# PALEOLIMNOLOGICAL STUDY OF CHIRONOMIDAE (DIPTERA) ASSEMBLAGE RESPONSES TO MULTIPLE STRESSORS IN LAKE NIPIGON (NORTHWESTERN ONTARIO)

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## ABSTRACT

Lake Nipigon is a large, deep lake in northwestern Ontario that has experienced over a century of environmental disturbance linked to hydroelectric development, mining, and climate change. This thesis applied paleolimnological methods to reconstruct historical chironomid assemblages and their responses to multiple stressors in nearshore areas, as limnological data for Lake Nipigon is only sparsely available. Core surveys conducted show generalist and moderately tolerant taxa that suggest moderately warm, vegetated, and nutrient-rich conditions in the nearshore areas, with generally favourable oxygen conditions. Chironomid subfossil assemblage changes in Gull Bay were relatively muted, while recent increases in the warm-adapted taxon *Stempellina* in South Bay suggest a response to warming in the nearshore environment. South Bay also documented declines in cold, oxygen-sensitive *Heterotrissocladius grimshawi*-type. This study established the first paleolimnological baseline for Lake Nipigon, which is intended to provide a foundation for future Indigenous-led lake monitoring efforts that reflect both western scientific and Indigenous knowledge systems.

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## TABLE OF CONTENTS

## LIST OF TABLES

## LIST OF FIGURES

# CHAPTER ONE: INTRODUCTION

## 1.1 Background

Lake Nipigon is the largest inland lake entirely within Ontario's borders, the 13th largest lake in North America, and the 34th largest lake in the world (Herdendorf, 1990). It is also the headwaters to Lake Superior, and is sometimes referred to as the 6th Great Lake of North America (Annin, 2018). Despite being perceived as pristine waters surrounded by untapped wilderness, past mining and hydroelectric developments have significantly impacted Lake Nipigon and its watershed. The full extent of ecosystem impact is not well documented because of the near absence of western science data available for Lake Nipigon.

Biinjitiwaabik Zaaging Anishnaabek (BZA; also known as Rocky Bay First Nation) are one of several First Nation communities whose traditional territory encompasses Lake Nipigon. Historic stressors have affected their commercial fishing industry, which is one of their primary sources of livelihood. They are also routinely asked to consult on new development projects such as mines and a proposed new hydro development project on the Little Jackfish River. BZA has raised concerns about a lack of understanding of historic stressors and their cumulative impacts, leading to a new partnership between BZA, York University, and Lakehead University to analyze lake sediment cores to reconstruct the missing data (paleolimnology).

Paleolimnology is the science of using biological, chemical, and physical proxies preserved in sediments of inland freshwater systems to reconstruct historical environmental change. This thesis focuses on using Chironomidae (Diptera) subfossil remains isolated from lake sediment cores as paleoecological indicators of the cumulative impacts of climate change and past resource development on benthic habitat quality in Lake Nipigon nearshore areas. The findings produced from my research will contribute to the first (western science) insights into the ecological history of Lake Nipigon, and will help BZA to better communicate their concerns to external groups when consulted on management and land-use planning in the basin.

## 1.2 Research Objectives

The Lake Nipigon watershed has undoubtedly changed from its early limnological assessments since the 1920s, a decade or two before significant resource developments and mining activities have altered the ecosystem. My research aims to answer the question of how multiple stressors impacted the benthic habitat quality of Lake Nipigon nearshore areas, as inferred from chironomid assemblages. My research objectives are:

i. Assess spatial variation in chironomid assemblages in nearshore areas of Lake Nipigon.
ii. Determine if historical stressors have impacted the chironomid assemblages (indicators of benthic ecological health) of the lake.

# CHAPTER TWO: LITERATURE REVIEW

## 2.1 Climate change in freshwater ecosystems

Climate change poses significant challenges to freshwater systems worldwide, altering their physical, chemical, and biological characteristics. Rising global temperatures, shifting precipitation patterns, and changes in runoff dynamics directly impact freshwater availability, water quality, and ecosystem health (IPCC, 2014). These disruptions can have cascading effects on aquatic ecosystems, influencing species distribution, water temperatures, and nutrient cycling. For instance, changes in temperature regimes affect thermal stratification in lakes and rivers, altering oxygen levels and habitat suitability for aquatic organisms (Adrian et al., 2009). Between 1985 and 2009, global lake temperatures increased at an average rate of 0.34°C per decade (O'Reilly et al., 2015), with projections indicating an additional rise of 1–4°C by the end of the century (Woolway & Maberly, 2020; Grant et al., 2021). These temperature increases can exacerbate algal blooms, reduce oxygen levels, and disrupt food web dynamics, threatening freshwater biodiversity (Woolway & Maberly, 2020).

Shifts in precipitation patterns further compound these challenges, leading to more extreme weather events such as severe droughts and intense flooding. These changes disrupt freshwater ecosystems, alter sediment transport, and exacerbate water scarcity issues in many regions (Milly et al., 2008). In the Northern Hemisphere, lakes have been experiencing shorter winters and an accelerating loss of ice cover over the past several decades (Woolway et al., 2022). Over the past 165 years, ice duration has declined by an average of 31 days, with ice loss occurring at a rate six times faster in the last 25 years (Sharma et al., 2021a). Reduced ice cover can have significant ecological and hydrological consequences, including altered mixing regimes, increased evaporation rates, and changes in thermal habitat suitability for cold-water fish species (Woolway et al., 2022). Climate change is co-occurring with anthropogenic land-use pressures, and the compounding influence of multiple stressors, combined with unpredictable ecosystem responses, is expected to drive unprecedented transformations in freshwater ecosystems worldwide (Woolway et al., 2022).

These climate-driven impacts underscore the urgent need for adaptive management strategies to mitigate adverse effects and safeguard the ecological and societal services provided by freshwater ecosystems. This is apparent when examining the transboundary water quality management in the Laurentian Great Lakes system (Abdel-Fattah & Krantzberg, 2014), the largest unfrozen surface freshwater system in the world. The Laurentian Great Lakes hold an estimated 23 quadrillion litres of water and spans approximately 244,000 square kilometers (Canadian Geographic, 2024). The basin drains an area more than three times its own size, making it a critical freshwater resource for both Canada and the United States (Canadian Geographic, 2024; Kayastha et al., 2022). Climate models predict rising water levels in the Great Lakes due to climate change, with average annual water levels in Lake Superior, Michigan-Huron, and Erie projected to increase by +0.19, +0.44, and +0.28 metres, respectively, by 2040–2049 (Kayastha et al., 2022). These changes may lead to shoreline erosion, habitat alterations, and shifts in nutrient cycling, further stressing an already vulnerable system.

Although substantial research has been conducted on the implications of climate change within the Great Lakes basin, scientific advancements have not kept pace with the need for process-oriented research. A fundamental gap remains in the basic data required to accurately forecast environmental changes, develop mitigation strategies, and restore large lake ecosystems such as the Laurentian Great Lakes (Sterner et al., 2017). Addressing these knowledge gaps will be essential for developing effective policies and conservation strategies that ensure the long-term sustainability of freshwater resources in a rapidly changing climate.

## 2.2 Large lake ecosystems

Approximately 253 lakes contain nearly 90% of the world's inland surface water supply (Herdendorf, 1982). The "Great Lakes" (defined as having surface area greater than 500 $km^2$ (Herdendorf, 1982)) have a total surface area of 1,456,149 $km^2$ and contain an estimated 202,000 $km^3$ volume of water, accounting for 93% and 88%, respectively, of the total surface area and volume of water held in all lakes of the world (Herdendorf, 1982). The Northern Hemisphere contains over 87% (220 lakes) of all large lakes, with most of these occurring north of the 40th parallel latitude (64%, 162 lakes) (Herdendorf, 1990). North America alone has nearly half of all large lakes, but only one-third of the world's total surface area for large lakes (Herdendorf, 1990). Large lakes around the world play a crucial role in boosting economic activity driven by transportation, fishing, aquaculture, waterfowl habitat, recreation, and waste assimilation (Herdendorf, 1990).

Large lakes tend to have several unique characteristics. They tend to be deeper than small lakes, and their shoreline development is reduced (Tilzer, 1990). The ratio of the volume of water to its surface area is considerably higher than in smaller lakes, leading to a longer water retention time (Adrian et al., 2009). A larger surface area allows for increased wind fetch and a deeper mixing of near-surface water layers even during periods of stratification (Tilzer, 1990). Oscillations, or seiches, are a much more common phenomenon in large lakes than small lakes (Boyce et al., 1989). They may be visualized as the constructive reflection of a train of long, gravitational, free surface waves from the ends of standing waves (Boyce et al., 1989).

Oscillations are commonly driven by temporal changes in wind and pressure patterns from external and internal sources (Tilzer, 1990). Large lakes tend to have a large heat storage capacity and thereby lose smaller proportions of their total heat content during the winter, making them less likely to freeze over when compared to small lakes in the same climatic regime (Tilzer, 1990).

Biogeochemical cycles in large lakes are also quite unique. The internal cycling of nutrients plays a much greater role in ecosystem function, and large lake systems tend to be less susceptible to short-term perturbations like temperature and water fluctuations, wind, and biological perturbations than small lakes (Adrian et al., 2009; Tilzer, 1990). Once nutrients and toxins enter large lake ecosystems, they also stay considerably longer than in small lakes owing to longer water retention times and a more efficient internal regeneration (Tilzer, 1990). This causes significantly longer lasting effects in the ecosystem and slower recovery times (Carpenter et al., 1998). External and internal nutrient loading rates are much smaller in large lakes, and this would characterize large lakes in general to be more oligotrophic (Tilzer, 1990).

Large lake food webs are highly influenced by the hydrography and biogeochemical characteristics discussed above. Due to the smaller surface-to-volume ratio in large lakes, pelagic communities are usually of greater importance than benthic communities (Tilzer, 1990), although nearshore areas of large lakes have a disproportionate influence on the productivity of large lakes especially in deep, oligotrophic systems (Hampton et al., 2011; Rosenberger et al., 2008). The shallow, nearshore waters are the sites where most feeding and breeding habitats of organisms are found, particularly fish and amphibians (Rosenberger et al., 2008). At temperate latitudes, winter mixing in large, deep lakes tends to reach greater depths, leading to greater summer seasonal productivity in the pelagic communities (Tilzer, 1990. In contrast to the pelagic communities, the profundal communities exhibit less seasonality due to the lack of environmental variation at the bottom of deep lakes (Tilzer, 1990). For the littoral zone, internal seiches can cause highly variable environmental conditions in the slope areas over the range of the thermocline depth fluctuations. This area is also where most of the lake productivity is found, and can be where most biota can subsequently be found (Tilzer, 1990).

In Canada, there are a total of 89 large lakes accounting for a total area of 201,449 km$^2$, making up 41.3% of the North American continent and 14.4% of the world (Herdendorf, 1990). Most of these large lakes in Canada formed during the Pleistocene glaciation (Hutchinson, 1957). A majority of these large lakes are found in northern Canada (Evans, 2000). Despite the national significance, the large lakes in northern Canada are severely understudied (Jenny et al., 2020). There was some general interest at the end of World War II to conduct research on northern lakes and rivers, with the aim of surveying the fisheries economies around the country (Hachey, 1965). After several early papers were published on two of Canada's largest lakes, Great Slave Lake and Great Bear Lake, the public interest on northern ecosystems waned down after early scientists stated that northern lakes were unproductive depressions filled with cold, clear water (Hachey, 1965). Since the 1940s and 1950s, this lack of interest contributed to the research decline on Canada's northern lakes (Cott et al., 2016). While research efforts in Canada's northern lakes have increased during the turn of the century, there is still more work to be done for a greater understanding of how present-day issues on climate change, disturbance, and human impacts continue to affect the north (Cott et al., 2016). The research gap in Canada is not restricted to Canada's northern Arctic and sub-Arctic communities. The Laurentian Great Lakes region, one of the best studied for large lake limnology on the planet, has a notable data gap for Lake

Nipigon, the focus of this thesis, sometimes referred to as the “Mother of Lake Superior” and the “Headwaters of the Great Lakes” (see Chapter 3).

A lack of long-term research and monitoring data impedes our ability to assess the cumulative impacts of climate change and other anthropogenic stressors in large lakes, which is necessary for adaptive management to protect water quality and freshwater biodiversity. In lieu of long-term records, paleolimnological techniques can be used to reconstruct the missing data. Paleolimnology involves the analysis of lake sediment cores as paleoenvironmental archives, allowing for the reconstruction of historical environmental conditions and assessment of conditions as a foundation for recommendations related to future management strategies.

## 2.3 Paleolimnology

Paleolimnology is a multidisciplinary science that uses physical, chemical, and biological information preserved in lake sediments to reconstruct environmental and ecological histories (Gregory-Eaves & Smol, 2024; Smol, 2008). These natural archives can record changes that can span up to decades and millennia back (Gregory-Eaves & Smol, 2024). Aquatic ecosystems are constantly changing, often at a rapid pace, and while long-term environmental monitoring programs are important to track these changes, there are challenges and limitations for making these efforts successful (Smol, 2019). Most environmental monitoring programs today only span an average of a year, with the aim of assessing before and after the stressor impacts, but one year is not long enough to understand natural variability (Smol, 2019). Paleolimnology can provide that missing link to gain insights into past ecosystem conditions, shifts in aquatic composition, and changes in ecosystem services over time (Gregory-Eaves & Smol, 2024). Long-term studies can distinguish between the “environmental noise” and signals from sustained environmental changes (Smol, 2008).

Lake sediments continuously accumulate at the bottom of lakes throughout the year (Smol, 2008). Deep lake basins often function as natural sediment traps, as their low-energy conditions rarely allow for the resuspension or transport of settled material (Smol, 2008). According to the Law of Superposition, in an undisturbed sequence, older deposits lie beneath younger ones, creating a depth-time profile that gradually builds up over time (Gregory-Eaves & Smol, 2024). These accumulating sediments originate from both external (allochthonous) and internal (autochthonous) sources, preserving valuable environmental information (Smol, 2019). Ideally, sediment cores should be collected from the deepest part of the lake to obtain the most representative sample (Smol, 2008). However, practical limitations such as lake depth, remoteness, and complex bathymetry may necessitate coring in nearshore areas to better capture environmental changes, especially in large-lake systems where deepwater coring locations are unlikely to capture the impacts of land-use on sensitive nearshore areas that are often biological "hotspots," and where people interact most with the lakes.

Sediment coring is an essential technique in paleolimnology. The main objective is to recover an undisturbed sample, typically including the sediment/water interface, which is a representative sample of the material that makes up the lakebed (Glew et al., 2001). Different types of corers can be used to retrieve a sediment profile. A gravity core is the simplest and easiest corer to use due to its portability and reliability (Smol, 2008). These samplers can retrieve samples up to 1 metre in length, and primarily use gravity to penetrate the sedimentary profile (Smol, 2008). The corer consists of a tube that is driven vertically into the sediments. After the coring drive is finished, the top of the tube is closed, and forms a seal. The tube is now filled with sediment, and can be retrieved to the lake surface (Smol, 2008). Freeze-crust samplers are another type of gravity corer that is best for sampling materials that are susceptible to disturbance and mixing, such as those close to the sediment-water interface or gas that may disrupt the core profile (Glew et al., 2001). The freeze-crust sampler contains a weighted chamber that is filled with a coolant (i.e. dry ice) immersed in alcohol, which is then lowered on a rope or cable into the sediment (Smol, 2008). The supercooled apparatus is held in position for 10 minutes, and it freezes the crust of sediment on the surface, *in situ*. The device is then retrieved to the lake surface, where the frozen sediment crust is removed for subsequent sampling (Smol, 2008).

When longer sedimentary sequences are required, piston corers are used (Glew et al., 2001). A typical piston corer has a piston and cable assembly, the core tube, and the drive head and drive rods (Smol, 2008). The corer enters the sediment with the piston at the bottom of the tube, effectively preventing sediment from entering the tube until the correct sampling depth is reached. The piston is then held stationary, and the core tube is pushed past into the sediment. After the sample is taken, the corer is recovered, and the core sample is extruded from the core tube (Smol, 2008). Coring can be carried out sequentially in different sequences, but a very stable platform is required, ideally an ice-covered lake surface (Glew et al., 2001).

The most common modification to gravity and piston cores is the addition of a driving weight on top of the device to increase penetration of the core tube (Gilbert & Glew, 1985). The hammer core is lowered into the sediment where it penetrates by its own weight. The driving weight on top, also known as the hammer, can be lifted from the top of the corer, then released to drive the core tube into the sediment (Glew et al., 2001). The recovery line is used as a guide, and a second line is attached to the driving weight. Tension must always be maintained on the recovery line, and precise control of the lifting line must be exercised to avoid the lines twisting (Glew et al., 2001).

Once the core is collected, it must be sectioned to represent different timescales. The resolution of sectioning will depend on the temporal resolution of interest. For instance, sectioning a more disturbed core at a sequence of 0.25 cm will be unrealistic due to bioturbation (Last & Smol, 2001). A slightly larger resolution of 0.5 cm or 1.0 cm would be more ideal (Smol, 2008). Once a sediment core is successfully retrieved and sectioned, it must be precisely dated to establish a reliable chronological framework. One of the most commonly used dating techniques in paleolimnology involves the measurement of Lead-210 ($^{210}$Pb), a naturally occurring radioisotope (Appleby, 2001). This method allows for the reconstruction of environmental changes over approximately the past 150 years, as $^{210}$Pb has a relatively short half-life of 22.26 years, making it well-suited for tracking recent historical events (Smol, 2008). $^{210}$Pb is produced through the decay of Radon-222 ($^{222}$Rn) and exists in two forms within sediments: (i) supported $^{210}$Pb, which originates from the in-situ decay of the parent isotope Radium-226 ($^{226}$Ra) within the sediment, and (ii) unsupported $^{210}$Pb, which is introduced through atmospheric deposition (Appleby, 2001). To determine sedimentation rates and assign dates to sediment layers, two primary mathematical models are commonly applied: the constant rate of $^{210}$Pb supply (CRS) model and the constant initial concentration (CIC) model (Appleby & Oldfield, 1978; Robbins, 1978). The CRS model assumes a constant rate of atmospheric $^{210}$Pb input, while the CIC model assumes a uniform initial concentration of $^{210}$Pb across all sediment layers (Appleby, 2001). The choice of model depends on site-specific conditions and sedimentation dynamics. The CRS model is most commonly used (Baud et al., 2022).

In addition to $^{210}$Pb dating, paleolimnologists often use other isotopic markers to enhance chronological accuracy. One widely utilized isotope is Cesium-137 ($^{137}$Cs), a synthetic radionuclide first released into the environment in 1945 as a byproduct of nuclear weapons testing (Pennington et al., 1973). The presence of $^{137}Cs$ in sediment cores provides a clear stratigraphic marker for the mid-20th century, coinciding with the onset of nuclear weapons testing. A peak in $^{137}Cs$ concentrations is often observed around 1963, corresponding to the period of highest atmospheric nuclear fallout before the signing of the Partial Nuclear Test Ban Treaty (Appleby, 2001). Another notable event that contributed to $^{137}Cs$ deposition was the Chernobyl nuclear disaster in 1986, which released significant quantities of radioactive material into the atmosphere. However, the fallout from Chernobyl was largely concentrated in regions of the former Soviet Union, Europe, and parts of Turkey, with minimal global dispersal compared to earlier nuclear testing (Smol, 2008). Over time, as international treaties restricted atmospheric nuclear testing, $^{137}Cs$ concentrations in sediments declined, providing a useful marker for dating post-1963 sediment layers (Appleby, 2001).

Paleolimnologists use a range of geochemical proxies to reconstruct past environmental conditions. Advances in analytical instrumentation now allow for precise measurement of inorganic geochemistry in sediment profiles (Boyle, 2001). Mineralogical analyses, through chemical precipitates in the water column or sediment pore spaces, offer insights into historical water chemistry, temperature, and other environmental variables (Last & Smol, 2001). Magnetic properties have also proven useful in establishing chronologies of sediment accumulation (Lowenstein & Brennan, 2001). Sediments are frequently used to track contaminant loading and persistent organic pollutants (POPs) across time and space (Blais & Muir, 2001). Near-infrared spectroscopy (NIRS) enables rapid, non-destructive analysis of organic molecules, providing reconstructions of past water chemistry and environmental conditions (Korsman et al., 2001). Among the most widely used techniques are stable isotope analyses, which provide paleoclimatic and paleohydrologic information (Last & Smol, 2001). For instance, $\delta^{13}C$ differentiates between aquatic and terrestrial sources of organic matter, $\delta^{15}N$ reveals nitrogen cycling and anthropogenic inputs, and $\delta^{18}O$ traces historical changes in precipitation, evaporation, and temperature (Wolf et al., 2001).

