond water $\delta^{18}O$ records from Wolfe et al. (2011) indicate this was a period when Left Lake started experiencing increased evaporative stress and a decrease in water level.

From 1960 CE – ~1990 CE, abundances of *Psectrocladius sordidellus*-type and *Cricotopus intersectus*-type continued to decrease while the abundance of most of the Tanytarsini taxa either increased or changed little. This was also during the time when the abundances of *Dicrotendipes nervosus*-type and *Procladius* began to steadily increase. Taxonomic diversity was lower while reconstructed temperatures were higher. The C/N ratio also continued to decline in this zone as did $\delta^{13}C_{org}$, while $\delta^{15}N$ began decreasing from 0‰. This shift was potentially caused by a combination of factors including a continued decrease in water levels and further increases in primary production. *Dicrotendipes nervosus*-type and *Procladius* are both indicators of shallower lakes (Brooks et al., 2007; Porinchu et al., 2009) and the cellulose-inferred pond water $\delta^{18}O$ record from Wolfe et al. (2011) showed that Left Lake was continuing to experience high amounts of evaporative water loss at this time. Further reductions in the relative abundance of *Cricotopus intersectus*-type and *Psectrocladius sordidellus*-type indicates that macrophytes were continuing to disappear, likely because of the reduction in littoral aquatic habitat as the lake basin evaporated. An increase in algal biomass is also expected as a result of the increased evaporative water loss as some algae have been shown to proliferate more effectively in shallower waters (Qin et al., 2013; Cantonati and Lowe, 2014) and decreased water levels means available nutrients were also more concentrated (Gajewski et al., 2005).

From 1990 CE to 2010 CE, abundances of *Psectrocladius sordidellus*-type and *Cricotopus intersectus*-type continued to decrease, while the abundance of *Dicrotendipes nervosus*-type and multiple Tanytarsini taxa increased. For example, *Cladotanytarsus mancus*-type*,* an indicator of higher nutrient availability and higher temperature (Medeiros and Quinlan, 2011), increased from ~8 to 20% abundance during this period. *Dicrotendipes nervosus*-type and *Chironomous* also increased in abundance in this zone and are considered thermophilic (Brooks et al., 2007). The C/N ratio continued to decrease, likely indicating further increases in algal production and lower macrophyte abundance. The increase in abundance of *Dicrotendipes nervosus*-type as well as Tanytarsini, such as *Tanytarsus glabrecens*-type and *Tanytarsus pallidicornis*-type*,* indicate that Left Lake was continuing to desiccate with the margins of the lake likely representing flooded grassy areas rather than a typical littoral zone (Brooks et al., 2007; Bouchard et al., 2013). This is confirmed in the cellulose-inferred pond water $\delta^{18}O$ record, which indicated that there were several periods of time when water levels were lower in Left Lake than they had been since at least the start of the 20th century (Wolfe et al., 2011). This is also a period in which modern climate warming was affecting the region. A shift in biological assemblages attributed to warming ~1990 CE has also previously been documented in Hudson Bay Lowlands due to feedback mechanisms related to melting sea ice in Hudson Bay (Friel, 2011; Friel et al., 2014; Rühland et al., 2013; Rühland et al., 2015; Hadley et al., 2019).

2.6.1.2 Larch Lake

During the beginning of the 20th century the dominant chironomid taxa present in Larch Lake consisted mainly of *Limnophyes, Cricotopus intersectus*-type*, Psectrocladius sordidellus*-type, and various Tanytarsini taxa. These are all indicators of a lake with a large littoral zone; *Limnophyes* is an indicator for an especially shallow lake (Brooks et al., 2007). The temperature reconstruction for Larch Lake indicated that this period was the coldest in the core; species richness values were slightly higher than Left Lake. The high C/N ratio in this time indicates that most of the organic matter input into Larch Lake was from terrestrial sources, which were likely macrophytes considering the high abundance of *Cricotopus intersectus*-type and *Psectrocladius sordidellus*-type (Meyers, 1997; Brooks et al., 2007; Cummins et al., 2022). $\delta^{15}N$ values were also close to 0 ‰, suggesting nitrogen limiting conditions (Meyers, 2003). In ~1938 CE, Larch Lake underwent a fairly significant change in its chironomid assemblage, with *Limnophyes, Psectrocladius sordidellus*-type*, Cricotopus intersectus*-type, and *Metriocnemus* all decreasing in abundance or disappearing from the assemblage entirely. At this time C/N ratios also decreased, while $\delta^{15}N$ values increased. This shift was potentially triggered by the opening of the channel fen previously suggested by Wolfe et al. (2011). Based on cellulose-inferred pond water $\delta^{18}O$ records, Wolfe et al. (2011) concluded that Larch Lake started receiving water closer to the 1960s; however, our chironomid inferences and geochemical data indicate that this event may have occurred a couple decades prior. A sudden increase in water level caused by this channel would explain why *Limnophyes* disappeared, as it is an indicator of very shallow systems (Brooks et al., 2007). Higher water levels may also reduce macrophyte abundance which would explain why *Psectrocladius sordidellus*-type*, Cricotopus intersectus*-type, and *Metriocnemus* all decreased in abundance rather suddenly, while various Tanytarsini taxa were for the most part unaffected (Brooks et al., 2007; Rühland et al., 2014). A loss of macrophytes would also explain why C/N ratios fell rapidly (Meyers, 1997). The transition from a macrophyte dominated community to an algal dominated one is often a prolonged process though that can take decades (Sayer et al., 1999; Brodersen and Quinlan, 2006; Qin et al., 2013). The major transition seen here appears to happen over the course of only ~6-10 years. If there was a relatively sudden influx of water into Larch Lake it could be deleterious to the macrophyte community as macrophyte abundance and distribution are dependant on sunlight, which becomes less available as lake depth increases (Gasith and Hoyer, 1998; Greenway, 2007; Bornette and Puijalon, 2011).

From 1966 to 1990 CE, abundances of *Cricotopus intersectus*-type and *Psectrocladius sordidellus*-type continued to decrease, while some Tanytarsini, such as *Tanytarsus mendax*-type, increased in abundance. Similar to Left Lake, this assemblage shift may have been triggered by Larch Lake becoming more algal dominated as rising δ^{15}N values, and a declining C/N ratio, indicates an increase in aquatic production under less nitrogen limited conditions (Meyers, 2003). Unlike the previous shift in this lake, the changes in chironomid assemblages were more gradual as were the changes to the C/N ratio, which are more indicative of an algal related shift (Brodersen and Quinlan, 2006, Macdonald et al., 2014). A similar shift occurred in a thermokarst lake in the Seward Peninsula of Alaska in the 1960s, where C/N ratios fell while δ^{15}N values rose which indicate increased nitrogen availability and algal production (Medeiros et al., 2014).

Like Larch Lake, this shift also led to a decrease in the average abundance of *Psectrocladius sordidellus*-type while Tanytarsini taxa were for the most part unaffected (Medeiros et al., 2014). This shift is also likely not related to a depth change, as shallow indicator taxa such as *Dicrotendipes nervosus*-type or *Procladius* did not significantly increase in abundance like they did when Left Lake started getting shallower (Brooks et al., 2007). Further to that point, Wolfe et al. (2011) found that it was during this time period that cellulose-inferred pond water $\delta^{18}O$ records started to stabilize indicating that Larch Lake was no longer being as heavily influenced by evaporation.

After 1990 CE, the abundance of *Cricotopus intersectus*-type and *Psectrocladius sordidellus*-type changed little as did the abundances of most of the Tanytarsini taxa, except *Cladotanytarsus mancus*-type, which experienced its highest average abundance in the core. The C/N ratio and $\delta^{15}N$ values also did not change much nor did diversity values, reconstructed temperatures however did consistently increase. As was the case with Left Lake, this shift was likely temperature based. A significant climate shift ~1990 CE has been previously documented and warm water taxa like *Cladotanytarsus mancus*-type have been known to have increase in abundance likely as a result (Brooks et al., 2007; Rühland et al., 2014; Hadley et al., 2019).

2.6.2 Context within Hudson Bay Lowlands and other subarctic regions

In Hudson Bay Lowlands, chironomid-based paleolimnological studies have been limited (Rühland et al., 2014; Hargan et al., 2020). Rühland et al. (2014) performed an exploratory "top-bottom" survey of the ecological changes in several lakes in the eastern Lowlands region that occurred throughout the 20th century. As this was a study only utilizing the surface sediments and the bottom-core sediments, there can be no exact comparison to our study as they did not pinpoint exactly when chironomid assemblage changes may have taken place. That being said, most of the lakes in that study had chironomid assemblages similar to those seen here and had similar assemblage shifts including a loss of both *Psectrocladius sordidellus*-type and *Cricotopus,* and an increase of *Cladotanytarsus mancus*-type. Other Hudson Bay Lowland chironomid studies also had similar assemblages to those seen here with *Psectrocladius,* *Cricotopus* and Tanytarsini having similar abundances (Dickinson et al., 2014). The chironomid assemblages in Larch Lake and Left Lake are also similar to those in subarctic lakes in Europe (Olander et al., 1997; Porinchu and Cwynar, 2000; Brodersen et al., 2001) with Olander et al. (1997) stating that the chironomid fauna in those lakes represented those of a typical shallow Canadian Arctic lake and Brodersen et al. (2001) having similar assemblage shifts in response to rising nutrient levels and a loss of macrophytes. The biggest difference between the assemblages in Larch Lake and Left Lake and those in other lakes is that many lakes in other studies have much higher abundances of *Microtendipes* and *Corynocera ambigua*. The low abundances of *Microtendipes* may be due to the fact that Churchill is too far north for this taxon to proliferate; *Microtendipes* is known to be found in warmer lakes further south of tree line (Porinchu and Cwynar, 2000, Porinchu et al., 2009). Reasons for the absence of *Corynocera ambigua* are more difficult to determine as this taxon is believed to prefer cold, oligotrophic conditions, but has also been found in warmer, more nutrient rich lakes and has been seen to have a somewhat sporadic distribution in regions where it is found (Brodersen and Lindengaard, 1999; Brooks et al., 2007; Porinchu et al., 2009). Some of the major ecological shifts seen in Larch Lake and Left Lake since the beginning of the 20th century are also in line with those observed in other Hudson Bay Lowland and subarctic lakes. The increase in primary production seen in Left Lake in ~1920 CE has been documented in other Hudson Bay Lowland lakes (Ruhland et al., 2013; MacDonald et al., 2015; Shinneman et al., 2016). There were no paleolimnological studies that had any kind of ecological shift that occurred at the same time as the one seen in Larch Lake in 1938 CE, which highlights the unique nature of the opening of a channel fen. A number of lakes in Hudson Bay Lowlands experienced a shift in ~1965 CE that included increased primary production characterized by a more substantial decrease in C/N ratios (MacDonald et al., 2015; Friel, 2011;

Hargan et al., 2020). A similar shift occurred in a shallow tundra-taiga boundary lake in Alaska in the 1960s where C/N ratios fell and $\delta^{15}N$ values increased, in response *Psectrocladius* abundance fell while Tanytarsini taxa were mostly unaffected (Medeiros et al., 2014). This increase in primary production could have been caused by degrading permafrost (Medeiros et al., 2014), increased nutrients from bird feces (Macdonald et al., 2015), or potentially from increased fossil fuel burning (Galloway et al., 2008). Hadley et al. (2019) note that it is difficult to determine the exact cause of an increase in primary production as this increase is likely a mix of causes that could include a longer ice-free season brought on by previously recorded early 20th warming (Hughen et al., 2000; Keatley et al., 2006; Medeiros et al., 2012).

2.6.3 Differences between Left Lake and Larch Lake

Prior to 1938 CE, Left Lake and Larch Lake had similar chironomid assemblages, geochemical composition, and were under similar levels of evaporative pressure. Considering their close proximity and similar depth, the similarity between the two lakes was expected; however, some taxa did differ as well as the onset of assemblage changes through time. These differences can be heavily attributed to the divergent hydrological trajectories of these lakes.

Prior to the opening of the channel fen, Larch Lake had higher C/N ratios than Left Lake and lower $\delta^{13}C_{org}$ values, which indicates that Larch Lake likely had a higher abundance of macrophytes and less algal production, which may be due to the fact the Larch Lake and Left Lake occupy different vegetation zones (Meyers, 1997; White, 2012). After Larch Lake started receiving water, C/N ratios were on average lower than those in Left Lake indicating that aquatic primary production was greater in Larch Lake (Meyers, 2003). Higher $\delta^{15}N$ values suggest less nitrogen limiting conditions in Larch Lake than in Left Lake (Hadley et al., 2019; Macdonald et al. 2015). A greater depth was also likely deleterious for the macrophyte communities in Larch

Lake, which would have further reduced C/N ratios (Meyers, 1997; Gasith and Hoyer, 1998; Greenway, 2007; Bornette and Puijalon, 2011).

After ~1960 CE evaporative pressure seemingly has little influence on the water level of Larch Lake based on the fact that cellulose-inferred pond water $\delta^{18}O$ values only changed marginally after this time (Wolfe et al., 2011). In Left Lake however, $\delta^{18}O$ values continued to rise for the remainder of the core indicating that evaporative pressure was still having a significant impact on Left Lake's water levels. As a likely result of the lower habitat availability caused by decreasing the water levels, Left Lake generally has lower taxonomic richness than Larch Lake with taxa such as *Metriocnemus* and *Mesocricotopus* that are present in Larch Lake being rare or entirely absent in Left Lake. This may be due to the fact that Larch Lake had more littoral habitat and a higher surface area to depth ratio than Left Lake (Brooks et al., 2007; Luoto, 2009). Left Lake also had higher abundances of Tanytarsini taxa, which may also be due to the fact that Tanytarsini are considered to be a more generalist than most others and could better survive the loss of aquatic habitat (Rühland et al., 2014; Brooks et al., 2007).

Larch Lake also did not experience as significant of an ecological shift after 1990 CE as Left Lake did. This may be because at this point in time Larch Lake was likely much larger and deeper than Left Lake and larger deeper lakes have been shown to be somewhat less influenced by the effects of daily weather and have usually have a greater thermal capacity (Medeiros et al., 2012a). Left Lake also had negative $\delta^{15}N$ values, which are potentially related to variations in post depositional processes as the timing of the chironomid assemblage and C/N changes and $\delta^{15}N$ deviation from zero are still indicative of nitrogen increases (Wolfe et al., 2006).

