ASSESSING LONG-TERM ECOLOGICAL CHANGES IN LAKE SCUGOG (SOUTHERN ONTARIO, CANADA) FROM ~1700 to 2019 USING CLADOCERA (BRANCHIPODA, CRUSTACEA) SUBFOSSIL REMAINS AS PALEOECOLOGICAL INDICATORS

by

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ABSTRACT

Lake Scugog is a shallow impoundment located in southern Ontario that is encountering several anthropogenic stressors, such as the introduction of invasive species, eutrophication, periodic algal blooms, and climate change. This thesis used a paleolimnological approach to assess the long-term (~200 years) ecological changes in the west and east arms of Lake Scugog using Cladocera (crustacean zooplankton, Class Branchiopoda) remains as bioindicators. *Bosmina* and *C. brevilabris* were the dominant cladocerans in Lake Scugog throughout the last several hundred years. Measured *Bosmina* body sizes were small, indicating high fish planktivory pressure on *Bosmina*. The changes in subfossil Cladocera and *Bosmina* in sediments suggest that the stress levels associated with eutrophication, climate change and invasive species have not been large enough to significantly alter predation levels or this part of the zooplankton community in Lake Scugog.

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CHAPTER ONE: GENERAL INTRODUCTION

1.1 Background

Ontario lakes provide a number of ecosystem services, including tourism and recreation opportunities, habitat to support biodiversity, and in some cases provide a source of drinking water. Southern Ontario lakes are mostly found in the vicinity of urbanized locations, and hence receive high nutrient and road salt inputs, disturbances by invasive species, and erosion and siltation. These stressors can often result in negative ecological changes in aquatic ecosystem (Carpenter & Brock, 2006). In order to monitor the health of aquatic ecosystems impacted by multiple stressors, scientists and lake managers examine ecological indicator species that are relatively easy to monitor, such as macro invertebrates, algae, and zooplankton (Lu et al., 2015). Scientists use indicators species as early-warning signals of ecological problems, and as a barometer for trends in ecological resources (Niemi & McDonald, 2004). Historically, ecological indicators of water bodies were simple, such as observing colouration and measuring oxygen content (Rapport, 1992). With time it has developed from visual cues to using concentrations of specific chemicals and specific species to determine the health of ecosystems. Various indicators, such as plants, fish, plankton, benthic macro-invertebrates, and mammals can be used to monitor environmental changes and assess the efficacy of management and remediation practices (Siddig et al., 2016). By studying ecological indicator species, we can determine if a lake system is vulnerable to damage, and take necessary steps to remediate harm before critical ecosystem thresholds are crossed (Yang et al., 2015).

Lake Scugog is impacted by multiple stressors such as eutrophication, invasive species, road salt runoff, and high demand for water usage, with important implications for sport fishing in the lake. A recent 4-year lake-wide monitoring program on Lake Scugog, including water quality, algae, and macrophytes, indicated that Lake Scugog may be experiencing a period of ecological transformation due to climate change and invasion by starry stonewort (Harrow- Lyle & Kirkwood, 2020a). My MSc thesis will complement the findings of the 4-year monitoring program to provide a long-term perspective on ecological change in Lake Scugog over its entire history since it was created following the construction of the Lindsay Dam in 1834, to support lake management decision-making. My research objective is to reconstruct changes in Cladocera assemblage, diversity, and *Bosmina* size structure through time to infer changes in lake habitat quality and fish predation in Lake Scugog. This provides a unique opportunity to use paleolimnological techniques to examine changes in ecological indicator taxa over much longer timelines (centuries to millennia) than would otherwise be possible with traditional monitoring approaches. Insights into lake ecosystem dynamics, including the role that humans have played in altering lake ecosystems, is essential for developing feasible and effective lake management and remediation plans (Renberg et al., 2009). Cladocera are commonly used paleolimnological indicator taxa for assessing food web changes and water quality degradation (Smol, 2008). Lake Scugog is an economic driver in the Scugog Township, and it is crucial to understand threats to water quality posed by climate change, shoreline development, and invasive species.

1.2 Research Objectives

This thesis will use paleolimnological techniques to understand how the interaction of these stressors has driven limnological change in Lake Scugog over the last several hundred years of its history, including tracking the ecological impacts of reservoir formation, eutrophication, invasive species, and food web changes over time.

My objective is to provide a historical perspective on Cladocera assemblages and diversity for Lake Scugog, and how they’re altered by historical events. Another objective is to examine food web changes in Lake Scugog using size measurements of *Bosmina* remains.

CHAPTER TWO: LITERATURE REVIEW

Canada has an abundance of freshwater lakes, rivers, streams, and wetlands, with over 9% of the surface area of the country covered by freshwater (Meybeck, 1995; Lehner & Doll, 2004). Freshwater ecosystems and the riparian habitats are considered to be the most biologically diverse on Earth (Albert et al., 2021). Wetlands are vanishing three times faster than terrestrial ecosystems such as forest, and freshwater vertebrate populations have decreased more than twice as steeply as terrestrial and marine populations (Tickner, 2020). Human have impacted the lakes, rivers, streams, wetland and groundwater, the biodiversity is declining rapidly on every continent and every major river basin on Earth (Albert et al., 2021). These ecosystems are declining worldwide due to hydrological alterations, habitat degradation and loss, overexploitation, invasive species, pollution and the multiple impacts of climate change (Darwall et al., 2018).

These bodies of water are susceptible to a variety of anthropogenic stressors such as deforestation, hydrological alternation by dams, agricultural activities and urbanization (Li, 2021). These lakes provided numerous ecosystem services such as the provision of drinking water, hydrological regulation, habitat for conservation of aquatic and terrestrial biota, and contributions to the local and regional economies, through recreation and fisheries (Schallenberg et al., 2013). They also suppose a great proportion of the global biodiversity by discovering many new species discovered every year, including endemic species that are confined to small ranges of the ecosystem (Abell et al., 2008).

When lakes are subjected to anthropogenic stressors, it alters the lake ecosystems for hundreds of years through land uses changes, water-level manipulation, acidifying emissions, climate change, point-source pollution, and the introduction of non-native species, among others (Saulnier- Talbot, 2016; Quinlan et al., 2008; Palmer & Yan, 2013). In southern Ontario, anthropogenic climate change, eutrophication, urban development, and exotic species invasions are significant threats to lake water quality and the provision of ecosystem services (Harrow-Lyle & Kirkwood, 2020; Winter et al., 2011).

Lake Scugog is located in the Kawartha region of Ontario, a region which has experienced changes in aquatic environment from multiple stressors since the arrival of European settlers in the early 1800s (Mallory, 1994; Adams & Taylor, 2009; Conolly, 2016; Berger, 2018). The major consequence of nutrient enrichment from human activities is enhancing primary production known as cultural eutrophication. Additionally, climate warming and enhanced precipitation can increase primary and secondary producers and other direct and indirect changes associated with climate (e.g changes in ice cover, seasonality, wind intensity) (Woolway et al., 2021). Other stressors like changes in fishery practices and regional deposition of acidifying emissions that can also contribute to changes in the aquatic communities. These stressors are common to lakes around the world, not only in Lake Scugog. It is vital to understand how the lakes are responding to stressors so that we can formulate remediation plans that will help preserve lake water quality.

2.1 Dams and Reservoir Creation

Hydrological alteration from dams and lock systems are a major factor for ecosystem function in Lake Scugog, which is a reservoir system that was created to provide improvements to navigation, irrigation and flood controls, but have resulted in negative changes to the aquatic environment (Li, 2021). Dams lead to changes in water level, and flooding adjacent terrestrial areas that can re-shape the aquatic community structure, terrestrial environment and the landscape around the lake system (Baxter, 1997; French & Chambers, 1997; Paragamian, 2002; Hambright et al., 2004). Deforestation and hydrological alteration can lead to enhanced nutrient loads, further exacerbated by agriculture and industrialization, with impacts on the natural environment (Hasler, 1947; Kira, 1993).

The trophic surge hypothesis (TPH) predicts increased phosphorus loading from the leaching and decomposition of organic matters from new terrestrial areas, causing a period of elevated productivity in the years immediately following reservoir creation (Ostrofsky, 1978; Ostrofsky & Dutchie, 1980; Grimard & Jones, 1982). Based on this hypothesis, newly created reservoirs undergo three main phases: a non-equilibrium trophic surge, a non-equilibrium trophic depression and a new equilibrium phase.

2.2 Cultural Eutrophication of Lakes

Cultural eutrophication is human-mediated nutrient enrichment of water bodies. It is the biggest threat to water quality worldwide, and it causes significant negative effects on water quality as a result of excessive algal or aquatic plant growth (Khan & Ansari, 2005; Smith & Schindler, 2009; Birk et al., 2020). This can lead to harmful cyanobacteria blooms, increased frequencies of anoxic events and associated fish kills, increased costs of water purification, declines in fish and wildlife populations, and loss of recreational amenities (Carpenter, 2005). Within the scientific community, there has been debate over the ultimate cause of excessive algal or plant growth, with the focus on nitrogen and phosphorus as potential limiting nutrients. A limiting nutrient is a substance which is limiting to biological growth in a lake or reservoir due to short supply or availability (Argen et al., 2012).

Nitrogen and phosphorus are essential elements in the ecosystem as plants and animals require them for maintaining growth and metabolic processes (Khan & Ansari, 2005). Phosphorus and nitrogen enter into the lakes through various entry points such as diffuse (non-point) run-off from agricultural or urban lands, and point source sewage and industrial discharges

(Carpenter, 2005). Phosphorus is a chemical element found in the natural world and it controls growth in many ecosystems as limiting nutrients (Pasek et al., 2014). Phosphate mineral are the ultimate source of phosphate in biosphere and they are poorly soluble and they are slow at dissolving at neutral pH at room temperature (Pasek et al., 2014). The atmospheric phosphorus (P) is very limited and part of the P available in soluble and particulate form is removed from the land to body of water is through erosion and runoff (Mekonnen & Hoekstra, 2017). Phosphorus dissolves and stimulates the growth of phytoplankton and aquatic plants (Carpenter, 2005). Nitrogen is a chemical element that flows from the atmosphere into terrestrial and marine ecosystems through biological nitrogen fixation (BNF), which is largely unreactive molecular nitrogen that is reduced to ammonium compounds (Fowler et al., 2015). Nitrogen (N) pollution has also increased over the past several decades as a result of increased agricultural run-off and atmospheric deposition of reactive N from fertilizers and the combustion of fossil fuels (Galloway et al., 2004).

In the 1970s, phosphorus reduction was widely accepted as the most crucial strategy to reduce eutrophication of North America Great Lakes and other lakes in Europe and North America (Schindler et al., 2016). The conclusion to reduce phosphorus was based on three lines of evidence: a whole-lake experiment, a well-documented history of successful ecological recovery following phosphorus abatement measures, and historical monitoring of eutrophication (Schindler et al., 2016). A whole-lake eutrophication experiment has been ongoing in Lake 227 in the Experimental Lake Area (ELA) of northwestern Ontario, Canada, since 1968, to examine the drivers of algal blooms (Schindler, 1974; Schindler et al, 2008). Lake 227 has two basins, there are no inlet streams; the lake is fed via surface runoff, primarily due to snowmelt and occasional heavy rainstorms. Phosphate, nitrate, and inorganic carbon were added to one of the basins of Lake 227, while only nitrate and inorganic carbon were added to the second basin (Schindler, 1974). The basin that had phosphate added was transformed into a green swamp within weeks of adding the nutrient, indicating phosphorus was the limiting nutrient (Schindler, 1974).

Oxygen depletion at the lake bottom often occurs as a result of eutrophication (Schindler, 1974). Dead and decaying organic matter from excess aquatic plant and algae growth sinks to the bottom of the lake where it is decomposed by microbes, creating a high Biological Oxygen Demand (BOD) in the sediments. This can lead to deep water oxygen depletion, especially during periods of stratification when bottom waters do not mix with overlying waters receiving oxygen from photosynthesis and diffusion from the atmosphere (Brodersen & Quinlan, 2006). Oxygen is one of the most important environmental factors influencing fish assemblages, and different fish have different oxygen requirements (Brodersen & Quinlan, 2006). Oxygen is also vital for nutrient dynamics and biogeochemical cycling. Anoxia at the sediment-water interface can remobilize phosphorus in the sediments and exacerbate problems with eutrophication via internal phosphorus loading (Brodersen & Quinlan, 2006). Oxygen depletion at the sediment-water interface is thought to be an important contributor to toxic cyanobacteria blooms (Molot et al., 2014; Molot et al, 2021).

2.2.1 Shallow Lakes and Alternative Stable States

In shallow lakes, aquatic macrophytes are considered the key component for lake ecosystems (Jeppesen et al, 1998). Eutrophication and other anthropogenically driven processes (e.g., the introduction of exotic species and habitat fragmentation) over the last two centuries has led to a shift in ecological condition from a macrophyte-dominated lake to an algal dominated lake in many shallow lakes (Jeppesen et al., 2000 & Scheffer et al., 2001). The concept of alternative stable states in shallow lakes was first put forward in the 1960s (Williams, 2005). One state consists of clear water dominated by submerged macrophytes, whereas the other is a turbid phytoplankton- dominated states (William, 2005). Each stable state has a number of buffers (top-down and bottom-up processes) that help maintain that state (William, 2005). When these buffers are overrun, the shallow lake can shift from one stable state to another.

2.3 Climate Change Impacts on Lakes

Climate change has become a significant global issue in the 20th century, and the impacts of climate changes are expected to increase global air temperatures, reduce sea ice coverage, and increase global sea levels (McGinn, 2002). In the Experimental Lakes Area of northwestern Ontario, long-term environmental monitoring showed that mean annual air temperature rose by 2ºC and evaporation rates increased by 30% from the late 1960s to the mid-1980s (Schindler et al., 1990). The effects of climate change on freshwater ecosystems varies latitudinally and regionally. In Ontario, climate change is reducing lake ice coverage and increasing summer thermal stability in lakes (Sharma et al, 2019).

Lake ecosystems and organisms that live within them are vulnerable to temperature changes, especially during thermal extremes (Woolway et al., 2021). For shallow lakes, heatwave intensity is greater, and heatwaves extend across multiple seasons with lakes (Woolway et al., 2021). Lake heatwaves could alter species composition by pushing aquatic species and ecosystems to their limit, threatening biodiversity (Woolway et al., 2021). Another study conducted by Jane et al (2021), examined concentration of dissolved oxygen in aquatic system. Their finding illustrated that there is a decline in dissolved oxygen in surface and deep-water habitats. The decline in surface water is associated with reduce solubility under warmer water temperatures.

2.3.1 Climate Change and Cyanobacteria Blooms

Eutrophication impacts lakes within a multiple-stressor context (Smol, 2010), and may be exacerbated by climate change in particular (Schinder et al., 2016). Eutrophication is linked to increases in cyanobacteria, but increasingly blooms are being reported in oligotrophic lakes (Reinl et al., 2021). Climate change processes, include lake warming, increasing water column stability, and increased frequency and intensity of storm events, may promote cyanobacterial blooms in both oligotrophic and eutrophic lakes. Cyanobacteria requires four factors to thrive in oligotrophic lakes: an inoculum, adequate light, temperature, and nutrients (Reynolds, 2006).

Climate change provide ideal conditions for cyanobacteria to succeed. Cyanobacteria thrive in warm waters, and when the global temperatures rise so do the global water temperatures (Bennett, 2017). Cyanobacteria thrive on the surface and the water column, absorbing more sunlight, nutrients that are released through internal loading and growing more easily and denser (Bennett, 2017). The increased rainfall and storms cause more frequent nutrient pollution into water bodies causing more cyanobacteria to bloom and thrive (Bennett, 2017). The change in climate change also affects precipitation rates, intensifying the Earth's water cycle increasing evaporation (Bennett, 2017).

The diversity of cyanobacterial species that produce blooms and the factors that interact with them, make predicting and managing blooms challenging (Sterner et al., 2020). The study done by Sterner et al (2020) documented the presence of nearshore cyanobacterial blooms along a portion of the southern shoreline of Lake Superior, where regions of human recreational contact is high contributed to high blooms.