Biological proxies are powerful tools for reconstructing past ecological conditions, offering long-term insights that complement short-term water chemistry datasets (Smol, 2008). These indicators are typically preserved as morphological remains such as diatom frustules, invertebrate exoskeletons, or pollen grains. To be effective, a biological proxy must be identifiable in the sediment record, have a well-understood ecological affinity, and respond rapidly to change (Smol, 2008). Pollen and spores are commonly used to track terrestrial vegetation changes, aquatic macrophyte dynamics, and shoreline vegetation shifts (Birks & Birks, 2001). Charcoal particles—tiny residues from past fires—can indicate both natural fire regimes and anthropogenic wood-burning activities (Smol, 2008). Diatoms (Bacillariophyta), siliceous algae found in freshwater systems, are particularly useful due to their diverse ecological tolerances and rapid colonization abilities. Diatom assemblages can reconstruct variables such as pH, nutrient levels, lake productivity, and water-level changes (Battarbee et al., 2001). Similarly, siliceous remains of Chrysophytes (scales and cysts), which often dominate in oligotrophic systems, can be useful ecological indicators (Smol, 2008). Fossil pigments, composed of organic molecules like lipids, carbohydrates, and proteins, provide biochemical evidence of past primary production. They are often used alongside carbon-to-nitrogen (C/N) ratios to distinguish between algal (C/N = 4–10) and terrestrial plant input (C/N > 20) (Meyers & Teranes, 2001; Smol, 2008).

Invertebrate remains also serve as key indicators. Chitinized body parts of Cladocera (water fleas) have been widely applied to studies of eutrophication, acidification, salinity shifts, metal contamination, invasive species, and trophic changes (Korhola & Rautio, 2001; Jeppesen et al., 2001). Chironomidae (Diptera), another group of macroinvertebrates, are especially valuable in paleoenvironmental reconstructions, and are the primary focus of this thesis (see Section 2.3.2).

### *2.3.1 Paleolimnology of the Great Lakes*

The paleolimnology of large, deep lakes is well studied, though gaps in knowledge remain. Paleolimnological records from Great Slave Lake, North America's deepest lake, has narrowed down research gaps with evidence of Arctic warming on the aquatic ecosystem (Rühland et al., 2023). By analyzing algal regains from four high-resolution sediment cores from the lake's west basin and comparing them with historical phytoplankton data from the 1940s to 1990s, Rühland et al. (2023) observed a significant shift in algal community composition in ~2000. The combined impacts of increased regional air temperatures, reduced ice cover, and decreased wind speeds have altered the aquatic habitats of a major freshwater ecosystem in the Canadian sub-arctic (Rühland et al., 2023). In Europe, perialpine lakes have shown signs of pressure from climate warming, human-induced effects, and eutrophication (Perga et al., 2015; Tolotti et al., 2018). Climate warming is affecting freshwater ecosystems globally, and high-resolution paleolimnology offers better context than monitoring surveys by providing more detailed insights to inform management actions (Perga et al., 2015). A combined approach integrating paleolimnology with contemporary limnological surveys has been highlighted as one of the best ways to inform management strategies from ongoing environmental changes (Tolotti et al., 2018).

In the Laurentian Great Lakes of North America, O'Beirne et al. (2017) analyzed eight sediment cores from Lake Superior, and showed that anthropogenic climate change has contributed to extended seasonal stratification, longer ice-free periods, and increased primary productivity. European settlement further drove productivity increases, amplified by rising surface water temperatures. Beyond climate change, human activities such as urbanization and resource extraction have significantly shaped Lake Superior's ecology. Diatom community shifts indicate nutrient enrichment following European settlement and agricultural expansion, while trace metal profiles reflect increased mining and ore processing in the mid- to late-20th century (Chraïbi et al., 2014). More recently, the dominance of oligotrophic diatoms suggests ongoing physical and chemical shifts linked to climate change, underscoring the complex interplay of anthropogenic and natural factors in shaping the lake's long-term ecological trajectory (Chraïbi et al., 2014).

Similarly, Sgro & Reavie (2018) observed significant alterations in diatom communities and sediment geochemistry in Lake Huron following major anthropogenic disturbances across its multiple basins. A surge in population growth, along with industrial and mining pollution, led to severe sediment contamination, particularly evident by the 1930s (Sgro & Reavie, 2018). Diatom assemblages also began indicating climate-related changes as early as the 1970s (Sgro & Reavie, 2018). The combined impact of these environmental stressors has led to a sustained decline in phytoplankton abundance, a trend that is expected to persist unless effective remediation and management strategies are implemented to restore ecological balance within the lake basin (Sgro & Reavie, 2018).

Paleolimnological records reveal that Lake Ontario has undergone nutrient enrichment due to stressors such as human settlement and urbanization (Schelske, 1991). Before European settlement in the early 1800s, phosphorus levels were near pristine but gradually increased with population growth. Phosphorus loading peaked between 1940 and 1970 as communities expanded but declined sharply in the mid-1970s following phosphorus abatement efforts.

Organic carbon production closely mirrored historical phosphorus trends. Before mid-1800s deforestation, silica levels were sufficient for diatom growth but became limited in the summer epilimnion after 1865 and throughout the water column after 1950 due to increased diatom production and sedimentation. Elevated phosphorus loading further depleted silica reserves (Schelske, 1991). Diatom community shifts in sediment records further reflect the impacts of human activity (Pilsbury et al., 2021). Over the past 300 years, Lake Ontario has undergone multiple ecological transitions, beginning with an oligotrophic/mesotrophic diatom-dominated phase from 1700 to 1920, characterized by inorganic sediment accumulation. This was followed by rapid eutrophication between 1920 and 1980. Since 1980, diatom communities have returned to background levels in response to pollution reduction efforts. However, despite signs of recovery, Pilsbury et al. (2021) suggest that Lake Ontario remains vulnerable to the ongoing effects of climate change, underscoring the need for continued monitoring and management. Similar trends are also observed in The Bay of Quinte, a large embayment on Lake Ontario (Alward et al., 2025). After the combined stressors of eutrophication from anthropogenic nutrient loading in the late 1800s and an increase in post-dreissenid invasion, chironomid assemblage shifted from a hypoxia-tolerant *Tanytarsus lugens* and *Tanytarsus mendax* dominance to the more hypoxic-tolerant *Chironomus* sp. Nutrient abatement actions resulted in a decrease in *Chironomus* sp. and *Tanytarsus* spp. after 1977. Alward et al. (2025) highlighted that nutrient abatement actions were a stronger driver of overall chironomid taxonomic and functional composition than the dreissenid invasion.

Lake Erie has also shown similar paleolimnological shifts following industrialization, agricultural development, and changing population dynamics, with a noticeable shift in the geochronological record after 1935 (Sgro & Reavie, 2018). The diatom assemblage also shifted after 1985 attributing to ongoing climate warming and pollution (Sgro & Reavie, 2018). In support of these findings, Perlov et al. (2023) conducted a chironomid-based palolimnological assessment of several basins in the meso-eutrophic Lake Erie. Long-term trends across all coring sites suggested that oxygen concentrations have been on a decline over the past ~150 years due to the combined stressors of (1) increased land use due to population and agricultural development, (2) climate change resulting in earlier spring warming and increased duration of stratification, and (3) an increase in total phosphorus (TP) loadings in the watershed. The mid-1800s signalled the first shift to an oxygen-poor chironomid assemblage, which was followed by another shift in the 1950s marked by an assemblage confirming anthropogenic hypoxia (Perlov et al., 2023). These combined findings show that a Great Lake like Lake Erie requires a multi-scale recovery plan for a long-term goal in improving water quality, hypolimnetic dissolved oxygen (DO), and an overall healthy ecosystem.

The nearshore areas of the Great Lakes have also seen some advancement in research. In Lake Superior, Janssen et al. (2025) employed sediment core geochemistry, including total mercury (Hg) and isotopic signatures, to trace mercury inputs from tributaries such as the St. Louis River. Their results revealed that Hg is differentially sequestered between nearshore and offshore depositional zones, emphasizing the role of hydrodynamics and particle transport processes in the spatial distribution of contaminants. In a nearby freshwater estuary, Alexson et al. (2018) integrated diatom stratigraphy, organic geochemistry, and sediment dating (using $^{210}$Pb and $^{137}$Cs) to reconstruct nutrient enrichment trends and support nutrient delisting within an Area of Concern. The study demonstrated the utility of diatom-inferred total phosphorus (DI-TP) models and sedimentary chlorophyll-a as proxies for eutrophication. Moir et al. (2022) applied diatom teratology and multivariate ordination techniques (e.g., non-metric multidimensional scaling, NMDS) to assess industrial effluent impacts along the St. Lawrence River. The presence of malformed diatom frustules and shifts in community composition provided clear evidence of historical contamination, particularly with polycyclic aromatic hydrocarbons (PAHs) and heavy metals.

In a follow-up study, Armstrong et al. (2023) used subfossil chironomid assemblages and redundancy analysis (RDA) to evaluate the efficacy of remediation actions in the same region. Their findings indicated a partial recovery of benthic communities, with shifts from tolerant taxa (e.g., *Chironomus*) to more sensitive ones (e.g., *Micropsectra*), corresponding with reductions in sediment contaminant concentrations. In coastal wetlands of north-central Lake Ontario, Hoskin et al. (2024) highlighted how physical landscape features, particularly barrier beach morphology and water level fluctuations, influence the distribution of paleoecological indicator taxa such as diatoms and cladocerans. Their study employed sediment stratigraphy, loss-on-ignition (LOI), and high-resolution microfossil analysis to detect changes in salinity gradients and organic matter content, revealing how hydrological connectivity shapes ecosystem structure and function. Taken together, these studies illustrate the power of integrating biological proxies (e.g., diatoms, chironomids), geochemical markers (e.g., trace metals, isotopes), and stratigraphic dating methods to diagnose historical pollution, quantify environmental change, and inform evidence-based watershed management across complex freshwater systems.

### *2.3.2 Chironomids as Paleoecological Indicators*

Chironomidae (Diptera) are a diverse group of insects with an aquatic larval stage that serve as key indicators of freshwater ecosystem changes and an important food source for aquatic and terrestrial organisms (Epler, 2001; Smol, 2008). Chironomid larvae range from 2-30 mm in length, and come in a variety of colours, mostly pale yellow, green, or red. They are worm-like in appearance, with an elongated, nine-segment abdomen and three thoracic segments (Brooks et al., 2007). Chironomids are the most abundant macroinvertebrates in freshwater habitats, occurring on all continents and undergoing complete metamorphosis through a four-stage life cycle (Epler, 2001; Walker, 2001; Brodersen & Quinlan, 2006). Their life cycle begins with a brief egg stage, followed by a planktonic first-instar larva that eventually settles to the lake bottom. Over three successive instars, chironomids undergo growth and ecdysis, shedding their head capsule and cuticle, with most species becoming benthic (Walker, 2001). Chironomid larvae, which appear maggot-like in form, become larger as they proceed through the instar stages (Porinchu & MacDonald, 2003). The larvae exhibit diverse feeding habits, including filter-feeding, detrivory, algal grazing, and predation. Some burrow into wood or aquatic plants, while others prey on smaller invertebrates (Walker, 2001). The chironomid larval stage usually lasts 3-4 days. Once developed, a pair of paddles at the apex of the pupal abdomen assists the fully metamorphosed adult to swim to the surface, where it emerges from the water meniscus to breathe (Brooks et al., 2007). Adult chironomids appear like mosquitoes, but they lack scales on their wings and do not have a lengthy proboscis (Porinchu & MacDonald, 2003). Adults of most species live for just a few days, with some living up to a few weeks (Pinder, 1986). They do not usually feed, but may consume some honeydew or nectar (Brooks et al., 2007). Male adults typically congregate in large swarms, and can be present around trees, bushes, and prominent landmarks (Armitage et al., 1995). Females are usually attracted into the swarms to mate, where species are segregated by their differences in the height above ground and distance from water of the swarm (LeSage & Harrison, 1980).

Due to their sensitivity to environmental changes, chironomid assemblages in sediment cores provide essential insights into historical shifts in water quality, climate, and nutrient dynamics (Walker, 2001). Chironomid head capsules are highly resistant to decay due to their chitinized structure, make them valuable paleoindicators for reconstructing past environmental changes and exposure to toxic substances (Walker, 2001; Xu, 2014). During each molting phase, chitin concentration in their exoskeleton increases, enhancing their preservation in sediment layers (Brooks et al., 2007).

Chironomids are widely utilized in paleoenvironmental reconstructions to assess past changes in aquatic productivity, temperature, water depth, and hypolimnetic oxygen levels (Walker, 2001; Brodersen & Quinlan, 2006). Their resilience to environmental stressors, including pollution and habitat alterations, makes them valuable indicators of ecological change over centuries to millennia (Ferrington, 2008). Temperature is a key factor governing chironomid life cycles, influencing egg development, hatching success, and population dynamics (Williams, 1981). Chironomids have emerged as one of the most reliable biological proxies in paleolimnology for reconstructing past climatic conditions, particularly temperature, due to their strong ecological sensitivity (Porinchu & MacDonald, 2003). Hofmann (1988) was an early proponent of using chironomids in paleolimnological research, highlighting their potential for reconstructing not only temperature, but also other limnological variables such as lake depth, oxygen status, and productivity. Since then, numerous studies have refined and validated the chironomid-temperature relationship. For instance, Eggermont and Heiri (2012) provided a comprehensive review of chironomid ecology, confirming that chironomid assemblages display strong and predictable responses to temperature gradients in modern ecosystems. This ecological signal is sufficiently stable and statistically robust to form the basis for quantitative reconstructions using transfer functions, typically based on weighted averaging or modern analog techniques.

The geographic breadth of chironomid application has also expanded, from temperate regions to the Arctic and tropics. Brooks (2006) demonstrated that Eurasian chironomid communities exhibit distinct biogeographic patterns and thermal preferences, making them particularly valuable for paleoclimate studies across the Northern Hemisphere. In western Alaska, Kurek et al. (2009) used fossil chironomid assemblages to reconstruct Late Quaternary temperature changes and linked them with shifts in vegetation and glacial-interglacial cycles. Their findings showed clear faunal transitions corresponding to major climatic shifts, with cold-adapted taxa dominating during glacial periods and warm-water species appearing during interglacials. Similarly, Eggermont et al. (2010) applied chironomid-based paleothermometry in tropical Africa, proving that these proxies are also effective in low-latitude regions, where traditional paleoclimate proxies are often scarce or ambiguous. Luoto et al. (2014) further validated the use of chironomids in boreal and subarctic Finland by showing that chironomid-inferred lake water temperatures are closely correlated with measured summer air temperatures, supporting the assumption that chironomid communities reflect mean summer climate conditions during their growth period.

To make chironomid-based climate reconstructions more accurate and widely applicable, paleolimnologists have developed large reference datasets, known as training sets, which link modern chironomid communities to measured environmental conditions—especially temperature. These datasets allow researchers to identify patterns in how specific chironomid species respond to different climates and use that information to estimate past temperatures from fossil remains. Medeiros et al. (2022), for example, created a detailed training set for Arctic regions, which has significantly improved the ability to reconstruct past temperatures in those cold environments. By comparing fossil chironomid assemblages from sediment cores to modern datasets, researchers can calculate past summer temperatures and track how climate has changed over time. While some uncertainty is always present, these reconstructions include built-in estimates of error, helping scientists understand how confident they can be in the results. Overall, the strength of using chironomids lies in their clear link to temperature, their widespread presence in lakes around the world, and the excellent preservation of their remains in sediments. This makes them a powerful tool for revealing how climates have changed over thousands of years and for placing recent warming trends into a longer-term historical context.

General trends show that warmer temperatures generally lead to greater species richness and head capsule abundance, except in oxygen-limited environments, where assemblages are dominated by Chironomini taxa (Brooks et al., 2007). Cold-water lake sediments, by contrast, tend to have lower taxonomic diversity and head capsule abundance, with dominant taxa including Diamesinae, *Paracladius*, *Heterotrissocladius*, *Sergentia*, *Paracladopelma*, *Stichtochironomus*, *Micropsectra*, *Corynocera*, and *Tanytarsus lugens*-type (Brooks et al., 2007). Chironomid assemblage composition also varies by substrate morphology, with Orthocladiinae and Diamesinae more common in rocky and gravel substrates, while oxygen-poor silty sediments support higher proportions of Tanypodinae and Chironominae (Pinder, 1986; Brooks et al., 2007).

Chironomids have also been used for quantitative reconstructions for a wide range of environmental variables (Juggins, 2013). Quantitative reconstructions in paleolimnology are made possible through the development of robust statistical models that relate biological indicators, like chironomids, to specific environmental variables (Juggins, 2013). These reconstructions rely on the principle that certain chironomid taxa have well-defined ecological preferences, particularly for variables like temperature, total phosphorus (TP), oxygen availability, and habitat complexity. By analyzing fossil chironomid remains preserved in sediment cores and comparing them to modern datasets, researchers can infer past environmental conditions in a statistically meaningful way (Juggins, 2013). While quantitative reconstructions remain scientifically valid models, they must be used cautiously using statistically rigorous approaches. Juggins (2013) emphasized the importance of validation techniques, such as bootstrapping and cross-validation, to ensure model reliability and guard against overfitting.

One foundational example of this approach is found in the work by Brooks et al. (2001), who developed a chironomid–total phosphorus inference model to reconstruct historical nutrient levels in lakes. By examining surface sediment samples from 44 lakes in the United Kingdom and identifying the relationships between chironomid assemblages and measured TP concentrations, they created a weighted averaging model that could predict past TP with high accuracy. Reconstructed TP models from the mid-19th century to present day showed results that closely matched previous results from a diatom-based model, indicating gradual increase in TP since 1850, with a more pronounced upward trend post-1974. Chironomids are thus a useful tool in understanding past nutrient dynamics in aquatic systems.

Medeiros et al. (2015) further reinforced the reliability of quantitative reconstructions by evaluating the extent to which chironomid-based paleotemperature estimates are influenced by secondary environmental gradients. Their study of 61 lakes across northern North America confirmed that July air temperature was the dominant factor shaping chironomid distributions, but also identified the influence of variables such as lake depth and conductivity. Despite these secondary effects, the authors concluded that chironomid-based reconstructions remain robust when modern calibration datasets are regionally appropriate and sufficiently broad.

Complementing this, Stewart et al. (2013) conducted a paleolimnological experiment in High Arctic ponds, isolating the effects of temperature, nutrients, and oxygen on chironomid assemblages. Their findings demonstrated that chironomids respond to multiple environmental stressors simultaneously, but temperature and oxygen were the most consistent drivers. The dominance of oligotrophic, cold stenotherms in nutrient-rich ponds suggested that oxygen, not nutrients, was the primary control on chironomid assemblage composition in high latitude Arctic ponds. This reinforces the need to account for multivariate influences while also validating chironomid sensitivity to key climatic and limnological changes.

Brodersen and Lindegaard (1999), as well as Brodersen and Anderson (2002), used chironomids to classify and assess lake conditions in Denmark and West Greenland, respectively. These studies demonstrated that chironomid assemblages could effectively distinguish lakes by trophic status and thermal regimes, supporting their use in environmental reconstruction across varying geographic contexts. The latter study showed that chironomid distributions aligned well with both temperature and nutrient gradients in Arctic lakes, highlighting their versatility as indicators. Additional examples reinforce this point. Jeziorski et al. (2021) used chironomids to track deepwater oxygen recovery in Peninsula Lake, Ontario, a Precambrian Shield lake that has experienced many environmental stressors since the arrival of European settlers in the mid-1800s, including forest clearance, water-level management, sewage inputs, and land-use changes. The chironomid assemblage indicates faunal shifts that corresponded with improved oxygen conditions following management interventions from the 1970s onwards. Similarly, Lim et al. (2021) linked changes in chironomid assemblages and chitin degradation to seabird population dynamics over 1,700 years in Baccalieu Island, Newfoundland and Labrador, illustrating the sensitivity of these insects to both biotic and abiotic factors. Chironomid community changes closely tracked seabird population trends – especially colony growth beginning in the early 1800s and decline in the 1980s. Nutrient-rich N and P inputs from bird guanos resulted in the changes in lake chemistry, pH, metal concentrations, and oxygen levels (Lim et al., 2021). Antczak-Orlewska et al. (2021) added further ecological nuance by exploring chironomid morphological types and feeding groups as indicators of habitat complexity. Their functional approach underscores the potential of using chironomid traits—not just taxonomic identities—for reconstructing past environments with greater ecological depth. There are strong correlations between certain mentum types and environmental factors such as macrophyte cover, substrate type, and water chemistry, all of which suggest that morphological and feeding traits of chironomids are reliable indicators of past habitat structure and ecological function (Antczak-Orlewska, 2021). Together, these studies collectively demonstrate that quantitative reconstructions are possible and increasingly sophisticated through the use of chironomid-environment calibration models. When supported by rigorous statistical methods, region-specific datasets, and careful consideration of confounding environmental gradients, chironomid-based paleolimnology offers a powerful tool for reconstructing climate, nutrient dynamics, and ecological change in freshwater systems.