2.7 Conclusion

The hydrological responses of ponds in Hudson Bay Lowlands to 20th century warming have been quite variable as warming can cause different responses in lakes based on their morphometry and landscape position with shallow lakes being especially vulnerable to this variation. Decades of increasing evaporative stress have significantly reduced the water levels in Left Lake which potentially desiccated completely in 2022 CE, which may have led to an increased loss of aquatic biodiversity. A similar fate may have befallen Larch Lake if the channel fen that opened in the mid-20th century was not there feeding this pond more water to offset the evaporative stress. Despite their divergent hydrology, warming has elicited a common ecological response from these ponds. Decreased ice cover, a longer growing season and increased nutrient inputs that may also be related to temperature increases have started shifting both ponds from being macrophyte dominated to being dominated by algae. This shift has been well documented in Hudson Bay Lowlands with many lakes experiencing extremely high levels of primary production. The rise in primary productivity seen here and in other Hudson Bay Lowland studies could be an indication that these lakes are starting to lose their ecological resilience and are shifting to becoming more eutrophic. The combination of eutrophication and decreasing depth due to evaporative pressure means that lakes like Larch Lake and Left Lake are at risk of either disappearing entirely or potentially undergoing an ecological transition to algal dominated systems that can only support a few generalist taxa. With how important these kinds of shallow ponds are to the ecology of Hudson Bay Lowlands it is important we have a solid understanding of the potential consequences of their continued degradation.

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CHAPTER 3: INFERRING THE INFLUENCE OF URBAN DEVELOPMENT ON FRESHWATER SYSTEMS IN A WARMING ARCTIC; THE CASE OF IQALUIT, NUNAVUT

3.1 Statement of Student Contribution

Study design by Connor Nishikawa and Andrew Medeiros, limnological data collection by Connor Nishikawa, Andrew Medeiros, Roberto Quinlan and Milissa Elliot, chapter writing by Connor Nishikawa with editorial contributions by Andrew Medeiros, Jennifer Eamer and Roberto Quinlan.

3.2 Abstract

Arctic ecosystems are considered to be especially vulnerable to the effects of environmental change, but the combined influence of a warming Arctic and expanding urban development is less clear. While temperature-driven change has long been the principal focus of studies on Arctic lakes, increasing local human populations, inadequate municipal infrastructure, and expansion of the resource extraction industry may now have a larger influence. Here, we present a chironomid-based paleolimnological assessment of two lakes within the urban boundary of Iqaluit, Nunavut, to determine if local human activities have become a primary driver of ecological stress to urban Arctic lakes. Iqaluit is one of the largest urban centers in the Canadian Arctic with a population that has been increasing for decades and a history of pollution that has affected nearby freshwater systems. Despite being exposed to urban development, both lakes were found to respond primarily to climate.

In particular, assemblage shifts during recent decades favored generalist taxa, including *Psectrocladius* and those of the Tribe Tanytarsini*,* over cold-water stenotherms, such as *Heterotrissocladius*, and *Corynocera oliveri*-type*,* that were concurrent with documented regional warming. However, these lakes do differ in landscape position; IQ04 is a large kettle lake with a pronounced profundal zone, whereas IQ01 is oblong with a larger shallow littoral reach. As a result, the chironomid assemblages differ due to lake morphometry and have responses to warming related to that difference. While human activities did not have a noticeable effect on the chironomid assemblages of these lakes, it is still important that these ecosystems be monitored given continued environmental stress expected with future warming and further urban development.

3.3 Introduction

The influence of the Anthropocene on freshwater ecosystems is multifaceted, with a number of interacting threats that may influence biodiversity, especially in a warming future. Due to Arctic amplification, which is the enhanced change in temperature in polar latitudes caused by factors such as the positive feedback loop associated with reduced albedo (Serreze and Barry, 2011), the effects of climate change are more pronounced than in most other regions in the world (Serreze et al., 2008). Near surface temperatures in the Arctic have increased by more than twice that of the global average since the 1970s (Hansen et al., 2010). Arctic freshwater lakes and ponds are under an especially great degree of environmental stress (Schindler and Smol, 2006) as temperature has been shown to play a critical role in mediating their chemical and biological processes (Michelutti et al., 2007). In comparison, temperate ecosystems can be affected by a more diverse set of environmental factors, such as catchment-mediated nutrient inputs (Blanco, 2014), terrestrial vegetation (Pleskot et al., 2022), and forest fires (Bixby et al., 2015).

While the influence of warming on the ecology of Arctic lakes has been previously observed (Medeiros et al., 2012a; Rühland et al., 2014; Beaudoin et al., 2016; Hadley et al., 2019), studies on the impact of local human populations on freshwater ecosystems, such as Belle et al. (2017) and Antoniades et al. (2011), have been limited in Canadian Arctic communities.

Arctic ecosystems have long been considered to be some of the least impacted by humans in the world (Weckstrom et al., 2010). Yet, human influences have been known to extend to polar regions, for example, through the atmospheric deposition of volatile pollutants (Fitzgerald et al., 2005) including the transport of mercury into northern food-chains (Gouin et al., 2004; Steffen et al., 2014; Obrist et al., 2017). Even though the influence of human populations may be small, northern regions are quickly developing, and northern communities have grown in recent decades expanding direct local influences on the environment. Several Arctic regions are expected to experience a substantial population increase over the next few decades (Heleniak, 2020). For example, the northern territories of Nunavut and the Yukon had the highest per capita population increases in Canada in 2020-21 (Statistics Canada, 2022). Environmental stress from increased populations is exacerbated by limited and aging infrastructure. For example, the inefficiency of generating electricity from diesel fuel combustion results in northern communities generating considerably higher emissions per capita than many of those in the rest of Canada (Das and Canizares, 2019). Urbanized Arctic settlements in particular seem to have a notable influence as inputs from these settlements including sewage effluent, increased sediments from road construction, and industrial runoff have been shown to have noticeable effects on nearby freshwater systems including eutrophication, higher heavy concentrations, and significant species turnover leading to a loss of taxonomic richness (Medeiros et al., 2011; Sivarajah et al., 2022). Arctic ecosystems are considered to be especially vulnerable to the influence of any kind of environmental stressor (Schindler and Smol, 2006), and climate change is considered to be a threat multiplier that has the potential to make the effects of other environmental stressors worse (Smol, 2010).

Arctic freshwater ecosystems are often considered to be sentinels of environmental change as their increased sensitivity to factors such as climate change and increased terrestrial inputs allows researchers to infer past environments from physical, chemical, and biological indicators preserved in their sediment history. The examination of lacustrine sediment records has enabled inferring the influence of Thule whalers from centuries ago (Hadley et al., 2010), despite the fact that their environmental impact was quite small by modern standards (Smol, 2016), as well as the impact from modern local human influences (Thienpont et al., 2013; Stewart et al., 2014).

Here, we analyze the lacustrine sediment history of two urban lakes and infer influences of environmental change across the Anthropocene from subfossil chironomid assemblages (Diptera: Chironomidae) in Iqaluit, Nunavut, Arctic Canada. Designated the capital of the territory in 1999, Iqaluit has had a relatively long history of local human influence for the Canadian Arctic. Iqaluit was founded in the 1940s as the United States airbase “Crystal Two”. Over time this base transitioned into a major settlement, emerging as one of Canada’s most rapidly growing municipalities (Shields and Weber, 2013). Urban development over the last 60 years has been well documented (Samuelson, 1998; Zahara, 2015), including significant impairment of local freshwater resources (Dick et al., 2010; Medeiros et al., 2011; 2021; Liboiron et al., 2021). Since biodiversity in Arctic ecosystems is often seen as having a bottom-up control due to the harsh climate limiting the abundances of several warm water taxa, small-scale influences from pollution may have a larger effect on species that already exist at thresholds of tolerance. For example, Medeiros et al. (2011) noted that species that are explicitly known as pollution tolerant in southern ecosystems were the first to disappear in impaired streams in Iqaluit due to severe climate-related stresses on fecundity. As such, we aim to determine whether increased urban development over the Anthropocene has superseded climate as the primary driver of ecological change in local urban lakes. As populations in Arctic communities continue to grow, it is important to understand how direct anthropogenic activities impact the trajectory of climate stressed ecosystems.

3.4 Methods

3.4.1 Site description

Iqaluit is located in Frobisher Bay, Baffin Island, and has grown in population from 6700 in 2011 to 7429 in 2021 (Statistics Canada, 2022). Iqaluit has an average July temperature of 8.2 °C and annual total precipitation averaging 404 mm (Station 2402590, Environment Canada). We examined two lakes within a short distance of the urban core of Iqaluit; IQ01 and IQ04 (Figure 3.1). The surficial geology of both lake catchments is Precambrian bedrock overlain by glacial deposits (Miller et al. 2005). There is little vegetation around the catchment area of either of the lakes, with a predominance of Arctic grasses, sedges, and a few species of moss. The primary difference between IQ01 and IQ04 is their proximity to urban development, with the catchment of IQ01 having experienced larger disturbances due to its location near the former American military site known as ‘Upper Base’ and the road was built across its catchment due to recent gravel-haul operations.

IQ04 is located near residential areas that have expanded into the area, and is within walking distance of the urban core, and as such IQ04 is often used as a camp-site by locals during the summer months.

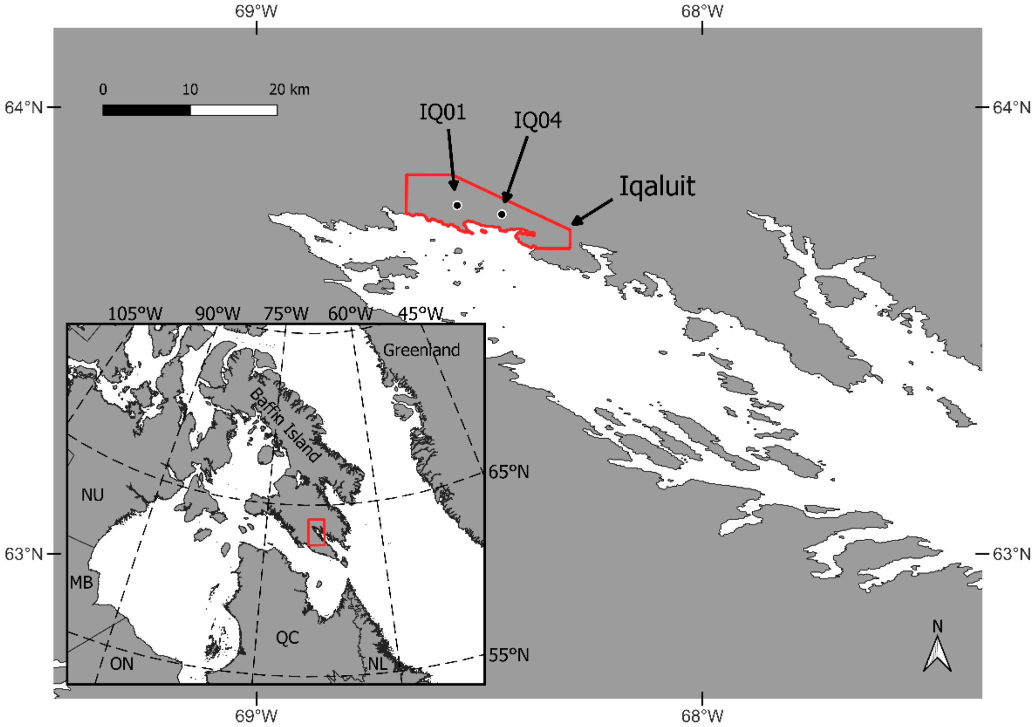

Figure 3.1: Location of the IQ01 and IQ04 coring sites near Iqaluit, Nunavut, Canada. The red box represents the National Topographic System (NTS) urban boundary for the City of Iqaluit.

3.4.2 Field sampling

Sediment cores were collected from the mid-basin of IQ01 and IQ04, Iqaluit, Nunavut, August of 2007, with the use of an Uwitec gravity corer (8.4 cm diameter) deployed from an inflatable boat. Cores were extruded at 0.5 cm intervals for the entire core lengths collected; 25 cm from IQ01, and 20 cm from IQ04. Each subsample was placed in labelled Whirl-Pak® bags, transported in a cooler, and stored at 4 °C at Dalhousie University in Nova Scotia Canada for processing and analysis.

At the same time as core collection, depth and environmental parameters (Temperature, Oxygen Reduction Potential, Conductivity, and pH) were measured with the use of a YSI multi-parameter probe (Table 3.1). Water samples for analysis of chlorophyll-*a*, major ions, nutrients, and dissolved and particulate organic carbon were also collected and transported to the Canadian Centre for Inland Waters for analysis following Environment Canada (1994) and are reported in Medeiros et al. (2012b).

Table 3.1 Environmental characteristics of sampled lakes IQ01 and IQ04 collected from Iqaluit, Nunavut, Canada in August of 2007.

Parameter	Units	IQ01	IQ04
Lat	DD°	63.78	63.76
Long	DD°	-68.55	-68.45
Area	Ha	1.8	8.8
Elev	m.a.s.l.	175	158
Depth	M	4.0	9.0
Temp	°C	10.3	10.2
pH		8.0	7.9
COND	$\mu S\ cm^{-1}$	19	43
ORP	Mv	123	165

(Lat = latitude, Long = longitude, Area = lake surface area, Elev = lake surface elevation, Depth = maximum depth, Temp = surface water temperature, COND = conductivity at 25 C, ORP = oxidation-reduction potential

3.4.3 Laboratory analysis

Sediment intervals for IQ01 were measured for ^{210}Pb in 2008 by alpha-spectrometry, and sediments were later measured for ^{137}Cs in 2010 by gamma-spectrometry at the Center for

Canadian Inland Waters, Burlington, Ontario. A logarithmic decline in excess ^{210}Pb determined sedimentation rates. A dating chronology was established for each core using a Constant Rate of Supply (CRS) model (Appleby, 2001). Loss-on-ignition was conducted on both cores following Heiri et al. (2011), where each sample was burned at 550 and 950 °C for two hours to calculate weight-loss derived %Organic and %Carbonate from sediments. The elemental and isotopic composition of organic carbon and nitrogen from sediments of IQ04 was completed at the University of Waterloo Environmental Isotope Laboratory following Wolfe et al. (2001) and are reported in the supplemental materials (ESM1). Samples for the analysis of elemental and isotopic composition of IQ01 were lost due to storage infrastructure failure at the University of Waterloo and could not be recovered.

Sediment cores were analyzed for subfossil chironomids at Dalhousie University following standard methods (Walker, 2001), where each sample was treated with potassium hydroxide (KOH) and heated at 75 °C for 30 minutes, with stirring for 15 minutes to deflocculate the sediment so that the head capsules could be more easily extracted. The treated sediment was then poured through nested sieves of 212 and 106 μm and rinsed with 95 % ethanol. Residues were then rinsed with distilled water and backwashed into Bogorov counting chambers and then sifted by hand using a dissecting microscope and specimens extracted with fine tipped forceps. Specimens from each subsample were mounted on glass slides using Entellen® and identified using a compound microscope to the most specific taxonomic resolution possible following Brooks et al. (2007) and Medeiros and Quinlan (2011). A minimum of 50 head capsules were enumerated from each sediment core interval (Quinlan and Smol, 2001); sediment intervals with fewer than 50 enumerated head capsules in the initial sediment subsample were resampled for more sediment until the minimum head capsule count was surpassed.