2.3.2 Climate Change and Fish

Climate change also has implications for fish and fisheries in Ontario lakes. Impacts on fisheries in Ontario lakes may include variation in year-class strength of different species (Casselman, 2002), change in bioenergetics and growth (Berndt et al., 2002), a shift in terrestrial species distributions (Rahel, 2002), and altered trophic dynamics (Jackson & Mandrak, 2002). Climate change can also result in changes in productivity, at both the ecosystem and population levels and by driving species from one location to other location, thereby creating a shift in the biogeographical boundaries (Shuter et al., 2002). At the population level, productivity changes such as age-specific growth, maturity and survival schedules are altered (Shuter, 1990). With the increasing sensitivity to climate, some of these processes, especially recruitment, is likely a precursor of a shift in zoogeographic boundaries (Shuter et al., 2002).

2.4 Invasive Species

As global transport has become more efficient, our planet has been increasingly connected (Dukes & Mooney, 2004). Commodities and people can travel easily from continent to continent, and this provides opportunities for thousand and plants and animals to move or to hitchhike into new environments beyond their normal dispersal capabilities (Dukes & Mooney, 2004). Many of these hitchhiking organisms can thrive in new environments causing ecological harm (Dukes & Mooney, 2004). Invasive species can outcompete native species, leading to population decline and loss of biodiversity, and modify entire ecosystems, including altered food web structures (Dukes & Mooney, 2004; Invasive Species Centre, 2016). Invasive aquatic species pose a significant threat to all Ontario water bodies, from small streams to Great Lakes (Invasive Species Centre, 2016).

Human activity is the primary reason for invasive species to invade terrestrial or aquatic environments (Invasive Species Centre, 2016). Aquatic invasive species move into Ontario through various means such as shipping, recreational and commercial boating, live bait, aquarium and water garden trade, live food fish, canals and water diversion and terrestrial species pathways (Government of Canada, 2017). Lakes with higher connectivity and higher boat traffic often have a greater risk of invasive species introduction. They commonly enter the aquatic ecosystem through improper equipment cleaning, lack of rising recreational boat before moving between waterbodies, and illegal transportation of fish (Invasive Species Centre, 2016). In Ontario lakes, Asian Carp (a collective term that refers to four species: bighead, black, grass and silver crap) is native to China and southern Russia but were brought into the U.S. in the early 1960s (Invasive Species Centre, 2016). The U.S uses Asian Carp as a biocontrol method in aquaculture; they escaped into the wild and have become established in the eastern United States (Invasive Species Centre, 2016). Sea lamprey (*Petromyzon marinus*), eel-like fish that are cylindrical and about 30 to 80cm long, are another invasive species in Ontario (Hansen et al., 2016). The sea lamprey invaded the Great Lakes from the Atlantic Ocean (Christie & Goddard 2003; Eshenroder, 2014). They were first found in Lake Ontario in 1835, Lake Erie in 1921, Lake Michigan in 1936, Lake Huron in 1937, and Lake Superior in 1938 (Applegate, 1950; Lawrie, 1970; Smith, 1979; Smnith & Tibbles, 1980; Smith, 1985). The sea lamprey has reduced the number of sporting fish in the Great Lakes, by attacking the fish with its ring of sharp teeth, infecting the fish, and causing the fish to die (Hansen et al., 2016). Zebra mussels (*Dreissena polymorpha*) are the most aggressive freshwater mussels and are now common throughout the Great Lakes (Invasive species center, 2016). Invasive species continue to be a problem in North America and around the world.

2.5 Paleolimnology to Infer Lake Responses to Stressors

Long-term lake monitoring is important, but long-term monitoring data is very limited for most lakes. It is important to understand the changes that have altered the aquatic ecosystem throughout time so that it can help guide restoration and lake management strategies. Where long-term high quality monitoring data is lacking, paleolimnological techniques can be used to fill in some of the gaps. Paleolimnology examines physical, chemical, and biological proxies preserved in lake sediment cores to reconstruct long-term environmental change over decades, centuries, and millennia (Smol, 2008). Paleolimnological studies are highly reproducible, meaning that one single core collected from the deepest basin of a waterbody can be replicated with additional sediment cores collected from the same location (Heggen et al., 2012). Lake ecosystem responses to vegetation development, land-use change, eutrophication, invasive species, and climate factors are all examples of environmental changes that can be reconstructed from species subfossil assemblages preserved in lake sediment cores, providing the missing long-term perspective, and helping to determine the timing and magnitude of changes in response to the stressors (Smol, 2008). For these reasons, paleolimnology can be a powerful tool for lake ecosystem management (Paterson et al., 2020).

Sediment cores are usually collected from the deeper basin(s) in lakes, which experience only minimal erosion or bioturbation, allowing the sediments to be well-preserved (Burge et al., 2017). The sediment cores are stratigraphic, meaning the new materials sit on top of the older material. Water turbulence moves sediment from shallower areas to the deepest zones within the lake, this is known as sediment focusing (Blais & Kalff, 1995). This allows for a single sediment core extracted from the deepest basin in the aquatic ecosystem to represent changes that are occurring across the lake (Blais & Kalff, 1995). Sediment cores collected from the deepest section of the lake can be sectioned into 0.5-2.0 cm intervals, to provide a high-resolution (decadal to sub-decadal) perspective on lake ecosystem changes over hundreds to thousands of years.

The two primary coring methods that are used to collect sediment records are gravity corers and piston corers. The gravity corer is mainly used for watery surface sediments dating back ~300 years. The gravity corer relies on gravity to penetrate the sediment of the sea, lake or river bottom (Axelsson & Haakanson, 1978). For the gravity corer, a tube open at the top and the bottom sinks vertically into the sediment and the tube closes pulling the sediment upwards (Axelsson & Haakanson, 1978). The piston corers are used for much longer millennial time spans (Glew et al., 2001). The piston corer is a coring system that combines heavy gravity corer with an interior piston that work to expel water and draw sediment into the corer (USGS Pacific coastal and marine science centre, 2020). This corer is mostly used in location that has soft sediment such as clay (USGS Pacific costal and marine science centre, 2020).

Establishing reliable sediment core chronologies is crucial for determining the timing of changes that have occurred in a lake, and for establishing cause and effect. For the dating of recent (deposited since ~1850) sediments, lead-210 (^{210}Pb) radioisotope activity is used (Appleby, 2001). ^{210}Pb dating is considered to be a powerful tool for estimating sediment accumulation rates and providing a timeline to evaluate the environmental history of a lake and its catchment (Rowan et al., 1995). ^{210}Pb enters the lake through precipitation and runoff, and is incorporated into the sediments over time (Appleby, 1998). To use ^{210}Pb for dating, there has to be “unsupported” or excess ^{210}Pb, which is produced from radium located outside the sediment and is not at isotopic equilibrium. The total supported ^{210}Pb in the sediment is that which is at isotopic equilibrium (Appleby, 1998). ^{210}Pb has a short half-life (22.5 years), and so the dating technique is limited to the last ~150 years (Appleby, 1998). ^{137}Cesium, produced during atmospheric bomb testing, can complement ^{210}Pb analysis. ^{137}Cs activity in sediments traces back to the history of atmospheric bomb testing. Its firsts appeared in the environment in the 1950s, and peaked in 1963 when a ban on nuclear bomb testing was implemented (Pravalie, 2014). Once in contact with the soil, ^{137}Cs becomes adsorbed to the finer organic and soil particles and is transported from the drainage basin to the lake, where it eventually is deposited in the lake sediments (Ritchie et al., 1973). ^{137}Cs has a half-life of 30.2 years (MH Paller et al., 2014). For older sediments, radiocarbon (^{14}C – the half-life of 5,730 years) dating is the most widely applied chronological tool (Blakemore, 2019; Lee, 2011).

Sediment cores contain an array of physical, chemical and biological proxies. Physical proxies examine characteristics of the sediment itself, such as sediment composition, texture, colour, density, and magnetic properties (United States Geological Survey, 2016). These physical proxies can be applied to understand past climate conditions, land-use changes, hydrological changes, and among others (Last & Smol, 2006). Chemical proxies examine the chemical composition of bulk sediments and subfossil remains of aquatic organisms, which are affected by the chemistry of water in which they form (USGS, 2016). Notable examples of chemical proxies are stable isotopes of carbon, nitrogen, and oxygen, and biomarkers (Last & Smol, 2006). Biological proxies are the subfossil remains of living organisms and include pollen, plant macrofossils, charcoal, foraminifera (amoeboid protists), ostracodes and Cladocera (microscopic crustaceans), diatoms (siliceous algae), and chironomids (benthic macroinvertebrates) (Smol, 2008).

The most widely used biological proxies to assess past aquatic conditions in paleolimnological studies are from primary producers (eg diatoms, chrysophyte scales and pigments) and assemblages of invertebrates (eg cladocerans and chironomids) (Li, 2021). Biological proxies provide information on how ecological communities have changed over time (eg biodiversity, production, nutrients, and changes in habitat), and may be able to infer past changes in limnological changes in water chemistry (eg salinity, acidity, deep water oxygen, and other sources of pollution) and physical conditions (eg temperature and lake depth) (Last & Smol, 2001; Smol et al., 2001a, 2001b). This is because different species have unique environmental optima and tolerances and are only found under certain environmental conditions. Characteristics that make organisms good biological proxies include high abundance, good preservation, rapid life cycle, and are responsive to environmental change (Gornitz, 2009).

Diatoms are golden-brown algae whose cellular contents are enclosed between two valves or shells composed of silica (Bradbury & Krebs, 1995). When diatoms die, the values are incorporated in sediments and they are the most abundant and common microfossils of lake sediments (Bradbury & Krebs, 1995). Their remains are durable and well preserved in accumulated sediments (Smol & Stoermer, 2010; Bradbury, 2004). Diatoms have been used as reliable environmental indicators due to their abundance in lakes and high species diversity (Smol & Stoermer, 2010). Diatoms are ecologically diverse and widely distributed, there is well over 1,000 common freshwater diatoms species in temperate North America and Europe (Bradbury & Krebs, 1995). Diatoms are sensitive to many environmental variables, including light, temperature regime, salinity, pH, oxygen, and nutrients (Van Dam et al., 1994).

Chironomids (Insecta, Diptera) are non-biting midges that spend the majority of their life cycle as aquatic larvae living in benthic habitat zones (Larocque et al., 2001). They are widely distributed worldwide and are often the more common insects in freshwater system (Walker, 2001). The larvae of chironomids are important benthic invertebrates in freshwater systems and usually live either in littoral or profundal zones (Walker, 2001). Chironomid larvae are important to the aquatic environment as they play an important role in the food webs of aquatic communities, linking primary producers and secondary consumers (Tokeshi, 1995). Certain parts of chironomid larvae are well preserved in lake sediments. The chitinous head capsules shed by the third- and fourth- instar- larval stages during ecdysis (or moulting) represent the bulk of chironomid sub-fossils (Walker, 2001; Larocque & Rolland, 2006). The head capsules can be identified and for chironomid assemblages a powerful paleoenvironmental tool used for both qualitatively and quantitively inferring changes in environmental conditions through time (Larocque & Rolland, 2006). Chironomid species distributions are strongly related to lake depth, where the depth preferences among chironomids arise from related environmental gradients including temperature, substrate stability, food quality, and oxygen availability (Brodersen & Quinlan, 2006). In a lake or reservoir, the most essential lake management application of chironomids as bioindicators is their use in inferring deepwater oxygen levels (Quinlan & Smol, 2005). Individual chironomid taxa respond differently to hypolimnetic oxygen condition as a result to adaption such as possession and concentration of haemoglobin (Cranston, 1988; Int Panis et al., 1966), behavioural ventilation (Heinis et al., 1994), body size (Brundin, 1951, Int Panis et al., 1996) and metabolic rate (Heinis et al., 1994). Chironomid show a wide range of tolerance to oxygen conditions and their head capsules are preserved well in lake sediments (Little & Smol, 2001).

This thesis uses Cladocera subfossil remains to reconstruct long-term ecological change in Lake Scugog. Cladocera subfossils (headshields, postabdomens, and carapaces) within the sediment provide information on various environmental events and disturbances that affect lake status, such as climate change, water level changes, and acidification (Korhola & Rautio, 2001).

Their community composition reflects the physical, chemical, and biological characteristics of the ecosystems they inhabit (Frolova, 2014). Their susceptibility to both ‘bottom-up’ (resources regulated) and ‘top-down’ (predation stressed) food-web effects that make them useful in studying both processes in a lake system (Hessen et al., 1995). The response in species composition, relative abundance, size structure can be used to assess changes in water quality variables as well as predation intensities (Korhola, 1999; Hann & Turner, 2000; Chen et al., 2010; Jeppesen et al., 2002; Alexander & Hotchkiss, 2010).

2.6 Cladocera as Paleolimnological Indicators

Cladocera (Class Branchiopoda) are a superorder of small crustaceans commonly known as water fleas (Li, 2021). Cladocera are generally smaller than 1 mm and include 16 to 18 family lineages (Adamczuk, 2016). Most Cladocera species are herbivorous filter feeders (Adamczuk, 2016). They have to the ability to inhabit almost any kind of freshwater habitat, ranging from ponds and larger bodies of water (Korhola & Rautio, 2001; Adamczuk, 2014). Within lakes, the greatest abundance and diversity of cladocerans is usually found in the vegetation at margins of lakes and ponds (Korhola & Rautio, 2001). For example, the species-diverse Family Chydoridae are located in three distinct habitats: littoral muds, submerged vegetation and open water (Whiteside, 1974). Pelagic (open water) cladocerans can migrate up and down in the water column on a diurnal cycle, typically coming to the surface at night and moving to deeper water depths in the daytime in order to avoid visual feeding predators such as fish (Whiteside, 1974).

Cladocera are an essential component of lakes and reservoirs, as they play a significant role in the functioning of ecosystems (Geraldes & Boavida, 2004). They are active grazers on algae and detritus, regenerating nutrients back to the primary producers and bacteria. They are also important prey species for fish and other planktivorous organisms (Korhola & Rautio, 2001). Thus, Cladocera represent an important intermediate trophic level in lakes, and cladoceran communities are responsive to critical bottom-up and top-down ecological drivers (Daly, 2019). This also means that changes to the cladoceran community can cascade through lake trophic webs (Carpenter et al., 1985).

Cladocerans disarticulate (break apart) upon death and can be identified in lake sediments from their chitinous body fragments that preserve well (Figure 2.1), including headshields, carapaces, postabdomens, and ephippia, which is a structure that protects diapausing eggs (Nykänen et al., 2009). Chitin is a fibrous substance consisting of polysaccharides, and forms the exoskeleton of arthropods (Korhola & Rautio, 2008). Not all chitinous remains are equally resistant to degradation, and various mechanical, chemical and biological factors can all contribute to partial decomposition of exoskeletal fragments (Korhola & Rautio, 2008). Because of this, preservation in the sediments differs among cladoceran species, although most species will leave some form of identifiable remain preserved in the sediments (Nykänen et al., 2009). Chydorids and bosminids are usually well represented in lake sediments, while *Daphnia* and other larger cladocerans are typically only represented by smaller body parts such as post abdominal claw, caudal cerca, and mandibles (Jeppesen et al., 2001). Most studies on Cladocera focus on open-water taxa such as *Bosmina* and *Daphnia*, and comparatively little ecological data exist for the Chydoridae and other cladocerans that inhabit near-shore areas (Jeppesen et al., 2001).