Given the numerous examples of using chironomids as paleoenvironmental indicators of ecological change, the chironomid subfossils can provide useful information on changes to benthic habitat quality in Lake Nipigon. In limnological studies conducted in the 1920s, chironomids were the most abundant macroinvertebrates in shallow and benthic habitats (Adamstone, 1923). Gut content analysis of fish species confirmed their ecological importance as primary food source for fish, highlighting their role in sustaining the Lake Nipigon aquatic food web, and contributing to ecosystem productivity. Furthermore, anthropogenic stressors that have impacted Lake Nipigon over the last century (explained further in Chapter 3) are likely to have an impact on benthic habitat.

# CHAPTER THREE: STUDY SITE DESCRIPTION

## 3.1 Lake Nipigon Physiography

Lake Nipigon is situated approximately 170 kilometers northeast of the city of Thunder Bay and 50 kilometers north of the town of Nipigon (Herdendorf, 1990). Lake Nipigon was first measured in the 1870s to be 70 miles long (~113 kilometres) and 50 miles wide (~80 kilometres) (Bell, 1870). Lake Nipigon's drainage basin structural depression was shaped by volcanic and intrusive mountain-building processes, interspersed with extended periods of differential erosion (Ministry of Natural Resources, 2004). The surface relief in the region is primarily governed by bedrock features. Lake Nipigon, along with most of its shoreline and islands, is situated within the Nipigon Plate, a sub-province of the Southern Province in the Canadian Shield (Ministry of Natural Resources, 2004).

The surficial geology of the Lake Nipigon area is largely shaped by events from the late Wisconsin glacial period and the early Holocene (Scott, 2015). The region features an eroded, glacially sculpted bedrock surface interspersed with localized deposits of glacial till, peat, and glaciolacustrine sands and silty clays (Ministry of Natural Resources, 2004; Scott, 2015). During the late Wisconsin period, small, shallow pockets of ground moraine—composed of unstratified and unsorted sediment—were deposited among numerous rock outcrops (Ministry of Natural Resources, 2004). The moraine material, mostly composing of till and ranging in size from clay to large boulders, was also eroded by glacial meltwater (Scott, 2015). This led to the creation of vast amounts of sand and gravel (aggregate), with notable deposits in Innes Township, south of

Lake Nipigon, and north of Windigo Bay (Scott, 2015). Four major moraines, the Kaiashk, Onaman, Nipigon, and Crescent moraines, highlight the significant glacial events in the area (Ministry of Natural Resources, 2004).

The watershed was mostly ice covered until the late stages of the Wisconsin glacial retreat (Scott, 2015). The Kaiashk interlobate moraine, oriented northeast-southwest and located southwest of Lake Nipigon, formed during a significant retreat of glacial ice from northwestern Ontario (Ministry of Natural Resources, 2004). This retreat enabled extensive drainage of Glacial Lake Agassiz into the Superior Basin (Scott, 2015). The ice subsequently retreated further to the north and east before re-advancing around 10,000 years ago during the Marquette stadial, forming the Nipigon Moraine. This north-south trending moraine runs roughly parallel to, but inland from, Lake Nipigon's present western shore (Ministry of Natural Resources, 2004; Scott, 2015). As the ice retreated again, the Onaman interlobate moraine developed along a glacial weakness northeast of Lake Nipigon. Around 9,500 years ago, Glacial Lake Kelvin emerged along the ice margin, eventually covering the entire Lake Nipigon Basin (Scott, 2015). Lake Kelvin later drained into Lake Superior, leaving behind what is now Lake Nipigon (Scott, 2015). Shallow-water sand deposits from Glacial Lake Kelvin, including plains and beaches, are most prominent along Lake Nipigon's northern and northeastern shores, particularly north of Windigo Bay (Ministry of Natural Resources, 2004). Glaciolacustrine features are also visible along the Kabitotikwia River floodplain (Ministry of Natural Resources, 2004). Deeper water silty clay deposits from Lake Kelvin can occasionally be found along the southwestern shore and near Humboldt and Ombabika Bays in the South Peninsula area (Ministry of Natural Resources, 2004).

Within the Lake Nipigon watershed, the overburden of Lake Nipigon is characterized by morainic material comprised of silt and sand (Ministry of Natural Resources, 2004). The erosive action by the retreating glacier accompanied the deposition of outwash sand and gravel (Ministry of Natural Resources, 2004). These deposits are sorted and stratified in various resource areas around the lake. Fine sand, silt, and clay were deposited over much of the Lake Nipigon area adjacent to Gull Bay, English Bay, Windigo Bay, Ombabika Bay, and Humboldt Bay (Scott, 2015). These sedimental deposits indicate historical deep water and inundation by a glacial lake (Scott, 2015).

Lake Nipigon, often referred to as the "Mother of the Great Lakes," is part of the larger Great Lakes Basin in North America (Herdendorf, 1990; Scott, 2015). The Nipigon basin is the largest drainage system within the Lake Superior watershed and intersects the Arctic-Atlantic Continental Divide located north of the lake (Scott, 2015). Numerous rivers and streams form a radial network of tributaries around Lake Nipigon, draining an area of 24,650 square kilometers (Scott, 2015). The watershed was further expanded by 50% in the 1940s with the construction of the Ogoki Diversion by the U.S. Army Corps of Engineers, which significantly increased water flow and enhanced hydroelectric power generation downstream at Niagara Falls (Scott, 2015). Key tributaries include the Wabinosh, Gull (Kiashke), and Poshkokagan Rivers on the west shore; the Onaman, Namewaminikan (Sturgeon), and Blackwater Rivers on the east shore; and the Pikitigushi, Ombabika, and Little Jackfish Rivers to the north (Scott, 2015). Most of these rivers have moderate gradients, except for the Gull, Little Jackfish, and Namewaminikan (Sturgeon) Rivers, which contain whitewater rapids (Scott, 2015). Among them, the Little Jackfish River, which channels the Ogoki Diversion, is now the most turbulent and carries the highest water volume (Scott, 2015).

Lake Nipigon has a climate characterized by cold winters and mild summers, with average temperatures of -18.5ºC in January and 16.8ºC in July (Environment and Climate Change Canada, 2025). During the open water season, the lake's waters remain clear and cool, with surface temperatures ranging from 15°C to 25°C, while shallower bays can reach 20°C-25°C (Environment and Climate Change Canada, 2025). Summers are long and sunny, with up to 16.5 hours of daylight at the summer solstice on June 21, whereas the winter solstice in January provides only eight hours of daylight (Environment and Climate Change Canada, 2025). The summer season typically lasts from mid-June to early September (approximately 89 days), whereas winter extends from late October to late April, spanning 175-190 days (Environment and Climate Change Canada, 2025). Due to its northern position near the 50th parallel and proximity to the Eastern-Central time zone meridian, the sun remains visible well past 11:00 pm during the summer solstice (Environment and Climate Change Canada, 2025).

The lake's prevailing winds generally blow from the west to northwest in autumn, winter, and summer, while shifting to north-northwest in spring (Environment and Climate Change Canada, 2025; Scott, 2015). Wind speeds often exceed 30 km/h during the open water season, creating large swells and rough conditions (Environment and Climate Change Canada, 2025). Lake Nipigon also plays a significant role in regional precipitation patterns, as prevailing winds pick up moisture from the lake, particularly in late fall and early winter before the lake freezes. This results in heavy snowfall, especially along the east shore, where accumulations surpass those of surrounding areas (Environment and Climate Change Canada, 2025). Historical average snowfall between 1991-2020 ranges from 64 to 79 cm, though in exceptional years, totals can exceed 102 cm (Environment and Climate Change Canada, 2025). Ice cover typically begins forming in early to mid-December, with the lake freezing over completely. Spring ice breakup varies by year but generally occurs between late April and mid-May (Environment and Climate Change Canada, 2025).

## 3.2 A Brief History of the Lake Nipigon Basin

Six First Nations communities—Biinjitiwaabik Zaaging Anishinaabek (Rocky Bay), Gull Bay, Animbiigoo Zaagi'igan Anishinaabek (Lake Nipigon Ojibway), Red Rock, Sand Point, and Whitesand—have long inhabited the Lake Nipigon Basin (Ministry of Natural Resources, 2001). Among the Great Lakes, Lake Nipigon was the last to be reached by European settlers in the early 1600s, with French explorers, Jesuit priests, and fur traders first to arrive in the region (Waters, 1987). French explorer Daniel Greysolon named the lake "Lac Alemipigon", meaning "water you cross to get somewhere else" (Waters, 1987).

Ojibwa communities played a key role in the fur trade, acting as intermediaries between the French and Indigenous groups such as the Assiniboine and Cree, exchanging trade goods for furs (Higgins, 1979). In the 1700s, the Hudson's Bay Company was established to support the expanding fur trade industry (Chisholm & Gutsche, 1998). From 1670 to 1820, Lake Nipigon was at the center of a prolonged conflict between England and France over trading rights (Chisolm & Gutsche, 1998). This rivalry ended in 1763 when England took control of all French territories in North America following the signing of the Treaty of Paris (Higgins, 1979). In 1779, former Hudson's Bay Company employee Edward Umfreville established the North West Company (Dawson et al., 1969). Intense competition between the two companies resulted in the overexploitation of fur in the Nipigon region (Scott, 2015). Eventually, in 1821, they merged under the Hudson's Bay Company (Higgins, 1979). By the mid-1800s, the fur trade began to decline due to shifting fashion trends in Europe (Higgins, 1979). Most trading posts had closed by 1870, with Nipigon House, located on Jackfish Island near English Bay, being the last remaining one (Dawson et al., 1969). In 1937, it was relocated to Gull Bay, at the site of the present-day Gull Bay Indian Reserve, before ultimately closing in the 1990s (Dawson et al., 1969).

The construction of railways between the late 1800s and early 1900s significantly boosted economic activity in the Nipigon region by improving access, encouraging settlement, and supporting development (Higgins, 1979). This expansion also brought a surge in tourism, with Indigenous communities serving as fishing guides for American anglers who traveled to Lake Nipigon in search of trophy trout after the Nipigon River was declared the "finest trout stream of the world" in 1887 (Higgins, 1979). However, by 1888, overfishing and exploitation had already begun to deplete fish populations (Waters, 1987). To address these concerns, fishing restrictions—including licenses, size limits, and catch limits—were introduced in 1892 (Waters, 1987). The situation worsened between 1920 and 1950 with the construction of three dams on the Nipigon River, which further harmed the fishery (Dawson et al., 1969). Today, while the Nipigon River still holds some prized trout, it is largely defined by a series of dammed lakes interspersed with short, turbulent sections of moderate current (Scott, 2015).

### 3.3 Mining in the Lake Nipigon Watershed

The Lake Nipigon watershed has a long and complex history of mining, dating back to the early 1900s (Ministry of Natural Resources, 2004). The first significant mining discovery in the region was made in 1925 by Powers and Silam, a mining company that identified valuable mineral deposits east of Lake Nipigon (Speed & Craig, 1992). This discovery set the stage for the development of the region's mining industry, leading to the establishment of the first operational mines—the Northern Empire Beardmore Mine and the Pan-Empire Mine—in 1934

(Speed & Craig, 1992). The initial success of these mines attracted further exploration and investment, prompting the expansion of mining operations to surrounding areas, including the towns of Long Lake and Nakina (Speed & Craig, 1992).

Between 1934 and 1968, the Beardmore-Geraldton-Longlac mining belt became one of Ontario's richest sources of gold, silver, and other metals. At its peak, this region supported a thriving mining industry, with a total of 21 mines producing an estimated 4,115,000 ounces of gold and 318,500 ounces of silver (Speed & Craig, 1992). Mining in the area not only shaped the economic landscape, but also contributed to the rapid development of local communities, fostering infrastructure growth and workforce migration (Scott, 2015). However, as ore deposits were gradually depleted and global economic conditions fluctuated, mining operations began to decline. By the late 1980s, most mining activities had ceased, marking the end of an era for the region (Speed & Craig, 1992).

Despite the cessation of large-scale mining, the legacy of these operations remains. While many mine claims are still active, the long-term environmental impacts, including potential contamination from tailings, waste rock, and heavy metals, remain largely unstudied (Speed & Craig, 1992). The extent of legacy mining contamination within the Lake Nipigon watershed is still not well understood, and its potential effects on water quality, aquatic ecosystems, and Indigenous communities reliant on the watershed for subsistence remain critical areas for future research. Understanding and addressing these historical mining impacts is essential for ensuring the long-term sustainability and health of the Lake Nipigon watershed.

### 3.4 Ogoki River Diversion

Lake Nipigon has experienced significant water level manipulations due to hydroelectric projects that began in the early 1900s. These large-scale interventions were primarily driven by the growing demand for hydroelectric power in southern Ontario, particularly during the height of World War II (Scott, 2015; Annin, 2018). As industrial production ramped up and energy needs surged, Ontario Power Generation (OPG) sought new sources of power to prevent shortages (Annin, 2018). The urgency of the situation led to a wartime diplomatic agreement between Canadian and U.S. legislators, formalizing concerns over an impending power crisis in communities surrounding the Great Lakes region (Annin, 2018). This agreement laid the groundwork for the construction of two major water diversion projects—Ogoki and Long Lac—to address energy shortfalls, with Canada securing hydro rights over both systems (Annin, 2018). These diversions were later codified under the Niagara Treaty of 1950, which updated international hydro rights at Niagara Falls (Day et al., 1982). The potential economic benefits of these diversions were widely publicized, with a 1925 op-ed in the Chicago Tribune boldly proclaiming, “Every cubic foot of water diverted from the relatively useless basin of Hudson Bay into the lakes will prove liquid gold” (Canadian Geographic, 2024). While these diversions played a critical role in providing wartime hydroelectric relief, they also fundamentally altered water levels, introduced extreme fluctuations, and disrupted fish habitats, not only within the diversion systems but across the entire Great Lakes basin (Annin, 2018; Canadian Geographic, 2024).

The Ogoki Diversion, one of the largest water management projects in Canadian history, was constructed in the 1940s to redirect billions of gallons of water from the Ogoki River, which naturally drains into Hudson Bay, into the Atlantic watershed via Lake Nipigon (Annin, 2018).

This complex engineering feat involved the construction of two major dams in remote, roadless regions to regulate the inflow of diverted water. The first, Waboose Dam, was a massive 50-foot-high, 1,700-foot-long structure designed to block the upper reaches of the Ogoki River and effectively reroute its natural course (Annin, 2018). The second, Summit Dam, serves as the main outflow point, releasing billions of gallons of reversed Ogoki River water daily into Lake Nipigon through the Little Jackfish River at a rate of approximately 4,000 cubic feet per second (cfs) (Annin, 2018). Today, the Ogoki Diversion remains the largest river diversion project in North American history, surpassing even the more widely recognized Chicago Diversion in Illinois, which has an estimated flow rate of 3,200 cfs (Annin, 2018).

In addition to the Ogoki Diversion, a network of dams was constructed along Lake Nipigon's southeastern region, including those on the Namewaminikan River and Nipigon River, to further regulate water flow into Lake Superior (Annin, 2018). Although these hydroelectric developments were designed to bolster Canada's energy economy, they were undertaken without consultation with the Indigenous communities directly impacted by fluctuating water levels and environmental changes (Annin, 2018). First Nations communities living along Lake Nipigon have long voiced concerns about the cumulative effects of these water level manipulations, citing their profound impacts on traditional livelihoods, local economies, and cultural practices (Annin, 2018). Some communities were forcibly displaced due to rising water levels and shoreline erosion, which resulted in flooding of traditional lands and sacred burial sites (Higgins, 1979). The Whitesand First Nation, historically settled along the northwest shores of Lake Nipigon, was forced to relocate near Armstrong, Ontario, in 1942 as water level fluctuations made their lands increasingly uninhabitable (Canadian Geographic, 2024).

Beyond displacing communities, these hydroelectric projects also introduced significant ecological disruptions. The increased turbidity caused by heightened streamflow from the Ogoki Diversion eroded sediments along the Little Jackfish River, degrading water quality and damaging fish habitats, particularly in biologically productive nearshore areas (Day et al., 1982). Moreover, the creation of multiple dams along the Ogoki River inadvertently exacerbated mercury contamination in the watershed (Annin, 2018). When soil is flooded due to dam construction, the release of organic carbon can trigger the conversion of elemental mercury into toxic methylmercury—a potent neurotoxin that bioaccumulates in aquatic food webs (St. Louis et al., 2004). This process led to dangerously high mercury concentrations in local fish populations, posing serious health risks to Indigenous communities who traditionally relied on fish as a dietary staple (Annin, 2018). As a result, many First Nations communities were forced to limit their fish consumption to mitigate health risks (Hollinger, 2024). Whitesand First Nation members, for example, reduced their fish intake to just two fish per month from the Little Jackfish River beneath the reservoir in an effort to minimize exposure to mercury (Canadian Geographic, 2024). The long-term ecological and health effects of these hydroelectric interventions remain poorly understood.

### 3.5 Limnology of Lake Nipigon

Present-day Lake Nipigon is characterized to be large, deep, oligotrophic (nutrient poor) lake, with its water quality typical of lakes located within the Canadian Shield (Scott, 2015). There are over 500 islands within Lake Nipigon, most of them remote and far from the mainland (Scott, 2015). These islands serve as important fish-spawning habitats, woodland caribou feeding sites, and provide wind breaks. The sheltered bays also provide recreational tourists opportunities for mooring and camping (Scott, 2015). Lake Nipigon is composed of 90% water and 10%

islands, with many of these islands having their own distinct ecosystems. Shakespeare Island, the 2nd largest island within Lake Nipigon, is its own community with its estimated size of 1,200-2,700 hectares (Scott, 2015). Lake Nipigon has one of the best pristine, cold-water fisheries within Superior Country, known for its prized sport fish like Brook Trout, Walleye, Lake Trout, and Northern Pike (Superior Country, 2023).

In the 1980s, Lake Nipigon was described to be the only Great Lake with its original flora and fauna intact (Hecky, 1989). As such, Lake Nipigon is ideal to serve as the reference system for the rest of the Great Lakes, most of which are in more deteriorated conditions, and still suffers biotic nutrient and contamination perturbations (Hecky, 1989). However, during her frequent trips to Lake Nipigon during the 1980s as part of her work with the Ontario Ministry of Natural Resources, Scott (2015) regularly met with community leaders who observed significant changes in Ombabika Bay, as characterized by greater historical siltation due to the Ogoki Diversion. Despite community concerns regarding the impacts of resource developments occurring in the watershed, limnological research and monitoring of Lake Nipigon has historically been lacking. The last time a detailed limnological study was conducted for Lake Nipigon was prior to the onset of most industrial development (described below), and those study did not include Ombabika Bay.

A series of in-depth limnological studies in Lake Nipigon were conducted by a group of researchers from the Department of Biology in the University of Toronto between 1921-1924. Clemens (1923) characterized Lake Nipigon to be a large, deep, cold-water lake with high oxygen, low free carbon dioxide, high bicarbonate contents, and a hydrogen-ion concentration between 8.0 to 8.3. The central, main portion of the lake extended up to a depth of 123 metres (Clemens, 1924). There are many shallow waters surrounding the main shores of many islands, and there are numerous tributary streams connected to many sections of Lake Nipigon that bring in dissolved materials and plant debris to support healthy fish spawning activities (Clemens, 1924). Dissolved oxygen (DO) was consistently monitored during the study period, and results show that it never went below 86% saturation (Clemens, 1923). DO saturation remained high even in deep waters (Clemens, 1923). The amount of free carbon dioxide was small at all depths, bicarbonate content was mostly over 100 parts per million in surface and bottom sampling, and the total acidity of the water ranged from 2 to 5 parts per million of $CaCO_3$. Lake Nipigon exhibited good mixing patterns, and was thoroughly mixed down to a depth of 46 metres (Clemens, 1924). Heat distribution was measured at a depth of 23 metres in all sampling years, and the values showed a consistent yearly trend from 1921-1923 (Clemens, 1924). The large surface area of the lake makes it possible for severe storms to happen, and as such has left many shorelines largely barren, rocky, or sandy (Clemens, 1924). Large amounts of sand are also brought in from the streams surrounding the rocky drainage basin, and has created some large, desert-like areas within the lake (Clemens, 1924).