3.4.4 Statistical Analysis and Climate Reconstructions

Data analysis was completed using R statistical software v.4.1.2 (R Core Team, 2021). For each core, taxa were represented by their relative abundances, calculated as the percentage of identifiable chironomids in each sample for each interval, and plotted stratigraphically over depth and chronological time. Zonation was established using Constrained Cluster Analysis (CONISS) with the number of significant zones determined using a broken-stick model (Bennett, 1996). The taxonomic richness of each interval was determined using rarefaction analysis, where richness is E(Tn) or the number of each taxon present in each sample of n=50 (Birks and Line, 1992). A principal components analysis (PCA) was used to reduce the dimensionality of the assemblage dataset for each core. To aid in interpretation of dominant trends, rare taxa, i.e., those whose relative abundance never exceeded 1% of the total identifiable chironomids in any interval, were removed from the stratigraphic diagrams.

A chironomid-based paleotemperature model (Fortin et al., 2015) was applied to the relative abundances of chironomid taxa from each interval of both cores to estimate mean-July air temperature. A weighted-averaging partial-least squares (WA-PLS) transfer function was applied to the chironomid assemblages of each core interval using taxa-environmental relationships from a training set of 403 lakes from across northern North America (Fortin et al., 2015).

The chironomid assemblages from each core were compared through analogue matching to the training set data and inferred reconstructions were validated with a goodness-of-fit test using the "palaeoSig" package v.2.0-3 (Telford, 2019). The core trajectories of both lakes were passively plotted across training set space using the "analogue" package (Simpson, 2007).

3.5 Results

3.5.1 Core Chronologies

The ^{210}Pb record for IQ01 had an initial value of 30.7 Bq kg^{-1} at 0.5 cm and 6.0 Bq kg^{-1} at 6.0 cm where ^{210}Pb background was estimated as 1880 CE using the constant rate of supply model (Figure 3.2a). The model was also supported by the ^{137}Cs peak of 15.8 Bq kg^{-1} at 3.0 cm, corresponding to 1990 CE based on the ^{210}Pb profile. Interval ages below 6.0 cm depth were estimated through linear extrapolation with the basal interval of 25.0 cm being estimated to 1560 CE. The ^{210}Pb record for IQ04 had an initial value of 72.0 Bq kg^{-1} at 0.5 cm and 0.0 Bq kg^{-1} at 3 cm where ^{210}Pb background was estimated as 1900 CE using a constant rate of supply model (Figure 3.2b). The model was also supported by the ^{137}Cs peak of 139.5 Bq kg^{-1} at 2.0 cm, corresponding to 1960 CE based on the ^{210}Pb profile. Interval ages below 4.0 cm depth were estimated through linear extrapolation, such that the basal interval of 20.0 cm is estimated to 1320 CE. Extrapolated dates prior to 1885 CE should be interpreted with caution.

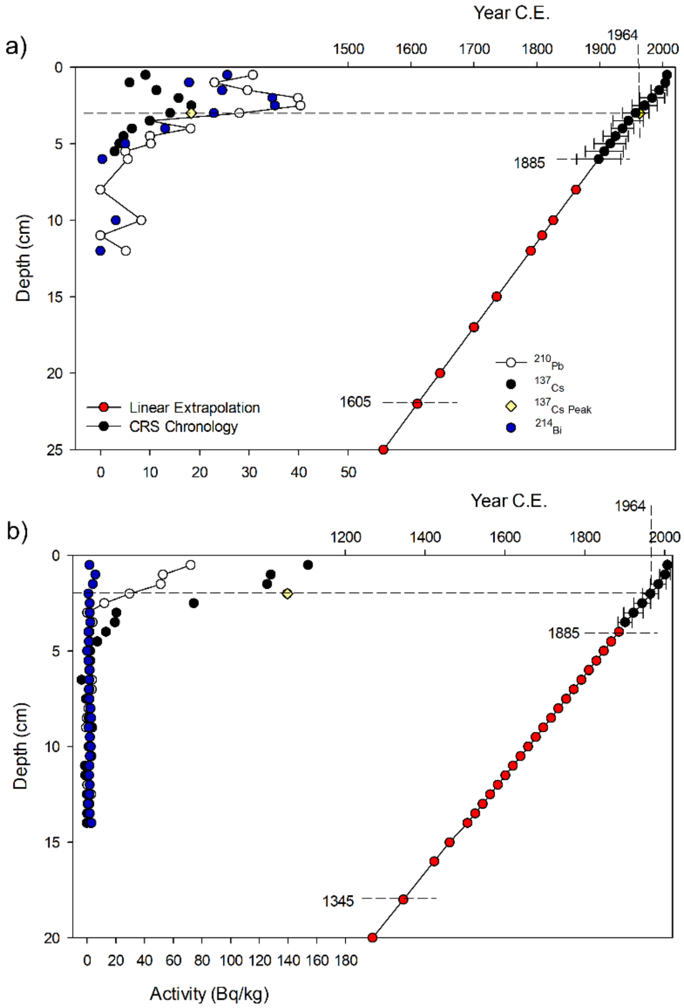

Figure 3.2: Sediment core chronologies based on the CRS ^{210}Pb model and ^{137}Cs peak for (a) Lake IQ01, and (b) Lake IQ04. Linear interpolation of chronology is indicated by red dots. Error bars represent the standard deviation of calculated sediment core interval age based on a CRS model chronology. The dotted line across the upper portion of each sediment core chronology represents the core depth of the ^{137}Cs peak.

3.5.2 Chironomid stratigraphy and PCA scores

3.5.2.1 IQ01

A total of 1269 chironomid head capsules were extracted from the IQ01 sediment core comprised of 59 different taxa. PCA axis 1 explained 33.8 % of the variance and axis 2 explained 20 %. Based on the compositional similarity of samples, 4 zones were identified as significant (Figure 3.3). Zone 1 (25 – 15 cm; 1560 - 1740 CE) was comprised of high abundances of *Stictochironomous rosenschoedi*-type*, Microspectra contracta*-type, and *Corynocera oliveri*-type, with maximum abundances of 45, 19, 17, and 11 %, respectively. *Stictochironomous rosenschoeld*-type declined in abundance in this zone from 45 % at 25 cm (1560 CE) to a minima of 3.5 % at 17 cm (1700 CE) before rebounding to 21 % at 15 cm (1740 CE). *Microspectra contracta*-type increased from 3 % at 25 cm (1556 CE) to 17 % at 22 cm (1610 CE) and then decreased to ~6 % for the remainder of the zone (20 – 15 cm; 1650 - 1740 CE). The abundance of *Corynocera oliveri*-type increased gradually through this zone from 2 % at 25 cm (1560 CE) to its maximum abundance of 11 % at 15 cm (1740 CE). The average species richness in this zone was 21.

Zone 2 (15 - 6 cm; 1740 -1900 CE) had high abundances of *Stictochironomous rosenschoeldi*-type*, Corynocera oliveri*-type and *Zalutschia lingulata pauca*-type. *Stictochironomous rosenschoeldi*-type reached a maximum abundance of 21 % at 15 cm (1740 CE), but rapidly decreased in abundance for the rest of the zone to a low of 6 % in 1900 CE. The abundance of *Corynocera oliveri*-type had an average of 14 % with the only substantial deviation occurring at 12 cm (1790 CE) when abundances increased to 22 %.

Zalutschia lingulata pauca-type, which never had an abundance over 7 % in the previous zones, was 0 % at 12 cm (1790 CE), but increased to 11 % from (15 – 6 cm; 1830 - 1900 CE). This zone was slightly more diverse than Zone 1 with an average species richness of 22.

Zone 3 (6 - 3.5 cm; 1900 - 1950 CE) had high abundances of *Stictochironomous rosenschoedi*-type*, Zalutschia lingulata pauca*-type*, Psectrocladius sordidellus*-type and *Corynocera oliveri*-type. *Stictochironomous rosenschoedi*-type increased in the first few intervals before reaching its peak at 24 % at 5 cm (1920 CE) but decreased to 8 % by 3.5 cm (1950 CE). The average abundance of *Zalutschia lingulata pauca*-type in this zone was 11 %, with one notable interval, 5.5 cm (1910 CE), where the abundance was 15 %. The average abundance of *Psectrocladius sordidellus*-type during this period was 7 %. *Corynocera oliveri* had an average abundance of 8 %, which is lower than the 14 % seen in Zone 2. Species richness for Zone 3 was 23.

Zone 4 (3.5 - 0.5 cm; 1950-2007 CE) featured high abundances of *Zalutschia lingulata pauca*-type*, Corynocera oliveri*-type*, Zalutschia*-type C and *Stictochironomous rosenschoedi*-type. The average abundance of *Zalutschia lingulata pauca*-type in this zone was 5 %, significantly lower than in Zone 3. The average abundance of *Corynocera oliveri*-type was also lower than Zone 3, decreasing to 5 %. In contrast, the average abundance of *Zalutschia*-type C was higher, rising to 14 %, with a high of 26 % occurring 0.5 cm (2007). *Stictochironomous rosenschoedi*-type had an average abundance of 10 % in this zone with no significant deviations from that average. Zone 4 had an average species richness of 23, very similar to the previous zone.

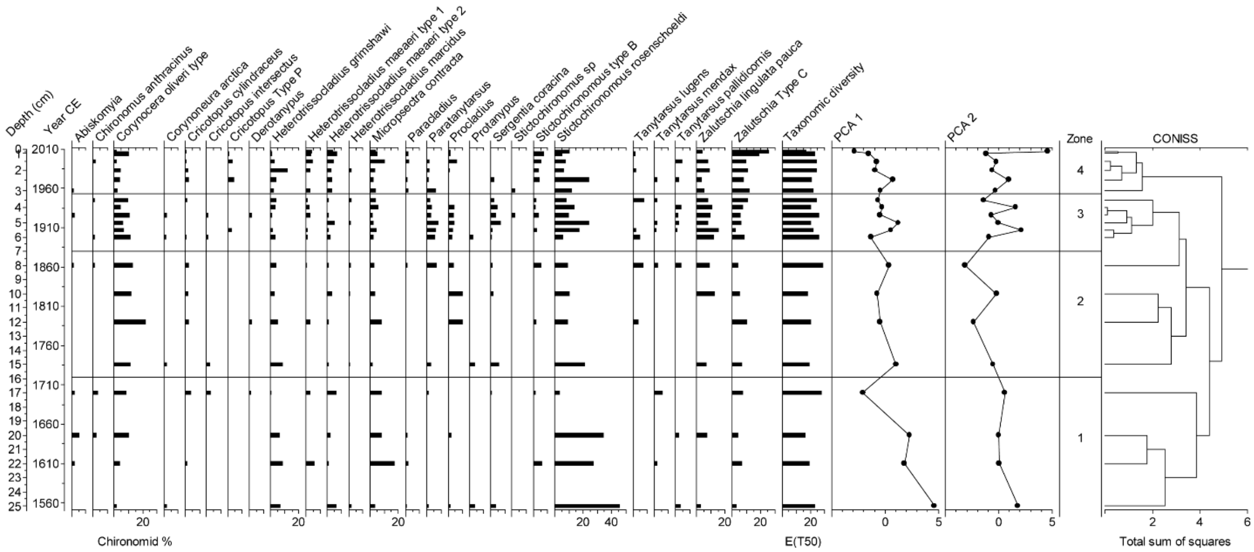

Figure 3.3: Stratigraphy of chironomid taxa and PCA sample scores for the IQ01 core, with only taxa present at a minimum of 1 % in 2 lakes are shown. Taxonomic richness was calculated through rarefaction analysis using all taxa from the respective core interval. Horizontal lines denote the boundaries between chronological zones of the sediment core assemblages, based on a constrained cluster analysis with significant zones identified through a broken stick model.

3.5.2.2 IQ04

A total of 1564 subfossil chironomid head capsules were extracted from the sediment core comprised of 64 taxa. Principal components analysis (PCA) was used to reduce the dimensionality and display significant trends in chironomid assemblage through time. For IQ4, Axis 1 explained 41.9 % of the variance and Axis 2 explained 12.9 %. Based on the compositional similarity of samples 4 zones were identified as significant (Figure 3.4). Zone 1 (19 - 13 cm; 1320 - 1540 CE) was mainly dominated by *Heterotrissocladius maeaeri*-type 1, *Zalutschia*-type C*,* and *Heterotrissocladius maeaeri*-type 2.

Heterotrissocladius maeaeri-type 1 had an average abundance of 29 % in this zone with a low of 23 % occurring at 19 cm (1320 CE), and a high of 35 % at occurring at 16 cm (1430 CE). *Zalutschia*-type C had an average abundance of 9 % with a high of 17 % occurring at 16 cm (1430 CE) and a low of 5 % occurring at 19 cm (1320 CE). *Heterotrissocladius maeaeri*-type 2 had an average abundance of 8 % with a high of 9 % occurring at 15 cm (1470 CE) and a low of 6 % occurring at 16 cm (1430 CE). The average taxonomic richness in this zone was 25, increasing from 20 at 19 cm (1320 CE) to 24 at 15 cm (1470 CE) to 27 at 13 cm (1540 CE).

Zone 2 (11– 5.5 cm; 1620 - 1830 CE) was mostly dominated by *Heterotrissocladius,* especially *Heterotrissocladius maeaeri*-type 1, which had an average abundance ~25 %. *Heterotrissocladius maeaeri*-type 2 decreased from 19 % at 11 cm (1620 CE) to 7 % at 7 cm (1770 CE), rebounded to 14 % at 6.5 cm (1790 CE), and then fell again to 7 % at 5.5 cm (1830 CE). Abundances of *Heterotrissocladius grimshawi*-type in this zone fluctuated between 3 and 13 % from 11 – 8 cm (1620 – 1730 CE) before falling to 2% at 5.5 cm (1830 CE). Zone 2 had an average richness of 24.

Zone 3 (6 – 3 cm; 1850 - 1900 CE) was dominated by *Heterotrissocladius maeaeri*-type 1, *Heterotrissocladius maeaeri*-type 2, *Paracladius*, *Tanytarsus pallidicornis*-type, and *Zalutschia*-type C. The average abundance of *Heterotrissocladius maeaeri*-type 1 in this zone was 27 %. Average abundance of *Heterotrissocladius maeaeri*-type 2 in this zone was 12 % with a high of 16 % occurring at 5 cm (1850 CE) and a low of 10 % occurring at 4.5 cm (1870 CE). The average abundance of *Zalutschia*-type C in this zone was 11 % with a high of 15 % occurring at 4.5 cm (1870 CE) and a low of 6 % occurring at 4 cm (1890 CE). The average abundance of *Tanytarsus pallidicornis*-type in this zone was 11 %, with high point occurring at 5 cm (1850 CE) where abundance was 18%, and low of 6 % occurring at 4.5 cm (1870 CE).