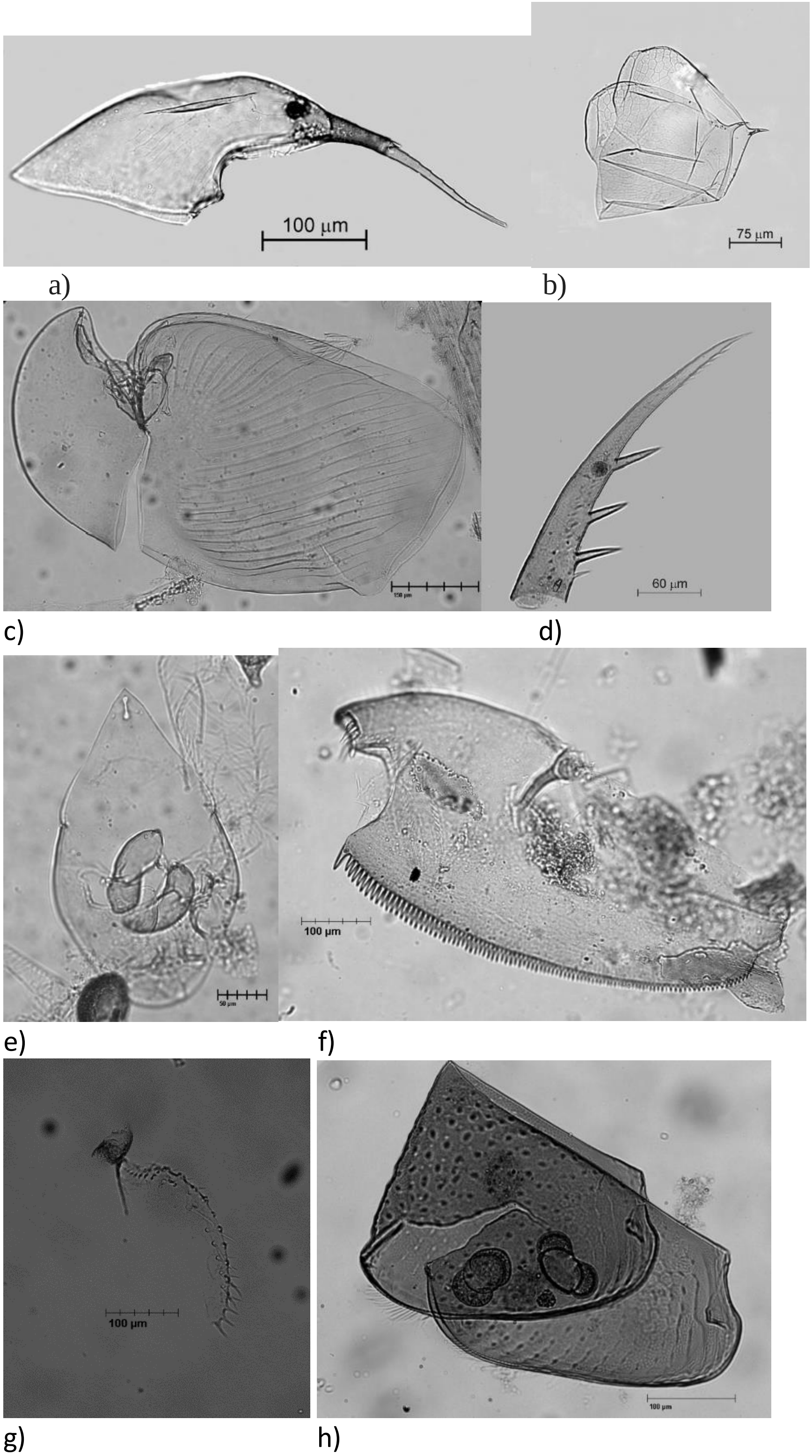


Figure 2.1. a) subfossil *Bosmina* headshield; b) subfossil *Bosmina* carapace; c) *Acroperus harpae* whole; *Sida crystallina* claw; e) *Alona affinis* headshield; f) *Eurycercus* post abdomen; g) *Leydiga* post abdomen; h) *Chydorus* carapace.

Cladocera community composition reflects the physical, chemical and biological characteristics of the ecosystems they inhabit (Frolova, 2014). From this information, Cladocera fossil remains can be used in paleolimnological studies to reconstruct past environmental conditions, inferring lake level changes, climate change impacts, acidification, and eutrophication (Luoto et al., 2011; Korhola & Rautio, 2001; Daly, 2016). Cladocera assemblages have been used to reconstruct changes in TP and tropic status in paleolimnology as qualitative and sometimes quantitative bioindicators (Boucherele & Züllig, 1983; Holfmann, 1987; Whiteside & Swindoll, 1988; Korhola, 1999; Bos & Cumming, 2003; Chen et al., 2010). For example, several experiments suggest that the higher the phosphorus level can cause higher growth rates for *Daphnia* (Urabe et al., 1997; Elser et al., 2000). The changes in trophic levels can affect Cladocera assemblages in direct and indirect ways such as changes in food quantity and quality or changes in habitat between pelagic and littoral conditions (Li, 2021). The changes in TP levels can indirectly affect cladoceran assemblages by changing the food quality (Sterner et al., 1993; Plath & Boersma, 2001) and quality (Geller & Müller, 1981; Gliwicz, 1990), and habitat availability of cladocerans (Whiteside, 1970; Hofmann, 1987; Tremel et al., 2000, Musher et al., 2019). For example, habitat of pelagic and littoral cladoceran can be largely altered by eutrophication. This alteration can lead to increases in phytoplankton abundance and detrital particle while decreases in rooted macrophytes densities and water transparency (Whiteside, 1970).

Lakes that are dominated by *Chydorus brevilabris* have been routinely observed in environments under eutrophic and hypereutrophic conditions (Bos & Cumming, 2003; Navalainen & Luoto, 2013). As a littoral Cladocera taxa, *C.brevilabris* attaches to substrate and prey as an algal scraper in less trophic systems while expanding the habitat to open water by filter-feeding on phytoplankton, detritus and bacteria in more eutrophic systems (de Eyto, 2001).

Other species that show greater abundance with increased nutrients and biological production in some lakes are *Daphnia ambigua*, *D. pulex,* and *Bosmina longirostris,* and other species that are found in less productive lakes are *Acroperus harpae*, *Alonella nana*, and *A. excise* (Bos & Cumming, 2003; Luoto et al., 2008; Liu et al., 2009, Nevalainen & Luoto, 2013).

Cladocera assemblages' changes can be summarized by changes in habitat use, such as benthic, littoral and pelagic (Li, 2021). The shifts from benthic to pelagic production occur when lakes become enriched in nutrients (Jeppesen et al., 2000, 2001; Davidson et al., 2011; DeSellas et al., 2011). Shifts in littoral, benthic, and pelagic cladocerans is also sensitive to lake depth, with higher abundances of pelagic species including *Daphnia* and *Bosmina* spp., in deeper lakes (Korhola et al., 2000; Amsink et al., 2006). Changes in fish density through top-bottom effects and changes in macrophytes coverage that provide habitat and refuge from predation can also drive changes in the relative proportions of littoral, benthic, and pelagic cladocerans (Jeppesen et al, 2001; Davidson et al., 2010; Bennion et al., 2015).

The cladoceran taxa *Bosmina* and *Daphnia* have been used in several studies as being effective indicators of predation intensity in paleolimnological studies (Nilssen & Sandøy, 1990; Palm et al., 2005; Korosi et al., 2008, 2010; Alexander & Hotchkiss, 2010). *Bosmina* and *Daphnia* subfossil size structure can also be used to track changes in food web structure, including the strength of planktivorous fish predation (Korosi et al., 2013). Planktivorous fish tend to select for larger, more visible cladocerans such as large-bodied *Daphnia* (Brooks & Dodson, 1965). In contrast, invertebrate predators such as copepods are limited by their own gape size, and tend to prey on small-bodied *Bosmina* spp that can be easily handled and manipulated to shorten prey handling time and save energy (Kerfoot, 1975; Zaret & Kerfoot, 1975).

Cladocera exhibit cyclomorphosis in response to predation (Li, 2021). Cyclomorphosis the occurrences of cyclic or seasonal morphological changes to an organism (Pijanowska, 1990). Individual size in Cladocera is one of the most important morphological features that determines the vulnerability of an organism to vertebrate and invertebrate predation (Pijanowska, 1990). *Bosmina* and *Daphnia* populations exhibit changes in body size and appendage length seasonally when there is a change in the dominant predators and predation intensity (Li, 2021). Small-bodied *Bosmina* have also been shown to alter their morphology in response to different predation regimes, which can be tracked through time by measuring *Bosmina* subossil remains in lake sediments (Korosi et al., 2013). *Bosmina* exhibit extensive variation in their morphology, notably in the size of their body, carapace shape and the length and curvature of the mucro and antennules (Korosi et al., 2013). Elongated appendage lengths and carapace heights play a crucial role in the defence against gape-limited, grasping invertebrate predators like copepods (Korosi et al., 2013). *Bosmina* with long murcos and antennules have a greater possibility of escaping an encounter with a predatory copepod (Kerfoot, 1975a, 1977b, 1978,1999). When *Bosmina* have long antennules that curve towards the ventral margin, they have the added advantage of shielding the vulnerable swimming antennae and soft ventral body surface from damage during an attack (Kerfoot, 1975; Sakamoto et al., 2007; Sakamoto & Hanazatoo, 2008). These morphological features are typically reduced in the absence of predation (Korosi et al., 2013).

CHAPTER THREE: STUDY SITE DESCRIPTION

Lake Scugog is located in southern Ontario, within the Durham and Kawartha Lakes regions of Ontario (Figure 3.1). The Lake Scugog basin was formed following the retreat of the Laurentide ice sheet ~ 12,000 years ago (Fisher & Alexander, 1993). It is a headwater lake for the greater Kawartha Lakes system and part of the Trent-Severn Waterway. Lake Scugog was initially a wetland until the construction of a downstream dam in the Town of Lindsay in 1830s caused water levels to increase by approximately four feet to create Lake Scugog, a shallow, human-made impoundment and a navigable waterway. Today the depth, volume, and area of the lake are still controlled by operations at the Lindsay Dam.

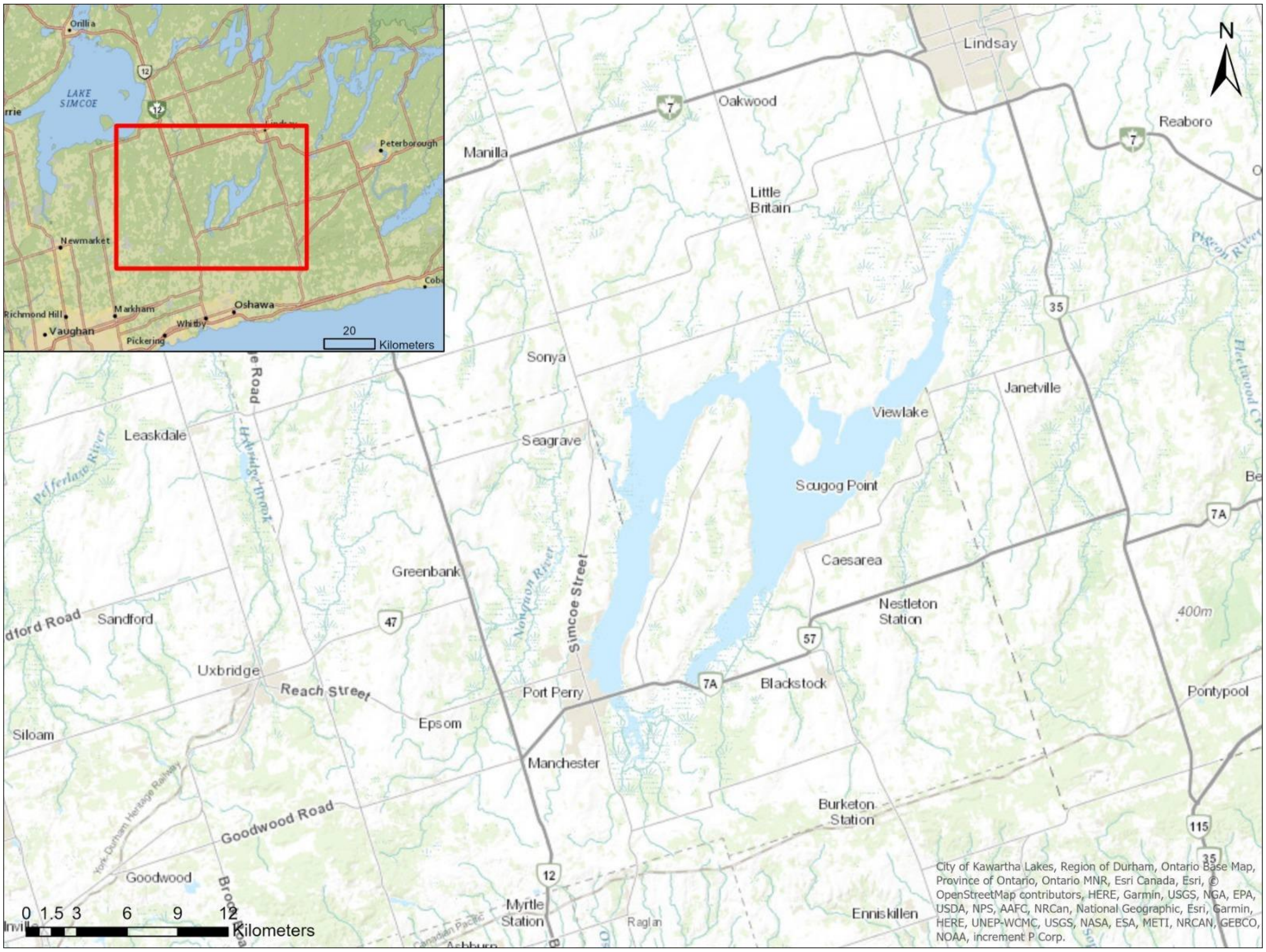
Present-day Lake Scugog is large (surface area = 68 km^2) and shallow, having an average depth of 1.4 m and a maximum depth of 7.6 m. The total watershed area is 141km^2 or 14,129 ha. The entire lake has a volume of approximately 95.7 million cubic metres. Lake Scugog watershed has about 37% of natural cover, with deciduous forests as the dominant forest type (Kawartha Conservation, 2010). The forests interior is about 441 ha and has a deep interior of 51 ha (Kawartha Conservation, 2010). The lake consists of two “arms” (an elongated western and eastern arm) that are subjected to different land-use pressures in the watershed. The west arm is primarily impacted by agriculture, as well as by urbanization in the Town of Port Perry. The surrounding lands of the east arm are less developed, with primarily agricultural land-use. The west arm is shallower than the east arm (max depth 5.4 m versus 7.6 m). It receives nutrient runoff from the Nonquon River and stormwater runoff from the Town of Port Perry Bay (Kawartha Conservation, 2010).

The Lake Scugog watershed can be subdivided into three distinct regions: The Oak Ridges Moraine, the Peterborough Drumlin Field, and the Schomberg Lowland Clay Plains

(Kawartha Conservation, 2010). The geology of the region is dominated by thick sequences of Quaternary glacial sediments that overlie the Paleozoic bedrock formations (Kawartha Conservation, 2010). The substrate composition of the lake is mainly composed of silt and clay that originated from the glaciolacustrine deposits in the Lake Scugog watershed and erosional inputs. The Lake Scugog watershed covers 529.7 km², with 78.5% of the drainage area located within the Region of Durham, and the other 21.5 % within the City of Kawartha Lakes (Kawartha Conservation, 2010). The water that enters into the lake comes through sub-watersheds flowing northward off of the Oak Ridges Moraine (Kawartha Conservation, 2010). The water exits the lake through the northeastern arm, at the Scugog River, eventually flowing northward through the Lindsay Dam and into Sturgeon Lake. One of the major tributaries to Lake Scugog is the Nonquon River, which drains primarily agricultural lands.

The dominant land use in Lake Scugog watershed is mainly composed of agricultural land-use (67%), followed by natural land cover (mostly wetlands and forest) (30%) and the remainder is composed of urban developed land-use (Smith & Kirkwood, 2020). Table 3.1 provides a detailed history of Lake Scugog. The developed land is a small proportion when compared to the total area of the watershed, it is highly concentrated along the shoreline (Smith & Kirkwood, 2020). Nearshore sites that are impacted by urban development had the most degraded water quality and ecological conditions in Lake Scugog when compared to locations that are dominated by agricultural land-used or natural cover (Smith & Kirkwood, 2020). Developed areas also had the highest chloride and phosphorus concentrations, which in some cases have exceeded the Provincial Water Quality Objectives (Smith & Kirkwood, 2020). The developed area of Port Perry hosts 43% of Scugog populations and continues to expand along shorelines (Smith & Kirkwood, 2020).