Sediment samples were collected by Clemens (1924) to assess the health of the bottom fauna. Chironomid larvae were noticeably one of the most common bottom organisms across all sampling years, and they usually reached their maximum numbers at a depth between 15-20 feet at a temperature of 5°C (Adamstone & Harkness, 1923; Clemens, 1924). A total of 228 dredgings extending to a depth of 180 feet yielded approximately 3723 chironomid larvae (Adamstone & Harkness, 1923). Chironomid abundances decreased between the depths of 30 to 60 feet, except for a marked increase between 60 to 75 feet. Larvae catch beyond a water depth of 75 feet was relatively small, but some Chironomidae larvae were being obtained at a depth of 178 feet (Adamstone & Harkness, 1923). Within the Lake Nipigon ecosystem, Adamstone

(1923) highlighted that chironomid larvae are one of the most important food organisms in the lake, not only because of their abundance, but also because of their widespread distribution. Clemens (1924) also emphasized that the abundance of macroinvertebrate organisms is a good indicator in support of a healthy, natural fish-cultural body of water.

In addition to chironomids, other early limnological investigations of Lake Nipigon focused on the diversity and distribution of various aquatic organisms, including diatoms, plankton, mollusks, and phytoplankton (Adamstone & Harkness, 1923; Adamstone, 1923; Bigelow, 1923; Clemens et al., 1923). These studies collectively highlighted the richness of Lake Nipigon's freshwater biodiversity, reinforcing the lake's ecological significance as a habitat for a diverse array of aquatic fauna (Adamstone & Harkness, 1923; Adamstone, 1923; Bigelow, 1923; Clemens et al., 1923).

Although Lake Nipigon remains one of Ontario's largest and most ecologically significant inland freshwater systems, contemporary research has largely focused on its fish populations rather than on comprehensive environmental monitoring. Notably, studies have examined the ecomorphological and genetic polymorphisms of *Coregonus* species (ciscoes) throughout the lake, revealing intraspecific diversity that suggests habitat partitioning and potential evolutionary divergence (Turgeon et al., 1999). In more recent years, attention has shifted toward assessing the legacy impacts of mercury (Hg) contamination, particularly its effects on commercial fisheries and local Indigenous communities (Hollinger, 2024). Following the construction of the Ogoki Diversion, which altered hydrological inputs to the lake, Indigenous commercial fishers reported a sharp decline in fish stocks—attributed in part to elevated mercury inputs from the Little Jackfish River (Hollinger, 2024). The collapse of the Walleye (*Sander vitreus*) fishery in the mid-1990s—particularly in Ombabika Bay—highlighted the ecological vulnerability of the lake. Early concerns raised by the Rocky Bay Fisheries Unit resulted in the implementation of a moratorium and the eventual discontinuation of commercial harvest quotas (Hollinger, 2024). In response, the Rocky Bay First Nation has since developed and maintained its own independent dataset to monitor contaminants and assess emerging environmental threats. Despite these efforts, the absence of long-term ecological data and the lack of comprehensive environmental baselines limit the community's ability to fully understand the drivers of change affecting their traditional fisheries.

A multi-proxy paleolimnological reconstruction can address this gap by offering a long-term perspective on ecological change. Such an approach integrates biological, geochemical, and isotopic indicators preserved in sediment cores to infer historical fluctuations in water quality, productivity, contaminant loading, and climate-related stressors (Gregory-Eaves & Smol, 2024). This is especially critical given the lack of coordinated environmental monitoring in Lake Nipigon. As noted by the Ontario Ministry of Natural Resources (2003), standardized sampling protocols and continuous monitoring data for key limnological variables—including phytoplankton, zooplankton, benthos, and water quality—remain unavailable. Data exchange between academic institutions, the Ministry of the Environment, and other regulatory agencies has also been suboptimal, further hindering efforts to assess lake health in a coordinated and integrated fashion.

Despite its importance as the headwater of the Great Lakes drainage system, Lake Nipigon is not formally included in binational lake management frameworks instituted by organizations like the International Joint Commission and the Great Lakes Fishery Commission, thereby excluding it from long-term water quality goal-setting aligned with the Lake Superior basin plan. A robust paleolimnological assessment will not only reconstruct historical ecological conditions, but will also provide an essential baseline for anticipating future impacts from anthropogenic and climate-related stressors. This information is critical for informing adaptive fisheries management, supporting Indigenous stewardship, and guiding conservation planning for the long-term sustainability of Lake Nipigon.

### 3.6 Bays and Nearshore Areas of Lake Nipigon

Lake Nipigon is a vast and ecologically complex freshwater system situated in northwestern Ontario. Encompassing approximately 48,500 hectares of water surface area, the lake is characterized by a network of prominent bays encircling its expansive central basin (Ministry of Natural Resources, 2003). Management of commercial fishing and recreational angling across Lake Nipigon is governed by provincial regulations and community-level planning, with localized oversight by First Nations communities.

Among the lake's many embayments, Ombabika Bay—located in the northeastern sector—was a principal coring site for this research. This bay received inflows from the Ogoki Diversion via the Little Jackfish River, a hydrological alteration associated with documented concerns from Rocky Bay First Nation fishers regarding declines in the Walleye (*Sander vitreus*) population (Hollinger, 2024). Despite these community-reported impacts, no formal environmental monitoring programs have evaluated the ecological consequences of the diversion on this region of the lake.

South Bay, located in the southeast, lies directly west of Rocky Bay First Nation and was another focus of ecological assessment for this study. According to the Ministry of Natural Resources (2003), South Bay supports sensitive and representative fisheries habitats, including extensive aquatic vegetation, wetland complexes, and critical spawning grounds. These attributes also make it an important zone for recreational fishing. To the southwest lies McIntyre Bay, a relatively remote and uninhabited area that is used seasonally by Indigenous communities for recreational activities (Scott, 2015). This bay has also been classified by the Ministry as ecologically sensitive, hosting significant spawning habitats and aquatic biodiversity (Ministry of Natural Resources, 2003). McIntyre Bay is culturally significant to Biinjitiwaabik Zaaging Anishinaabek (Rocky Bay First Nation).

Gull Bay, situated on the western shore, is home to Gull Bay First Nation and provides access to the western expanse of Lake Nipigon. With an estimated surface area of 507 hectares, the bay is designated as an Enhanced Management Area, which includes a 200-metre shoreline zone prioritized for future access and development planning (Ministry of Natural Resources, 2003; Higgins, 1979; Scott, 2015). Chiefs Bay, also on the western shoreline, is considered relatively pristine with no documented environmental stressors. Known for its scenic beaches, it is a popular site for recreational angling (Scott, 2015). Humboldt Bay, located on the eastern side of Lake Nipigon, surrounds Livingstone Point Provincial Park and is largely uninhabited. It has been identified as an area of ecological significance, with vital spawning and aquatic habitats (Ministry of Natural Resources, 2003). Wabinosh Bay, in the northwest section of the lake, is notable for its rocky substrate and its proximity to the Wabinosh River and Kopka River Provincial Park Addition. The bay supports a population of American white pelicans (*Pelecanus erythrorhynchos*) and is also considered a key location for spawning and recreational fisheries (Scott, 2015; Ministry of Natural Resources, 2003).

Beyond the bays, Lake Nipigon is dotted with hundreds of islands, which hold significant ecological and recreational value. The northern islands serve as a major habitat and migratory corridor for woodland caribou (*Rangifer tarandus caribou*), while the southern islands support populations of moose (*Alces alces*) and other game species (Ministry of Natural Resources, 2003). These areas are protected under legislation such as the Crown Game Preserve Status and the Wilderness Area Act. The Ministry also encourages natural fire regimes to proceed unimpeded on these islands—provided that no human life or infrastructure is at risk—in order to promote ecosystem renewal and habitat diversity (Ministry of Natural Resources, 2003).

# CHAPTER FOUR: METHODOLOGY

## 4.1 Field Methods

Sediment cores were collected in nearshore areas and bays throughout Lake Nipigon every summer from 2021-2024 (Figure 4.1; Table 4.1) using a UWITEC gravity corer (Uwitec, Mondsee, Austria). Coring locations were recommended by BZA commercial fishermen, who also provided transport to coring locations aboard BZA's research vessel (the *Velma Linda*). Nearshore cores were mostly chosen due to community concerns, primarily surrounding their commercial fishery livelihood and other sites of cultural significance. As there is no bathymetry available for Lake Nipigon, several different locations were cored to identify areas that had retrievable sediment accumulation. We also used a sonar unit to explore the bathymetry near the coring location, to ensure the location had a relatively flat bottom profile. Cores collected in 2021 and 2022 were extruded into 1.0-cm intervals, while cores collected in 2023 and 2024 were extruded into 0.5-cm intervals (Figure 4.2). All cores were extruded on the same day of collection using a modified Glew (1988) extruder. Each interval sample was placed in individual labelled Whirl-Pak bags, and were placed in a cooler with ice for transportation back to York University, where they were stored frozen until analysis.

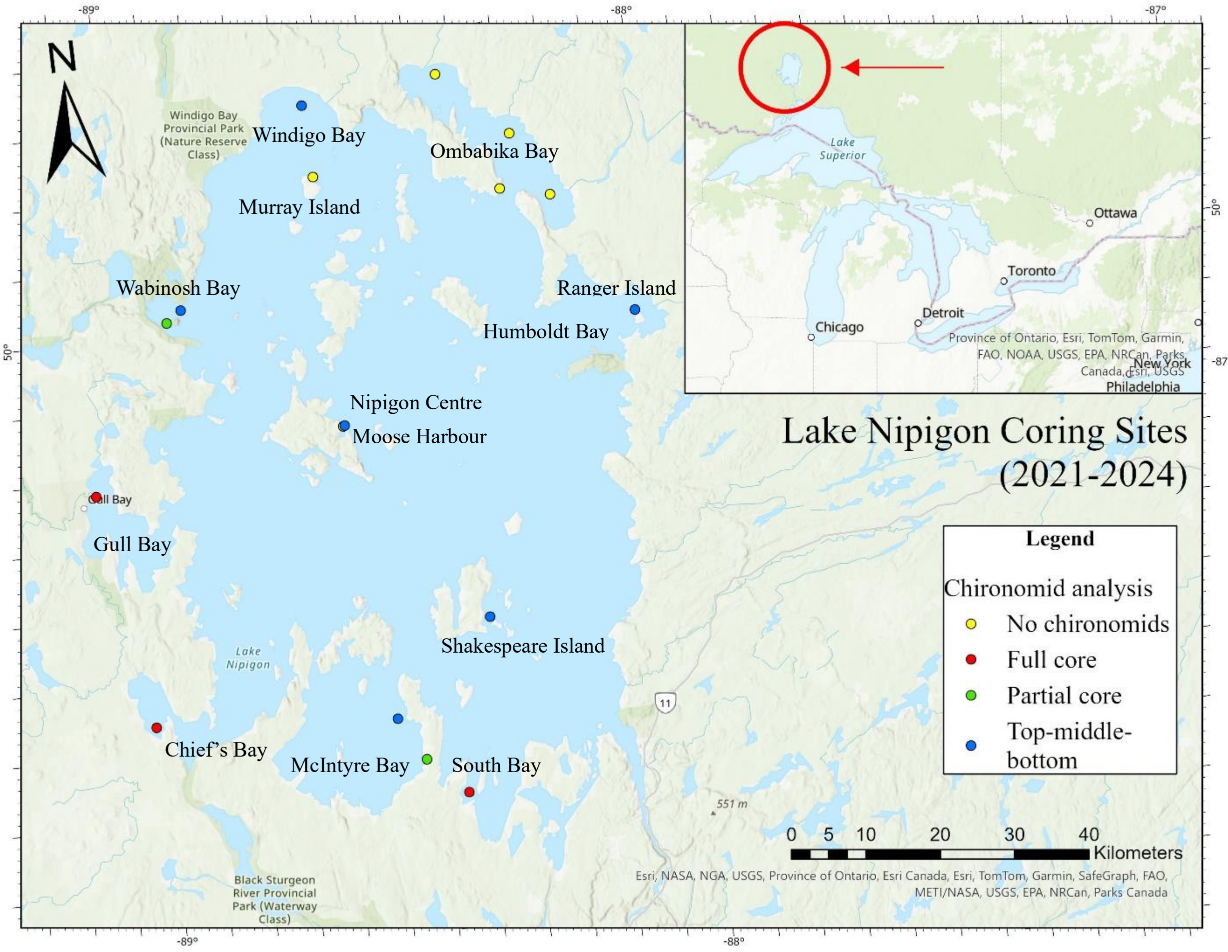

**Figure 4.1**. Sediment coring locations in nearshore areas of Lake Nipigon between 2021-2024. Chironomid analysis categories (full-core analysis, partial core analysis, “top-middle-bottom” survey, and no chironomid remains) are indicated by colour (see section 4.2). The inset map shows Lake Nipigon’s location, north of the entire Great Lakes system.

**Figure 4.2.** Sediment cores collected using a UWITEC™ gravity core on (a) a small fishing tugboat on Nipigon Centre and (b) the *Velma Linda* on Ombabika Bay.

**Table 4.1**. Sediment cores collected around Lake Nipigon with their respective lengths.

| Core | Sampling Date | Core Length (cm) |
|---|---|---|
| McIntyre Bay | June 21, 2021 | 37.0 |
| Chief's Bay | June 23, 2021 | 25.0 |
| Gull Bay | June 24, 2021 | 24.0 |
| Wabinosh Bay | June 25, 2021 | 40.0 |
| Ombabika Bay 1-1 | August 15, 2022 | 17.5 |
| Ombabika Bay 2 | August 15, 2022 | 12.5 |
| Ombabika Bay 3 | August 15, 2022 | 14.0 |
| Windigo Bay 2 | August 16, 2022 | 18.0 |
| Humboldt Bay – Ranger Island | July 31, 2023 | 24.0 |
| Ombabika Bay 4 | July 31, 2023 | 44.5 |
| Murray Island | August 1, 2023 | 59.5 |
| Nipigon Centre | August 3, 2023 | 26.5 |
| Wabinosh Bay 2 | July 24, 2024 | 40.0 |
| McIntyre Bay 2 | July 25, 2024 | 14.5 |
| South Bay 1 | July 25, 2024 | 52.0 |
| South Bay 2 | July 25, 2024 | 15.0 |
| Shakespeare Island | July 26, 2024 | 14.5 |

### 4.2 Lab Methods

All Whirl-Pak bags were weighed upon return to York University. Subsamples of every second centimetre across all the cores were freeze-dried and grounded using a mortar and pestle to achieve a uniform consistency. Approximately ~2g of freeze-dried sediment was measured in labelled gamma tubes, and sealed and stopped with a small, rubber septum. Gamma tubes were sent to the Paleoecological Environmental Assessment and Research Lab at Queen's University (Kingston, Ontario, Canada) for $^{210}$Pb dating using an Ortec High Purity Germanium Gamma Spectrometer (Oak Ridge, TN, USA) Certified reference materials were obtained from the International Atomic Energy Association (Vienna, Austra) for efficiency corrections. Sediment core chronologies were established using the constant rate of supply (CRS) model (Appleby and Oldfield, 1978) and verified using $^{137}$Cs radioisotopes.

Chironomid head capsules were isolated from the wet sediment matrix using standardized paleolimnological procedures outlined by Walker (2001) and Quinlan & Smol (2001). Initially, 1–2 grams of wet sediment were processed from the top, middle, and bottom intervals of each core to assess head capsule abundance. For cores with sufficient material, larger subsamples ranging from 4–12 grams (Table 4.2) were analyzed to allow for full core reconstruction. Full core analyses were conducted for Chiefs Bay, Gull Bay, and South Bay (Figure 4.1), with the amount of sediment processed per interval varying (Table 4.1) to meet the recommended minimum of 50 head capsules per sample (Quinlan & Smol, 2001a). Sediment subsamples were weighed, deflocculated in 10% potassium hydroxide (KOH), heated to 70–80°C for 20–30 minutes, and sieved through a 100-μm mesh. Head capsules were then hand-picked under a dissecting microscope using a Bogorov counting tray.

All cores were initially sampled for a top-middle-bottom survey. Those that yielded sufficient head capsules were selected for full core analysis. Some cores yielded insufficient head capsule densities across multiple intervals and were thus not suitable for full core analysis. In a few cases, initial top-middle-bottom surveys suggested potential for full analysis; however, intervening intervals failed to produce adequate head capsule counts, even after processing up to 10 grams of sediment (Table 4.2). For cores selected for full analysis, chironomid head capsules were sampled at 2-cm intervals from the surface to 10 cm depth, and every 4 cm thereafter, to enhance resolution in the upper sediments and better capture recent ecological changes. The chironomids were permanently mounted on slides using Entellan®. The microscope slides were identified under the Leica DM2500 LED microscope with 200x and 400x magnification and bright field optics, using the taxonomy of Brooks et al. (2007) and Andersen et al. (2013).

**Table 4.2**. Sediment weights (g) of the top-middle-bottom surveys and the full core analyses conducted on the lake sediment cores collected around Lake Nipigon. Clicker count indicates the number of chironomid head capsules individually picked, where one click represents a half head capsule and two clicks represents a full head capsule.

| **Core** | **Depth/Interval** | **Weight of Sediment Used (g)** | **Clicker Count** |
|---|---|---|---|
| Ombabika Bay 1-1 | 0.0-0.5 | 1.23 | 0 |
| | 10.0-10.5 | 1.12 | 0 |
| | 19.0-19.5 | 1.19 | 0 |
| Ombabika Bay 2 | 0.0-0.5 | 1.08 | 0 |
| | 6.0-6.5 | 1.2 | 0 |
| | 12.0-12.5 | 1.17 | 0 |
| Ombabika Bay 3 | 0.0-0.5 | 1.21 | 0 |
| | 6.0-6.5 | 1.03 | 0 |
| | 13.5-14.0 | 1.06 | 0 |
| Nipigon Centre (NC) | 0.0-0.5 | 1.62 | 3 |
| | 13.0-13.5 | 1.88 | 1 |
| | 26.0-26.5 | 1.10 | 0 |
| Ranger Island (RAN) | 0.0-0.5 | 1.08 | 2 |
| | 12.0-12.5 | 1.07 | 0 |
| | 23.5-24.0 | 1.11 | 0 |
| Murray Island | 0.0-0.5 | 1.13 | 0 |
| | 30.0-30.5 | 1.02 | 0 |
| | 59.0-59.5 | 1.09 | 0 |
| Ombabika Bay 4 | 0.0-0.5 | 1.23 | 0 |
| | 22.0-22.5 | 1.08 | 0 |
| | 44.0-44.5 | 1.16 | 0 |
| Wabinosh Bay (WAB) | 0.0-1.0 | 3.03 | 11 |
| | 18.0-19.0 | 5.02 | 8 |
| | 20.0-21.0 | 3.06 | 78 |
| | 22.0-23.0 | 3.17 | 58 |
| | 24.0-25.0 | 3.10 | 28 |
| | 38.0-39.0 | 3.06 | 13 |
| Windigo Bay-2 (Win-2) | 0.0-0.5 | 3.06 | 18 |
| | 8.0-8.5 | 3.04 | 5 |
| | 17.0-17.5 | 3.08 | 2 |
| Gull Bay (GB) | 0.0-1.0 | 4.07 | 200 |