Paracladius had an average abundance of 5 % with a high of 9 % occurring at 3.5 cm (1900 CE), and a low of 2 % occurring at 4.5 cm (1870 CE). Average taxonomic richness in this zone was 22.

Zone 4 (3 – 1 cm; 1920 - 2007 CE) was dominated by *Heterotrissocladius maeaeri*-type 1, *Heterotrissocladius maeaeri*-type 2, and *Zalutschia*-type C. In the first interval of this zone *Heterotrissocladius maeaeri*-type 1 abundance was 34 %, abundance then increased to over 50 % from 2.5 - 2.0 cm (1940 -1960 CE). At 1.5 cm (1980 CE) abundance of this taxon then decreased to ~30 % for the remainder of the interval. For the first 3 intervals of this zone (3 - 2 cm; 1920-1960 CE) the average abundance of *Heterotrissocladius maeaeri*-type 2 was 4 % which increased to 12 % at 1.5 cm (1980 CE) and then to 16 % at 1 cm (2007 CE). *Zalutschia*-type C showed a similar trend where the abundance was never above 10 % for the first 3 intervals but then climbed to 14 % at 1.5 cm (1980 CE) and to 11 % at 1 cm (2007 CE). Zone 4 had an average taxonomic richness of 19.

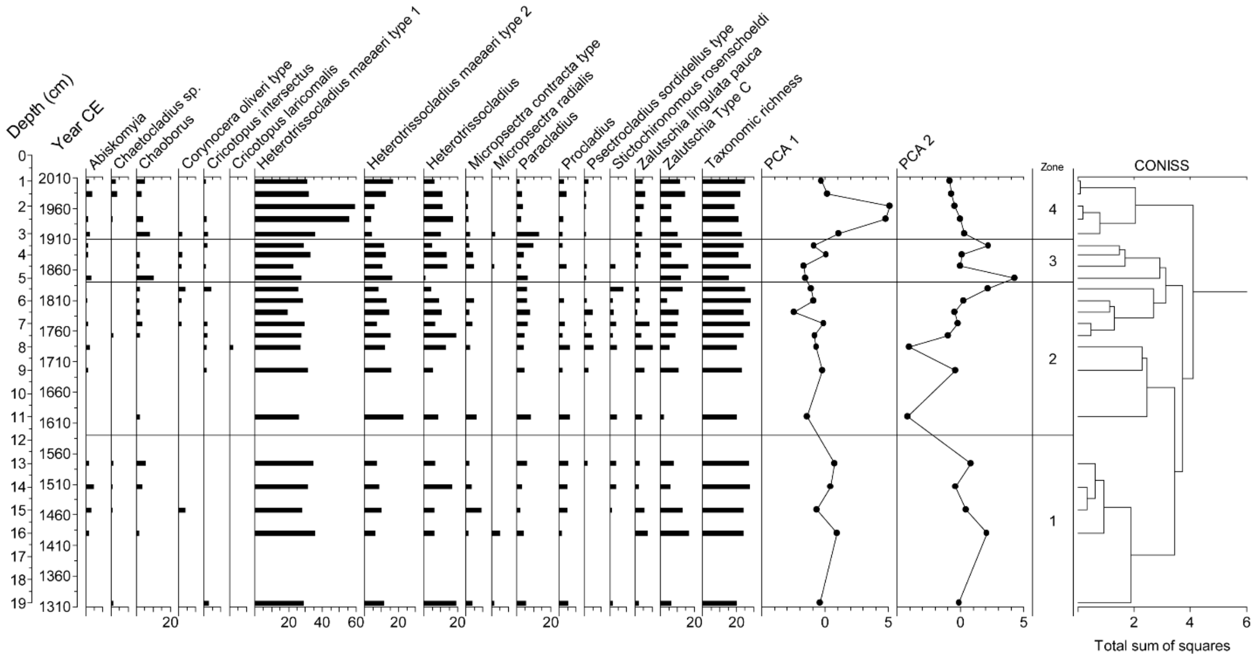

Figure 3.4: Stratigraphy of chironomid taxa and PCA scores for the core of IQ04, only those representing a minimum of 1 % in 2 lakes are shown. Species richness calculated through rarefaction analysis using all taxa from the respective depth interval. Horizontal lines denote the time zones within the respective cores based on a constrained cluster analysis with significant zones identified by a broken stick model.

3.5.3 Temperature reconstructions

Reconstructed July air temperatures from the core of IQ01 ranged from 7.4 to 9.9 °C (Figure 3.5a). Throughout the length of this core inferred reconstructed temperatures were relatively consistent with few notable outliers. Inferred temperatures were generally higher after Zone 1 (1560-1740 CE) by ~0.5°C, then declined by ~0.5 °C through Zone 2 before increasing again by ~0.4 °C in Zone 4. In contrast, inferred July air temperatures from the IQ04 were more variable, and lacked the same trend as IQ01. Inferred temperatures for IQ04 ranged from 6.0 to 9.6 °C (Figure 3.5b) with notable outliers that represented lower temperatures than the general trend.

The overall warming trend for IQ04 was ~0.4 °C, but recent intervals suggest a stronger warming signal in Zone 4 than was observed for IQ01.

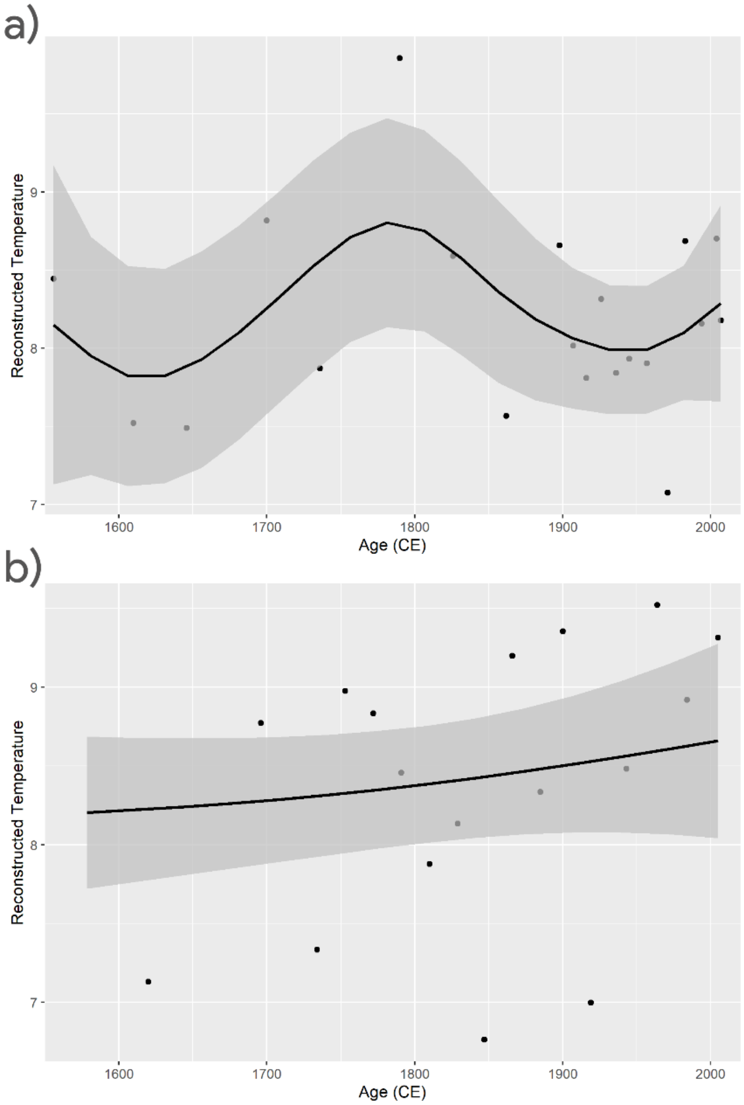

Figure 3.5: Reconstructed mean-July air temperatures for a) IQ01, and b) IQ04 using the Fortin et al. (2015) modern training set and the WA-PLS2 transfer function. A smoothing line and confidence intervals represents a generalized additive model (GAM) of the time series of inferred temperatures.

3.6 Discussion

While temperature is known to have a strong influence on the trajectory of Arctic freshwater ecosystems, less is known about how local human activities may impact these systems. For temperate lakes, anthropogenic stressors can influence both water quality as well as aquatic trophic systems (Garcıa-Rodrıguez et al., 2002; Amoatey and Baawain, 2019). However, climate change has also been shown to amplify the effects of other ecosystem stressors (Smol, 2010). Here, we assessed whether local anthropogenic influences from the relatively urbanized region of Iqaluit are observable, and whether these impacts have superseded climate-based forcings previously identified across northern Canada (Fallu et al., 2005, Smol and Douglas 2007, Campbell et al., 2018).

3.6.1 Climate related change

Chironomid taxa are known to have very specific temperature optima, therefore shifts in chironomid assemblages tend to be indicative of changes in regional climate, especially in the Arctic (Brooks et al., 2007). Zone 1 of IQ01 (1560-1740 CE) was dominated in its early stages by *Stictochironomous rosenschoedi*-type, but the abundance of this taxon fell dramatically, which coincided with an increase in the abundance of taxa with a higher thermal optima, such as those of the Tribe Tanytarsini, and an increase in taxonomic richness. This decrease in abundance may be primarily temperature driven as *Stictochironomous rosenschoedi*-type is a cold stenotherm (Brooks et al., 2007). *Stictochironomous rosenschoedi*-type also has a similar habitat preference to taxa of the Tanytarsini and *Psectrocladius sordidellus*-type, which both increased in abundance when *Stictochironomous rosenschoedi*-type started to disappear. This suggests that the shift in taxa was consistent with a change in overall temperature and not differences in lake level (Brooks et al., 2007).

This increase in temperature may have also led to an increase in available nutrients in IQ01, as taxa like *Corynocera oliveri*-type began increasing in abundance and are positively associated with higher nutrient conditions in cold regions (Gajewski et al., 2005, Keatley et al., 2006). Zone 2 (1740-1900 CE) had the highest inferred temperature of the core, a period when *Stictochironomous rosenschoedi*-type continued to decrease in abundance, replaced by generalist Tanytarsini and *Zalutschia lingulata pauca*-type, which has a higher thermal optima (Brooks et al., 2007). *Zalutschia* is also an indicator for dystrophic lakes with high humic content, which may also increase with warmer temperatures (Brooks et al., 2007; Lipczynska-Kochany, 2018). Zone 3 (1900-1950 CE) was similar to Zone 2 with similar taxonomic richness and inferred temperature. Zone 4 (1950-2007 CE) had similar taxonomic richness levels to Zone 3, but inferred temperatures were less consistent. This zone saw declines in the relative abundance of both Corynocera *oliveri*-type as well as *Stictochironomous rosenschoedi*-type. As these taxa declined, various warm water taxa such as Tanytarsini and Tanypodinae proliferated in this zone.

Lake IQ04 contrasted with Lake IQ01 in that it had higher abundances of *Heterotrissocladius* and lower abundances of *Stictochironomous rosenschoedi*-type*,* which is consistent with the fact that IQ04 is a deeper and colder lake. Prior to 1620 CE, IQ04 had the highest taxonomic richness of its record, primarily dominated by *Heterotrissocladius maeaeri*-type 1, a cold stenothermic taxon (Brooks et al., 2007). Over time taxonomic richness and abundance decreased; the interpolated timing of this decrease suggests that assemblages could have been responding to the early stages of the Little Ice Age, 1375-1820 (Moore et al., 2001). From 1850-1900 CE reconstructed lake temperatures changed little but taxonomic richness decreased slightly as cold-water taxa such as *Paracladius* and *Zalutchia*-type C became more prevalent as well as did some warmer water taxa like *Tanytarsus pallidicornis*-type (Brooks et al., 2007).

Following 1920 CE reconstructed temperatures consistently increased over time and species richness values were some of lowest of the entire core. The beginning of the 20th century has been inferred by other paleolimnological studies as a time when much of the eastern Canadian Arctic experienced a substantial shift in climate caused by anthropogenic warming (Perren et al., 2003; Keatley et al., 2006; Kaufman et al., 2009; Medeiros et al., 2012a). IQ01 responded to that warming in a similar way to IQ04 in that cold water adapted taxa, such as *Corynocera oliveri*-type, start to decrease in abundance, although reconstructed temperatures in the early 20th century do not rise as quickly in IQ01 as they do in IQ04, and species richness is also more variable during this period in IQ01 than IQ04. While the assemblage shift around 1950 CE is consistent with the timing of the development of the airbase in modern day Iqaluit, this shift is still most likely driven by temperature changes. Studies such as Kaufman et al. (2009) have stated that the early 20th century is when Arctic temperatures started to climb in response to anthropogenic warming, but also that the mid 20th century (~1950 CE) was when temperatures spiked to being, at the time, the highest in over 2000 years. This is consistent with the chironomid assemblage shifts for IQ01 at this time as the loss of cold-water taxa and increase in reconstructed temperatures are more pronounced than those in ~1900 CE. This temperature shift has also been observed in locations across the eastern Arctic including Baffin Island (Thomas et al., 2008), central Nunavut (Medeiros et al., 2012a) and Greenland (Axford et al., 2019). Based on the timing of the chironomid assemblage shift (~1950 CE), the notable decrease of cold-water taxa such as *Stictochironomous rosenschoedi*-type*, Corynocera oliveri*-type and *Tanytarsus lugens*-type*,* and the rising reconstructed temperatures in this zone, this is most likely when IQ01 started experiencing the more pronounced effects of anthropogenic global warming.

Therefore, the chironomid assemblage shifts in IQ01 can be seen as typical for an eastern Arctic lake responding to previously inferred regional temperature shifts. The major shifts in chironomid assemblages seen in IQ04 are consistent with previously reported climate shifts in the eastern Canadian Arctic. A gradual warming signal starting ~1850 CE has been observed in other Baffin Island lakes and may have been the result of the ending of the Little Ice Age (Moore et al., 2001, Hughen et al., 2000). During this period there may also have been a slight expansion of the littoral zone of IQ04 as evidenced by the increased abundance of taxa such as *Stictochironomous rosenschoeldi*-type and *Tanytarsus pallidicornis*-type. The shift following 1920 CE could be attributed to anthropogenic warming that began at the start of the 20th century, as the abundance of cold stenothermic taxa, such as *Corynocera oliveri*-type and *Stictochironomous rosenschoeldi*-type, decreased (Brooks et al., 2007). This early 20th century warming signal has also been regularly observed in reconstructions of lakes around Baffin Island (Hughen et al., 2000, Beaudoin et al. 2016) and across the Canadian Arctic (Perren et al., 2003; Keatley et al., 2006; Kaufman et al., 2009; Medeiros et al., 2012a); however, these studies were performed on remote lakes located long distances from any kind of urban setting. Therefore, as was the case for IQ01, all of the major assemblage shifts that occurred in IQ04 could be attributed to chironomid responses to shifts in regional climate.