Lake Scugog has historically been a eutrophic lake that supports abundant fish populations (Kawartha Conservation, 2010). Due to its plentiful fish populations and proximity to Toronto, the lake is an economic asset to the surrounding areas as a year-round tourist destination. Lake Scugog provided over 560,000 fishing hours in 2000 and is estimated to contribute over $200 million annually to the region's economy (Kawartha Conservation, 2010). Other municipal locations around Lake Scugog have been estimated to receive approximately $10-15 million annually from tourism and recreational activities (Kawartha Conservation, 2010). For many of the residents of Lake Scugog, one of the most pressing concerns is the decline in walleye populations over the past couple of years, the causes of which are unknown. The walleye fishery in Lake Scugog was established and repeatedly stocked between 1920 and 1940 to provide recreational fishing opportunities (Ontario Federation of Anglers and Hunters, 2019). In January 2016, the Ontario Ministry of Northern Development, Mines, Natural Resources & Forestry Service instituted a moratorium on walleye fishing in Lake Scugog (Table 3.1). Significant issues that are affecting the walleye population are recruitment failures and lack of juvenile fish surviving to adulthood (OFAH, 2019). Research is needed to understand the causes of recruitment failure.



City of Kawartha Lakes, Region of Durham, Ontario Base Map, Province of Ontario, Ontario MNR, Esri Canada, Esri, © OpenStreetMap contributors, HERE, Garmin, USGS, NGA, EPA, USDA, NPS, AAFC, NRCan, National Geographic, Esri, Garmin, HERE, UNEP-WCMC, USGS, NASA, ESA, METI, NRCAN, GEBCO, NOAA, increment P Corp.

Figure 3.1. Map showing the location of Lake Scugog in southern Ontario.

Table 3.1. Timeline of key events in the history of Lake Scugog**.**

Years	**Event**
~1700	The Mississauga's of Scugog Island First Nation re-settle in the basin of Lake Scugog.[1]
1821	The arrival of first European Settlers in Scugog.[2]
1827	The Town of Lindsay was founded by William Purdy, and his sons, who built the first dam to power a grist mill.[1]
1829	The Lindsay dam bursts due to high waters[2]
1837	A new Lindsay grist mill dam was built, raising upstream water levels by ~1.2 m.[2]
1838	A deadly fever swept through the area, killing a large number of people. The fever was linked to the increased numbers of insects in the newly flooded lands.[2]
1841	A group of young Mississaugas decided to take down the dam after petitioning the government to remove the dam.[1]
1844	The Purdy dam was replaced with a newer dam that had a lock system to control the flow of water.[1]
1856	A road connection across the lake to Scugog Island was built. The road was a floating bridge, and it increased sediment organic matter content and accrual rates in the lake.[2]
1870	First documented ice harvesting for personal and commercial use began.[3]
1876-1884	Construction of a permanent road to Scugog Island occurred.[2]
1883-1884	Fires in the Town of Port Perry (November 26, 1883, and July 3, 1884).[2, 3]
1885	A permanent roadway across Lake Scugog linking Reach with Scugog Township was completed.[3]
1891	Fire in the Town of Port Perry (October 22).[3]
1903-04	Severe winter caused many portions of Lake Scugog to freeze to the bottom, resulting in a massive fish kill.[3]
1906	An early-season freeze killed high numbers of frogs, which covered the shores of the lake.[3]
1910	The first record of efforts to control aquatic weed growth: log booms were dragged across the bottom of Lake Scugog.[3]
1912	Massive winter fish kill was reported due to under-ice oxygen depletion.[3]
1915	Additional efforts reported to combat the aquatic weed growth using rakes and scythes.[3]
1920	Stocking of walleye (*Sander vitreus)* in Lake Scugog occurred.[3]
1930	Lake Scugog became a tourist spot for good fishing and beautiful lakeshores.[1]
1940	Stocking of walleye in Lake Scugog occurred.[3]
1960	A disastrous flood in April caused the water to rise dramatically, with up to 30 inches of water covering sections of both the Port Perry and Cartwright Causeways.[3]
1960-1962	Construction of 7A Hwy occurred from Port Perry to Caesarea.[3]
1960	Major fish kill, almost 50,000 dead fish of several species were recorded.[3]
1970	Invasive Eurasian Watermilfoil (*Myriophyllum spicatum*) was introduced into the lake.[2]
1971-1977	The Nonquon River Water Pollution Control Centre was constructed.[3]

1980	Population peak recorded for Eurasian Watermilfoil (*Myriophyllum spicatum*) in Lake Scugog.[3]
1991	The first appearance Zebra mussels (*Dreissena polymorpha)* was recorded in Lake Scugog.[3]
2015	Invasive macroalga Starry Stonewort (*Nitellopsis obtusa*) became established.[3]
2016	Ontario Ministry of Northern Development, Mines, Natural Resources & Forestry Service declare a moratorium on walleye fishing in Lake Scugog.[3]

[1] McMullen, C. (2018, March 22). The history of water: Lake Scugog. Retrieved from https://chronicle.durhamcollege.ca/2018/03/history-water-lake-scugog/

[2] (n.d.). Retrieved from https://scugogheritage.com/history

[3] Cheney, P. (2007, June 23). At Lake Scugog, it smells like trouble. Retrieved from https://www.theglobeandmail.com/news/national/at-lake-scugog-it-smells-like-trouble/article688547/

3.1 Decline of Recreational Fisheries

Lake Scugog falls within Ministry of Northern Development, Mines, Natural Resources & Forestry Service, Fisheries Management Zone 17, and is an excellent location for sport and recreational fishing. The fish population of Lake Scugog contains abundant largemouth bass, smallmouth bass, perch, black crappie, catfish and carp. The planning area for FMZ 17 includes waters of the Trent-Severn Waterway from the mouth of the Trent River to Canal Lake, the Crowe River Watershed, and the eastern portion of the Oak Ridges Moraine (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009). The FMZ 17 management plan has been focused on declines in walleye (*Sander vitreus*) for over 30 years (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009). The walleye populations in all four lakes (Rice, Lake Scugog, Balsam and Buckhorn) that are routinely sampled by the Kawartha Lakes Fisheries Assessment Unit have been declining (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009). In Rice Lake and Lake Scugog, walleye have been in decline since the early 1990s (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009), and a moratorium on walleye fishing in Lake Scugog has been in place since 2016. The decline in walleye abundances has been shown to corresponds to an increase in largemouth or smallmouth bass, and the emergence of new species such as bluegill and black crappie (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009). The changes that are observed in the fish community were associated with increases in total phosphorus, and increases in summer water temperatures and water clarity (Robillard & Fox, 2006).

In recent surveys conducted by ‘on the water’ angler creel, walleye have been the most commonly targeted species, approximately accounting for one-third of all fishing effort (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009). With regards to sport fish species, the release rate for walleye is the lowest, with only one in four walleye released, including the ones that need to be released by regulation (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009). The management goal for the walleye population is to reverse declines and rebuild existing, self-sustaining walleye populations to maintain and enhance benefits to the public (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009). The Ministry of Northern Development, Mines, Natural Resources & Forestry Service has identified three challenges to meeting this goal. The first is spring water level, as flow fluctuations may cause recruitment failures at this critical spawning period (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009). The second challenge is that decreased adult abundances have reduced the productive capacity of walleye populations (Ministry of Northern Development, Mines, Natural Resources & Forestry Services, 2009). The third challenge relates to physical changes in the lakes that have made them less suitable to walleye and more suitable to other fish species, such as changes to the surrounding vegetation and increase in sediment loading causing the depth of the lake to decrease (Ministry of Northern Development, Mines, Natural Resources & Forestry Service, 2009).

3.2 Eutrophication of Lake Scugog

Based on a 2010 Management Plan published by Kawartha Conservation (Lake Scugog Environmental Management Plan, 2010), every year, an estimated 9,100 to 9,600 kg of phosphorus and 340 to 390 tonnes of nitrogen enters Lake Scugog, which contributes to extensive macrophyte growth. Total phosphorus concentrations reported in the 2010 report on Lake Scugog was 0.039mg/L, and provincial guidelines for TP are between 0.03mg/L and 0.02mg/L (Kawartha Conservation, 2010), where levels above these thresholds are considered to pose a risk for nuisance algae and aquatic plant growth. Areas of the lake that received the highest amounts of nutrient loading are in the western arm immediately offshore of the Port Perry urban area and at the inflow of the Nonquon and Layton rivers, which flow through agricultural land (Kawartha Conservation, 2010). The most abundant source of phosphorus in the lake (~23.5%) is runoff from agricultural fields and related activities (Kawartha Conservation, 2010). Urban development in the Town of Port Perry has increased impervious surfaces in the watershed, further exacerbating inputs of nutrients and other pollutants. The Town has multiple storms drains that high input levels of lawn fertilizer into the lake, also contributing to the excessive growth of macrophytes. These locations have been highlighted as requiring immediate remedial action (Kawartha Conservation, 2010). Blackstock Creek is another tributary that brings excessive phosphorus into the lake, contributing about 740 kg of phosphorus annually (Kawartha Conservation, 2010).

Eutrophication has been a significant water quality threat in Lake Scugog for the last ~30 years. In order to improve the water quality in the lake, reductions in total phosphorus from the current 9,400 kg to ~ 6,000-6,200 kg per year were recommended (Kawartha Conservation, 2010), but may not be large enough to lower internal phosphorus loading sufficiently to account for recent increases in internal phosphorus loading (Harrow-Lyle & Kirkwood, 2020a). In 1912 and 1960, major winter fish kills were reported that have been attributed to under-ice oxygen depletion (Table 3.1). Historically, eutrophication in Lake Scugog has resulted in excess aquatic macrophyte growth, but in recent years, cyanobacteria blooms, particularly *Microcystis*, have been reported (Harrow-Lyle & Kirkwood, 2020a). In the summers of 2017 and 2020, Durham Region Public Health Department put out water quality advisories for the Port Perry Bay area of Lake Scugog after cyanobacteria toxins were detected in the water. As watershed development around Lake Scugog intensifies in future, the percentage of impervious surfaces will increase and further enhance the runoff of nutrients and road salt into the lake.

A recent (2016-2020) four-year monitoring program during the open water months showed that the greatest contributions of nutrient inputs are from growing shorelines, urban developments, and internal loading (Harrow-Lyle & Kirkwood, 2020a). Other sources of nutrient inputs are from septic systems and fertilizer application on gardens are the major sources of nutrients from shorelines and subdivision (Harrow-Lyle & Kirkwood, 2020a). Port Perry Bay areas is the most degraded section of the lakes, with the highest amount of nutrient run-off, highest chloride levels, periodic algal blooms and prevalence of Starry Stonewort (Harrow-Lyle & Kirkwood, 2020a).

3.2.1 Invasive Species in Lake Scugog

Lake Scugog is threatened by several invasive species, including the Eurasian watermilfoil (*Myriophyllum spicatum*), the zebra mussel (*Dreissena polymorpha*) and Starry Stonewort (*Nitellopsis obtusa*). These invasive species all play a part in threatening the water quality, fish production and fish habitat in Lake Scugog. The Eurasian watermilfoil is an aggressive invasive aquatic plant that is native to Europe, Asia, and North Africa (Ontario's

Invading Species Awareness Program, 2020). The Eurasian watermilfoil prefers aquatic ecosystems that are shallow water about 1-3m deep but can root in up to 10m of water (Smith & Barko, 1990). This invasive species is a fast-growing perennial that forms dense underwater mats that shade out other aquatic plants (Smith & Barko, 1990). When it starts to die off in the fall, the decaying plant can reduce oxygen levels in the water (Ontario's Invading Species Awareness Program, 2020). These plants can interbreed with native milfoil, creating a more aggressive form of invasive species; the tiny plants have the potential to develop into new plants (Smith & Barko, 1990). Eurasian watermilfoil can be easily spread through water currents, boat propellers, trailers or fishing gear carrying plant fragments to new areas (Smith & Barko, 1990). It invaded Lake Scugog in 1970, to become the dominant aquatic macrophyte in the lake. Eurasian watermilfoil emerges earlier than native milfoil species, which allows it to shade out other native aquatic plants and inhibit them from growing (Parkison et al, 2011). Thick, dense Eurasian milfoil stands restrict recreational use (swimming, boating) of the lake and diminish the overall aesthetics.

In 1991, invasive zebra mussels were recorded in Lake Scugog. Zebra mussels were first discovered in the Laurentian Great Lakes in 1988 and from there spread to inland lakes (Hincks & Mackie, 1997; Herbert et al., 1989). Zebra mussels (*Dreissena polymorpha*) have negative consequences for the aquatic ecosystem, including reducing the biomass of phytoplankton and changing algal community composition, which can increase water transparency (Rolla et al., 2019) and alter nutrient cycling (Bastviken et al., 1998). Water chemistry of Lake Scugog is ideal for zebra mussel invasion; however, the soft-substrate common throughout the lake kept zebra mussel numbers under control (Hincks & Mackie, 1997; Harrow-Lyle & Kirkwood, 2020b). *Dreissena quagga* has invaded the great lake watershed and have been observed in Lake Simcoe and Lake Scugog, this species is able to colonize soft substrate. The newest invader to

Lake Scugog, the macroalga starry stonewort (*Nitellopsis obtuse*), may be facilitating the establishment of zebra mussels in Lake Scugog (Harrow-Lyle & Kirkwood, 2020b).

Starry Stonewort is native to Europe and Asia and is expanding its range into North America (Minnesota Aquatic Invasive Species Research Center, 2018). It was first identified in 1978 in the St. Lawrence River and was introduced to other lakes from ship ballast-water (Midwood et al., 2016; Larkin et al., 2018). Starry Stonewort has become a dominant aquatic macrophyte in many inland lakes across the east-central United States (Larkin et al., 2018). Starry stonewort in shallow ecosystems will grow taller than other native species and vascular plants, causing native species to become light-limited (Larkin et al., 2018). The dense mats that are formed by the Starry stonewort displace native macrophytes (Pullman & Crawford, 2010; Larkin et al., 2018) causing a decline in species richness (Brainard & Schultz, 2017). Starry Stonewort can negatively affect waterfowl that rely on native submerged aquatic vegetation for nesting, shelter, and foraging (Midwood et al., 2016). Fish populations can also be affected by Starry Stonewort, through a loss of foraging, nursery, and spawning habitat due to this macrophyte creating thick benthic mats (Midwood et al., 2016). In Lake Scugog, starry stonewort is associated with *Microcystis* and zebra mussels, suggesting it may promote their growth (Harrow-Lyle & Kirkwood, 2020b).

CHAPTER FOUR: METHODOLOGY

4.1 Sediment Core Collection

Sediment cores were collected from the east and west arms using a Uwitec^{TM} gravity corer. The deepest depositional area was chosen as coring sites for both locations. An east arm core was collected because it is the deepest site in Lake Scugog. Port Perry Bay in the west arm was chosen because it is the most heavily developed site. Gravity cores are capable of capturing recent deposited sediments with high water content, making this an ideal coring method for studying recent changes within the lake systems (Blomqvist, 1991; Glew et al., 2001). Gravity coring relies on gravity to penetrate the sediments of the sea, lake, or river bottom (Axelsson & Haakanson, 1978). Several open barrel gravity corers are small and easy to operate by a single person (Axelsson & Haakanson, 1978). It is considered the simplest form of operation, as a tube open at the top and bottom sinks) and driven vertically into the sediments (Last & Smol, 2006). Once the tube has sunk as far into the sediment that it will go, the top of the tube is closed, and the corer with tube and sediment is pulled upwards vertically from the lake bed to the surface of the water, where the bottom is then also capped before it is pulled out of the water (Last & Smol, 2006).