| | | | |
|---|---|---|---|
| | 1.0-2.0 | 4.99 | 208 |
| | 2.0-3.0 | 6.25 | 168 |
| | 3.0-4.0 | 5.33 | 125 |
| | 4.0-5.0 | 6.38 | 98 |
| | 6.0-7.0 | 6.05 | 138 |
| | 8.0-9.0 | 5.06 | 107 |
| | 11.0-12.0 | 4.64 | 138 |
| | 14.0-15.0 | 4.53 | 212 |
| | 17.0-18.0 | 7.60 | 131 |
| | 20.0-21.0 | 4.00 | 150 |
| | 22.0-23.0 | 4.71 | 156 |
| Chiefs Bay (CB) | 0.0-1.0 | 8.85 | 97 |
| | 1.0-2.0 | 10.15 | 22 |
| | 2.0-3.0 | 10.25 | 30 |
| | 3.0-4.0 | 10.65 | 86 |
| | 4.0-5.0 | 10.19 | 44 |
| | 6.0-7.0 | 12.28 | 16 |
| | 8.0-9.0 | 10.10 | 30 |
| | 12.0-13.0 | 11.43 | 27 |
| | 15.0-16.0 | 10.18 | 19 |
| | 18.0-19.0 | 10.27 | 19 |
| | 21.0-22.0 | 10.46 | 17 |
| | 24.0-25.0 | 10.83 | 23 |
| McIntyre Bay (MB) | 0.0-1.0 | 7.59 | 104 |
| | 1.0-2.0 | 8.06 | 63 |
| | 2.0-3.0 | 8.44 | 59 |
| | 3.0-4.0 | 3.08 | 10 |
| | 4.0-5.0 | 4.00 | 3 |
| | 5.0-6.0 | 5.28 | 30 |
| | 6.0-7.0 | 5.17 | 22 |
| | 7.0-8.0 | 5.63 | 26 |
| | 8.0-9.0 | 5.47 | 33 |
| McIntyre Bay 2 (MB2) | 0.0-0.5 | 5.03 | 5 |
| | 7.0-7.5 | 6.12 | 0 |
| | 14.0-14.5 | 2.01 | 0 |
| Shakespeare Island (SI) | 0.0-0.5 | 2.62 | 11 |
| | 10.0-10.5 | 2.48 | 0 |
| | 21.0-21.5 | 2 | 0 |
| Wabinosh Bay 2 (WAB2) | 0.0-0.5 | 3.07 | 11 |
| | 20.0-20.5 | 3.45 | 3 |

| | | | |
|---|---|---|---|
| | 39.5-40.0 | 3.21 | 5 |
| South Bay 1 (SB1) | 0.0-0.5 | 7.29 | 176 |
| | 1.0-1.5 | 4.35 | 127 |
| | 2.0-2.5 | 4.29 | 129 |
| | 3.0-3.5 | 7.31 | 158 |
| | 4.0-4.5 | 8.3 | 111 |
| | 5.0-5.5 | 7.45 | 145 |
| | 6.0-6.5 | 5.58 | 166 |
| | 7.0-7.5 | 9.26 | 132 |
| | 8.0-8.5 | 8.11 | 153 |
| | 9.0-9.5 | 7.84 | 130 |
| | 10.0-10.5 | 10.9 | 125 |
| | 12.0-12.5 | 8.21 | 153 |
| | 14.0-14.5 | 10.56 | 144 |
| | 16.0-16.5 | 10.17 | 134 |
| | 18.0-18.5 | 5.28 | 182 |
| | 20.0-20.5 | 3.82 | 139 |
| | 22.0-22.5 | 5.41 | 159 |
| | 24.0-24.5 | 3.2 | 134 |
| | 26.0-26.5 | 4.11 | 191 |
| | 30.0-30.5 | 2.83 | 174 |
| | 33.0-33.5 | 2.7 | 146 |
| | 36.0-36.5 | 2.74 | 175 |
| | 39.0-39.5 | 2.57 | 166 |
| | 42.0-42.5 | 2.78 | 139 |
| | 45.0-45.5 | 3.11 | 143 |
| | 48.0-48.5 | 6.26 | 196 |
| | 51.5-52.0 | 3.72 | 311 |
| South Bay 2 (SB2) | 0.0-0.5 | 3.06 | 87 |
| | 7.0-7.5 | 3.08 | 93 |
| | 14.5-15.0 | 4.08 | 91 |

### 4.3 Data Analysis

The relative abundances of chironomid taxa that reached at least 5% of the whole assemblage per interval were plotted as stratigraphies. For Gull Bay and South Bay (full-core analysis), the major stratigraphic groups of the relative abundance of chironomids were explored using a constrained incremental sum of squares (CONISS) analysis (Grimm, 1987). CONISS determinations were compared with a broken-stick model of random distributions to determine which groups are important, whether they occurred by random chance alone or otherwise (Bennett, 1996). All analyses were performed using the "riojaPlot: Stratigraphic diagrams in R" package version 0.1-24 (Juggins, 2025) using R version 4.4.1 (R Core Team, 2024) and RStudio 2023.06.0+421 "Mountain Hydrangea" Release (RStudio, 2023).

# CHAPTER FIVE: RESULTS

## 5.1 Top-Middle-Bottom Survey and Partial Core Analysis

A total of 14 sediment cores across Lake Nipigon were surveyed for chironomid subfossil abundance in a "top-middle-bottom" analysis (Figure 5.1). Chironomid remains were not recovered in the initial "top-middle-bottom" survey for Nipigon Centre, Ranger Island, Murray Island, and all four cores from Ombabika Bay. At least some remains were recovered for the remaining 7 sediment cores (Figure 5.1).

McIntyre Bay (MB), located on the south section of Lake Nipigon, was initially thought to be a candidate for a full core analysis, but many of its intervals did not yield enough head capsules. As a result, only the first 9 intervals were processed and mounted onto slides, and among those, only the top 3 intervals yielded count sizes larger than 10 remains .Species recovered most frequently included *Dicrotendipes nervosus*-type (DO optima 4.144 mg/L; Quinlan & Smol, 2001a), *Chaetocladius*, Tanytarsini (obscured), and *Stempellina* (DO optima 4.82 mg/L; Quinlan & Smol, 2001a).A second core, labelled as McIntyre Bay-2 (MB2), was collected in 2024. No remains were recovered from the middle interval (7.0-7.5 cm) and bottom interval (14.0-14.5). In the top interval (0.0-0.5 cm), one remain was recovered the taxon *Cricotopus*-*Orthocladius* (DO optima 3.86 mg/L; Quinlan & Smol, 2001a).

For Windigo Bay-2 (Win-2), 3 chironomid remains were recovered in the top (0.0-0.5 cm) interval (*Chaetocladius*, *Ablabmesyia*, and *Procladius*) and 1.5 remains were recovered from the middle (8.0-8.5 cm) interval (Chironomini (obscured) and *Chironomus anthracinus*-type). No remains were recovered at the bottom interval of 17.0-17.5 cm. The middle interval of Win-2 was composed of Chironomini (obscured) recovered at an abundance of 33%. Chironomini (obscured) is a category that represents broken and/or worn-down head capsules from the Tribe

Chironomini that cannot be identified further down to genus or species level. These head capsules could (1) have missing morphological features or (2) poor preservation quality.

Wabinosh Bay (WAB), located on the northwestern section of Lake Nipigon, was thought to be another ideal candidate for a full core analysis. Similar to McIntyre Bay, however, most of its intervals did not yield enough head capsules to characterize the assemblage. In the top interval (0.0-1.0 cm), 4.5 remains were recovered, belonging to *Glyptotendipes* (DO optima 3.313 mg/L; Quinlan & Smol, 2001a), Orthocladiinae (obscured), *Synorthocladius* (DO optima 4.093 mg/L; Quinlan & Smol, 2001a), and *Ablabmesyia.* In the 18.0-19.0 cm interval, 4.5 remains were recovered belonging to Chironomini (obscured), *Chironomus anthracinus*-type (DO optima 3.271 mg/L; Quinlan & Smol, 2001a), *Chaetocladius*, and *Cricotopus*-*Orthocladius* (DO optima 3.860 mg/L; Quinlan & Smol, 2001a). Remains were more abundant in the 20.0-21.0 cm and 22.0-23.0 cm intervals (38.5 and 21 remains, respectively. At 20.0-21.0 cm, *Dicrotendipes notatus*-type (DO optima 4.144 mg/L; Quinlan & Smol, 2001a), Orthocladiinae (obscured), *Synorthocladius* (DO optima 4.093 mg/L; Quinlan & Smol, 2001a), Tanytarsini (obscured) were recovered at abundances ≥10%. There were 9 other genera recovered at abundances <10%. In the interval 22.0-23.0 cm, *Endochironomus* (DO optima 2.425 mg/L; Quinlan & Smol, 2001a), Orthocladiinae (obscured), *Corynoneura* (DO optima 4.154 mg/L; Quinlan & Smol, 2001a), and Tanytarsini (obscured) were recovered at abundances ≥10%. There were 5 other genera recovered at abundances <10%. In the interval 24.0-25.0 cm, 9.0 remains belonging to *Dicrotendipes nervosus*-type (DO optima 4.144 mg/L; Quinlan & Smol, 2001a), *Dicrotendipes notatus*-type (DO optima 4.144 mg/L; Quinlan & Smol, 2001a), Orthocladiinae (obscured), *Corynoneura* (DO optima 4.154 mg/L; Quinlan & Smol, 2001a), and Tanytarsini (obscured), and *Paracladius* were recovered. In the bottom interval of 38.0-39.0 cm, 2.0 remains belonging to

*Glyptotendipes* (DO optima 3.313 mg/L; Quinlan & Smol, 2001a) and *Paratanytarsus* were recovered. A second core, labelled as Wabinosh Bay 2 (WAB2), was collected in 2024. No chironomids were recovered at the bottom interval of 39.5-40.0 cm. In the top interval 0.0-0.5 cm, 2.5 remains from *Cricotopus-Orthocladius* (DO optima 3.86 mg/L; Quinlan & Smol, 2001a) and Tanytarsini (obscured) were recovered, while 2.0 remains from Orthocladiinae (obscured) were recovered at 20.0-20.5,

Shakespeare Island, one of the largest islands within Lake Nipigon, was cored in its nearshore area in the southeastern section of the lake. No chironomid remains were recovered at the middle interval of 10.0-10.5 cm and the bottom interval of 21.0-21.5 cm. The top interval of 0.0-0.5 cm had 3.5 remains belonging to *Psectrocladius* (DO optima 4.34 mg/L; Quinlan & Smol, 2001a), Tanytarsini (obscured), and *Micropsectra* (DO optima 6.06 mg/L; Quinlan & Smol, 2001a).

South Bay 2 (SB2), located on the southeastern section of Lake Nipigon, had more than 30 remains recovered in the top-middle-bottom intervals, including *Cricotopus-Orthocladius* (DO optima 3.86 mg/L; Quinlan & Smol, 2001a) and Tanytarsini (obscured) recovered at abundances of ≥10% across the top (0.0-0.5 cm), middle (7.0-7.5 cm), and bottom intervals (14.5-15.0 cm). There are 9 genera that were recovered at abundances ≤5% across all three intervals.

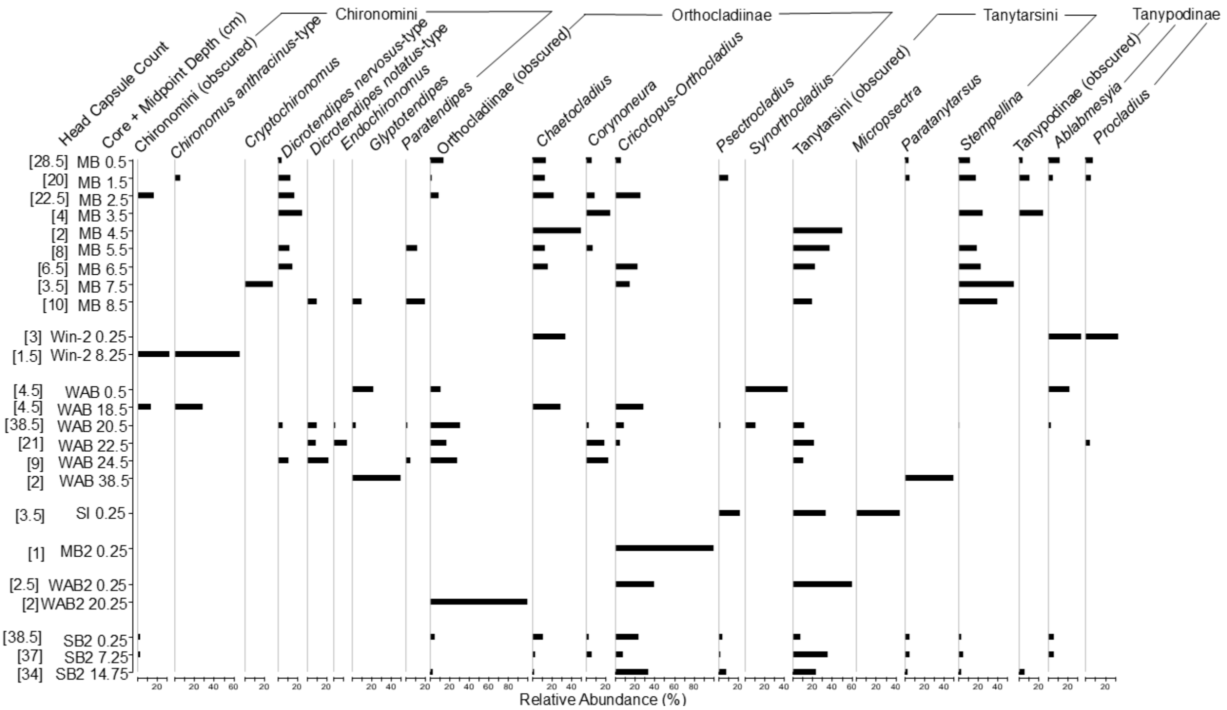

**Figure 5.1.** A relative frequency diagram of chironomid assemblages (as relative abundances) for the top-middle-bottom survey and partial core analysis conducted in Lake Nipigon, for McIntyre Bay (MB), Windigo Bay-2 (Win-2), Wabinosh Bay (WAB), Shakespeare Island (SI), McIntyre Bay 2 (MB2), Wabinosh Bay 2 (WAB2), and South Bay 2 (SB2). The assemblage is grouped into their respective tribes and subfamilies. Tribes and subfamilies with the label (obscured) represent chironomid head capsules that cannot be identified further due to missing or worn-down morphological features like mandibles, antennal pedestals, and ventromental plates. The y-axis represents the cores and their midpoint depths. The head capsule count recovered per core and interval is recorded in brackets. The recommended minimum count is 50 head capsules (Quinlan & Smol 2001a).

**Table 5.1.** A table of observed chironomid taxa in Lake Nipigon sediment cores, their oxygen optima (where data is available) from Quinlan & Smol (2001), and notes on their ecologies from Brooks et al. (2007) and Andersen et al. (2018).

| **Chironomid taxa** | **Oxygen optima (mg/L)** | **Notes on ecology: Brooks et al. (2007)** | **Notes on ecology: Andersen et al. (2018)** |
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| *Ablabmesyia* | N/A | Eurytopic and almost ubiqutuous in lake sediment samples where it can be abundant. Occurs in acidified lakes (Brodin, 1986; Pinder & Morley, 1995; Raddum & Saether, 1981; Buskens, 1987) | *Ablabmesyia* is eurytopic and cosmopolitan: larvae inhabit small and large standing and flowing waters from cold temperate to warm tropical climatic zones. Late instar larvae are predatory, perhaps especially on smaller chironomid larvae and oligochaetes which are engulfed. Lentic larvae live in shallow water as well as in deeper regions of lakes. Individual species may occur very varied habitats, and may include tolerant and sensitive species to acidity and humic content. |
| *Chaetocladius* | N/A | Most species of *Chateocladius* occur in flowing water or amongst vegetation in seepages. However, some do occur in lakes particularly in arctic and subarctic regions. The genus is distributed throughout the | Larvae of *Chaetocladius* are found in wet leaves, among plants and in mud in springs, wells, streams, ditches, sewage plants, ponds, and permanent and temporary pools. Most species can be characterized as semi-aquatic, but a |

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| | | Holarctic (Cranston et al., 1983) | few are truly aquatic and most commonly found in high mountains and in arctic and subarctic areas. |
| *Chironomus* | 3.27 | Abundant in warm, eutrophic lakes, although can occur in arctic lakes; mostly confined to the profundal but may also be present in the littoral; tolerant of low oxygen concentrations and even anoxia for a few weeks; tolerant of low pH (Henrikson et al., 1982; Brodin, 1986; Johnson & Wiederholm, 1989) and high salinity (Vallenduuk & Moller Pillot, 2002); opportunitistic, often an early colonizer after significant environmental change where it may occur in suboptimal conditions (Brooks, 1997); detrivores and filter-feeders; usually associated with soft sediments. | Larvae of *Chironomus* graze on detritus or are filter-feeders, predominantly in soft sediments of standing water, more rarely in flowing water. Perhaps all construct and dwell within tubes in or on the substrate. Has a worldwide distribution. |
| *Cladopelma laterlis*-type | 2.52 | A littoral taxon occurring in muddy and sand/gravel substrates (Hoffmann, 1984; Walker & MacDonald, 1995). A warm stenotherm (Walker et al., 1991) | Some species of *Cladopelma* are eurytopic and pollution tolerant. Larvae occur in streams and larger rivers, lakes, ponds, as well as brackish waters and hot |

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|  |  | but not particularly tolerant of high nutrient conditions, typical of mesotrophic lakes (Saether, 1979; Brodin, 1986) | springs. Worldwide distribution |
| *Corynoneura* | 4.15 | The genus is common in lake sediment samples. It occurs throughout the Holarctic. | *Corynoneura* larvae are found in nearly all types of aquatic habitats from standing water in ditches to fast flowing mountain streams. Because of their small size, they have been frequently overlooked and their distribution under-recorded. |
| *Cricotopus-Orthocladius* | 3.86 | At the genus level, *Cricotopus-Orthocladius* is eurytopic and can be found in flowing and standing waters. The genera are particularly ubiqutuous in lake sediment samples and are frequently common. Most lentic taxa between both genera are associated with temperate, relatively eutrophic conditions. | Larvae of *Cricotopus* inhabit all types of freshwater and to a lesser extent saline coastal and inland waters. They are frequently associated with aquatic plants, inclduing algae, and some mine living parts of aquatic macrophytes. *Orthocladius* occurs in all zoogeographical areas and inhabits all types of flowing waters. |
| *Cryptochironomus* | 4.65 | Occurs in the profundal, in nutrient rich waters (Kansanen, 1985; Saether, 1979) and apparently can be acidophilic (Halvorsen, 1999). | Larvae of *Cryptochironomus* occur in various substrata in lakes, small streams, large rivers. Worldwide in distribution. About |

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| | | Larvae occur in a variety of substrates in the littoral and sublittoral. | 30 species are named from the Holarctic. |
| *Dicrotendipes* | 4.14 | Occurs in the littoral of lentic environments (Pinder & Reiss, 1983) where it is often associated with macrophytes (Moller Pillot & Buskemd, 1990; Brodersen et al., 2001) and mesotrophic to eutrophic waters (Broden, 1986). Most temperature-based training sets indicate that *Dicrotendipes* is thermopilic. | Inhabit littoral sediments of standing waters and may be common in lentic habitats. Worldwide distribition, from the tropics to temperate zones. About 30 species are known from the Holarctic. |
| *Eukiefferiella-Tvetenia* | 3.33 | These genera are eurythermic and occur in flowing waters and the surf zones of lakes (Cranston et al., 1983; Lindegaard, 1992). They frequently occur in lake sediment samples although they are seldom abundant. | Larvae of *Eukiefferiella* are eurythermic and primarily inhabit flowing water of all types. *Tvetenia* occur in flowing water. Most species prefer cold, swift-flowing, well-oxygenated streams (Bode, 1983). |
| *Endochironomus* | 2.42 | Generally found in littoral sediments of mesotrophic and eutrophic lentic water bodies (Pinder & Reiss, 1983; Brodin, 1986) | Occur in "Aufwuchs" of living and dead substrata in almost all types of still water or mine in leaves and stems of macrophytes with a tendency to preferential occurrence in small, eutrophic standing waters. Distributed primarily in |

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| | | | temperate regions of the Holarctic. |
| *Glyptotendipes* | 3.31 | Occur in the littoral of warm (Walker & MacDonald, 1995), detritus-rich (Pinder & Reiss, 1983), mesotrophic to eutrophic lakes (Brundin, 1949; Brodin, 1986; Brooks et al., 2001), are often associated with macrophytes which are mined by many species of the genus (Brodin, 1982; Buskens, 1987; Vallenduuk, 1999; Brodersen et al., 2001) | Occurs in detritus-rich littoral sediments and in "Aufwuchs" of lakes, ponds, small water bodies and (less often) running water. A considerable proportion of species are "semi-miners" under the bark of submerged branches in colonies of Bryozoa and sponges or floating leaves and stems of submerged plants. |
| Heterotrissocladius | 6.97 | Species of *Heterotrissocladius* occur in the profundal of cold oligotrophic lakes (Brundin, 1956; Saether, 1979; Brodin, 1986; Walker & MacDonald, 1995). There are four motphotaxa described with different thermal optima, which makes them valuable indicators of climate change. | Larvae of *Heterotrissocladius* are found in the littoral to profundal zones of lakes and some species also in streams, springs, rivers, ponds, or puddles. The members of the *subpilosus* group and the *maeri* group are typical of ultraoligotrophic or strongly oligotrophic lakes. Also most members of the *marcidus* group are most common in oligotrophic lakes. |