3.6.2 Local Differences

Despite being located in relatively close proximity to each other, IQ04 and IQ01 had notably different chironomid assemblages and responses to climate change; however, these differences can be mostly attributed to differences in the morphology between the lakes. While chironomids are known to respond the temperature, it is important to note that lake morphometry also has a strong influence on chironomid community composition (Gajewski et al., 2005).

IQ04 is a relatively deep kettle lake with a pronounced profundal zone, and the chironomid assemblages of IQ04 are consistent with that type of lake morphometry. The assemblages for IQ04 were heavily dominated by *Heterotrissocladius,* which has been shown to be most commonly found in deeper colder lakes (Porinchu and Cwynar, 2000). *Heterotrissocladius* is commonly found across the eastern Canadian Arctic (Medeiros and Quinlan, 2011) and more specifically around Baffin Island (Thomas et al., 2008). IQ04 also has high abundances of *Paracladius* and *Abiskomyia* compared to IQ01, which is also consistent with IQ04 being a deeper and colder lake, as these taxa are often more abundant in large, cold lakes (Walker et al., 1997, Gajewski et al., 2005). The chironomid assemblages seen in IQ04 are mostly consistent with those seen in lakes around Baffin Island (Thomas et al., 2008), including those in more remote locations (Francis et al., 2006, Axford et al., 2009). The main difference between the assemblages of IQ04 and those seen in other Baffin Island studies, such as Francis et al. (2006), is that IQ04 had very low abundances of *Oliveridia* and *Pseudodiamesa*, which are also cold stenotherms; although, these taxa start to disappear when lake temperatures reach 7 °C, and most of the reconstructed temperatures for IQ04 are at or above that temperature threshold. IQ04 also had a less pronounced response to 20th century warming as it only experienced one assemblage shift after 1900 CE as opposed to the two that IQ01 experienced and the taxonomic richness and reconstructed temperatures of IQ04 changed less than they did in IQ01. This difference in climate response is also landscape-based as deeper and larger basins have been shown to be less influenced by the effects of variable daily weather and have a greater thermal capacity (Medeiros et al., 2012a). IQ04 also has a larger profundal zone than IQ01 and profundal chironomid taxa are known to be less influenced by temperature shifts than those that live in the littoral zone (Larocque-Tobler et al., 2010).

This difference in zone warming may also explain why the *Heterotrissocladius* taxa of IQ04 are less affected by warming than some of the other cold stenotherms in these lakes as *Heterotrissocladius* are known to occupy the profundal zone of Arctic lakes instead of the littoral (Walker et al., 1991; Brooks et al., 2007).

The chironomid assemblages of IQ01 are indicative of a shallower lake with a more pronounced littoral zone. The most notable difference between IQ01 and IQ04 is that IQ01 has much lower abundances of *Heterotrissocladius,* this is most likely due to IQ01 being smaller, shallow, and warmer than the optima for this taxon (Gajewski et al. 2005). Instead, the assemblages of IQ01 were primarily composed of *Stictochironomous rosenschoeldi*-type*,* several taxa of the Tribe Tanytarsini*, Psectrocladius sordidellus*-type*, Corynocera oliveri*-type*, Zalutschia,* and *Microspectra contracta*-type. *Stictochironomous rosenschoeldi*-type*,* the Tanytarsini and *Psectrocladius sordidellus*-type are all taxa that are commonly found in the littoral zone of lakes, with Tanytarsini and *Psectrocladius* also being associated with warmer lakes (Brooks et al., 2007). IQ01 also appears to have been more productive than IQ04, as IQ01 had higher abundances of *Corynocera oliveri*-type*,* which, aside from being a cold stenotherm, is also often found in lakes with higher dissolved organic content (Medeiros and Quinlan, 2011). Other taxa in IQ01 also show that this lake may have had higher nutrient levels as *Stictochironomous rosenschoeldi*-type is often associated with mesotrophic lakes while *Heterotrissocladius* is more associated with oligotrophic lakes (Luoto, 2011). This apparent difference in nutrient levels is once again most likely related to lake morphometry as larger, deeper lakes like IQ04 tend to be colder which can limit productivity and their large size can dilute the input of nutrients and organic matter from the surrounding landscape (Gajewski et al., 2005).

Like IQ04, the chironomid assemblages in IQ01 are mostly consistent with those seen in other studies around Baffin Island (Francis et al., 2006; Thomas et al., 2008; Axford et al., 2009), even though the majority of these Baffin Lakes are distant from any kind of urban settlement. The main differences between the assemblages of previous studies (Francis et al., 2006; Axford et al., 2009) and those of IQ01 are the lack of *Oliveridia* and *Pseudodiamesa*, and that IQ01 has a high abundance of *Stictochironomous rosenschoeldi*-type while most other studies (Francis et al., 2006; Thomas et al., 2008; Axford et al., 2009) have little to none of this taxon. The reason for this appears to be that the distribution of *Stictochironomous rosenschoeldi*-type on Baffin Island is restricted to the southern portion of the Island near Iqaluit (Sæther, 1975). Chironomid studies on Baffin Island, such as Briner et al. (2006) and Thomas et al. (2008), had little to no *Stictochironomous rosenschoeldi*-type in their chironomid assemblage data, but these studies also occurred far north of this taxon's documented range. However, Walker et al. (1997) did find *Stictochironomous rosenschoeldi*-type present on Baffin Island, with assemblages near Iqaluit. *Stictochironomous rosenschoeldi*-type is an indicator of cold-water littoral conditions (Brooks et al., 2007, Samartin et al., 2012) and likely represents a shift in lake morphology that occurred over time.

3.6.3 Influences of urbanization

In cases where local human activities have become a primary driver of environmental change in lakes, increased anthropogenic input would normally increase the abundance of certain pollution tolerant taxa, such as *Chironomous* (Al-Shami et al., 2010), *Cryptochironomous* (Canfield et al., 1994), as well as some taxa from sub-family Orthocladiinae including *Cricotopus bicinctus*-type and *Hydrobaenus* (Medeiros et al., 2011), while species richness would generally decrease (Medeiros et al., 2011; Zubova et al., 2020).

In IQ04 and IQ01 there is no evidence that this kind of shift has happened by the time the cores were collected. Taxa associated with pollution were low in abundance and never substantially increased. There were also no chironomid assemblage shifts that occurred after the development of Iqaluit that cannot be explained as resulting from previously inferred regional temperature shifts. The assemblages of IQ04 and IQ01 are also similar to lakes in nonurban regions around Baffin Island, which indicates that the chironomids of these lakes are not showing signs of being affected by urbanization. However, this is not to say that these lakes and other freshwater systems around Iqaluit are not being affected by their urban setting. Medeiros et al. (2011) found that the urbanization of Iqaluit has had a noticeable effect on the chironomid assemblages of nearby freshwater streams, and pollutants have been found in the tissues of fish in lakes near Iqaluit (Dick et al., 2010), bags of asbestos have been found in the lake and catchment of IQ01 (personal observation), and in 2021 some freshwater sources around Iqaluit became so contaminated they were not longer safe to drink from (CBC News, 2021). The results here are not indicative of these lakes being unaffected by local human disturbance, they instead indicate that despite their proximity to Iqaluit, these lakes are not yet experiencing substantive enough human disturbance to elicit an ecological response from chironomids that is greater than that of ecological response to a warming climate.

3.7 Conclusion

Climate is the primary driver of environmental change in two urban lakes examined in Iqaluit, Nunavut, despite the increased local human presence. Local disturbances were either not sufficient to have a noticeable effect on chironomid assemblages of two local lakes, or the influence of climate overshadowed these effects. Shifts in cold-stenothermic taxa were similar to shifts recorded in paleolimnological studies from other lakes on Baffin Island and other Arctic regions and assumed differences in taxonomic richness due to increased nutrient enrichment from urbanization were not observed. While other freshwater systems in the urban environment were observed to experience impairment, the lakes we examined had less direct point-source influences; proximity to point-sources of pollution may be a factor for having an impact on the ecology of urban lakes. Climate signals were observed in both systems that likely correspond to known climate shifts in the region, and the trajectory of both lakes is expected to follow climate in a warming future. Climate change and local influences are expected to only get more extreme over time and both are known to reduce the biodiversity of freshwater systems which could in turn affect the overall health of these systems. Continued monitoring and management of Arctic freshwater systems in the face of these potential dangers will be important as ecosystem health could be critical for the health of Arctic communities.

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CHAPTER 4: UNDERSTANDING THE INFLUENCE OF RECENT CLIMATE SHIFTS ON FRESHWATER LAKES IN THE HIGH ARCTIC; A CASE FROM CLYDE RIVER, NUNAVUT

4.1 Statement of Student Contribution

Study design by Connor Nishikawa and Andrew Medeiros, limnological data collection by Connor Nishikawa, Andrew Medeiros, Roberto Quinlan and Milissa Elliot, chapter writing by Connor Nishikawa with editorial contributions by Andrew Medeiros and Jennifer Eamer.

4.2 Abstract

Arctic lakes are sensitive to environmental change, which is especially true in the High Arctic where ice is the predominant factor that governs aquatic trophic systems. The direct relationship between temperature and aquatic biota makes high Arctic lakes especially useful for reconstructing long-term climate shifts in paleolimnological studies. Here, we present data associated with the paleolimnological history of a high Arctic lake near Clyde River, Nunavut, Canada. Subfossil chironomids (Diptera: Chironomidae) were used to infer recent climate shifts in the northern Baffin region. The chironomid assemblage was primarily represented by cold stenothermic taxa, such as *Heterotrissocladius* and *Abiskomyia*. Shifts in these assemblages at~1950 CE indicate a gradual warming signal likely caused by anthropogenic induced climate change. A long hiatus from 1320-1780 CE was observed where chironomid abundances were extremely low (0.5 HC/g DW). This period could represent the onset of the Little Ice Age. Prior to 1320 CE the assemblage was represented by higher abundances of taxa with higher thermal optima such as Tanytarsini and *Psectrocladius*.

This period may have been during the ending of the Medieval Warm Period when Arctic temperatures were relatively high which also led to higher levels of taxonomic richness and abundance.

4.3 Introduction

Northern freshwater systems are known to be sensitive to environmental change, due in part to temperature acting as a principal mediator for chemical and biological processes. This sensitivity is compounded by the fact that the Arctic has been warming to a far greater degree than most other regions on the planet (Hansen et al., 2010). Determining the degree to which these lakes have been impacted by this warming is difficult because most Arctic regions lack any kind of long-term environmental monitoring data (Luoto et al., 2019). Paleolimnology, however, can be used to make inferences on how Arctic lakes have responded to warming and reconstruct historic climate (Axford et al., 2009). Paleolimnology is a science whereby sediment cores are extracted from lakes and chemical, physical, and biological indicators downcore are used as proxies of the past (Smol, 1992). The analysis of these indicators often includes models derived from relationships generated from modern conditions, where the primary gradient is modelled in the context of a calibration-set and transfer function (Birks et al. 2003). Subfossil chironomids (Diptera: Chironomidae) have been repeatedly shown to be excellent bioindicators for reconstructions of temperature and have been used in several studies to reconstruct regional temperatures for Arctic regions (Thomas et al., 2008; Axford et al., 2009; Medeiros et al., 2012).

In the high Arctic lakes have been shown to lack any substantial catchment inputs or the local human influences that have been known to confound the results of paleolimnological studies in more temperate regions (Smol, 2016) and are therefore useful sentinels for inferring climate induced change. Many of the chironomid based paleolimnological studies that have been performed in the high Arctic have been able to reconstruct long periods of time (Francis et al. 2006; Thomas et al. 2008; Axford et al. 2009). As a result, these reconstructions can sometimes lack the fine scale temporal resolution needed to determine the precise timing and effects of more recent climate events such as the little ice age and modern anthropogenic warming. Here, we analyze the lacustrine sediment history of a high Arctic lake on Baffin Island, Nunavut, Arctic Canada, to infer the influences of environmental change in the high Arctic over several hundred years. As a result of their sensitivity to temperature, Arctic lakes are among the most at risk to be influenced by anthropogenic climate change. Understanding how these lakes have responded to the influence of previous recent shifts will be important in understanding how may respond as global temperatures continue to rise.

4.4 Methods

4.4.1 Site description

Patricia Bay Lake (70.46°/-68.50°) is located along the northeastern coast of Baffin Island, Nunavut near the Inuit hamlet of Clyde River. Baffin Island is the largest island in the Canadian Arctic and one of the largest islands in the world, covering over 1600 km in length and multiple climate zones (Miller et al. 2005). Average July temperatures in the area range from 8.8-1.2 °C with an annual total precipitation around 258mm (Station 2400800, Environment Canada).

This region is above treeline and in a zone of continuous permafrost with a geology consisting primarily of Precambrian bedrock (Miller et al. 2005). There is little human activity in the area and while the community of Clyde River is relatively close, there are no indications of any local human activities around the catchment area.

4.4.2 Field sampling

A sediment core was collected from the mid-basin of Patricia Bay Lake in Clyde River, Nunavut, August of 2007 with the use of a Uwitec gravity corer (8.4 cm diameter) deployed from an inflatable boat. The core was extruded at 0.5 cm intervals for the entire length of the core (19cm). Each subsample was placed in labelled Whirl-Pak® bags, transported in a cooler, and stored at 4 °C at Dalhousie University in Nova Scotia Canada for processing and analysis. At the same time as core collection, depth and environmental parameters (Temperature, Oxygen Reduction Potential (mV), Conductivity (µS cm-1), and pH) were measured with the use of a YSI multi-parameter probe (Table S.1). Water samples for analysis of chlorophyll-a, major ions, nutrients, and dissolved and particulate organic carbon were also collected and transported to the Canadian Centre for Inland Waters for analysis following Environment Canada (1994) and are reported in Medeiros et al., (2012).

Table 4.1 Environmental characteristics of Patricia Bay Lake, Nunavut Canada in August of 2007.

Parameter	Units	Patricia Bay Lake
Lat	DD°	70.46
Long	DD°	-68.50
Area	Ha	71.5
Elev	m a.s.l.	20
Depth	M	4.5
Temp	°C	8.6
Ph		8.1
COND	µS cm^{-1}	14

4.4.3 Laboratory analysis

A dating chronology was established for each core determined using a Constant Rate of Supply (CRS) model (Appleby 2001) applied to measured 210Pb isotopes by alpha-spectrometry at the Center for Canadian Inland Waters, Burlington, Ontario. A logarithmic decline in excess 210Pb determined sedimentation rates. Loss-on-ignition was conducted on both cores following Heiri et al. (2011), where each sample was burned at 550 and 950 °C for two hours to calculate weight-loss derived %Organic and %Carbonate from sediments.