Multiple (three) gravity cores were obtained from the western basin of Lake Scugog off of the Port Perry Marina (Figure 4.1) by researchers at the University of Waterloo in June 2016 (Wiklund & Hall, 2017) Cores were sectioned into 1-cm intervals using a Glew vertical extruder (Glew, 1988). Each of the slices represent an interval of time. The total lengths of the sediment cores were 54, 64, and 66cm. I used the 66-cm core for my thesis. In August 2017, I collected three sediment cores from the deepest spot of the lake, on the east arm near Caesarea (Figure 4.1), using a Uwitec^{TM} gravity corer. I sectioned the sediment cores into 0.5 cm intervals for the top 20 cm, and switched to 1 cm intervals from 20 to the bottom of the core. The sediment core lengths were 47, 64, and 71. I conducted all of my analyses using the 71 cm core.

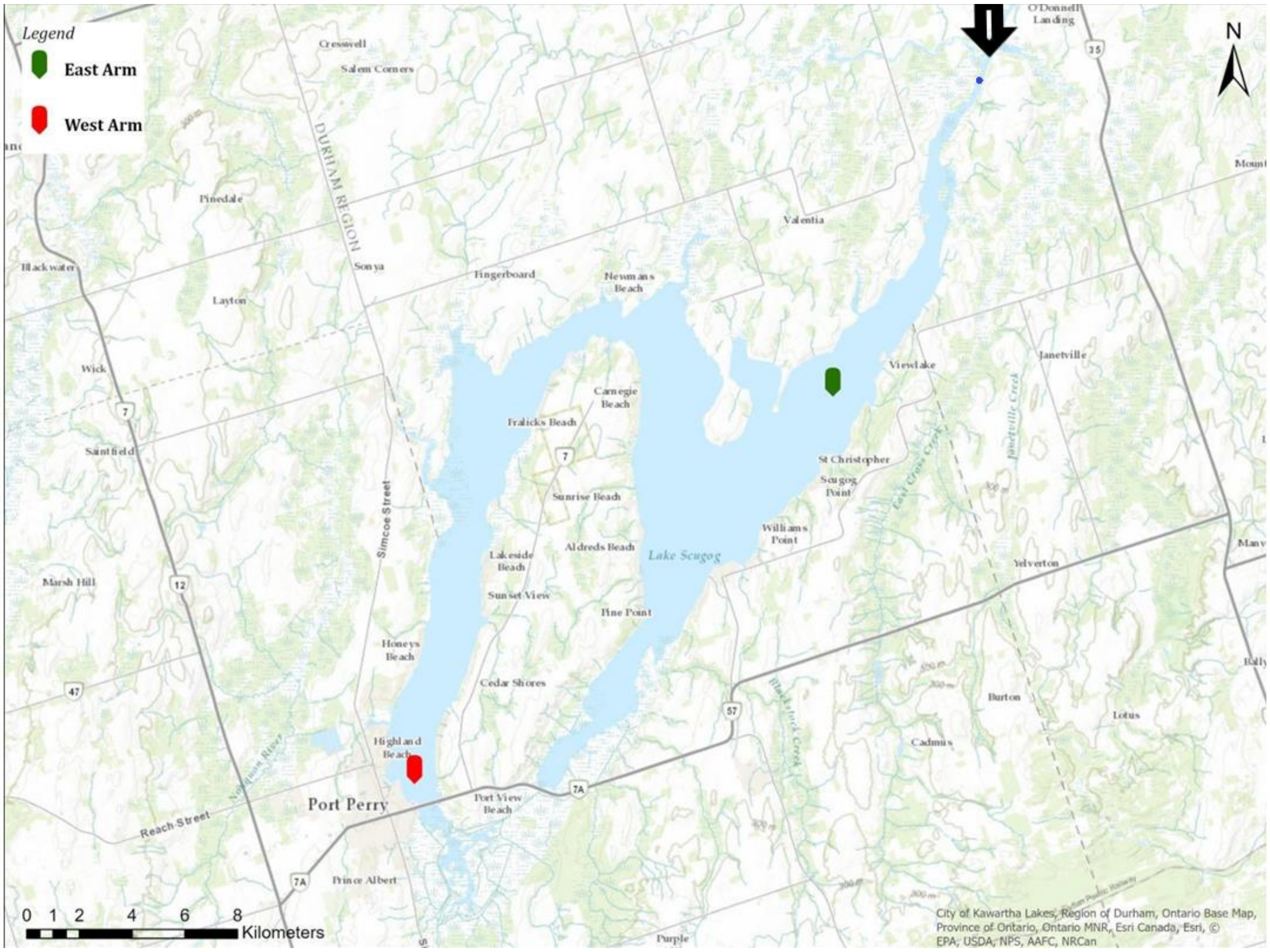


Figure 4.1. The locations of the two cores collected from Lake Scugog. The green arrow is the core for the east arm and the red arrow is for the west arm. The black arrow is where the Lindsay Dam located northeast of the lake on the Scugog River in the Town of Lindsay.

4.2 Sediment Chronologies

Sediment dating (a procedure to date deposits that are hundreds to thousands of years old) is a vital component of a paleolimnological approach to determine what sediment intervals correspond to which historical events. The core from the west arm (Port Perry Bay) was ^{210}Pb- and ^{137}C-dated using gamma spectroscopy at the University of Waterloo. The core from the east arm was ^{210}Pb-dated by alpha spectroscopy at ChronosScientific Inc. Gamma spectrometry measures the activity of other daughter products of ^{226} Ra (i.e. ^{214} Bi, ^{214}Pb) together with total ^{210}Pb on each sediment depth intervals (Zaborska et al., 2007). For the east arm (Caesarea) core,

^{210}Pb dating was performed using alpha spectroscopy by ChronosScientific Inc. Alpha spectrometry is determined by measuring ^{210}Pb “unsupported” ^{210}Pb, which is produced from radium located outside the sediment and the total ^{210}Pb in the sediment (Wetzel, 2001). ^{210}Pb has a short half-life (22.5 years), and so the dating technique is limited to the last 150 years (Wetzel, 2001).

^{137}Cesium, produced during atmospheric bomb testing, can complement ^{210}Pb analysis. Once in contact with the soil, ^{137}Cs becomes strongly absorbed on the finer organic and soil particles (Ritchie et al., 1974). The dating method assumes ^{137}C fallout with rain, attached to particles and transported from the drainage basin to lake deposits, where it eventually gets absorbed into the suspended particulate matter and deposits in the sediments (Ritchie et al., 1973). ^{137}Cs activity in sediments traces back to the history of atmospheric bomb testing, first appearing in the 1950s, and peaking in 1963 when a ban on nuclear bomb testing was implemented (Pravalie, 2014). Alpha spectroscopy does not provide simultaneous ^{137}Cs and ^{210}Pb radioisotopic activities, and so only ^{210}Pb-dating was completed on the eastern core.

The Constant Rate of Supply (CRS) model was applied to develop an age-depth model based on ^{210}Pb activities for cores from both arms. The CRS model assumes that there is a constant fallout of ^{210}Pb from the atmosphere to the water, which results in a constant supply of ^{210}Pb to the sediment regardless of any variation that may have occurred in the sediment accumulation rate (Lubis, 2008). The model therefore takes into consideration that there is variable sedimentation, i.e. that sedimentation rates change with depth (Lubis, 2008).

4.3 Loss on Ignition

Loss on ignition is a widely used method to estimate the organic and carbonate content of sediments (Heiri et al., 2001; Dean, 1974; Bengtsson & Enell, 1986). Dean (1974) evaluated the methods and concluded that LOI determines carbonate and organic contents in soil, sediments and rocks with precision and accuracy compared to other methods. We did loss on ignition for this study to verify the timing of the Lindsay Dam construction and the changes that we expect to see through time, where reservoir formation should correspond to an initial surge in % organic content and a relative decline in carbonates. The loss on ignition analysis was conducted by University of Waterloo for the west arm core, and I conducted the loss on ignition analysis for east arm core.

The percentage of water is calculated by weighing a certain amount of wet sediment, in this case about ~0.5g of sediment, that was then dried in the oven for about 24 hours at 90^{o}C (Smol, 2008; Wiklund & Hall, 2017). The samples need to be weighed again after the dry process, and percent water can be calculated by subtraction and division (Wiklund & Hall, 2017). The percent water information is useful as it provides knowledge of the amount of compaction the sediment has been subjected to, as well as clues to the paleoenvironmental history of the system (Smol, 2008; Wiklund & Hall, 2017). Usually, the surface sediments have the highest percent water, decreasing in the mid and bottom segments of the intervals (Smol, 2008). The dry sediments are then placed in a muffle furnace at 550^{o}C for 2 hours to burn off organic matter (Wiklund & Hall, 2017). The burning process of the dry samples will ensure that any organic matter in the sediments should be combusted (Dean, 1974; Smol, 2008). The sample is placed in the desiccator and re-weighed; the % organic matter is what is referred to as a

% loss on ignition (Smol, 2008). Sediments are then combusted at 950 ^{o}C for 2 hours to burn off carbonates and re-weighed to determine % loss on ignition for this fraction (Smol, 2008).

4.4 Cladocera-based Reconstruction of Ecological Change

Cladoceran remains were analysed at 1cm, 2 cm and 5 cm resolution for the full 64 cm for west arm and 71 cm for east arm sediment cores. For the west basin, 1 cm intervals were chosen from 0 cm to 5 cm of the core, transitioning to 2 cm from 5 cm to 20 cm and 5 cm interval from 20 cm to 64 cm. For the east basin, 1 cm intervals were chosen from 0 cm to 5 cm, transitioning to 2 cm from 5 cm to 24 cm and 5 cm intervals from 25 cm to 71 cm. The sediment layers on the surface are younger than those layers below (Watson, 1997). Higher sampling resolution was used at the top of the core (younger sediments) where we anticipated more changes to occur.

The cladoceran subfossils where isolated from the sediment matrix, ensuring that as much of the mud and other materials found in the sediment matrix were removed to make cladoceran subfossils more readily visible. About 2 g of wet sediment per interval were placed into a 100 mL beaker. The sample of wet sediment were deflocculated with 80 mL of 10% KOH (80 mL of deionized water and 8 g of KOH), heated for 30 minutes at 80ºC, and then passed through the 36 -µm sieve and washed numerous times (Korosi & Smol, 2012a). The deflocculation step is needed to break down the sediment matrix further so it would be able to pass through the sieve. The residue retained on the sieve is then transferred into a small vial and 1-2 drops of ethanol was added as a preservative and 1 drop of saffronin glycerol was added to stain the cladoceran remains for viewing under the microscope. About 50 microliters of the samples were pipetted onto microscope slides and air-dried. Glycerine jelly was used to mount a coverslip overtop the residue.

The slides were viewed for cladoceran remains under an Amscope B690-PL microscope at 200X magnification. Cladocera break apart into different body parts upon death and can be identified in lake sediments from their chitinous body fragments, including headshields, postabdomens, postabdominal claws, mandibles, and carapaces (Ejsmont-Karabin & Karabin, 2013). Body fragments for each species were counted, and the most abundant body part was used to determine the number of individuals in the same. Cladoceran remains were identified to species or genus level using Korosi and Smol (2012a, 2012b) as taxonomic guides. A minimum of 70 individuals was counted for each interval, to ensure that an accurate representation of the cladoceran assemblage was achieved (Kurek et al., 2010). Species relative abundance was calculated by dividing the number of species from one group by the total number of species from all groups.

Bosmina subfossil remains were measured following methods described in Korosi et al, (2010) using a digital camera interfaced to the microscope, and the program Toupeview. Specifically, the headshield antennule length (AL), carapace length (CL - representing the body size) and mucro length (ML) were measured on the *Bosmina* remains. The microscope was calibrated for measurement in μm, and a minimum of 35 headshields and carapaces were measured per interval (Brahney et al., 2010). The average of the 35 measurements were calculated for the headshield antennule length, carapace length and mucro length. Calculating the average size measurements allowed me to compare size structure between the east and west basins.

4.5 Statistical Analysis of Downcore Trends

I predict significant changes have occurred in Cladoceran assemblages, in response to the creation of the Lindsay dam, clearing of watershed for agriculture, urban land development in

Port Perry (Western core), and climate change (Eastern core). To test this, I used a stratigraphically-constrained incremental sum of squares (CONISS) cluster analysis (Grimm, 1987) to determine periods of change (zones) in cladocerans assemblages through time, and the broken stick model was used to identify the number of significant zones (Bennett, 1996).The statistical analysis testing was conducted using R studio version 1.1.463 using the Rioja and Vegan packages (Okasanen et al., 2012).

CHAPTER FIVE: RESULTS

5.1 Eastern Arm

5.1.1 Sediment Core Chronology

There was a steady decline in unsupported ^{210}Pb from 10 cm to 31 cm, while unsupported ^{210}Pb in the surface (0-10 cm) sediments was relatively consistent (i.e., no decline with sediment depth) (Figure 5.1). Unsupported ^{210}Pb reached zero at a core depth of 60 cm, indicating background ^{210}Pb was reached. The earliest date provided by the CRS age-depth model was ~1880 ±50 years at a core depth of 50cm (Figure 5.1). Sedimentation rates steadily increased from ~1880 (core depth 50 cm) to present-day, with the increase in sedimentation rate accelerating after ~1950 (core depth 20 cm) (Figure 5.1).

The CRS age-model has higher uncertainty at deeper core depths where unsupported ^{210}Pb is low. Loss-on-ignition (LOI) was used as an independent dating marker to determine the timing of the formation of present-day Lake Scugog following the creation of the Lindsay Dam, because reservoir creation should correspond to an increase in the relative proportion of sediment organic matter. The loss-on-ignition analysis showed a strong increase in organic content beginning at 40 cm and culminating at 30 cm (Figure 5.2). The increase in organic matter corresponded to a relative decline in carbonates. This indicates that a core depth of ~40 cm corresponds to the timing of the construction of the Lindsay dam in 1834, in contrast to the CRS age-depth model that provides a date of ~1900 ± 20 years at 40 cm. The percentage of water is steady throughout the period between 84-96%, until it reached 40cm depth where there is fluctuation in the water percentage between 75-89% from 40-63cm.

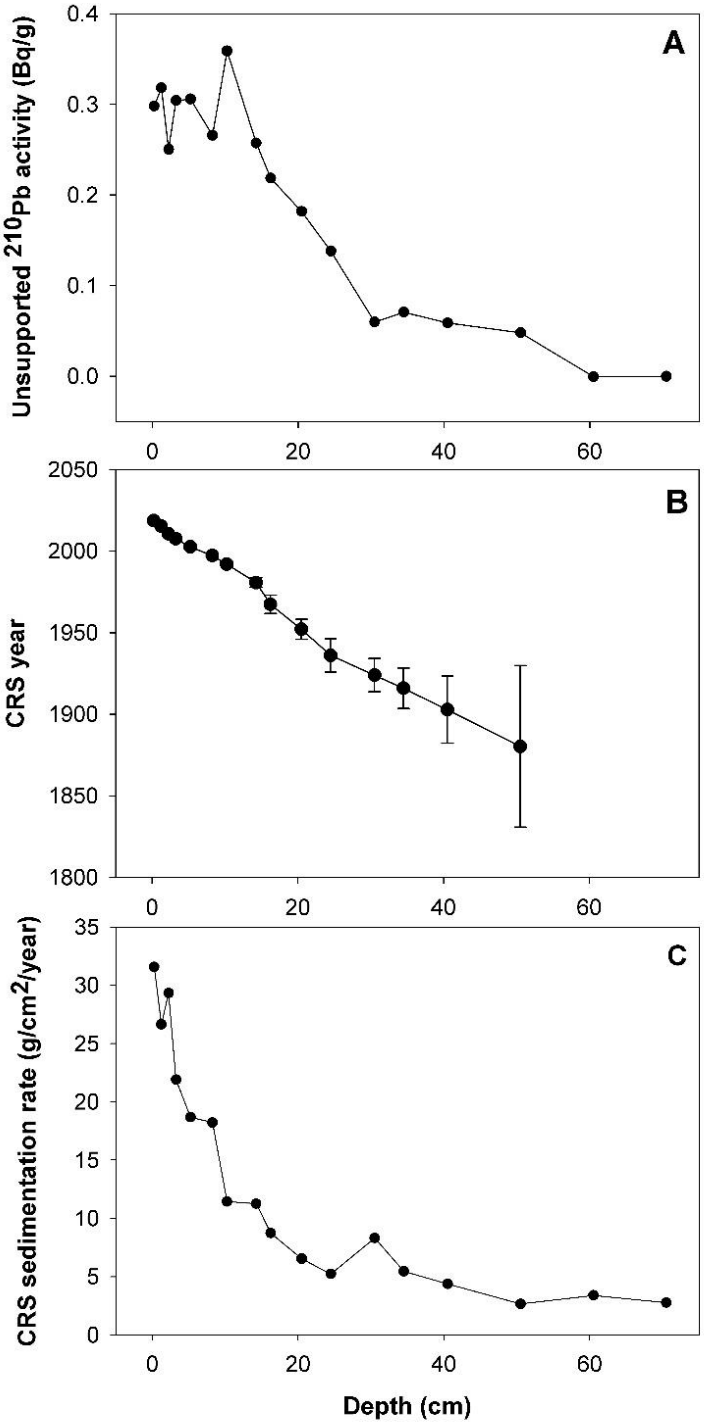


Figure 5.1. Results of ^{210}Pb dating (using alpha spectrometry) for the sediment core collected from the eastern arm of Lake Scugog: A) Unsupported ^{210}Pb activity with sediment depth; B) Sediment age with depth, based on the constant rate of supply (CRS) model; C) Inferred sedimentation rate with depth as inferred based on the CRS age-depth model.