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| Heterotrissocladius grimshawi-type | 6.97 | *H. grimshawi*-type occurs in the coldest lakes. | N/A |
| *Micropsectra radialis*-type | 6.06 | This is a cold stenothermic taxa and typically lives in ultraoligotrophic, cold arctic or alpine lakes where it may be abundant and dominate the chironomid fauna. This taxon often appears in lakes at lower altitudes during the Younger Dryas (Brundin, 1956; Brodin, 1986) | *Micropsectra* have been recorded from a wide range of habitats, inclduing hygropetric situations, thermal springs, moorland- and temporary pools. They are particularly characteristic of muddy deposits in slack regions of streams and small rivers and of mesotrophic-oligotrophic lakes, including the profundal zone. Many species are cold stenothermic and are recorded in or near coldwater springs. |
| Microtendipes pedellus-type | 3.87 | Common in littoral and sublittoral sediments of lentic waters (Pinder & Reiss, 1983), often in coarse sediment, low in organics (Hofmann, 1984; Brodersen & Lindegaard, 1999b). Often abundant in Holocene and Lateglacial sediments where it is an indicator of intermediate temperatures in northern Europe (Brooks & Birks, 2001). | Larvae of *Microtendipes* inhabit littoral and sublittoral sediments of larger bodies of still water; sporadically also in "Aufwuchs", also in sediments and submerged mosses in running water. |

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| *Neozavrelia* | N/A | It is a cold stenothermic and occurs in calcareous waters in montane regions (Pinder & Reiss, 1983; Ekrem, 2006) | Found in various habitats and at least some species appear to be cold stenothermic. Larvae are recorded from streams, rivers, lakes, and peat pits as well as in hygropetric situations. The genus is known from all geographical locations except Africa and the Neotropics (Ekrem, 2006). |
| Paracladius | N/A | Larvae of *Paracladius* are usually found during cold episodes in lake sediment cores and are present in some of the coldest lakes in Scandinavian training sets. As such, the genus is a useful indicator of cold, oligotrophic conditions in lake sediment samples where it may be abundant in these conditions (Walker et al., 1991). | Larvae of *Paracladius* inhabit springs and bodies of all sizes of flowing and still water. |
| *Paratanytarsus* | N/A | May be abundant in warm or cold lakes. | A eurytopic genus and has been recorded from a variety of habitats such as brackish ponds, cool streams, lakes, rivers, reservoirs, and marches. |
| *Paratanytarsus austriacus*-type | N/A | Occurs in cold oligotrophic lakes at | N/A |

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| | | high latitude or altitude. | |
| *Polypedilum* | 4.32 | *Polypedilum* is an indicator of temperate climatic conditions. It occurs in the littoral (Hoffmann, 1984) of eutrophic lakes (Brundin & Gransberg, 1993; Brodin, 1982; Brodin, 1986; Klink, 2002), and some species occur amongst vegetation. | Occur in all still and flowing waters, with exception of the arctic and high mountains. Sediments are the preferred substratum. The genus has a worldwide distribution, with several hundred species. |
| *Procladius* | N/A | Abundant in most sediment samples from mesotrophic to eutrophic lakes (Brodin & Gransberg, 1993; Brodin, 1986; Brundin, 1949), although may be excluded from the coldest lakes. *Procladius* occurs in fine sediments in the profundal (Kansanen, 1985; Brodin, 1986; Saether, 1979) and so may be elimated during periods of anoxia (Heiri & Lotter, 2003). *Procladius* will tolerate acidification (Brodin, 1986; Il'yashuk & Il'yashuk, 2000). *Procladius* is carnivorous. | Larvae of the majority of *Procladius* species prefer muddy substrata of standing or slowly flowing water bodies, especially ponds and small lakes. A few species also inhabit the profundal zone of large, deep lakes. |
| *Psectrocladius* | 4.34 | This genus is common and often abundant in lake sediment samples. This genus is usually associated with | *Psectrocladius* is eurytopic with a worldwide distribution, perhaps with the exception of the Australasian and |

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| | | temperate lakes (Brundin, 1949). | Antarctic regions. All subgenera and species groups occur in the Holarctic region. |
| *Stempellina* | 4.82 | A warm stenotherm (Brundin, 1949; Walker et al., 1991) most abundant in oligotrophic lakes (Brundin, 1949; Brodin, 1986). | Larvae of *Stempellina* occur in unpolluted springs and small streams as well as in lakes. Larvae construct straight transportable cases of fine sand, silt, often speckled with detritus, which are slightly wider at the anterior end. Worldwide distribution with 18 named species, most of which are found in the Holarctic (Ekrem, 2007). |
| *Synorthocladius*+ | 4.09 | The genus is not usually abundant in lake sediment samples. The genus occurs in streams and the littoral of lakes throughout the Holarctic (Cranston et al., 1983). | Larvae of *Synorthocladius* inhabit springs, small to large bodies of flowing water and small bodies or shallower parts of still water. |
| *Tanytarsus* | 4.71 | The genus occurs in the littoral of relatively warm, productive lakes and will tolerate acidic conditions (Brodin, 1986; Uutala, 1986; Bilyj & Davies, 1989). | A eurytopic genus, occurring in all types of types of freshwater, with some marine and at least 1 terrestrial species. The freshwater species usually construct long, soft tubes that are fixed to the bottom substrate. |

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| *Tanytarsus chinyensis*-type | 4.53 | A cold stenotherm from oligotrophic lakes (Brodin, 1986). | N/A |

## 5.2. Full Core Analyses

### *5.2.1 $^{210}$Pb dating*

Radioisotopic activity of $^{210}$Pb reached background at 4.5 cm for Chiefs Bay, and the oldest date returned by the CRS model was be 1974±10 years at 2.5 cm (Figure 5.2). A $^{137}$Cs peak was observed at 2.5 cm, with a predicted date of ~1963, which was roughly consistent with the CRS age-depth model. Sedimentation rate (inferred based on the CRS model) declined with depth until 2.5 cm, the limit of the CRS model. Dates cannot be extrapolated beyond this depth due to inconsistent sedimentation rates.

Radioisotopic activity of $^{210}$Pb declined with sediment core depth for Gull Bay, with background reached at 7.0 cm (Figure 5.2). The oldest date returned by the CRS model was 1956±66 years at 4.5 cm (Figure 5.2). A $^{137}$Cs peak (~1963) was observed at 3.5 cm, roughly consistent with the CRS age-depth model. CRS-inferred sedimentation rate declined with depth.

$^{210}$Pb radioactivity in South Bay declined with depth between 0-5 cm where it reached background levels (Figure 5.3). Unsupported $^{210}$Pb was observed again at 35 cm, 42 cm, and 52 cm, which the CRS model infers as a large sedimentation event between 5-30 cm (Figure 5.3). The oldest date returned by the CRS model to be 1924±10 years at 52 cm (Figure 5.3). There was no $^{137}$Cs peak observed for the core.

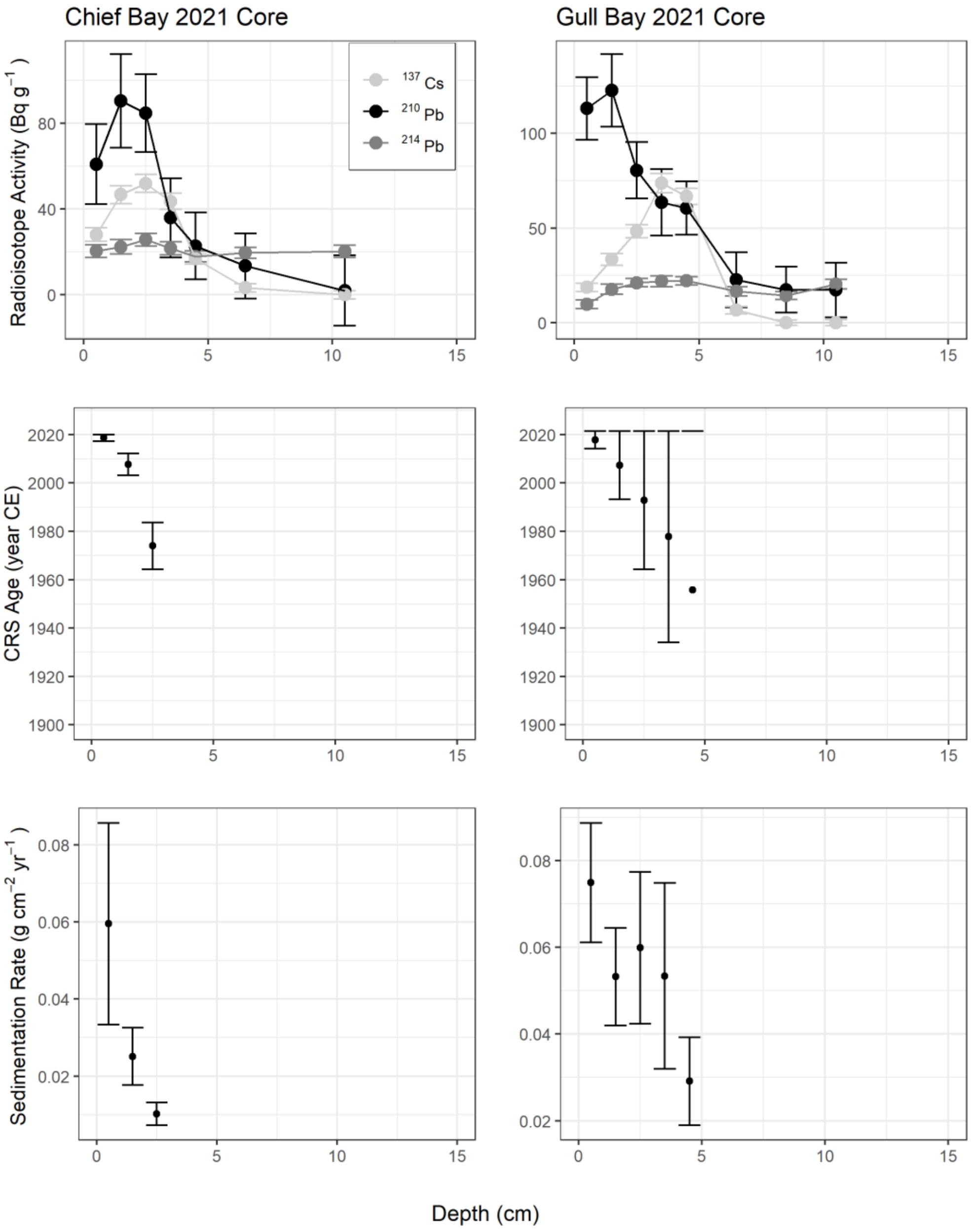

**Figure 5.2** Results of $^{210}$Pb and $^{137}$Cs dating for the sediment cores collected from Chief's Bay and Gull Bay. CRS Age is determined from the Constant Rate of Supply (CRS) model. Depth (cm) is the sediment core depth, where increasing core depth is read as increasing sediment age.

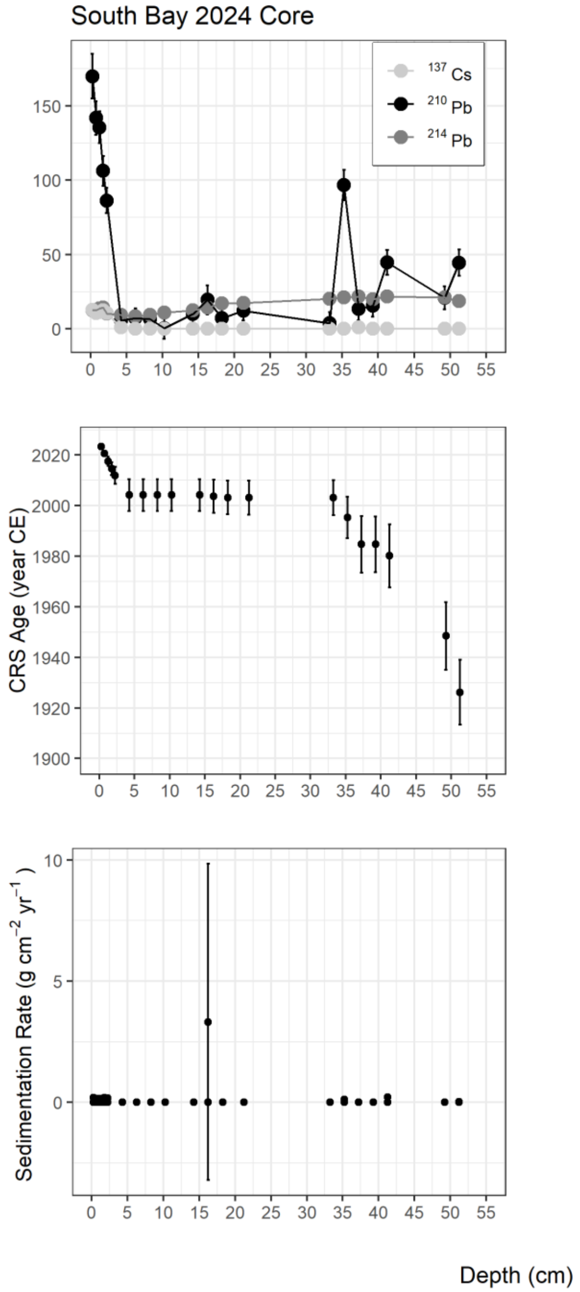

**Figure 5.3.** Results of $^{210}$Pb and $^{137}$Cs dating for the sediment core collected from South Bay. CRS Age is determined from the Constant Rate of Supply (CRS) model. Depth (cm) is the sediment core depth, where increasing core depth is read as increasing sediment age.

### *5.2.2 Subfossil Chironomids - Chiefs Bay*

All intervals sampled in Chiefs Bay did not reach the minimum headcount of 50 capsules recommended by Quinlan & Smol (2001). A stratigraphic diagram of assemblage changes was plotted with head capsule counts and $^{210}$Pb inferred dates as secondary y-axes (Figure 5.4). Head capsule count ranged between 2-27.5 chironomids per interval, even after processing 8-10 grams of wet sediment per interval. The highest head capsule count of 27.5 was observed at the surface interval of 0-1 cm, while the lowest head capsule count of 2 was observed at 20-21 cm. As such, the relative abundance results should be interpreted with caution, and inferences about changes over time are unreliable.

There was a total of 19 chironomid taxa found in Chiefs Bay (Figure 5.3). *Chaetocladius* and *Stempellina* (DO optima 4.82 mg/L; Quinlan & Smol, 2001a) are two taxa that were frequently recovered, with abundances ranging between 9-50% and 5-44% respectively. The rest of the assemblage was composed of 17 genera with abundances ≤20%, including 8 that were recovered at abundances greater than 5% in two or more intervals (Chironomini (obscured), *Polypedilum*, Orthocladiinae (obscured), *Corynoneura*, *Cricotopus-Orthocladius*, Tanytarsini (obscured), *Paratanytarsus*, and Tanypodinae (obscured). Subfamilies labelled as obscured represent head capsules that are not identifiable further down to genus or species level due to (1) missing morphological features or (2) poor preservation quality. The following taxa were only recovered in 1-3 intervals, at relative abundances <10% and are not discussed further: *Chironomus* (DO optima 3.271 mg/L; Quinlan & Smol, 2001a), *Cladopelma lateralis*-type (DO optima 2.524 mg/L; Quinlan & Smol, 2001a), *Dicrotendipes* (DO optima 4.144 mg/L; Quinlan & Smol, 2001a), *Glyptotendipes* (DO optima 3.313 mg/L; Quinlan & Smol, 2001a), *Paracladius*, *Psectrocladius* (DO optima 4.34 mg/L; Quinlan & Smol, 2001a), *Neozavrelia*, *Paratanytarsus* *austriacus*-type, *Tanytarsus* (DO optima 4.706 mg/L; Quinlan & Smol, 2001a), and *Ablabmesyia*.

Relative abundance of *Chaetocladius* exhibited considerable variation throughout the 25-cm core. There were 16.0 remains recovered for the whole core. There were 2.0 remains recovered at the bottom of the core, while no remains were encountered between 21-22 cm. There were 0.5-3.5 remains recovered from 0-19 cm. No *Stempellina* (DO optima 4.82 mg/L; Quinlan & Smol, 2001a) remains were encountered at the bottom of the core, and it first appeared between 18-19 cm with 1.5 remains recovered. No remains were recovered between 15-16 cm. Between 1.0-2.0 remains were recovered between 0-13 cm, except between 5-6 cm where no remains were recovered. There was a total of 12 *Stempellina* recovered for the whole core.

*Cricotopus-Orthocladius* (DO optima 3.86 mg/L; Quinlan & Smol, 2001a) was not recovered between 15-25 cm, and was first recovered 12 cm with 1 head capsule. No remains were recovered between 6-9 cm. Between 1-3.5 head capsules were recovered between 0-5 cm, except at 2-3 cm, where no remains were recovered. There was a total of 9.5 *Cricotopus-Orthocladius* recovered for the whole core. *Polypedilum* (DO optima 4.325 mg/L; Quinlan & Smol, 2001a) was recovered at the bottom of the core with 1.0 head capsule, while no remains were recovered between 9-24 cm. 1.0 head capsule was recovered per interval between 0-9 cm, except at 1-2 cm and 6-7 cm, where no remains were recovered. There was a total of 6 *Polypedilum* head capsules recovered throughout the core.

*Corynoneura* (DO optima 4.154 mg/L; Quinlan & Smol, 2001a) was not recovered at the bottom of the core (4.0 total remains). 1.0 remain was recovered at 18-19 cm, while 0.5 remain was recovered at 15-16 cm. No remains were recovered between 6-13 cm. 1.0 remain each was recovered at 4-5 cm and 3-4 cm. No remains were recovered between 1-3 cm. 0.5 remain was recovered at 0-1 cm.

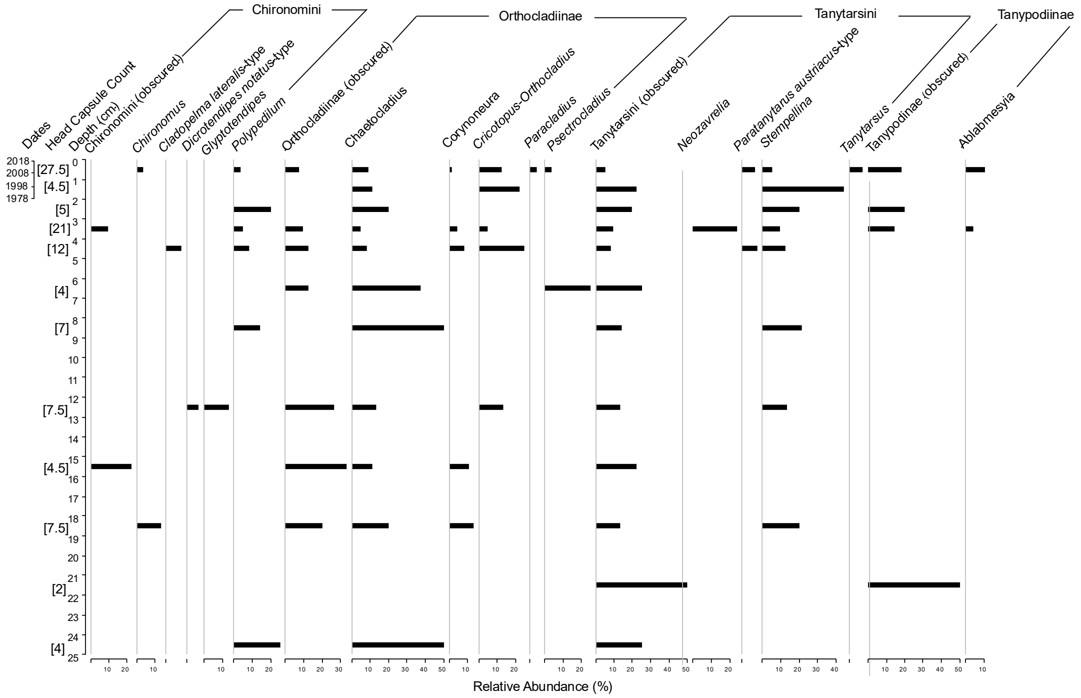

**Figure 5.4.** A stratigraphic diagram of chironomid assemblage reaching ≥5% relative abundance in a core collected from Chiefs Bay of Lake Nipigon (northwestern Ontario, Canada). Head capsule counts for the intervals sampled are indicated as a secondary y-axis. All intervals did not reach minimum headcount of 50 capsules recommended by Quinlan & Smol (2001). As such, these results should be interpreted with caution. $^{210}$Pb-inferred dates are shown on the left of the secondary y-axis. Chironomids from different genus and species names under the tribes Chironomini and Tanytarsini, and the subfamilies Orthocladiinae and Tanypodiinae are represented throughout the core. Tribes and subfamilies with the label (obscured) represent chironomid head capsules that cannot be identified further due to missing or worn-down morphological features like mandibles, antennal pedestals, and ventromental plates.