The sediment core from Patricia Bay Lake was analyzed for subfossil chironomids at Dalhousie University following standard methods (Walker, 2001) and at 0.5cm intervals for the entire length of the core. Each interval was subsampled in order to extract the chironomid head capsules from the sediment. Subsamples were treated with potassium hydroxide (KOH) and heated at 75 °C for 30 minutes, with stirring after 15 minutes. This was done to disconnect or deflocculate the sediment so that the head capsules could be more easily extracted. The treated sediment was then poured through nested sieves of 212 and 106 µm and rinsed with 95 % ethanol. The sediment was then sifted through for head capsules using a dissecting microscope and fine tipped forceps. Each sample was sifted three times to maximize the capture rate of all the available head capsules within a sample. Each interval from the three cores required a minimum of 50 head capsules to be considered statistically significant (Quinlan and Smol, 2001).

This minimum head capsule requirement was not meet between the intervals of 5.5 and 17.5 cm in the Patricia Bay core due to very low abundances. For the rest of the intervals, if less than 50 head capsules were found then more sediment was subsampled until the minimum head capsule count was surpassed. The head capsules from each subsample were then mounted on glass slides using Entellen® then identified using a compound microscope to the most specific taxonomic resolution possible following Brooks et al. (2007) and Medeiros and Quinlan (2011).

4.4.4 Statistical Analysis

Data analysis was completed using R statistical software v.4.1.2 (R Core Team 2021). For each core, taxa were represented by their relative abundances, calculated as the percentage of identifiable chironomids in each sample for each interval and plotted stratigraphically over depth and chronological time. Zonation was established using Constrained Cluster Analysis (CONISS) with the number of significant zones determined using a broken-stick model (Bennett 1996).

The taxonomic richness of each interval was determined using rarefaction analysis, where richness is E(Tn) or the number of each taxon present in each sample of n=50 (Birks and Line, 1992). A principal components analysis (PCA) was used to reduce the dimensionality of the assemblage dataset for each core. To aid in interpretation of dominant trends, rare taxa, i.e., those whose relative abundance was less than 1 % in 2 lakes of the total identifiable chironomids in each interval, were removed from the stratigraphic diagrams.

4.5 Results

4.5.1 Core Chronology

The ^{210}Pb record for the core of Patricia Bay Lake had an initial value of 26.7 Bq kg^{-1} at 0.5 cm and 15.0 Bq kg^{-1} at 3 cm where ^{210}Pb background was estimated as 1900 CE using a constant rate of supply model (Figure S.1). Interval ages below 4.5 cm depth were estimated through linear extrapolation, such that the basal interval of 19.0 cm is estimated to 1270 CE. Extrapolated dates prior to 1900 CE should be interpreted with caution.

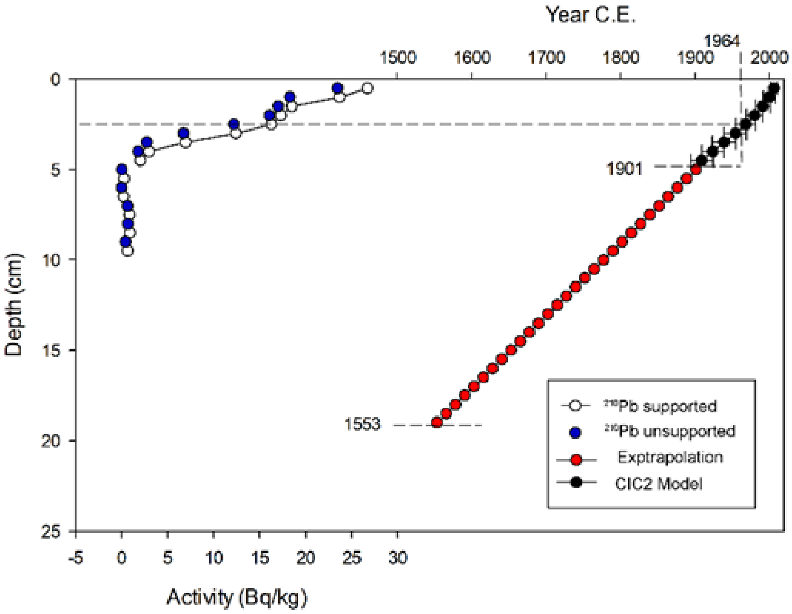

Figure 4.1 Sediment core chronology based on the CIC1 ^{210}Pb model for Patricia Bay Lake, Nunavut. Linear interpolation of chronology is indicated by red dots. Error bars represent the standard deviation of calculated sediment core interval age based on a CIC1 model chronology.

4.5.2 Chironomid stratigraphy

A total of 848 chironomid head capsules were extracted from the sediment core of Patricia Bay Lake. These head capsules were comprised of 46 different taxa that were found in 24 sampled depth intervals. Based on the compositional similarity of samples within statistically significant zones from the broken stick model of the cluster analysis, there were 3 zones identified for the Patricia Bay Lake data (Figure S.2).

Zone 1 (19.0-18.0 cm; 1320-1270 CE) was comprised of high numbers of *Abiskomyia*, *Heterotrissocladius grimshawi*-type, *Heterotrissocladius maeaeri*-type 1 and *Microspectra insignilobus*-type with maximum abundances of 25, 20, 11.5, and 14.5% in this zone, respectively. From 19.0-18.5 cm there were no notable assemblage changes in this zone.

From 18-18.5 cm the abundances of *Abiskomyia* and *Heterotrissocladius maeaeri*-type 1 decreased dramatically from 20 to 3 % and 11 to 5%, respectively. At the same time, the abundance of *Microspectra insignilobus*-type increased from 9 to 15%. Average taxonomic richness in the first two intervals of this zone (19-18.5 cm) was ~22.

Zone 2 (18-3.5 cm) included 2 distinct subzones, the first of these subzones (18-7.5 cm; prior to 1850 CE) was characterized as a period of extremely low abundances in the core. Regardless of additional subsampling, we were unable to reach 50 HC for several intervals. Most of the chironomid taxa that were present in the previous zone disappeared completely in this subzone. The most abundant taxa in this zone were *Abiskomyia*, *Heterotrissocladius grimshawi*-type and

Microspectra insignilobus-type. The second portion of this zone (7.5-3.5 cm; 1850-1950 CE) was almost entirely comprised of Tanytarsini. From 5 - 3.5 cm (1901-1950 CE) the taxonomic richness of this core increased to 13. The most dominant taxa were *Microspectra contracta*-type, *Heterotrissocladius grimshawi*-type, and *Heterotrissocladius maeaeri*-type 1 with maximum abundances of 19, 23, and 30%, respectively. *Heterotrissocladius grimshawi*-type and *Heterotrissocladius maeaeri*-type 1 increased from 5-3.5 cm by 8-15 % and 14 – 30 %, respectively. In contrast, *Microspectra contracta*-type decreased from 19 to 4%.

Zone 3 (3.5-0.5 cm; 1950-2006) was mostly dominated by *Heterotrissocladius* taxa including *Heterotrissocladius maeaeri*-type 1, *Heterotrissocladius maeaeri*-type 2, *Heterotrissocladius grimshawi*-type, and *Heterotrissocladius marcidus*-type. In this zone *Heterotrissocladius maeaeri*-type 1 decreased in abundance from 30 to 14 %, while *Heterotrissocladius maeaeri*-type 2 and *Heterotrissocladius marcidus*-type both increased in abundance from 2 to 16% and 3 to 9%, respectively. Average species richness in this zone was 17.

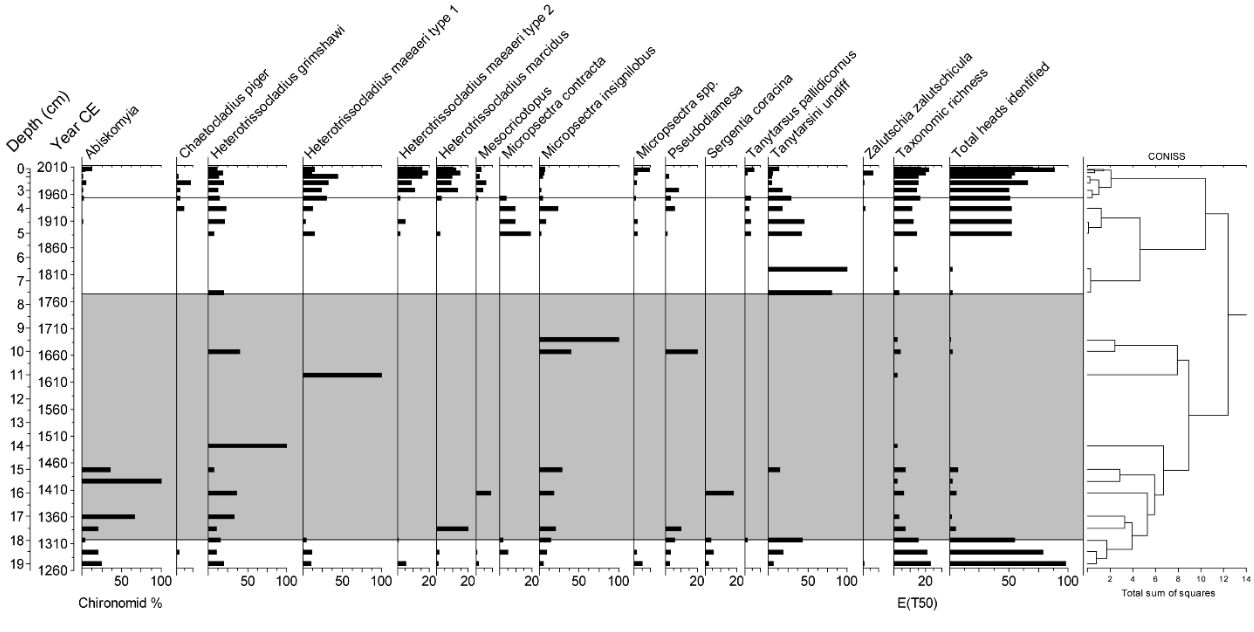

Figure 4.2: Stratigraphy of chironomid taxa for the core of Patricia Bay Lake, with only taxa present at a minimum of 1 % in each interval shown. Species richness calculated through rarefaction analysis using all taxa from the respective depth interval. Horizontal dashed lines denote the time zones within the respective cores based on a constrained cluster analysis with significant zones identified through a broken stick model. The grey region represents a period of very low abundance of chironomid head capsules when this lake may have been covered by a glacier

4.6 Discussion

The objective of this research was to create a reconstruction of the environmental history of a high Arctic lake to determine how climate shifts over the last several hundred years have affected Arctic freshwater systems in the region. Lakes in the high Arctic are mostly removed from the effects of local human disturbances or catchment related inputs that have the potential to confound paleolimnological reconstructions and therefore give a more accurate representation as to how Arctic lakes respond to changes in regional climate (Michelutti et al., 2007; Smol, 2016).

4.6.1 Inferences of historical environmental changes

During the early part of the Patricia Bay Lake record (below 18.0 cm; 1320 CE), the taxonomic richness was high relative to the rest of the core. This zone was dominated by cold-tolerant oligotrophic taxa, such as *Abiskomyia*, *Heterotrissocladius* and *Microspectra insignolobus*-type, as well as the generalist Tanytarsini (Brooks et al., 2007). After this period taxonomic richness and overall chironomid abundance started to decrease. By 17.5 cm (1340 CE) taxonomic richness and abundance dramatically decreased to the point that in many intervals chironomid abundance was too low to achieve statistical significance (Quinlan and Smol, 2001). This period of extremely low abundance may have been the result of glacial advancement over the catchment area of Patricia Bay Lake caused by the onset of the little ice age (~1300 to 1850 AD). Glaciation has been inferred in other paleolimnological studies as a reason for why some high Arctic lakes may experience significant hiatuses of bioindicators during relative cold periods of glacial advancement (Miller et al., 2002; Francis et al., 2006). This period of low taxonomic richness and abundance lasted until 5.0 cm; (1890 CE) when these values started to slowly increase. This increase in abundance may have been a result of reduced ice cover, which is known to be a major driver of ecological change in the Arctic (Quinlan et al., 2005). From 1950-2007 CE taxonomic richness and abundance increased to levels comparable to the most diverse periods of the first zone of this core (19.0-18.0 cm; 1320-1270 CE). This period was also when multiple taxa with higher temperature optima such as Tanytarsini and *Psectrocladius sordidellus*-type started to increase in abundance while some cold water stenotherms such as *Microspectra* began to slowly decrease.

This increase in temperature was likely a result of anthropogenic warming as the 1950s have been inferred by paleolimnological studies across the Canadian Arctic (Thomas et al., 2008; Kaufman et al., 2009; Medeiros et al., 2012) as a time when much of the eastern Canadian Arctic experienced a substantial shift in climate caused by anthropogenic warming. As a result of this warming taxonomic richness is expected to increase in lakes like Patricia Bay Lake as increasing temperatures will allow warm water chironomid taxa to disperse into regions and habitats that are currently too cold to support their development (Engels et al., 2020)

4.6.2 Context within the high Arctic

The chironomid assemblages of Patricia Bay Lake are very similar to those of other lakes on Baffin island. Lake CF8 (Thomas et al., 2008) which is located close to Patricia Bay Lake was also heavily dominated by *Heterotrissocladius* which is a taxon that is associated with cold oligotrophic lakes (Axford et al., 2009) and is known to dominate Arctic lakes (Gajewski et al., 2005). More southern Baffin Island lakes such as Fog Lake (Francis et al., 2006) and Lake IQ04 (Nishikawa et al., 2022) also had similar assemblages to Patricia Bay Lake with high abundances of *Heterotrissocladius* although these lakes had higher abundances of Tanytarsini than Patricia Bay Lake, but this may be due to those lakes being in a more southern warmer location (Medeiros and Quinlan, 2011). While CF8 did not have a drop in abundance during the little ice age, glacial advancement and recession were known to be somewhat variable in the region (Miller et al., 2005).

4.7 Conclusion

The results of this paleolimnological assessment of Patricia Bay Lake indicates a warming of the region since 1950 CE. Lakes in the high Arctic have little to no terrestrial catchment inputs and so many species are primarily restricted by temperatures. As regional temperatures continue to increase as by product of anthropogenic climate change, the taxonomic diversity of High Arctic lakes such as Patricia Bay Lake could begin to increase as the environment becomes more hospitable to more temperate taxa. However, if temperatures rise past a certain threshold many cold-water taxa may begin to disappear as temperatures pass their thermal tolerance, leading to local extinctions. Therefore, lakes in the High Arctic should continued to be monitored due to their increased environment sensitivity and the higher degree of warming that the Arctic will continue to receive.

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CHAPTER 5: CONCLUSION

5.1 Statement of Student Contribution

Chapter design and chapter writing by Connor Nishikawa with editorial contributions by Andrew Medeiros.