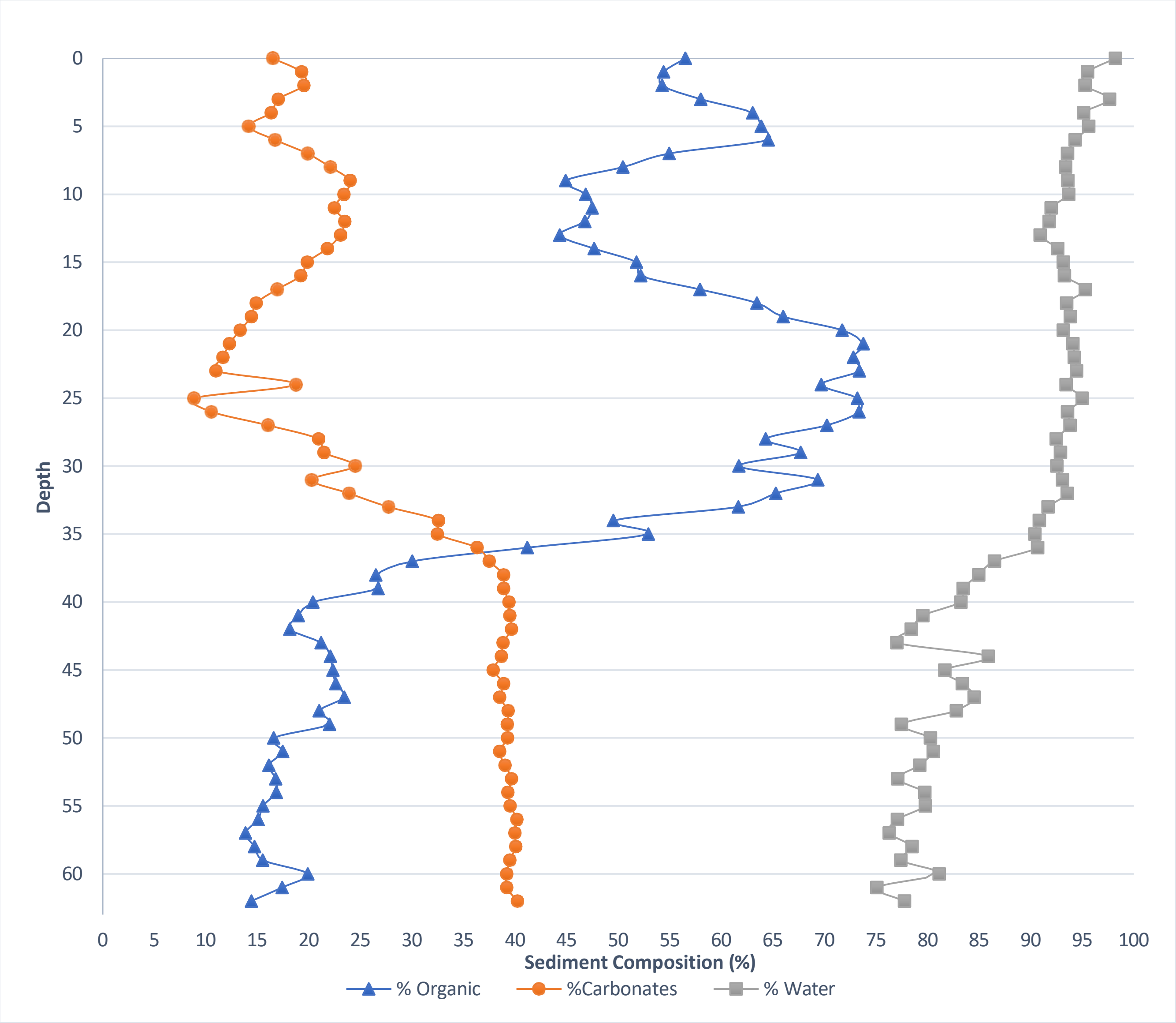


Figure 5.2. Loss-on-ignition data for the sediment core collected from the eastern arm of Lake Scugog, measuring changes in the percent water, organic matter and carbonate content of the sediments with core depth.

5.1.2 Cladocera Species Assemblages

A complete catalogue of the cladoceran species/genera found in the east arm of Lake Scugog includes: *Bosmina* spp., *Chydorus brevilabris, Alona affinis, Alona circumfimbriata, Acroperus harpae, Eurycercus, Leydigia leydigii, Paralona pigra, Pleuroxus trigonellus, Graptoleberis testudinaria, Alonella nana, Alonella excisa* and *Sida crystallina.* The dominant taxa were *Bosmina* and *Chydorus brevilabris*: *Bosmina* relative abundance ranged from ~25-50% and *Chydorus brevilabris* relative abundance ranged from ~25- 60%. Combined, *Alona* taxa ranged from ~2-33% relative abundance. The remaining taxa, including *Acroperus harpae*, *Alonella nana*, *Alonella excisa*, *Eurycercus*, *Leydigia leydigii*, *Paralona pigra*, *Pleuroxus trigonellus*, *Graptoleberis testudinaria*, *Sida crystallina*, were all found at relative abundances less than 10%.

Cladocera assemblages remained consistent throughout the sediment record (Figure 5.3). The only significant shift in cladoceran species assemblage based on a broken stick model of a constrained incremental sum of squares (CONISS) cluster analysis occurred at a core depth of 55 cm, marked by an increase in *Chydorus brevilabris* from 30% to 60% relative abundance and a decrease in *Alona* spp from 40% to 10% relative abundance before 1880. *Bosmina* relative abundance remained constant at ~20-25%. A later shift in cladoceran species assemblage was detected by the CONISS analysis consisted of an increase in *Bosmina* from 25% to 50% relative abundance and a corresponding decrease in *Alona* spp from 30% to 10%, but this shift was not significant based on the broken stick model.

5.1.3 Bosmina size structure

The overall length of carapace length for *Bosmina* ranged from 200-240 µm; the overall length of antennules length ranged from 100-125µm; and the overall length of mucro length ranged from 25-45 µm. *Bosmina* antennules length was stable throughout the core, though the longest average antennule lengths were measured at core depths 20 cm, 35 cm and 65 cm. The carapace length was also stable throughout the core, and the shortest averages are measured at core depth 12 cm, 20 cm and 25 cm. The longest carapace length was measured at a core depth of 65 cm, below which carapace length decreased again. The mucro length (the spine that forms on the carapace) was constant between 25-30 µm until a core depth of 60 cm depth, below which mucro length increased to 45 µm between 60-65 cm, but the change was not significant.

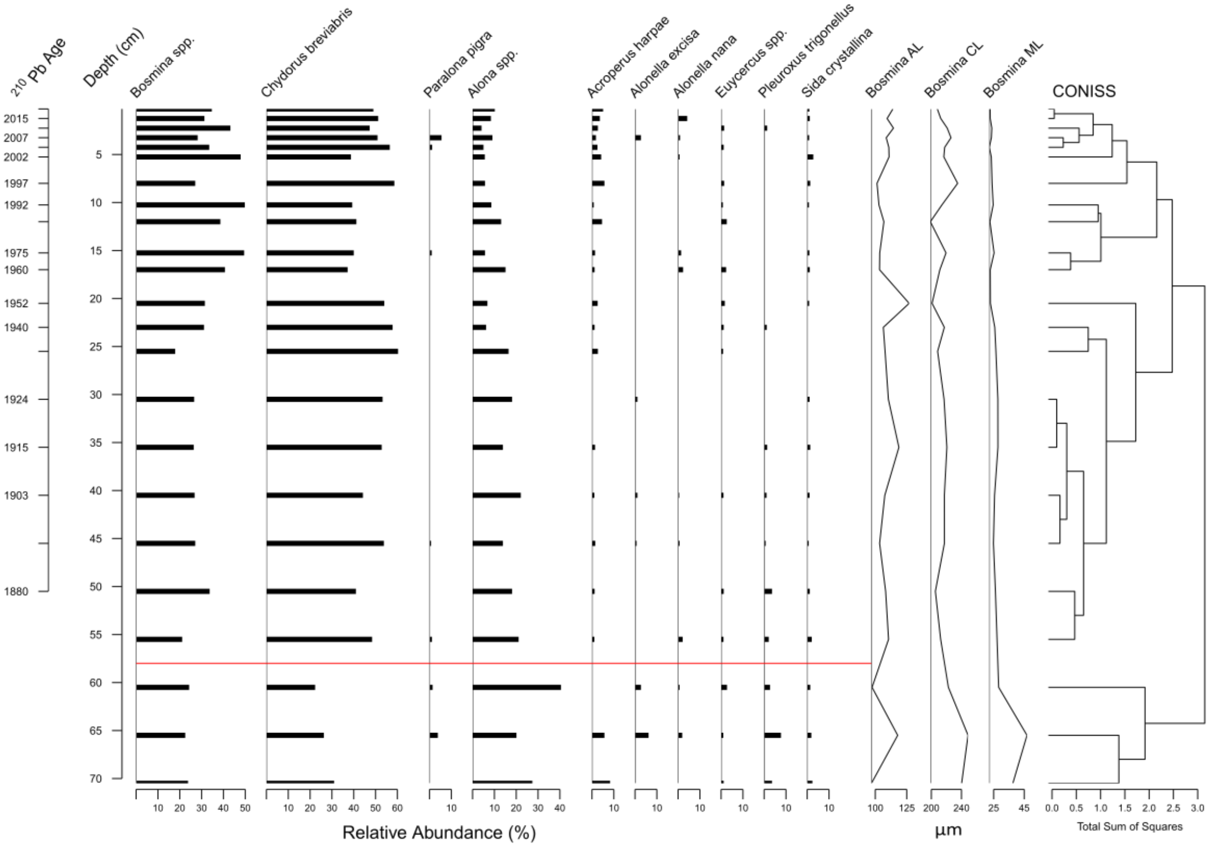


Figure 5.3. Stratigraphic profile showing changes in cladoceran species relative abundances (%) and *Bosmina* size measurements (AL = antennule length; CL = carapace length; ML = mucro length) with sediment core depth (in cm) for the eastern arm of Lake Scugog. Sediment age shown on the left is based on the ^{210}Pb Constant Rate of Supply model (Appleby 2001) but note that loss-on-ignition analysis infers a date of 1834 at 40 cm corresponding to the construction of the Lindsay Dam and the formation of present-day Lake Scugog. The results of the constrained incremental sum of squares analysis are shown on the right, and the red line indicates a significant assemblage shift assessed using the broken stick model (Grimm 1987).

5.2 Western Arm

5.2.1 Sediment Core Chronology

The results I present here for ^{210}Pb dating in the western arm is summarized from the Wiklund and Hall (2017) report commissioned by Kawartha Conservation. There was a decline in ^{210}Pb from 0 cm to 2 cm, followed by an increase from 2 cm to 8 cm, and then steady decline from 8 cm to 40 cm. At 40 cm depth, unsupported ^{210}Pb reached background. There was a steady decline in sedimentation rate from the surface of the core to a core depth of ~20 cm, which corresponds to about 1940, and sedimentation rates are slightly elevated between 30 to 40 cm (Figure 5.4). Based on the CRS model, the core dates back to ~1860 ± 50 years at a core depth of 39 cm (Figure 5.4). The formation of the Lindsay dam occurred at a core depth of ~44 cm based on loss-on-ignition from Wiklund and Hall (2017), which showed a transition in sedimentation composition at 44 cm (Do, 2021) similar to the eastern arm.

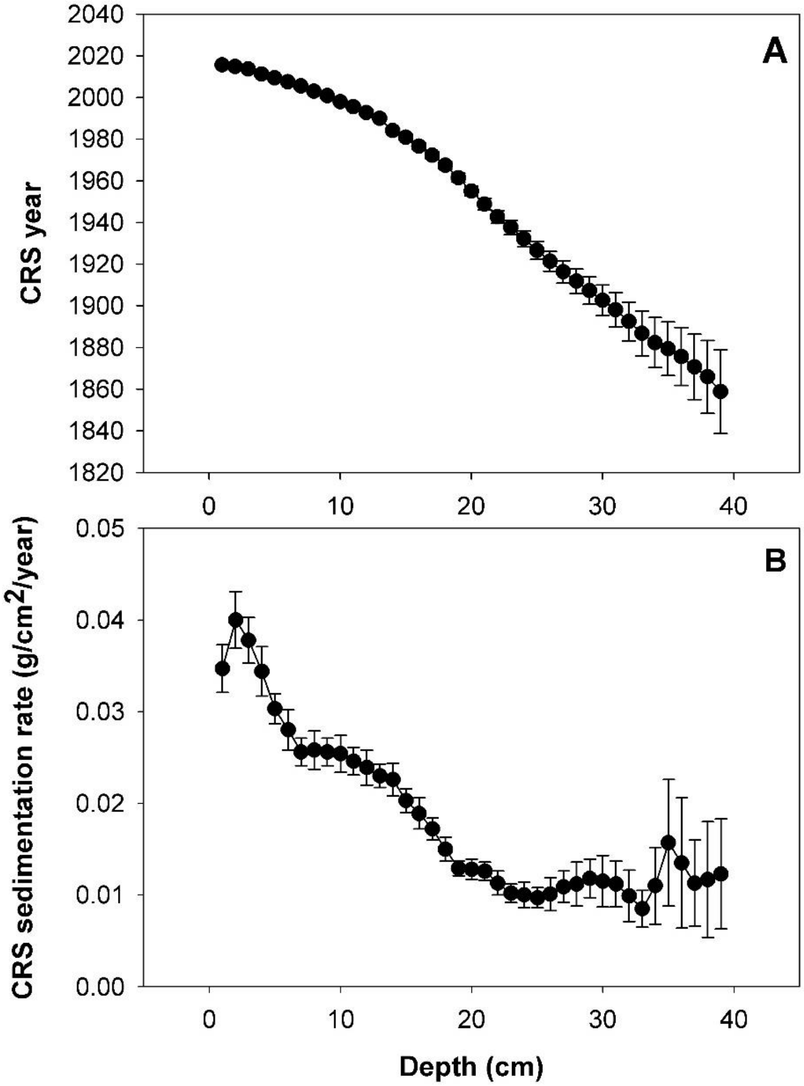


Figure 5.4. Results of the sediment core collected from the western arm of Lake Scugog: A) Sediment age with depth, based on the constant rate of supply (CRS) model; and B) Inferred sedimentation rate with depth as inferred based on the CRS age-depth model. Recreated from data presented in Wiklund and Hall (2017).