### *5.2.3 Subfossil Chironomids - Gull Bay*

There was a total of 20 chironomid taxa found in Gull Bay (Figure 5.4). All intervals sampled reached the minimum head count requirements recommended by Quinlan & Smol (2001). The assemblage is primarily made up of *Cricotopus-Orthocladius* and *Ablabmesyia,* with abundances ranging 5-17.5% and 5-16.5% respectively. The rest of the assemblage is composed of 18 genera with abundances ≤15%, including 15 that were recovered at abundances greater than 5% in two or more intervals. The CONISS analysis and broken stick model detected 6 major groups. Relative abundances of *Cricotopus-Orthocladius* (Subfamily: Orthocladiinae; DO optima 3.86 mg/L; Quinlan & Smol, 2001a) remained at 10% at the bottom of the core from 14-23 cm. Abundance decreased <10% between 8-12 cm. Peak abundance of 17.5% is reached at 6.5 cm. Abundance declined to 2% between 3-5 cm. The taxon remains stable between the range of 5-7% abundance from 0-3 cm. Relative abundance of *Ablabmesyia* (Subfamily: Tanypodiinae) remained between 5-9% at the bottom of the core between 17-23 cm. Peak abundance of 16.5% is reached at 14 cm. Abundances remained between 8-11% from 4-12 cm. An abundance of 3% was recovered at 3.5 cm. Abundances slightly increased and remained around 8% from 0-3 cm.

Three taxa from the tribe Chironomini were identified, including *Cryptochironomus*, *Dicrotendipes*, and *Polypedilum*. *Cryptochironomus* (DO optima 4.65 mg/L; Quinlan & Smol, 2001aa) was not recovered at the bottom of the core between 21-23 cm. It is present at an abundance of 5% at 20.5 cm. No remains were recovered between 15-20 cm. It reappeared at an abundance of 5% at 14.5 cm. No remains were recovered between 9-14 cm. It reappeared at an abundance of 5% at 6.5 cm. It reached a peak abundance of 9% at 3.5 cm. It declined to trace abundances of 1.5% between 1-3 cm. No remains were recovered at the surface interval. *Dicrotendipes* (DO optima 4.144 mg/L; Quinlan & Smol, 2001a) was recovered at abundances ranging from 1-3% between 17-23 cm. No remains were recovered between 9-17 cm. It reappeared at an abundance of 5% at 8.5 cm. No remains were recovered at 6.5 cm. The abundance remains stable at abundances ranging from 1.5-3% from 0-5 cm. *Polypedilum* (DO optima 4.325 mg/L; Quinlan & Smol, 2001a) was recovered at abundances 1-3% at the bottom of the core. It reached steady abundances of 4-7% between 8-17 cm. The abundance slightly declined to 1.9% at 6.5 cm. It was recovered at abundances of 4-7% between 3-5 cm. It declined to 1.6% at 2.5 cm. Peak abundance of 10.4% was recovered at 1.5 cm, then it declined to 1.6% at the surface interval.

In the subfamily Orthocladiinae, there are 6 other taxa (*Chaetocladius, Corynoneura, Cricotopus-Orthocladius, Eukiefferiella-Tvetenia, Heterotrissocladius,* and *Psectrocladius*) with subtle changes throughout the core. Orthocladiinae (obscured), represents head capsules that are cannot be identified further down to the genus or species level due to (1) missing morphological features or (2) poor preservation in the sediment record. This group was recovered at trace abundances of 1-4% at the bottom of the core, and 0.5-5% between 3-14 cm. No remains were recovered at 17 cm. Peak abundance of 6.4% was recovered at 2 cm. The relative abundance declined slightly to 3.5% at 1 cm, then increased to 6.3% at the surface level. *Chateocladius* was recovered at a peak abundance of 15% at the bottom of the core. Abundances ranged between 7-13% between 11-20 cm. No remains were recovered at 8 cm. Remains were recovered at ~8% between 4-6 cm. The relative abundances declined to 2% between 2-4 cm. Peak relative abundance of 12.5% was recovered at 1 cm. No remains were recovered at the surface interval. *Corynoneura* was recovered at abundances ranging from 5-14.5% between 2-23 cm, with peak relative abundance observed at 11 cm. Relative abundance declined to 2% at 1 cm, then increased to 11% at the surface interval. *Cricotopus-Orthocladius* (DO optima 3.86 mg/L;

Quinlan & Smol, 2001a) was recovered at abundances of 5-12% between 8-23 cm, with peak abundance of 17.6% observed at 6 cm. Abundances increased from 2% to 8% from 4 cm to the surface interval. *Eukiefferiella-Tvetenia* (DO optima 3.331 mg/L; Quinlan & Smol, 2001a) was not recovered between 14-23 cm. It was recovered at 2% at 11 cm, and 9% at 8 cm, and at 7-8% abundance between 1-4 cm. *Heterotrissocladius* (DO optima 6.971 mg/L; Quinlan & Smol, 2001a) was only recovered at abundances ≤5% in 3 intervals across the core. *Psectrocladius* (DO optima 4.34 mg/L; Quinlan & Smol, 2001a) was recovered at abundances of 1-8% between 11-23 cm, with peak abundance of 12.5% recorded at 8 cm. Abundance declined to 3% at 6 cm, then increased to 11% at 4 cm, with abundances ranging between 1-8% from 0-4 cm.

In the tribe Tanytarsini, Tanytarsini (obscured) represents head capsules that are cannot be identified further down to the genus or species level due to (1) missing morphological features or (2) poor preservation in the sediment record. Peak abundance of 10% was recovered at the bottom 23 cm and again at 6 cm. Relative abundances ranged between 3-7% between 8-21 cm, and remained between 2-6% between 0-5 cm. *Micropsectra radialis*-type (DO optima 6.06 mg/L; Quinlan & Smol, 2001a) was not recovered at the bottom of the core or the top 12 cm of the core. It reached a peak abundance of 5% at 17 cm and 20 cm. Remains were recovered at 1% at 14 cm. *Neozavrelia* was recovered at a peak abundance of 12% at the bottom of the core, and was recovered at abundances ranging from 3-5% between 3-21 cm, except at 4 cm and 8 cm where no remains were recovered. No remains were also recovered between 1-3 cm, but an abundance of 3% was recorded in the surface interval. *Paratanytarsus* was recovered at abundances of 1-4% between 4-23 cm, except at 6 cm, 17 cm, and 20 cm where no remains were recovered. Peak abundance of 8% was recovered at 3 cm. Abundances ranged between 1-3% from 0-3 cm. *Stempellina* (DO optima 4.82 mg/L; Quinlan & Smol, 2001a) was recovered at abundances of 1-5% between 20-23 cm. Peak abundance of 9% was recovered at 17 cm. Abundances between 1-7% were recovered between 1-15 cm, except at 2 cm, where no remains were recovered. No remains were also recovered at the surface interval. *Tanytarsus* (DO optima 4.706 mg/L; Quinlan & Smol, 2001a) was recovered at abundances of 3-5% between 3-12 cm, except at 3 cm, 8 cm, 14 cm, and 20 cm where no remains were recovered. Peak abundance of 8% was recovered at 2 cm. Relative abundances remained steady at 5% between 0-2 cm. *Tanytarsus chinyensis*-type (DO optima 4.529 mg/L; Quinlan & Smol. 2001a) was not recovered at the bottom of the core. It was first recovered at an abundance of 1% at 14 cm. No remains were recovered between 1-12 cm. Peak abundance of 6% was recovered at the surface interval.

In the subfamily Tanypodinae, Tanypodinae (obscured) represents head capsules that are cannot be identified further down to the genus or species level due to (1) missing morphological features or (2) poor preservation in the sediment record. Tanypodinae (obscured) was recovered at abundances ranging between 3-9% between 11-23 cm. No remains were recovered at 8 cm. Abundances ranged between 4-8% from 1-7 cm. Peak abundance of 11% was recovered at the surface interval. *Ablabmesyia* was recovered at abundances of 5-9% between 17-23 cm. Peak abundance of 16% was recorded at 14 cm. Abundances ranged between 8-11% between 4-12 cm. The relative abundance declined to 4% at 3 cm, then increased to 8% from 0-3 cm. *Procladius* was recovered at abundances of 3-7% between 17-23 cm. Abundances declined to 1% at 14 cm, and remained between 3-5 % from 8-12 cm. No remains were recovered at 6 cm. Remains were recovered at abundances of 6% at 3 cm and 4 cm. Peak abundance of 11% was recovered at 2 cm. Abundances of 5% was recovered at 1 cm, and 1% at the surface interval.

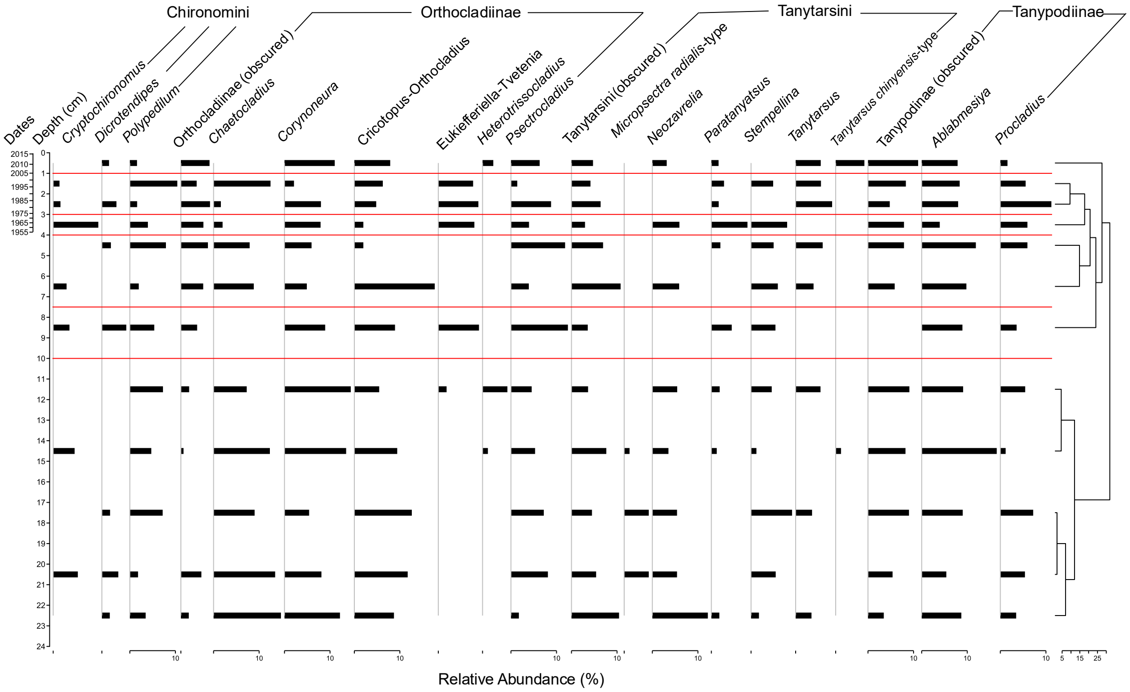

**Figure 5.5.** A stratigraphic diagram of chironomid assemblages (as relative abundances) in a core collected from Gull Bay of Lake Nipigon (northwestern Ontario, Canada). Results from the Constrained Incremental Sum of Squares (CONISS) cluster analysis (Grimm, 1987) of Gull Bay is shown on the far right. $^{210}$Pb-inferred dates are shown on the left as a secondary y-axis. Chironomids from different genus and species names under the tribes Chironomini and Tanytarsini, and the subfamilies Orthocladiinae and Tanypodiinae are represented throughout the core. Tribes and subfamilies with the label (obscured) represent chironomid head capsules that cannot be identified further due to missing or worn-down morphological features like mandibles, antennal pedestals, and ventromental plates.

### *5.2.4 Subfossil Chironomids - South Bay*

There were a total of 16 chironomid taxa found in South Bay. All intervals sampled reached the minimum head count requirements recommended by Quinlan & Smol (2001). *Heterotrissocladius grimshawi*-type and *Stempellina* are two taxa that dominated the core, with relative abundances ranging between 15-37% and 1-21% respectively. The rest of the assemblage is composed of 14 genera with abundances ≤10%, including 10 that were recovered at abundances greater than 5% in two or more intervals (*Dicrotendipes, Microtendipes pedellus*-type, *Chaetocladius, Cricotopus-Orthocladius, Paracladius, Psectrocladius,* Tanytarsini (obscured), *Paratanytarsus, Tanytarsus,* and *Tanytarsus chinyensis*-type. Six major groups were detected by the CONISS broken stick model (Figure 5.6).

In the tribe Chironomini, there are three taxa with notable changes. *Chironomus* was recovered at abundances of 1-5% throughout the core, except between 1-7.5 cm, 14 cm, 22 cm, 26 cm, 36 cm, 42 cm, and 45 cm, where no remains were recovered. *Dicrotendipes* (DO optima 4.144 mg/L; Quinlan & Smol, 2001a) was recovered at abundances of 2-3% between 45-52 cm. No remains were recovered at 42 cm. Abundances of 2-7% were recovered between 26-39 cm. No remains were recovered at 24 cm. Abundances of 3-7% were recovered between 12-22.5 cm. Peak abundance of 8% was recovered at 8 cm and 10 cm, with the 9 cm interval in between at an abundance of 1%. Abundances of 1-5% were recovered at the upper section of the core 0-7.5 cm. *Microtendipes pedellus*-type (DO optima 3.866 mg/L; Quinlan & Smol, 2001a) was recovered at abundances of 1-6% between 30-52 cm. Peak abundance of 9% was recovered at 9 cm. Abundances ranged between 1-4% between 22-24.5 cm. Peak abundance of 9% was again recovered at 20 cm. Abundances of 4-6% were recovered 14-18.5 cm. No remains were recovered at 12 cm. Abundances of 0.8-7% were recovered from 0-11.5 cm.

In the subfamily Orthocladiinae, there are six taxa with notable changes. Orthocladiinae (obscured) represents head capsules that are cannot be identified further down to the genus or species level due to (1) missing morphological features or (2) poor preservation in the sediment record. No remains were recovered at the bottom of the core. Remains were recovered in abundances of 3-4% between 42-48.5 cm. No remains were recovered at 39 cm. Remains were recovered at 1-2% between 30-36.5 cm. No remains were recovered at 26 cm. Peak abundance of 7% was recovered at 24 cm. No remains were recovered between 16-22.5 cm, except for 20 cm where remains were recovered at a trace abundance of 0.9%. Remains were recovered at 1-2% abundance 12-14.5 cm. No remains were recovered between 8-10.5 cm. Remains were steadily recovered at abundances of 2-4% from 0-7.5 cm, except for 5 cm where no remains were recovered. *Chaetocladius* was recovered at abundances of 2-11% throughout the core from 0-52 cm. Peak abundance was recorded at 15% at 2 cm. *Cricotopus-Orthocladius* (DO optima 3.86 mg/L; Quinlan & Smol, 2001a) was recorded at a peak abundance of 15% at the bottom of the core. Abundances of 5-9% were recovered at 26-48.5 cm. No remains were recovered at 24 cm. Abundances of 1-8% were recovered at 8-22.5 cm. No remains were recovered at 7 cm. Abundances of 1-7% were recovered between 0-6.5 cm. *Heterotrissocladius grimshawi*-type (DO optima 6.97 mg/L; Quinlan & Smol, 2001a) dominated the core at abundances of 14-37%, with peak relative abundance recovered at 42 cm. *Paracladius* was not recovered at the bottom of the core. Abundances of 1-4% were recovered between 14-42.5 cm, except at 14 cm, 16 cm, 22 cm, and 33 cm, where no remains were recovered. Peak abundance of 9% was recovered at 12 cm. Remains were recovered at abundances of 1-5% between 4-10.5 cm, except at 6 cm, 8 cm, and 9 cm, where no remains were recovered. No additional remains were recovered at the top of the core from 0-3.5 cm. *Psectrocladius* (DO optima 4.34 mg/L; Quinlan & Smol, 2001a) was recovered at abundances of 1-3% between 48-52 cm. No remains were recovered at 45 cm. Remains were recovered between 1-8% between 16-42.5 cm. No remains were recovered at 14 cm. Peak abundance of 11% was recovered at 12 cm. Abundances ranging between 1-9% were recovered from 1-10.5 cm. No remains were recovered at the surface interval.

In the tribe Tanytarsini, there are five taxa that show subtle changes throughout the core. Tanytarsini (obscured) represents head capsules that are cannot be identified further down to the genus or species level due to (1) missing morphological features or (2) poor preservation in the sediment record. Tanytarsini (obscured) was recovered at higher abundances of 15-33% for most of the core between 8-52 cm. Lower abundances of 9-15% were recovered at the upper section of the core from 0-7.5 cm. *Paratanytarsus* was recovered at abundances of 1-7.5% between 24-52 cm. No remains were recovered at 22 cm. Remains were recovered at abundances of 1-9% from 0-20.5 cm, with peak abundance observed at 2 cm. *Stempellina* (DO optima 4.82 mg/L; Quinlan & Smol, 2001a) was recovered at abundances of 1-10% at the lower part of the core between 10-52 cm. Higher abundances ranging between 5-21% were recovered at the upper part of the core between 0-9.5 cm, with peak abundance recorded at 8 cm. *Tanytarsus* (DO optima 4.706 mg/L; Quinlan & Smol, 2001a) was recovered at abundances of 1-5% between 26-52 cm, except at 36 cm and 48 cm, where no remains were recovered. Peak abundance of 11% was recovered at 18 cm. Abundances of 1-7% were recovered at the upper section of the core from 0-22 cm, except at 16 cm and 20 cm, where no remains were recovered. *Tanytarsus chinyensis*-type (DO optima 4.529 mg/L; Quinlan & Smol, 2001a) was not recovered at the bottom of the core between the 42-52 cm. Remains were recovered at 1-2% between 26-39.5 cm, except for 33 cm and 36 cm, where no remains were recovered. No remains were also recovered at 24 cm. Peak abundance of 6.5% was recovered at 22 cm. Abundances were recovered at 1-5% between 7-20.5 cm, except at

9 cm, where no remains were recovered. No remains were also recovered between 1-6.5 cm. An abundance of 3% was recovered at the surface level.

In the subfamily Tanypodinae, Tanypodinae (obscured) represents head capsules that are cannot be identified further down to the genus or species level due to (1) missing morphological features or (2) poor preservation in the sediment record. Tanypodinae (obscured) was recovered at abundances of 1-3% between 33-52 cm, except at 48 cm. where no remains were recovered. No remains were also recovered between 7-26.5 cm, except at 12 cm, where remains were recovered at 1.5%. Remains were recovered at abundances of 1-5% between 0-6.5 cm, except at 1 cm, where no remains were recovered. *Ablabmesyia* was recovered at trace abundances of 1% from 36-52 cm, except at 39 cm, 45 cm, and 48 cm, where no remains were recovered. No remains were also recovered at 33 cm. Peak abundance of 7% was recovered at 24 cm and 26 cm. No remains were recovered at 24 cm. Relative abundances ranging from 1-3% were recovered at the upper section of the core from 0-22.5 cm, except at some intervals 4-6.5 cm, 9 cm, and 14-18.5 cm, where no remains were recovered.