5.2 Introduction

As global warming becomes more severe the influence that a changing climate has on the Arctic is only going to become more significant. Multiple studies have shown just how sensitive environments in the Arctic are to climate related changes with some stating that the changes imposed by anthropogenic warming are now irreversible (Luoto et al., 2019). As the influence of climate change becomes more significant it is important to monitor how this increased influence affects of the biodiversity of Arctic environments including the various freshwater systems as biodiversity is often a proxy for ecosystem health (Mayfield et al., 2020). The results of this thesis show that climate change has had noticeable effects on freshwater systems across the Arctic no matter their location or prevailing environmental conditions. However, the ecosystem and by extension biodiversity responses to climate change were variable between lakes in different Arctic regions. Understanding the reasons for these variations and what effect prevailing environmental conditions have on freshwater responses to warming will be important for determining how these systems will change as the Arctic continues to warm.

5.3 Responses to warming around the Arctic

The lakes in the 3 Arctic regions explored in this thesis all demonstrated a susceptibility to climate shifts during the Anthropocene with chironomid assemblage shifts that indicate a variety of ecosystem responses. In the High Arctic there are very few local people and little terrestrial influence on freshwater lakes therefore, climate is virtually the only mediator of ecological change (Michelutti et al., 2007). The analysis from Patricia Bay Lake shows that regional climate shifts were shown to have drastic impacts on biodiversity. When the region was especially cold, species richness was extremely low as the harsh conditions limited the proliferation of a variety of taxa. As temperatures increased at the start of the 20th century biodiversity also increased as warm water taxa were able to disperse into regions and habitats that were initially too cold to support their development (Engels et al., 2020). Ecological communities of the Arctic tundra and the boreal forest overlap creating a dynamic environment that is very susceptible to change (Zhu et al, 2011; Smith and Goetz, 2021). The analysis from the shallow ponds of Hudson Bay Lowlands showed a mix of responses to warming. Heightened warming in 20th century increased the amount of evaporative pressure Left Lake experienced which has led to desiccation of the pond to the point that it may have dried up completely. Larch Lake on the other hand experienced an increase in water level due to permafrost melt creating a channel fen by which this pond receives water. Despite this divergent hydrological response both ponds seem to be experiencing a similar proliferation of algae that reduced each pond's macrophyte community. In response to these ecosystem changes the overall taxonomic richness in Left Lake decreased throughout the 20th century as its lower water level likely limited the amount of available habitat.

In Larch Lake species richness was relatively consistent although this pond did experience some species turnover with a loss of macrophyte related taxa that were replaced by species that expanded with the increased temperatures and higher nutrient levels.

In southern Baffin Island a growing local population and outdated infrastructure have been shown to have had a noticeable effect on the regions freshwater systems (Medeiros et al., 2011) to the point this effect may eventually be greater than that of a warming climate. The analysis of the lakes near Iqaluit, Nunavut however indicate that this is not the case, yet. These lakes have been shown to primarily respond to regional shifts in climate with little indication of a local influence. This indicates that either local influences are not significant enough to affect chironomid assemblages or that the influence of a warming climate is so significant it masks any local impacts. Taxonomic richness in IQ01 was relatively consistent though there was a loss of some cold stenothermic taxa that were replaced with warm water species. IQ04 also had relatively consistent richness levels except this lake did not experience as much taxonomic turnover. This is likely because IQ04 was a deeper lake with a larger profundal zone which is more insulted from the effects of temperature shifts than a lake like IQ01 which has a larger littoral zone (Olander et al., 1997).

4.4 The intermediate disturbance hypothesis revisited

The results of this thesis show just how varied the responses of Arctic freshwater systems to climate change have been. The secondary goal of this thesis was to test if the intermediate disturbance hypothesis could be applied to these systems in order to model how their taxonomic richness will change as global temperatures continue to increase. It was expected that as temperatures increased the species richness of a lake in the High Arctic would also increase as the higher temperatures would promote greater opportunity for warmer species to proliferate.

In the subarctic it was expected that biodiversity would decrease as warming temperatures would reduce habitat availability and the abundance of cold water and specialist taxa. This result was also expected of lakes that are affected by local human influences as climate change would put further stress on taxa that already have to contend with various local impacts. In terms of the intermediate disturbance curve (Figure 1.1) it was expected that each set of lakes would shift to the right of the curve.

The historical shifts in taxonomic richness that occurred in each of the study lakes in this thesis partially aligned with our expectations based on the intermediate disturbance hypothesis (Figure 5.1). The richness of Patricia Bay Lake shifted along the curve as expected as temperatures increased throughout the 20th century. Left Lake also shifted as expected with richness decreasing as higher temperatures led to less available habitat. Larch Lake did not have a pronounced shift in richness although the species turnover and continued evaporative stress in Hudson Bay Lowlands are still experiencing may indicate that a reduction in richness could be coming in the future. IQ01 and IQ04 also did not experience a reduction in richness although the results of this thesis indicate that these lakes are not experiencing enough of an influence from the local population to be consider a human disturbed lake. This was a potential concern alluded to by Fox (2013) when testing the intermediate disturbance hypothesis that the most disturbed lakes might not be disturbed enough to reach the peak of the intermediate disturbance let alone the point when richness might start to drop off. As a result, IQ01 and IQ04 are more comparable to an Arctic lake with little to no human disturbance such as Patricia Bay Lake only slightly more diverse, likely due to higher regional temperatures.

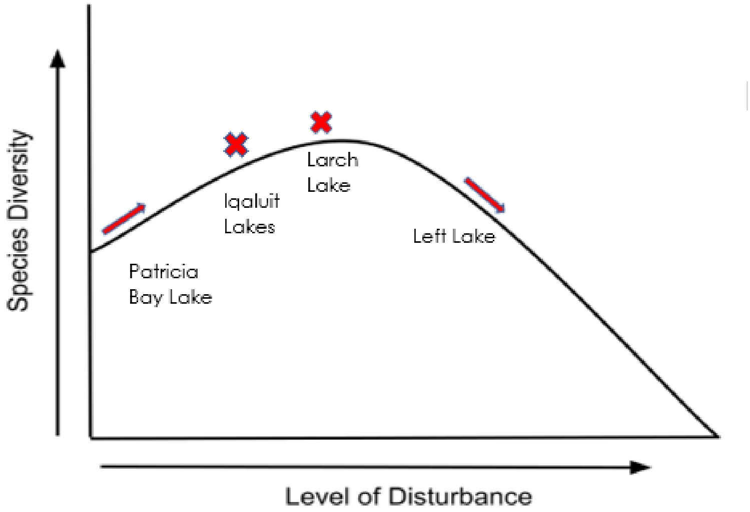

Figure 5.1: Intermediate disturbance hypothesis curve with the all the observed study lakes and their actual position on the curve. Red arrows indicate the directional shift of a lake along the curve under continued stress from climate change while red Xs indicate a lack of movement along the curve

5.5 Final thoughts and future directions

The results of this thesis show the effect that climate change has had on the freshwater systems of the Arctic. As warming becomes more intensive, it is important that these sensitive systems are continuously monitored. All of the lakes analyzed in this thesis were collected in 2010 CE or earlier and have likely changed to varying degrees as warming has become more intensive in the Arcticover the past 12 years. In the Clyde river region, where Patricia Bay Lake is located, average July temperatures have increased by 1.2°C since this sediment core was collected (Station 2400800, Environment Canada). For a region that is so mediated by climate, that much of an increase could have a significant impact on the freshwater systems such as Patricia Bay Lake.

It is possible that temperatures have crossed a threshold by which many cold stenothermic chironomids are now starting to go locally extinct, and richness has actually decreased with a warming climate (Engels et al., 2019). In the western Hudson Bay Lowlands, where Larch Lake and Left Lake are located, average July temperatures have increased by 0.6°C since the cores were collected (Station 5060600, Environment Canada). As temperatures have warmed, the concentration of algae may have increased to the point that shallow ponds like Larch Lake and Left Lake have transitioned entirely to algal dominated systems with little to no macrophyte abundance or diversity, which could limit the variety of available habitat and decrease overall species richness (Qin et al., 2013; Mayfield et al., 2020). It is also entirely possible that Left Lake is no longer even a pond as continued evaporative stress may have caused the pond to desiccate completely (Bouchard et al., 2013). In Iqaluit, temperatures have increased by 1.3°C (Station 2402590, Environment Canada) since core collection, this increased temperature could have a similar effect on IQ01 and IQ04 like it may have on Patricia Bay Lake in terms of a loss of cold-water chironomid taxa. On top of the temperature increase, the population in Iqaluit has increased by over 1,000 people or 20% since the cores were collected (Statistics Canada, 2022). With that increase in population likely comes an increase in the local influence on the nearby freshwater systems. A good example of this came in 2021 where fuel contamination made the much of the city's water supply undrinkable (CBC News 2021). Future research could be directed at comparing how all of the lakes analyzed in this thesis have changed since they were initially collected to give a more up to date interpretation of how they have been impacted by climate change as the 21st century has progressed.

Future research could also be directed towards revisiting the intermediate disturbance hypothesis and how it could be used to forecast how climate change will affect the biodiversity of lakes across the Arctic.

The results of this were not fully in line with expectations based on the intermediate disturbance hypothesis but they also do not indicate that the hypothesis is invalid as there are some explanations as to why the results deviated from expectation that future research could make note of. Patricia Bay Lake changed exactly as expected of a High Arctic lake with richness increasing as temperatures warmed. Future studies could likely use other High Arctic lakes to represent the first stage of the intermediate disturbance, although picking a lake that has less of a history of glaciation may be wise as it would limit any chances of a depositional hiatus. The results of the Left Lake analysis were also in line with expectations as the ecotonal Hudson Bay Lowlands served as an appropriate peak for the intermediate disturbance curve. The results of the Larch Lake analysis were not in line with expectation although this lake may be an outlier. It is likely that Larch Lake would have been similar to Left Lake if not for the channel fen that was created by permafrost melt. Whether an event like that is an outlier or something that will become more common as temperatures in the region warm is also a possible subject of future research. The results from the Iqaluit analysis were also not in line with expectations based on the intermediate disturbance hypothesis but this was likely because the local population was not having a significant enough impact. Future research could revisit these lakes now that the local population has increased or analyze lakes that are in even more populated Arctic settlements such as Whitehorse or Yellowknife.

The threat posed by climate change on the freshwater systems in the Arctic is of serious concern as water security has become a major issue in many Arctic communities (Medeiros et al., 2017;

Cincio et al., 2021). As the Arctic warms, many ponds and lakes are either desiccating (Abnizova and Young, 2009; Wolfe et al., 2011; Koch et al., 2014; Campbell et al., 2018), or draining into the ground through permafrost melt (Kirpotin et al., 2009; Lantz and Turner, 2015). As shown in the Churchill region, climate change has also led to the proliferation of algae in Arctic ponds which has the potential to make drinking water sources toxic to humans and wildlife (Brooks et al., 2015; Carmicheal and Boyer, 2016; Kashulin et al., 2021). The issue of water security is further compounded by the fact that the population in the Arctic is expected to continue to increase in the future (Heleniak, 2020) and that the existing infrastructure is too outdated to be able to even properly support the current population let alone a larger one (Medeiros et al., 2017). As the planet continues to warm these issues are only going to become more severe and so greater attention and efforts from more temperate locations are likely required if the millions of people who inhabit the circumpolar Arctic are to maintain their way of life.

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APPENDIX: SUPPLEMENTAL MATERIAL FOR UNDERSTANDING THE INFLUENCE OF RECENT CLIMATE SHIFTS ON THE FRESHWATER LAKES IN HUDSON BAY LOWLANDS

S.1 Statement of Student Contribution

Study design by Connor Nishikawa and Andrew Medeiros, limnological data collection by Connor Nishikawa, Erin Light, Roland Hall, Brent Wolfe, Merrin Macrae, LeeAnn Fishback and Johan Wiklund, chapter writing by Connor Nishikawa with editorial contributions by Andrew Medeiros.

S.2 Extended Stratigraphies for Left Lake and Larch Lake

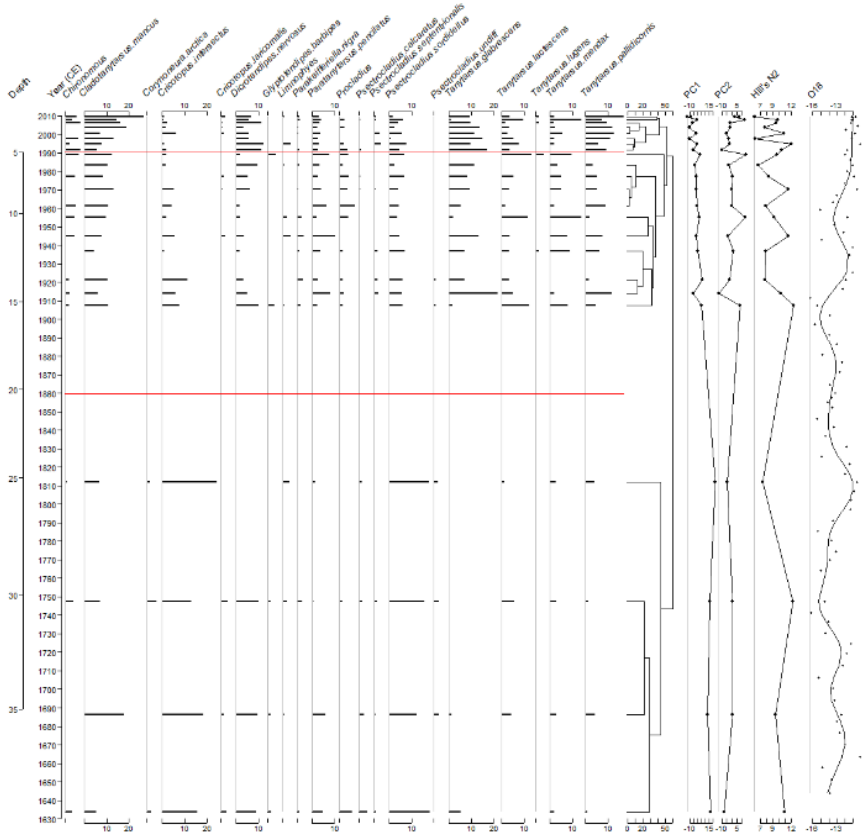

Figure S.1: Stratigraphy of chironomid taxa and PCA sample scores as well as a cellulose-inferred pond water δ^{18}O (‰ VSMOW) record for the core of Left Lake, with only taxa present at a minimum of 1 % in 2 lakes shown. Species diversity calculated through Hills N2 analysis using all taxa from the respective depth interval. Horizontal red lines denote the time zones within the respective cores based on a constrained cluster analysis with significant zones identified through a broken stick model. Cellulose-inferred pond water δ^{18}O record extracted from Wolfe et al. 2011.