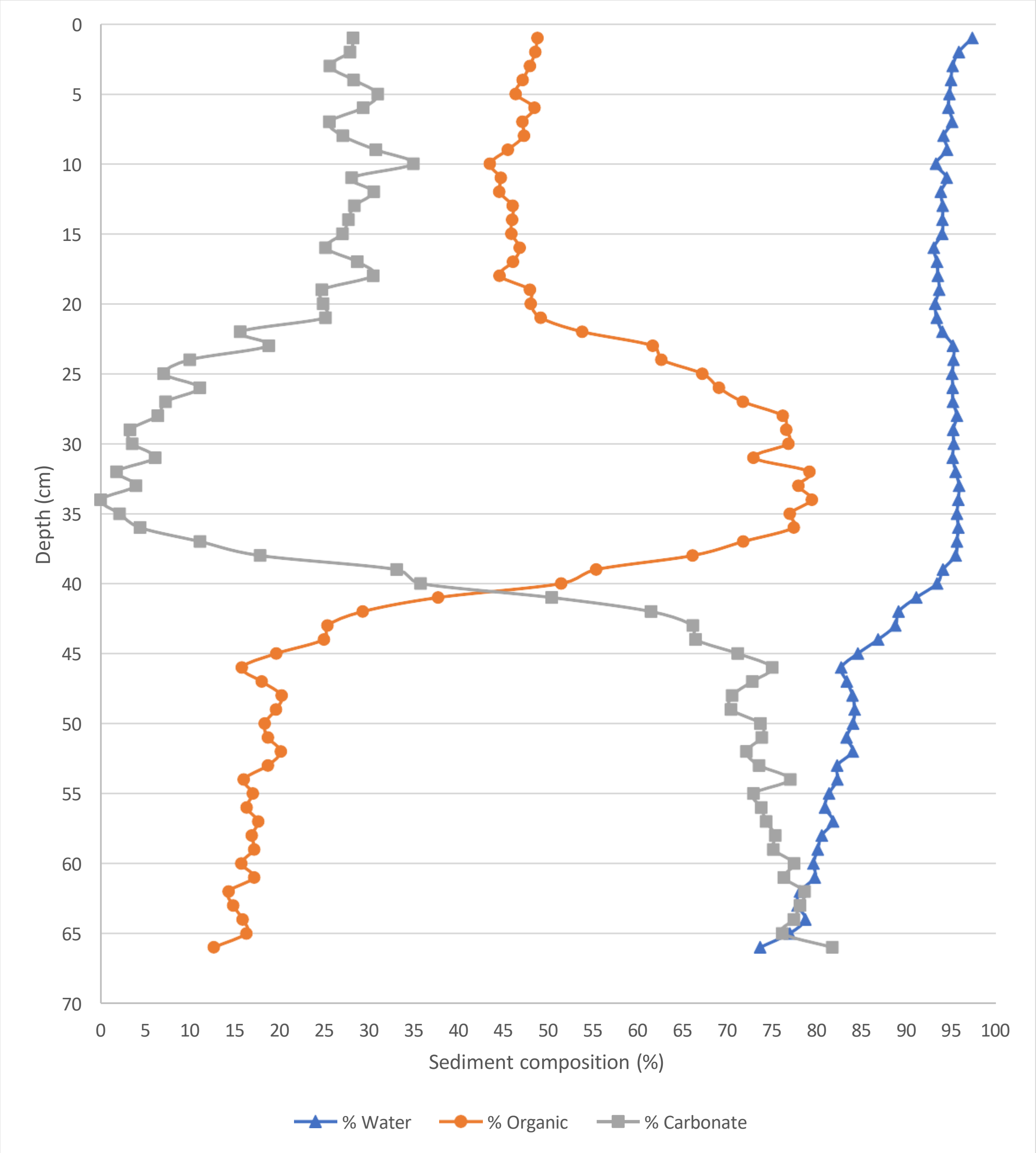


Figure 5.5. Loss-on-ignition data for the sediment core collected from the western arm of Lake Scugog, measuring changes in the precent of water, organic matter and carbonate content of the sediments with core depth. The figure was recreated from data provided in Wiklund & Hall (2017).

5.2.2 Cladocera Species Assemblages

A complete catalogue of cladoceran species/genera found in the western arm of Lake Scugog includes: *Alona affinis*, *Alona quandrangularis*, *Alona circumfimbriata*, *Acroperus harpae*, *Bosmina*, *Chydorus brevilabris*, *Eurycercus*, *Leydigia*, *Paralona pigra*, *Pleuroxus trigonellus*, *Sida crystallina*, *Latona setifera*, *Graptoleberis testudinaria*, *Kurzia latissima*, *Disparalona*, *Camptocercus*, *Ophryoxus gracilis*, *Alonella nana* and *Alonella excisa.* Of these taxa, *Alona quandrangularis*, *Latona setifera*, *Kurzia latissima*, *Disparalona*, *Camptocercus*, and *Ophryoxus gracilis* were not encountered in the sediment core collected from the eastern arm (Table 5.1).

Bosmina and *Chydorus brevilabris* were the dominant taxa. *Bosmina* relative abundance ranges from ~30-60% throughout the sediment intervals and *Chydorus brevilabris* relative abundance ranged from ~4-50% (Figure 5.6). Combined, *Alona* taxa ranged from ~1-23% relative abundance. The remaining taxa *Acroperus harpae*, *Alonella excisa*, *Alonella nana*, *Eurycercus* spp, *Kurzia latissimi*, *Leydigia* spp, *Paralona pigra*, *Pleuroxus trigonellus*, *Sida crystallina*, *Latona setifera*, *Graptoleberis testudinaria*, *Ophryoxus gracilis*, *Disparalona* and *Camptocercus*, were all found at relative abundance less than 10%.

The constrained incremental sum of squares (CONISS) cluster analysis identified one significant break in the cladoceran assemblage at core depth 35 cm, circa CRS year 1880. The first shift occurred pre- 1858, corresponding to the approximate timing of the construction of the Lindsay Dam; the arm was dominated by *Bosmina* from 40-60% and *Alona* spp from 10-20%. After the construction of the Lindsay dam, where there was a shift in dominance between *Bosmina* and *Chydorus brevilabris*, ranging between 30-50% for *Bosmina* and 30-50% for *Chydorus brevilabris*. *Alona* spp, *Acroperus harpae*, *Alonella excisa*, *Alonella nana*, *Eurycercus* spp, *Kurzia latissimi*, *Leydigia* spp, *Paralona pigra*, *Pleuroxus trigonellus, Sida crystallina*, *Latona setifera*, *Graptoleberis testudinaria, Ophryoxus gracilis, Disparalona* and *Camptocercus* were all encountered regularly both before and after the creation of the Lindsay Dam.

5.2.3 Bosmina *Size Structure*

The overall size of *Bosmina* subfossil remains was smaller than *Bosmina* measurements in the sediment core from the eastern arm. Average carapace length ranged from 190-220 µm (compared to 200-240 µm in the eastern arm), average antennule length ranged from 90-110 µm (compared to 100-125 µm), and average mucro length ranges from 21-27µm (compared to 25-45 µm). The three size measurements were fairly consistent throughout the core, with no significant changes with time.

Table 5.1. List of the cladoceran taxa encountered in Lake Scugog.

	Relative Abundance in Lake Scugog	Basic Ecology
Acroperus harpae	East basin: 3-8% West basin: 0-8%	Small taxon commonly in the littoral zones of lakes, and exhibits high habitat heterogeneity (Korosi & Smol, 2008)
Alona affinis	East basin: 2- 32% West basin: 1-18%	Small taxon strongly associated with macrophyte beds, although it has been found in all littoral and benthic habitats (Tremel et al., 2000; Korosi & Smol, 2008)
Alona circumfimbriata	East basin: 2-18% West basin: 1-18%	Small littoral/benthic taxon able to tolerate a wide range of environmental conditions (Chengalath,1982; Sinev, 2009)
Alona quandrangularis	West basin: 1-18%	Large mud dwelling taxon abundant in muddy areas with vegetation (Evans, 1984; Tremel et al., 2000)
Alona rustica	West basin: 1-18%	Small littoral species is often associated with a sphagnum type substrate (Walseng, 2016)
Alonella excisa	East basin: 0-3% West basin: 0-8%	Small taxon that is commonly and widely distributed across Canada and relatively acid-tolerant (Chengalath, 1982)
Alonella nana	East basin: 0-5% West basin: 0-8%	Small taxon commonly found in the littoral zones of lakes in Canada (Chengalath, 1982)
Bosmina (includes *Bosmina longirostris* and *Eubosmina longispina*)	East basin: 25-50% West basin: 30-60%	Planktonic taxon
Camptocercus	West basin: 0-8%	Small taxa that are poorly known in North America and represented by four species (Sinev, 2018). The taxa is similar to *Acroperus harpae*.

Chydorus brevilabris	East basin: 25-60% West basin: 4-50%	Small taxon that lives in nearshore and littoral zones, has some weak swimming abilities (Korosi & Smol, 2008)
Disparalona	West basin: 0-8%	Small taxa that is found in the littoral zone of the lake (Tremel et al., 2000)
Eurycercus	East basin: 0-4% West basin: 0-8%	Largest taxon of the chydorid community, and lives attached to macrophytes (Hann, 1982; Bekker et al., 2012)
Graptoleberis testudinaria	East basin: 0-1% West basin: 0-8%	Small taxon strongly associated with macrophyte beds commonly found northeastern North America and water bodies with dense vegetation (Korosi & Smol, 2008)
Kurzia latissimi	West basin: 0-8%	Small taxon that is located in the weedy margins of lakes (Chengalath, 1982)
Latona setifera	West basin: 0-8%	Small taxon that is located in littoral zones in lakes and ponds and are typically associated with vegetation (Korovchinsky, 1992; Korosi & Smol, 2012)
Leydigia leydigii	East basin: 0-3% West basin: 0-8%	Small taxon that is an obligate mud dweller that are widely distributed but rarely abundant in Canadian lakes (Chengalath 1982, 1987)
Orphryoxus gracilis	West basin: 0-8%	Small taxon that inhabits the littoral and benthic zones of the oligo and mesotrophic lakes; they are typically attached to the substrate and also capable of some swimming (Sergeev, 1970)
Paralona pigra	East basin: 0-5% West basin: 0-8%	Littoral taxon that is widely distributed in North America (Korosi & Smol, 2008)

Pleuroxus trigonellus	East basin: 0-4% West basin: 0-8%	Littoral taxon that lives associated with macrophytes (Korosi & Smol, 2008)
Sida crystallina	East basin: 0-3% West basin: 0-8%	Filter feeder commonly found in the littoral zones of lakes attached to plant surface (Korosi & Smol, 2008)

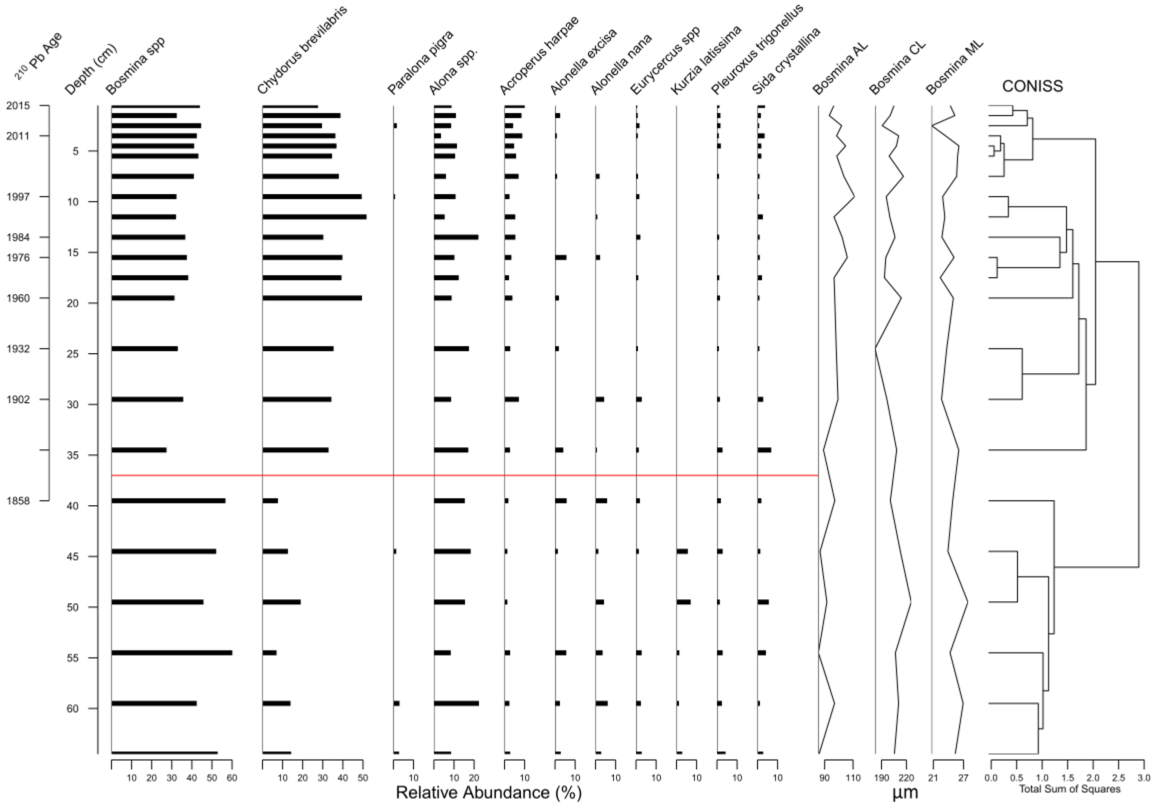


Figure 5.6. Stratigraphic profile showing changes in cladoceran species relative abundances (%) and *Bosmina* size measurements (AL = antennule length; CL = carapace length; ML = mucro length) with sediment core depth (in cm) for Port Perry Bay (western arm). Sediment age shown on the left is based on the ^{210}Pb Constant Rate of Supply model (Appleby 2001) but note that loss-on-ignition analysis infers a date of 1834 at 40 cm corresponding to the construction of the Lindsay Dam and the formation of present-day Lake Scugog. The results of the constrained incremental sum of squares analysis are shown on the right, and the red line indicates a significant assemblage shift assessed using the broken stick model (Grimm 1987).

CHAPTER SIX: DISCUSSION

6.1 Overview

My study aimed to provide an historical snapshot of Cladocera assemblages and changes through time for Lake Scugog. I used paleolimnological techniques to understand how the interaction of these stressors such as eutrophication, invasive species and food web has driven ecological change in Lake Scugog over the last several hundred years of its history, including the period prior to its formation following the construction of the Lindsay Dam. My results showed that *Bosmina* spp. and *Chydorus brevilabris* were the dominant taxa throughout the entire sediment core from both the east and west arms. In a pan-Canadian study of Cladocera in surface sediments, these taxa were associated with lakes that had high human impact in the watershed (Paquette et al., 2022), similar to Lake Scugog that has a watershed heavily impacted by agriculture, as well as urbanization in the Town of Port Perry.

Bosmina was the only true planktonic cladoceran species found in the sediments of Lake Scugog. *Bosmina* is a species complex of small-bodied filter feeders that includes *Eubosmina longispina* and *Bosmina longirostris* in this region (Adamczuk, 2015). They are one of the most recognizable species of Cladocera due to the round shape of the carapace and notable appendages (Adamczuk, 2015). Zooplankton surveys performed in the nearshore areas of Lake Scugog similarly found that *Bosmina* were dominant in the zooplankton community, along with nauplii copepods and cyclopoids (Smith & Kirkwood, 2022); However, the large-bodied planktonic cladoceran *Daphnia* was also recovered in nearshore areas of Lake Scugog (Smith & Kirkwood, 2022), though I did not recover their subfossil remains in the sediments. This is probably because *Daphnia* abundance is low, and *Daphnia* only leave small remains behind in the sediments, meaning that they are often underrepresented in the sediment core (Szeroczyfiska

& Sarmaja-Korjonen, 2007; Nykänen et al., 2009). Hard-shelled cladocerans such as *Bosmina* and chydorids are well preserved in sediments from multiple body parts, including headshields, carapaces and postabdomens (Nykänen et al., 2009). In contrast, taxa such as *Daphnia* are represented only by smaller fragments (postabdominal claw, caudal cerca and mandibles (Jeppesen et al., 2001)).