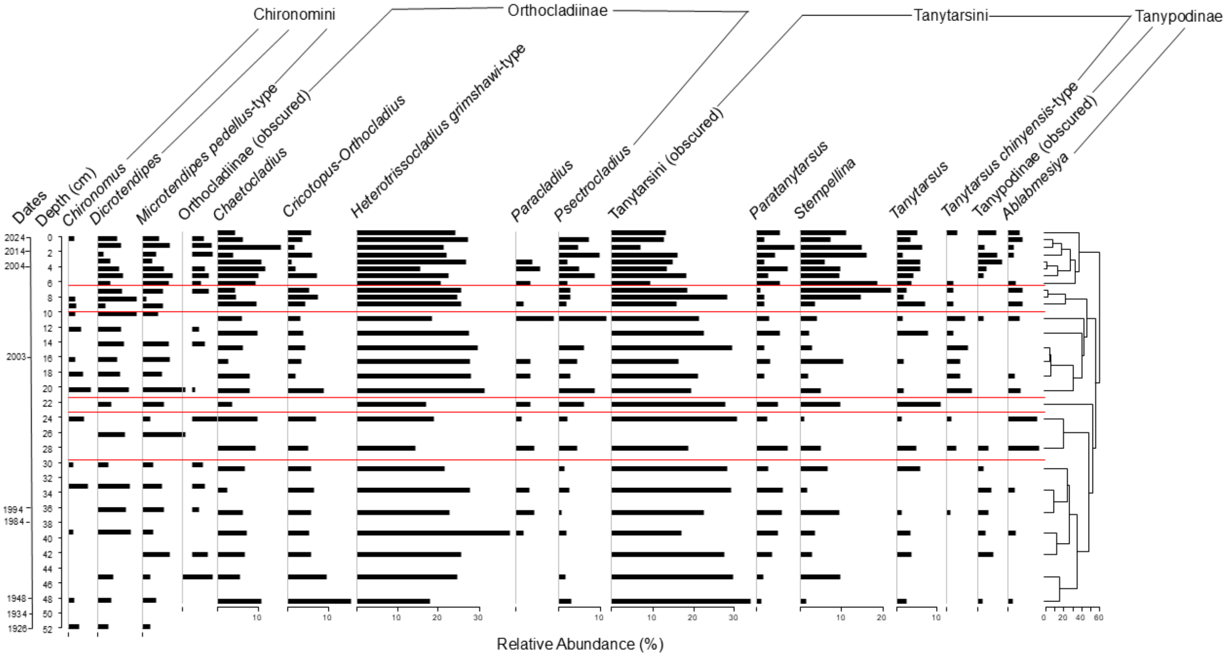

**Figure 5.6.** A stratigraphic diagram of chironomid assemblages (as relative abundances) in a core collected from South Bay of Lake Nipigon (northwestern Ontario, Canada). Results from the Constrained Incremental Sum of Squares (CONISS) cluster analysis (Grimm, 1987) of South Bay is shown on the far right. $^{210}$Pb-inferred dates are shown on the left as a secondary y-axis. Chironomids from different genus and species names under all subfamilies Chironomini, Orthocladiinae, Tanytarsini, and Tanypodiinae are represented throughout the core. Tribes and subfamilies with the label (obscured) represent chironomid head capsules that cannot be identified further due to missing or worn-down morphological features like mandibles, antennal pedestals, and ventromental plates.

# CHAPTER SIX: DISCUSSION

## 6.1 The Chironomidae of Lake Nipigon

This thesis provides the first assessment of chironomid assemblages in Lake Nipigon, as the original survey of benthic fauna in Lake Nipigon did not identify Chironomidae to the species-level (Adamstone and Harkness, 1923). Species recovered most frequently include *Chaetocladius, Dicrotendipes* nervosus-type, *Cricotopus-Orthocladius*, and *Glyptotendipes.* Species recovered sporadically include *Stempellina*, *Micropsectra*, *Synorthocladius*, and *Procladius.*

Several taxa were consistently abundant across multiple sites, providing valuable insights into prevailing environmental conditions. *Chaetocladius* was one of the most frequently recovered genera, found in McIntyre Bay, Windigo Bay-2, and Wabinosh Bay. It typically inhabits vegetated littoral zones and tolerates a wide range of environmental conditions (Brooks et al., 2007; Andersen et al., 2013). *Dicrotendipes nervosus*-type was another commonly occurring taxon, especially in McIntyre Bay and Wabinosh Bay. It is a thermophilic littoral taxon with global distribution, often associated with warmer, lentic environments (Brooks et al., 2007; Andersen et al., 2013). Similarly, *Cricotopus-Orthocladius*, a eurytopic and widespread genus, was found in Wabinosh Bay and South Bay, reflecting generalist conditions across various aquatic habitats (Brooks et al., 2007). *Paratanytarsus*, also eurytopic, was recovered from Wabinosh Bay and South Bay, further supporting the abundance of taxa that thrive in varied ecological niches (Andersen et al., 2013). *Glyptotendipes*, a warm-adapted genus common in mesotrophic to eutrophic lakes (Walker & MacDonald, 1995), was found in both the upper and lower intervals of Wabinosh Bay. The frequent presence of these broadly distributed, generalist taxa suggests that many nearshore areas of Lake Nipigon support habitats that are moderately warm, vegetated, and somewhat nutrient-rich, with assemblages reflecting intermediate dissolved oxygen levels.

Less frequently occurring taxa were only found at some bays. *Stempellina* was found in McIntyre Bay and South Bay, and though not dominant, its presence is noteworthy as it is a warm stenotherm typically associated with oligotrophic lakes—implying subtle warming trends or persistently low nutrient conditions (Brundin, 1949). *Micropsectra*, found only in Shakespeare Island, is a cold stenotherm with a high oxygen requirement, often restricted to ultraoligotrophic environments (Brodin, 1986). Its local abundance suggests that nearshore waters at Shakespeare Island remain cold and well-oxygenated. *Synorthocladius*, recovered in small numbers in Wabinosh Bay, is typically rare in sediment records but indicates littoral habitats when present (Andersen et al., 2013). *Procladius*, found in Windigo Bay-2, is more associated with mesotrophic to eutrophic lakes but may decline during anoxic events, highlighting episodic oxygen stress (Brodin, 1986; Brundin, 1949). These rarer taxa, while limited in abundance, provide critical evidence of localized environmental gradients and can indicate warming, oxygenation status, or nutrient shifts that may not be apparent from dominant taxa alone.

The chironomid assemblage from Chiefs Bay should be interpreted with caution since only 2-27.5 head capsules were recovered at all sampled intervals. Two mid-oxygen tolerant taxa, *Chaetocladius* and *Stempellina*, were more frequently recovered than others. *Stempellina* has a dissolved oxygen (DO) optima near 4.82 mg/L (Quinlan & Smol, 2001), suggesting that Chiefs Bay historically maintained moderate oxygen levels atypical of oligotrophic or mesotrophic shield lakes (Smol, 2008). The fluctuating presence of *Chaetocladius*, from its higher abundance at the bottom of the core, to its absence mid-core, and its slight resurgence in upper intervals, may reflect variable oxygen availability or organic inputs, potentially driven by changes in lake mixing regimes, catchment vegetation, or hydrological conditions (Lotter et al., 1998; Brodersen & Lindegaard, 1999). The presence of *Polypedilum* (DO optima 4.325 mg/L; Quinlan & Smol, 2001a), *Corynoneyra* (DO optima 4.154 mg/L; Quinlan & Smol, 2001a), and *Cricotopus-Orthocladius* (DO optima 3.86 mg/L; Quinlan & Smol (2001a) in multiple core zones further supports the inference of generally oxygen-rich benthic conditions (Quinlan & Smol, 2001a). These genera are often found in well-oxygenated littoral habitats, and are responsive to changes in macrophyte abundance, sediment grain size, and detrital food availability (Rosenberg & Resh, 1993; Heiri & Lotter, 2001). Several rare taxa like *Chironomus*, *Cladopelma lateralis*-type, and *Glyptotendipes* are known indicators of low-oxygen or eutrophic conditions (Quinlan & Smol, 2001a; Brooks et al., 2007), but their presence in low abundances in only some zones of the core suggests that severe hypolimnetic oxygen depletion or persistent nutrient enrichment was not a dominant feature of Chiefs Bay during the period represented by the sediment core.

The results from the surveys offer a taxonomical assessment of chironomid assemblages in Lake Nipigon, filling a critical knowledge gap left by earlier surveys in the 1920s that did not identify Chironomidae to the species level (Adamstone & Harkness, 1923). The consistent presence of generalist, moderately tolerant taxa such as *Chaetocladius*, *Dicrotendipes nervosus*-type, *Cricotopus–Orthocladius*, and *Glyptotendipes* suggests that many nearshore habitats in Lake Nipigon are characterized by warm, vegetated, and moderately nutrient-rich conditions, with benthic environments that remain reasonably well-oxygenated. The recovery of more specialized taxa like *Micropsectra* and *Stempellina* further indicates localized gradients in a range of environmental variables, reflecting a complex and heterogeneous nearshore environment.

However, the low abundance of chironomid remains recovered from several sediment cores, particularly from the partial survey and from Chiefs Bay, significantly limits both the temporal resolution and the statistical robustness of ecological interpretations. This scarcity is noteworthy, as large, deep lakes typically preserve abundant chironomid subfossils. For instance, Luo et al. (2023) recovered 2,756 chironomid head capsules from 35 surface sediment samples in Fuxian Lake, a deep (max depth: 158.9 m) oligotrophic lake in China's Yunnan-Guizhou Plateau, using methods from Brooks et al. (2007), with all samples yielding >50 head capsules (Quinlan & Smol, 2001). Similarly, earlier surveys in Lake Nipigon by Adamstone & Harkness (1923) recovered a total of 3,654 chironomid larvae from benthic dredge samples taken between 0 and 20 metres depth, highlighting how the low subfossil yields observed in this study may reflect taphonomic processes, sampling limitations, or sedimentological variability. In addition, the scarcity of remains from the northern part of the lake, an area most affected by the Ogoki diversion, also prevents meaningful evaluation of possible diversion-related ecological changes in that region.

Taken together, these findings offer a valuable, though preliminary, baseline of chironomid diversity and distribution in Lake Nipigon. The lack of dominance by hypoxia-tolerant taxa such as *Chironomus* spp., which are commonly associated with nutrient-enriched and degraded habitats in the lower Great Lakes (Perlov et al., 2023; Alward et al., 2025), suggests that Lake Nipigon's benthic environments remain comparatively intact. Despite the overall scarcity of remains, sediment cores from Gull Bay and South Bay contained enough chironomid material to allow for temporal reconstruction of assemblage changes, offering valuable insights into how benthic habitat conditions have shifted in these specific areas.

## 6.2 Temporal trends in chironomid assemblages in Gull Bay and South Bay

### *6.2.1 Gull Bay*

The chironomid assemblage recovered from the Gull Bay sediment core reflects a largely stable benthic community, with minimal signs of ecological disruption across the sediment profile. Twenty taxa were identified, and while six CONISS zones (Figure 5.5) suggest minor stratigraphic variability, these transitions were gradual and lacked clear compositional shifts. This consistency suggests that Gull Bay has maintained relatively stable environmental conditions over time, with only subtle ecological responses to external stressors.

Key taxa such as *Cricotopus–Orthocladius*, a genus associated with temperate, eutrophic conditions (Brooks et al., 2007), and *Ablabesmyia*, a eurytopic taxon that occurs in acidified lakes (Brodin, 1986) remained consistently present, exhibiting modest fluctuations of 5-9% relative abundance throughout the core rather than abrupt changes. Slight mid-core increases in *Cricotopus–Orthocladius* and *Eukiefferiella–Tvetenia*, a genus that prefers cold, well-oxygenated streams (Bode, 1983), may reflect short-term fluctuations in oxygen availability, though these shifts could also be influenced by other environmental factors. The slight increase of 5-8% relative abundance of low-oxygen-tolerant and eutrophic taxa in upper intervals—such as *Polypedilum*, an indicator of eutrophic, temperate lakes (Brooks et al., 2007; Brodin, 1986), *Dicrotendipes*, a thermophilic taxon that is an indicator of mesotrophic to eutrophic conditions (Brooks et al., 2007; Brodin, 1986) and *Procladius*, a taxon that can be eliminated in periods of anoxia and can tolerate acidification (Heiri & Lotter, 2003; Brodin, 1986)—appears episodic and does not indicate a transition to degraded conditions.

Overall, the assemblage composition indicates that Gull Bay has undergone only gradual and minor ecological changes, likely reflecting a response to climate change, rather than pronounced or abrupt environmental shifts. The persistence of a diverse assemblage further supports the interpretation of ecological stability through time.

### *6.2.2 South Bay*

The sediment core from South Bay was recovered at a deeper depth (25m) than the other bays and nearshore areas (10-15 m). The chironomid assemblage in South Bay exhibited more pronounced temporal changes compared to Gull Bay. The assemblage reflected a system historically characterized by well-oxygenated conditions throughout the core, as evidenced by the dominance of *Heterotrissocladius grimshawi*-type. The high abundance of *Heterotrissocladius* indicates the persistence of cold, oxygen-rich bottom waters because this taxon, with a high dissolved oxygen (DO) optimum of 6.97 mg/L (Quinlan & Smol, 2001a), is widely regarded as a sentinel of a well-ventilated hypolimnia. *Heterotrissocladius* is also typical of low-nutrient, temperate Boreal Shield lakes with a well-oxygenated hypolimnia (Quinlan & Smol, 2001a; Larocque et al., 2006; Verbruggen et al., 2011). Its prevalence suggests that South Bay has avoided chronic hypoxia.

Additional oxygen-sensitive Orthocladiinae taxa support this interpretation. *Cricotopus-Orthocladius*, with a DO optimum of 3.86 mg/L (Quinlan & Smol, 2001a), was most abundant in the lower core, reaching 15% at the bottom—evidence of particularly favourable oxygen conditions during earlier periods. The abundance of *Psectrocladius* at 12% at the 12 cm interval, further suggests that well-oxygenated conditions persisted into more recent intervals.

The recent rise of *Stempellina*, alongside the subtle decline of cold-adapted taxa like *Cricotopus-Orthocladius* and a slight tapering of *H. grimshawi*-type toward the surface, suggests a community in transition. *Stempellina* is a taxon often associated with warmer, more productive, and occasionally lower-oxygen environments (Brundin, 1949; Brodin, 1986), exhibited a marked increase in the upper 10 cm, with the highest abundance of 22% at 8 cm. While *Stempellina*'s increase (DO optimum 4.82 mg/L; Quinlan & Smol, 2001a), is typically caused by warming-driven stratification, diminished oxygen penetration, or increased nutrient loading (Brundin, 1949; Brodin, 1986), a wide range of biotic and abiotic factors may contribute to its higher abundance in South Bay. Nonetheless, these shifts align with broader regional climate trends within the Great Lakes system, including warmer air temperatures and reduced ice cover duration over the 20th and 21st centuries (O'Beirne et al., 2017; Sgro & Reavie, 2018; Pilsbury et al., 2021; Alward et al., 2025). Together, these findings suggest that while oxygen conditions in South Bay have generally remained supportive of cold stenotherms, warming-related stratification may now be beginning to drive gradual ecological restructuring in the benthic invertebrate community.

# CHAPTER SEVEN: CONCLUSIONS AND FUTURE WORK

## 7.1 General Conclusions

This core surveys provided the first species-level assessment of chironomid assemblages in Lake Nipigon, filling a longstanding gap in the lake's ecological record since the early 20th-century surveys that did not identify Chironomidae beyond broad taxonomic levels (Adamstone & Harkness, 1923). The consistent recovery of generalist and moderately tolerant taxa, such as *Chaetocladius, Dicrotendipes nervosus*-type, *Cricotopus–Orthocladius*, and *Glyptotendipes*, suggests that many nearshore habitats in Lake Nipigon are moderately warm, vegetated, and nutrient-rich, with generally favourable oxygen conditions. The presence of rarer, more specialized taxa like *Micropsectra* and *Stempellin*a points to localized environmental variability, indicating cold, well-oxygenated waters in some areas and oligotrophic or warming trends in others. While the survey offers a valuable taxonomic baseline, low subfossil abundances—particularly from partial cores and Chiefs Bay limit temporal resolution and restrict the detection of lake-wide spatial patterns. This is notable given that large, deep lakes typically yield abundant chironomid remains, as demonstrated in other systems such as Fuxian Lake (Luo et al., 2023). Additionally, limited data from the northern portion of the lake, where diversion-related impacts may be most pronounced, hinder assessments of anthropogenic influence in that region. Nevertheless, the absence of hypoxia-tolerant taxa like *Chironomus* spp., commonly found in degraded Great Lakes habitats (Perlov et al., 2023; Alward et al., 2025), suggests that Lake Nipigon remains relatively unimpacted by the multiple stressors, with sediment records from Gull Bay and South Bay providing early insight into historical shifts in benthic habitat conditions.

This thesis further investigated subfossil chironomid assemblages preserved in sediment cores from Gull Bay and South Bay in Lake Nipigon to reconstruct past ecological conditions and assess spatial variation in limnological responses to environmental change. Stratigraphic changes in Gull Bay reveal a steady chironomid composition over time. Gull Bay exhibited the highest taxonomic richness (20 taxa). Assemblages show declining abundances of cold- and oxygen-preferring taxa, such as *Heterotrissocladius grimshawi*-type, alongside increases in warm-adapted, low-oxygen-tolerant taxa, such as *Stempellina* and *Tanytarsus*. These changes likely reflect a wide range of biotic and abiotic factors, with regional climate warming and possible shifts in local nutrient dynamics. While South Bay exhibited a slightly lower taxonomic richness (16 taxa), the assemblage showed more pronounced compositional shifts. While deep-water taxa like *Heterotrissocladius grimshawi*-type remained dominant in the lower sections of the core, a distinct rise in warm- and low-oxygen-tolerant taxa in upper intervals—such as *Stempellina*, *Microtendipes pedellus*-type, and *Cricotopus–Orthocladius*—points to recent ecological changes. These changes are consistent with increased thermal stratification and declining hypolimnetic oxygen levels, which may be driven by warming and enhanced productivity from recent land-use changes or nutrient loading. Together, these records suggest that climate-driven stratification and associated oxygen depletion are impacting Lake Nipigon's benthic ecosystems, with some similar trends observed in both full core analyses conducted in Gull Bay and South Bay. In particular, the decline of *H. grimshawi*-type and increase of *Stempellina* in the more recent intervals of South Bay along with the limited presence of *H.* grimshawi-type and the more consistent abundance of *Stempellina* throughout Gull Bay are indicative of changing climatic conditions in multiple areas of the lake, specifically in the west and southeast of Lake Nipigon.

This study provides one of the first paleoecological baselines of Lake Nipigon's ecological change since the 20th century, and the first characterization of the chironomid species assemblage, and offers valuable context for monitoring, management, and restoration. Strengthening partnerships with Indigenous communities, such BZA, will be key to integrating these findings with long-standing place-based knowledge, co-developing monitoring frameworks, and guiding future stewardship of the lake.

## 7.2 Future Directions

To build a more comprehensive record of past environmental change in Lake Nipigon, a multi-proxy paleoenvironmental approach is needed, especially considering the complex sedimentation history of Lake Nipigon in general, and nearshore areas of large lakes in particular. The insights from this study should be strengthened by using complementary paleolimnological indicators such as diatoms, cladocerans, and sediment geochemistry (e.g., C/N ratios, stable isotopes, metals analysis), as well as further inquiries to better resolve the core chronologies. Together, these proxies should provide a more detailed and refined reconstruction of past temperature trends, changes in productivity, and nutrient dynamics to interpret the sedimentary chironomid assemblages.

No remains of chironomids were recovered from the northern portion of the lake, with the exception of a few remains from taxa that are tolerant of low-oxygen conditions in Windigo Bay. Further study is needed to investigate whether this is an indicator of benthic habitat degradation due to inputs of eroded sediment from the Little Jackfish River (Ministry of Natural Resources, 2004). This thesis only analyzed a small amount of sediment from the northern bays, and was not designed to rigorously investigate possible impacts from the Ogoki. Furthermore, future coring sites can focus on more strategic areas where a higher focus should be expended with the aim of recovering a higher chironomid head capsule count size. For example, Windigo Bay can be a potential site for longer coring, with the aim of capturing a longer sediment record. The recovery of a few low-oxygen tolerant remains from the surveys may possibly point to a more productive, eutrophic assemblage, if a successful longer core were to be taken in the future. In addition, Chiefs Bay can be explored further given that the conditions are relatively pristine and there are no documented environmental stressors (Ministry of Natural Resources, 2004). A better site selection, further from any beaches or shallower nearshore areas can be cored to expend a more statistically robust chironomid assemblage representative of the bay's paleoenvironmental history.

Finally, building collaboration with Indigenous knowledge holders and communities is necessary to ensure that scientific interpretations enrich Indigenous place-based knowledge and values. Co-designing monitoring frameworks that include Western science and Indigenous knowledge will be key to informing ongoing restoration and conservation action on Lake Nipigon. The documented decline of cold stenotherms and increase in warmer taxa in more recent intervals may be direct evidence of climate warming affecting the lake. BZA can use these findings to advocate for better monitoring programs and management plans, focusing on how to protect the lake at present and for the future against the ongoing impacts of climate change. These findings, which are the first of its kind after nearly a century of ecosystem change driven by multiple stressors, will help BZA advocate for Lake Nipigon and communicate their management and remediation concerns to multi-government organizations and regional bodies like the International Joint Commission (Syed, 2025).

# CHAPTER EIGHT: REFERENCES

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