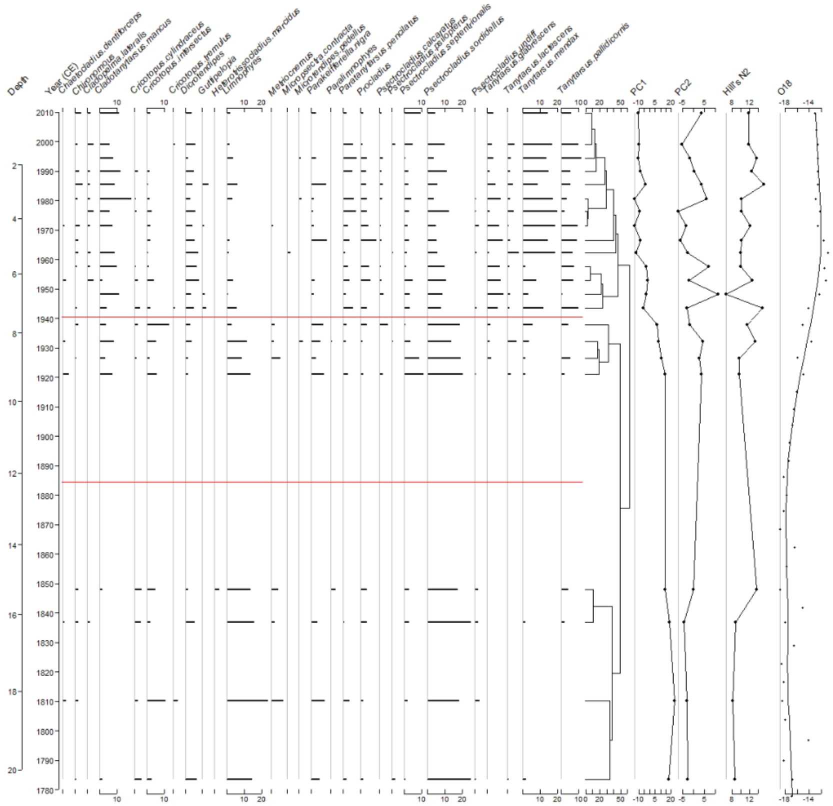

Figure S.2: Extended Stratigraphy of chironomid taxa and PCA sample scores as well as a cellulose-inferred pond water δ^{18}O (‰ VSMOW) record for the core of Larch Lake, with only taxa present at a minimum of 1 % in 2 lakes shown. Species diversity calculated through Hills N2 analysis using all taxa from the respective depth interval. Horizontal red lines denote the time zones within the respective cores based on a constrained cluster analysis with significant zones identified through a broken stick model. Cellulose-inferred pond water δ^{18}O record extracted from Wolfe et al. 2011.

S.3 Broken Stick figures for Left Lake and Larch Lake

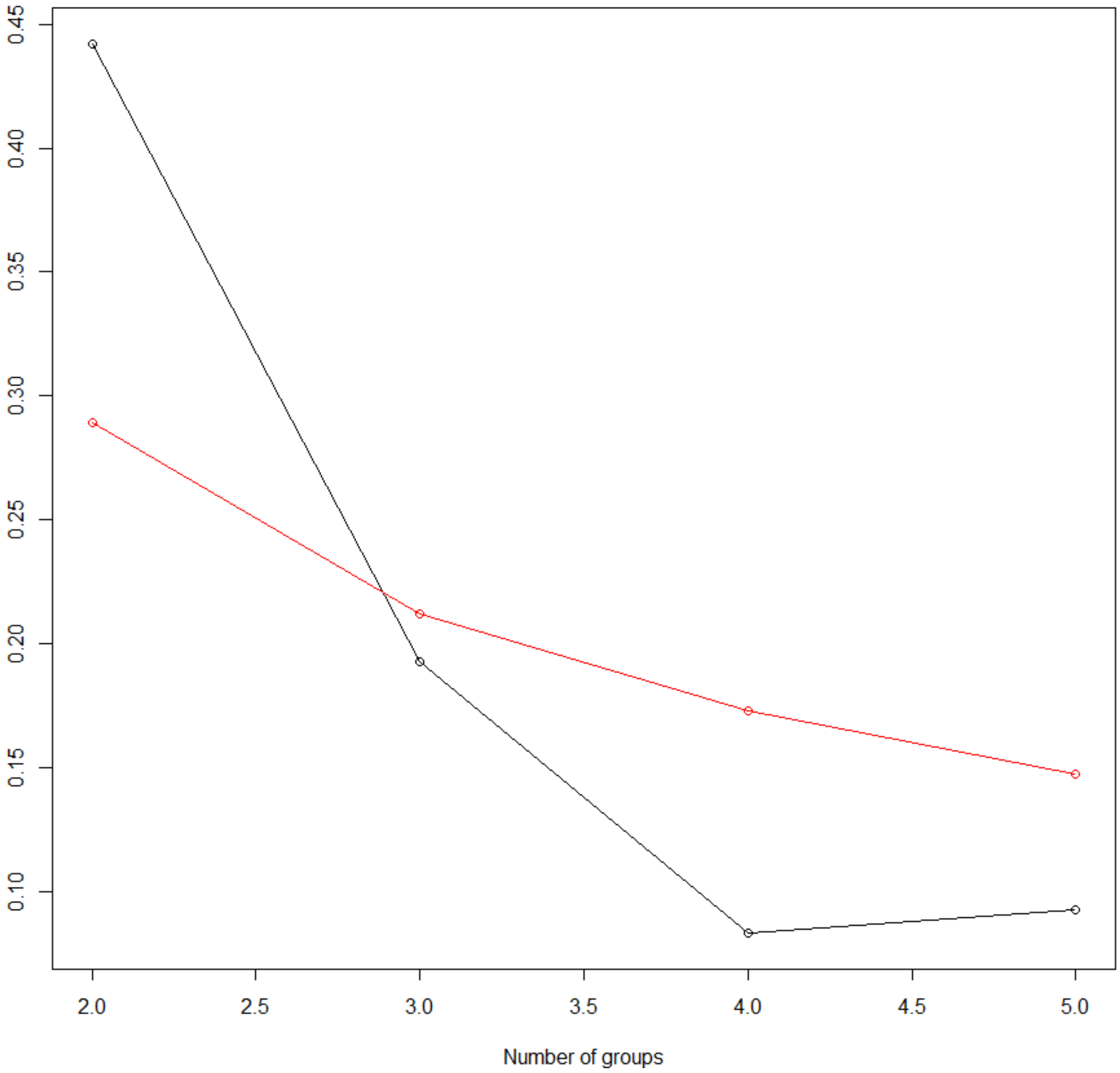

Figure S.3: Broken stick figure used in CONISS analysis of Left Lake to determine the number of significant zones based on chironomid assemblage data

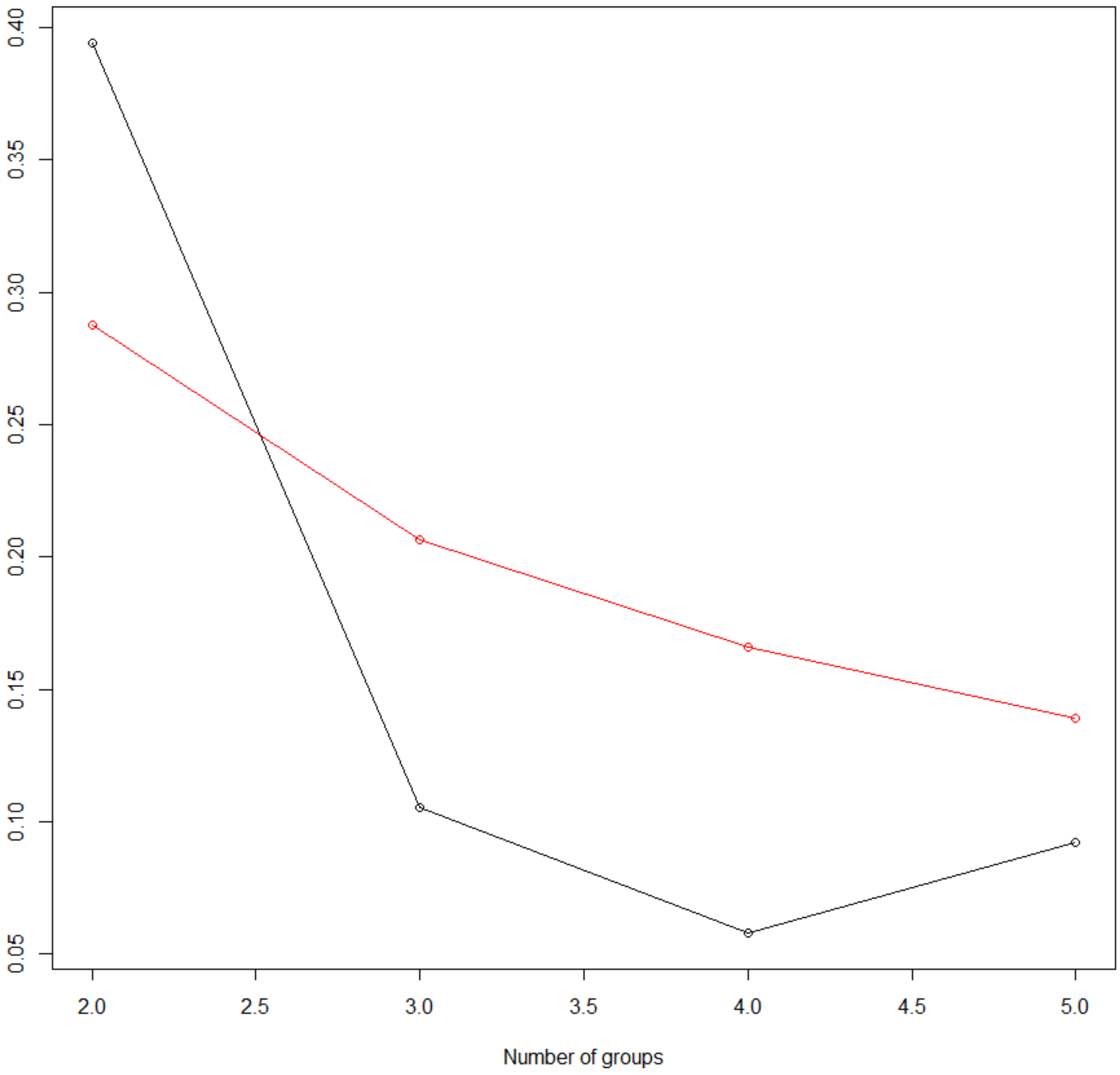

Figure S.4: Broken stick figure used in CONISS analysis of Larch Lake to determine the number of significant zones based on chironomid assemblage data

S.4: Historical Climate Record for Churchill, Manitoba, Canada

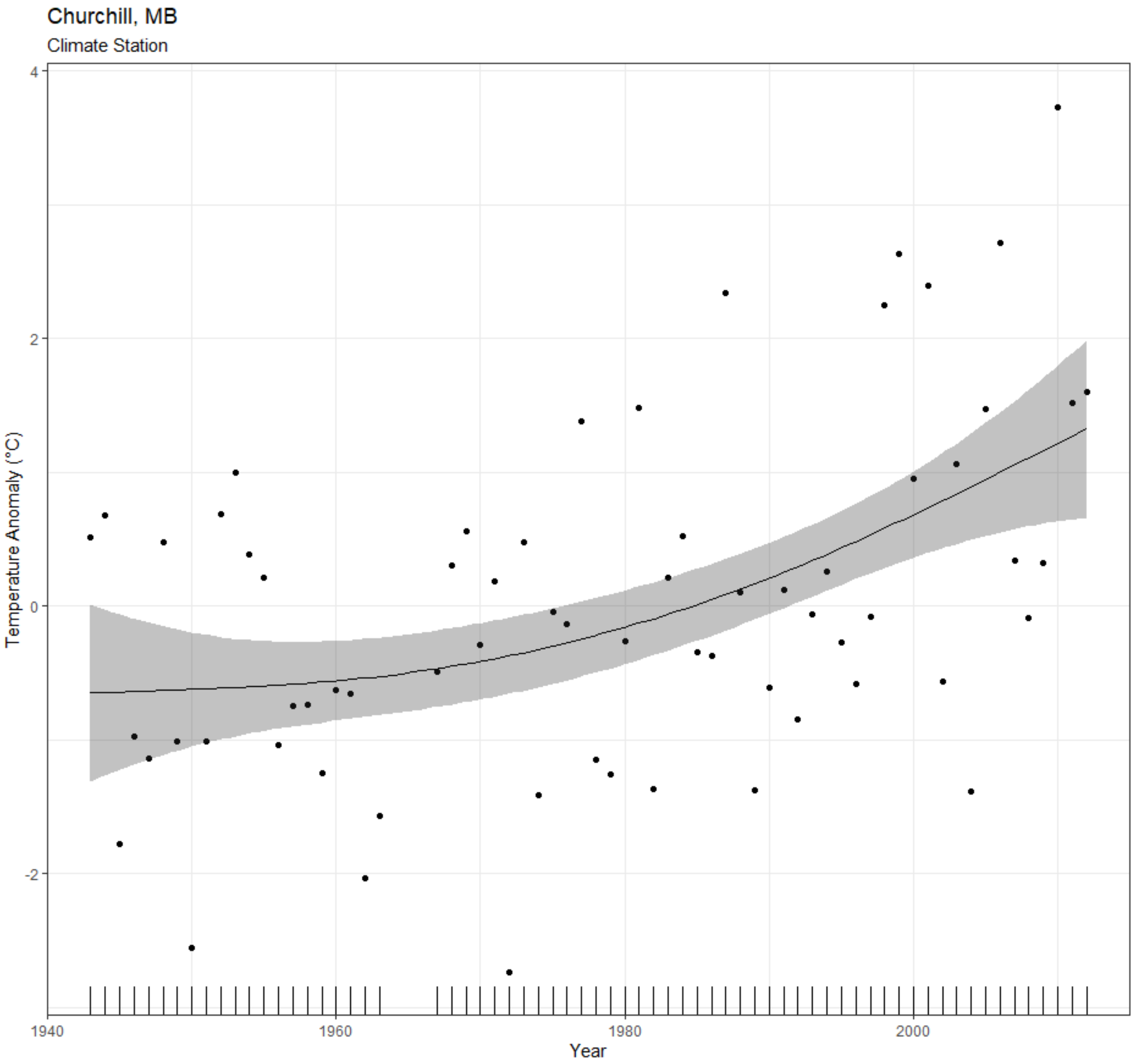

Figure S.5: Mean July temperatures from the Churchill weather station from 1943-2010. A smoothing line and confidence intervals represent a generalized additive model (GAM) of the time series of historical temperatures.