Daphnia abundances in Lake Scugog are likely low because Lake Scugog is a shallow lake that has a high intensity of fish planktivory. *Daphnia* are not typically abundant when fish planktivory is high because visual, non-gape limited predators like planktivorous fish preferentially feed on larger-bodied organisms like *Daphnia* (Brooks & Dodson, 1965). *Daphnia* were recovered in sediment cores from three other lakes (Cameron, Pigeon and Stoney) in the Kawartha region (Li, 2021), but these lakes are deeper than Lake Scugog (mean depth=3-6.3 m; maximum depth = 17.4-32m). *Bosmina* and *Daphnia* have different roles in energy transfer in the food web. *Bosmina* is thought to play a more minor role in energy transfer through aquatic food webs than *Daphnia* due to its small body size (Adamczuk, 2016). *Daphnia* are efficient consumers of most phytoplankton, and they are also a major source of food for planktivorous fish (Reynolds, 2011), although this may not be the case for Lake Scugog with abundant *Bosmina* populations and limited *Daphnia* abundance.

C. brevilabris attaches to substrates as an algal scraper, and also has weak swimming ability that allows them to expand into open-water habitats and filter-feed on phytoplankton, detritus and bacteria (de Eyto, 2001). *C.brevilabris* are often abundant in eutrophic and hypereutrophic lakes (Bos & Cumming, 2003; Nevalainen & Luoto, 2013; Jones et al., 2022), though as generalist taxa, they can also dominate in oligotrophic Arctic lakes (Thienpont et al., 2015). Jones et al. (2022) documented a shift from *Bosmina* spp to *C.brevilabris* and *D. pulex* spp and a decrease in specialist littoral species such as *Alona affinis* in several lakes located near mink farms in Nova Scotia, Canada, inferred as a response to increasing nutrient loading.

The remaining species in the cladoceran assemblage comprised several littoral and benthic taxa, with *Acroperus harpae*, *Alonella nana*, *Alonella excisa*, *Eurycercus*, *Kurzia latissimi*, *Paralona pigra*, *Leydigia* spp. and *Sida crystallina* being the most abundant. Species richness was higher in the sediment core from the west arm (19 species) compared to the east arm (13 species). The east and west arms of Lake Scugog differ in terms of depth (the east arm is more profundal than the west arm) and land development (the west arm has more intense urban and agricultural land development than the east arm), which influences water quality. The west arm is generally considered more degraded than the east (Kawartha Conservation, 2010). The west arm receives nutrients and other pollutants from the Nonquon River and stormwater runoff from the Town of Port Perry. Shallower lake depths and higher nutrient concentrations result in higher macrophyte biomass in the west arm, providing more habitat for littoral/benthic cladocerans that supports higher species richness compared to the east arm. This contrasts somewhat with a study of modern zooplankton communities in the nearshore environments of Lake Scugog, where zooplankton species diversity showed no relationship to macrophytes (Smith & Kirkwood, 2022). However, Smith & Kirkwood (2022) used net tows to collect zooplankton, rather than examining subfossils in sediment cores as I did. Net tows are biased towards planktonic species because they do not reliably recover littoral and benthic cladocerans, as they tend to live attached to substrates like macrophytes. Subfossil remains in lake sediments have been shown to provide a better representation of littoral and benthic Cladocera compared to plankton surveys, which is where most of the diversity for cladocerans exists (Frey, 1960).

6.2 Cladocera assemblage responses to reservoir formation

Reservoir formation causes water levels to increase, results in a trophic upsurge (an episode of increased productivity and biovolume of the plankton that is attributed to the release of nutrients from flooded soil and vegetations; Straškraba and Tundisi, 1999), and changes in temperature, and is expected to alter the species composition from riverine to more lake-like (lotic to lentic) (Agostinho et al., 2008). After the reservoir lake's formation, the water is usually turbid and nutrient-rich due to the decomposition of remnant vegetation, promoting plant growth and periodic algal blooms (Kennedy & Thorntom, 2001; Schmidt et al., 2020). Environmental changes that follow dam construction can persist for decades, and often overlaps with further watershed development activities. In Lake Scugog, a clear signal of the trophic upsurge is seen in the detection of okenone pigment at the bottom of the core (Korosi et al., 2023), a pigment produced by purple sulfur phototrophic bacteria (Family Chromatiaceae) that live in anoxic habitats such as stagnant waterbodies (Madigan & Jung, 2009). Okenone wasn’t detected above 30 cm (~1900), which probably indicates the end of the trophic upsurge (Korosi et al., 2023).

Bosmina and *C. brevilabris* were the dominant taxa in Lake Scugog before and after the construction of the Lindsay Dam and formation of present-day Lake Scugog. Cladocera assemblages in the east arm did not show any apparent changes following the construction of the Lindsay dam. Similarly, diatoms also did not show any assemblage changes in the east arm corresponding to the timing of the construction of the Lindsay Dam, although there was a significant change prior to this at 55cm (Do, 2021). In contrast, in the west arm, there was a shift in species assemblages towards increased relative abundance of *Bosmina* and decreased relative abundance of *C. brevilabris* at 35cm after the construction of the Lindsay dam. The different responses of *C. brevilabris* and *Bosmina* in the west and east arm of Lake Scugog may be due to differences in mean depth between the two locations. A similar shift from *C. brevilabris* to *Bosmina* and was documented in sediment cores from three other Kawartha Lakes along the after the creation of the Trent- Severn Waterway, which was interpreted to be a result to the increase in lake nutrient concentrations following reservoir flooding (Li, 2021).

Other groups of zooplankton that do not leave subfossil remains in the sediments may have been altered in Lake Scugog following the construction of the Lindsay Dam, such as rotifers and copepods. For example, a study by Schmidt et al (2020) examined the effects of damming on zooplankton community composition (rotifers, copepods, and cladocerans) before, during and after the construction of a reservoir from the hotspot Cerrado to Amazon in Central Brazil. The results found that 164 taxa were identified in all of the samplings. The most common were Rotifers (102 taxa), followed by cladocerans (41 taxa) and copepods (21 taxa). Before the formation of the dam, there was 71 taxa of rotifers that were present, 19 taxa of copepods and 30 taxa of cladocerans. After the formation of the dam, rotifer taxa decreased to 68, copepods decreased to 15 taxa and cladocerans to 27. The results indicate that rather than showing the well-described pattern of compositional changes to different species due to damming, the study indicates that changes are directional, promoting certain species during and after the lake's formation (Schmidt et al., 2020). The typical zooplankton species after the formation of the lake are *Asplanchna brightwellii* (Schmidt et al., 2020). A study by Missias et al (2017) suggests that rotifers, copepods and cladocerans respond differently to spatial ecological gradients in reservoirs. Copepods and cladocerans might respond the same as they are phylogenetically closer (both crustaceans) and they have more similar ecological niches.

6.3 Cladocera assemblage responses to multiple stressors in Lake Scugog

Multiple stressors have been impacting Lake Scugog since its formation following the construction of the Lindsay Dam, including climate change, eutrophication, invasive species, road salt, and competing demands for water usage (Kawartha Conservation, 2010). These multiple stressors can lead to enhanced lake primary production and associated changes in habitat, food supply, and water clarity, potentially resulting in compositional changes in cladoceran communities. In 1903-1904, the first reported massive fish kill occurred, 1970 was the first introduction of invasive Eurasian Watermilfoil into the lake. In 1971, the Nonquoin Water Treatment Plant was created. The first appearance of zebra mussels was in 1991, and in 2015, the invasive microalga Starry Stonewort became established in the lake.

Based on diatom subfossil assemblage changes, climate change has been altering the thermal regime of the east arm since ~1950. *Aulacoisera ambigua* was the dominant tychoplanktonic diatom prior to the 1930s, and the assemblage transitioned from *Aulacoisera* to *Cyclotella* starting in 1950, a hallmark response to enhanced thermal stratification and a longer ice-free period (Do, 2021). Cladocera assemblages do not appear to have responded to the changes in thermal structure and ice cover regime, and in fact Cladocera assemblages have remained stable over the entire history of present-day Lake Scugog in both the east and west arms in spite of multiple interacting stressors. *Bosmina* and *C. brevilabris* are both generalist taxa that thrive in eutrophic, human- impacted systems, which may account for the lack of assemblage changes I saw in Lake Scugog. In a meta-analysis of subfossil Cladocera assemblages across Canada and Alaska (Labaj et al., 2021) *Bosmina* spp and *C. brevilabris* were abundant in all eco-regions, thriving in all environmental conditions, although they did show that

C. brevilabris was the taxon associated with pH, Ca, TP and DOC, and *Bosmina* was mainly associated with larger and deeper lakes.

6.4 Temporal trends in Bosmina size structure

Bosmina and *Daphnia* subfossil size structure can also be used to track changes in food web structure, including the strength of planktivorous fish predation (Korosi et al., 2013). *Bosmina* subfossil remains in Lake Scugog are some of the smallest recorded in the literature, and *Bosmina* subfossils in the west arm were smaller than in the east arm. In the east arm, antennule length ranged from 120 µm- 125 µm, carapace length ranged from 200-240 µm and mucro length ranged from 20-45 µm. In the west arm, *Bosmina* antennules ranged between 90-110 µm, the carapace ranged between 190-220 µm, and mucro length ranged between 20- 27 µm. Size measures for *Bosmina* subfossils in other Kawartha Lakes (Cameron, Pigeon and Stoney Lake) were larger, they ranged from 40-60 µm for mucro length, 130-170 µm for antennule length, and 180-240 µm for carapace length (Li, 2021). Regional size ranges for *Bosmina* subfossil measurements in surface sediments of 50 south-central Ontario lakes were 80-200 µm for antennule length, 200-400 µm for carapace length, and 30-100 µm for mucro length (Korosi et al., 2008). Regional averages for *Bosmina* subfossil measurements in surface sediments of Nova Scotia lakes were 129.89 ± 22.27 µm for antennule length, 271.69 ± 43.02 µm for carapace length, and 90.82 ± 25.44 µm for mucro length (Korosi & Smol, 2011). *Bosmina* subfossil size measures from two Lake Ontario coastal wetlands ranged from 130-350 µm for carapace length, 10-50 µm for mucro length, and 55-140 µm for antennule length (Hoskin & Korosi, 2021).

Bosmina in Lake Scugog are likely small because fish planktivory is high, especially in the west arm. Predation on *Bosmina* occurs from two primary sources: macroinvertebrates and planktivorous fish. The fish are visual predators, who actively select the bigger-sized *Bosmina* to consume. Larger *Bosmina* with long appendages are expected to be more heavily preyed upon than smaller *Bosmina* when fish planktivory is high, theoretically inducing a decrease in overall bosminid size structure (Korosi et al., 2013). In lakes with abundant planktivorous fish, *Bosmina*, may experience some pressure for decreased size as a because it decreases their visibility, while lakes that are fishless may have a stronger selection for larger *Bosmina* sizes and longer mucro and antennule lengths that reduce vulnerability to gape-limited invertebrate predators (Korosi et al., 2013). For example, a study conducted by Labaj et al (2013) examined surface sediments from 29 remote small, shallow lakes in northwestern Ontario, and compared *Bosmina* size and morphology against fish presence/absence inferred from *Chaoborus* remains. The results showed that an increase in mucro length was associated with the presence of *Chaoborus* invertebrate predation and the absence of fish (Labaj et al., 2013).

Bosmina size measures in Lake Scugog did not change following reservoir formation, eutrophication, climate warming, or changes in fish communities. In a study by Korosi et al (2010), *Bosmina* size measures were compared between sediment intervals deposited recently and sediments deposited pre-1850 in 44 lakes in south-central Ontario. The study's results showed that the length of *Bosmina* body appendages (mucrones and antennules) was longer in pre-industrial times than at present (AL: ~125 µm versus 114 µm; ML: 75 µm versus 73 µm). The *Bosmina* appendage reduction was interpreted as reflecting a possible reduction in copepod predation pressures since pre-industrial times (Korosi et al., 2010). In Cootes Paradise, a highly impacted Lake Ontario coastal wetland, there was a statistically significant increase in the median *Bosmina* antennule length beginning prior to 1900, while in Jordan Harbour, another highly impacted Lake Ontario coastal wetland, there was a significant increase in antenntule length and a decrease in carapace length between ~1900 and 2017 (Hoskin & Korosi, 2021).

In contrast to the studies described above, *Bosmina* size measures in Lake Scugog did not show any statistically significant changes in either the east and west arm, despite the known changes to fish communities. This suggests that even though fish communities have changed, the relative strength of fish planktivory on *Bosmina* has stayed consistent.

CHAPTER SEVEN: CONCLUSION AND FUTURE WORK

7.1 Conclusion

The objective is to provide a historical snapshot of Cladocera assemblages and diversity for Lake Scugog before and after historical events. Another objective is to examine food web changes in Lake Scugog using *Bosmina* size measurement. Based on the results, Cladocera assemblages do not appear to have responded to the changes in thermal structure, ice-cover regime, and other ecosystem changes that have occurred over the time period encompassed by this study. In fact, Cladocera assemblages have remained stable over the entire history of present-day Lake Scugog in both the east and west arm in spite of multiple interacting stressors. *Bosmina* spp. and *C. brevilabris* are both generalist taxa that thrive in eutrophic, human-impacted systems, which may account for the lack of assemblage's changes in Lake Scugog.

When looking at food web changes in Lake Scugog, *Bosmina* size measures in Lake Scugog did not show any notable changes in either the east and west arm, despite the known changes to fish communities. *Bosmina* subfossil remains in Lake Scugog are some of the smallest and the *Bosmina* subfossils in the west arm were smaller than in the east arm. *Bosmina* in Lake Scugog are likely small because fish planktivory has always been high. *Daphnia* abundance in Lake Scugog are likely low because Lake Scugog is a shallow lake that has a high intensity of fish planktivory

Ecological indicators are indicators that are able to identify potential issues before they cause widespread ecological degradation. A key characteristic of good ecological indicators are that are sensitive to changes in the environment. Ecological indicators help monitor water quality and the ecological integrity in Lake Scugog, and would react to changes that are occurring in the lake. Overall, *Bosmina* and *C. brevilabris* do not appear to be sensitive paleoecological indictors of ecosystem changes in Lake Scugog, as they did not showcase changes to nutrient input temperature, eutrophication changes or any other stressors that they lake was subjected too. However, other indicators (e.g. diatoms, chironomids, pigments) have pointed towards significant changes in Lake Scugog (Korosi et al., 2023).

7.2 Future Work

There were no *Daphnia* subfossils in the sediment although they were found in low abundances in modern plankton sampling. Additional study comparing Cladocera in sediments to modern zooplankton assemblages would be helpful for understanding biases in preservation that may impact interpretations of changes in zooplankton communities over time. Additionally, by examining zooplankton in both water samples and sediment subfossils, a greater understanding of their responses to ecosystem stressors can be gained. Complementary sediment-based cladoceran studies with water column sampling of Cladocera, incorporating multiple years, months and seasonality, could provide a better picture of cladoceran community dynamics in Lake Scugog. This future work could also focus on the ecological reasons behind shifts between *Bosmina* and *C. brevilabris* dominance, as I did observe subtle changes between these two dominant groups. This would help understand where or not changes in Cladocera diversity and abundance occur with seasonality, aquatic plant biomass, and fish communities, which would improve the indicator value of Cladocera subfossils. Increasing the number of Cladoceran counts would result in more representation from the littoral and benthic species, where most of the diversity in Cladocera is found. This might show impacts to cladoceran assemblages that I was not able to pick up in my study following standard counting guidelines. Future work will also include multi-proxy study, examining cladoceran changes in relation to other biological and geochemical proxies to have a clearer understanding of multiple stressors in Lake Scugog.

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