CLADOCERAN SUBFOSSILS AS INDICATORS OF ECOSYSTEM RESPONSES TO MULTIPLE STRESSORS IN LAKE ONTARIO (CANADA) COASTAL WETLANDS

by

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Abstract

This thesis explores the use of Cladocera (Branchiopoda, Crustacea) subfossil remains preserved in sediment cores as potential ecological indicators of wetland health in three coastal wetlands of Lake Ontario (Canada). Great Lakes coastal wetlands are crucial for supporting wildlife and filtering pollutants and sediments from water. As watershed development intensifies, anthropogenic stressors impacting wetland health since European settlement in ~1850 remain a fundamental concern. Subfossil Cladocera were analyzed in McLaughlin Bay (Oshawa), Cootes Paradise (Hamilton), and Jordan Harbour (Lincoln). Subfossil Cladocera assemblage changes, particularly decreases in the abundances of littoral taxa, appeared to track declines in aquatic macrophyte coverage resulting from invasive carp, high turbidity, and poor water quality. Some evidence of recent ecosystem recovery was evident in Jordan Harbour, but not in McLaughlin Bay or Cootes Paradise. Overall, paleolimnological approaches can provide a historical context to guide future management and restoration of Great Lakes coastal wetlands.

Co-Authorship

This thesis examines the application of paleolimnology in Great Lakes coastal wetlands impacted by anthropogenic stressors along the Canadian shores of Lake Ontario, and is based on the following two manuscripts. Each chapter is presented in the format required by the journal to which it is to be submitted.

Chapter 2: Hoskin, G. N., Do, P. H. P., Coleman, K. A, Thienpont, J. R., Kirkwood, A. E. and Korosi, J. B. A paleolimnological approach for evaluating a proposed restoration strategy in a highly degraded Lake Ontario coastal wetland. Formatted for eventual submission to the Journal of Great Lakes Research.

I designed this study under the guidance of my supervisor, Prof. Jennifer Korosi, as well as Prof. Joshua Thienpont (Dept. of Geography, York University) and Prof. Andrea Kirkwood (Ontario Tech University). I conducted the analysis for ^{210}Pb sediment core dating, cladoceran subfossils, lab analysis, and historic air photos. The diatom analysis was conducted by Pham Ha Phuong Do, with scientific input from Kristen Coleman (PhD student, Dept. of Geography, York U), as part of Pham Do's undergraduate honour's thesis in the Dept. of Geography at York University. I co-designed and co-mentored Do's honour's thesis research in collaboration with Dr. Korosi and Kristen Coleman. I conducted all final analyses, produced the final figures, and wrote the manuscript, with editorial input from Dr. Korosi.

Chapter 3: Hoskin, G. N. and Korosi, J. B. Long-term ecosystem change in two highly degraded Lake Ontario (Canada) coastal wetlands. Submitted to the Journal of Paleolimnology.

I co-designed the project with my supervisor, Dr. Jennifer Korosi, and conducted all of the fieldwork and laboratory analysis for it. I wrote the manuscript and produced the final figures with editorial and scientific input from Dr. Korosi.

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List of Abbreviations

CL – *Bosmina* carapace length

ML – *Bosmina* mucro length

AL – *Bosmina* antennule length

PCA – Principal Components Analysis

CONISS – Constrained Incremental Sum of Squares cluster analysis

CRS – Constant rate of supply

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1.0 General introduction and literature review

1.1 Laurentian Great Lakes coastal wetlands

Great Lakes coastal wetlands are located along the shores of the Laurentian Great Lakes (Lake Superior, Lake Michigan, Lake Erie, Lake Ontario, and Lake Huron). Coastal wetlands are an integral component of the larger Laurentian Great Lakes ecosystem, and are directly influenced by the waters or connecting channels of the Great Lakes (United States Environmental Protection Agency, 2017). There are many types of coastal wetlands in the Laurentian Great Lakes region, however in southern Ontario they are classified primarily as barrier beach wetlands, or drowned river-mouth wetlands (Albert et al., 2005). Barrier beach wetlands are protected by a strip, or barrier, of land separating them from the wave action of the Great Lakes, although they can be either completely hydrologically isolated, or connected to the lake via a narrow channel (Albert et al., 2005). Drowned river-mouth wetlands are influenced by both the channels of the rivers flowing into them, as well as the Great Lakes waters (Albert et al., 2005). Both barrier beach and drowned river-mouth wetlands act as essential transition zones between upland ecosystems and the Great Lakes waters for many natural biological, physical, and environmental processes (United States Environmental Protection Agency, 2016).

Coastal wetlands are ecologically significant due to the beneficial ecosystem services they provide (Sierszen et al., 2012). For example, they act as habitat for key plants and animals, including many endangered avian, aquatic, and amphibian species that are unique to these environments (Mayer et al., 2007). Nutrient cycling, which is the movement of both organic and inorganic matter back into the production of living things, as well as hydrologic retention, which is the storage of water, are additional ecosystem services provided by coastal wetlands. Furthermore, Great Lakes coastal wetlands provide shoreline protection from Great Lakes wave action, and act as sediment traps by containing the sediment from surficial runoff heading towards the lakes (United States Environmental Protection Agency, 2018). They are also important for recreational and cultural activities, including kayaking, canoeing, fishing, and picking berries, among many others (United States Environmental Protection Agency, 2018).

Many of the Laurentian Great Lakes coastal wetlands are located within urban and suburban catchments, where they have been subject to multiple, intense anthropogenic stressors since European settlement. Anthropogenic activities have led to a loss of up to 90% of coastal wetlands in some areas (Environment Canada, 2002), and remaining wetlands are threatened by stressors such as lake-wide water level regulations which prevent some wetland vegetation from recolonizing, while promoting the spread of invasive alien species, as many plant species rely on the natural fluctuation of lake water to disperse seeds and grow (Wilcox and Nichols, 2008). This has occurred in many Laurentian Great Lakes coastal wetlands, and is very prominent in at least 14 northern Great Lakes wetlands where *Typha X glauca* was able to colonize the wetlands following multiple years of low water levels and the formation of barrier beaches in previous higher water level events, which then persisted during low water levels (Lishawa et al., 2010). High development pressures in urban wetlands include landscape and watershed alterations like agricultural practices and the damming of rivers (Bassi and Kumar, 2017), and habitat fragmentation as a result of urbanization (Faulkner, 2004; Johnson et al., 2012). Heavy motorized boat traffic can increase turbidity, and also tear up submerged and emergent aquatic vegetation (Sagerman et al., 2019), further exacerbating turbidity.

Increased sedimentation is a major threat to wetland ecosystem health resulting from watershed alterations that can increase turbidity and decrease aquatic macrophyte growth (van Hengstum et al., 2007). For example, in Frenchman’s Bay (Pickering, Ontario), increasing sedimentation rates of finer grained sediments like silts and clays was observed since ~1850 (± 56) in a paleolimnological study, attributed to the progressive loss of native vegetation in the surrounding watershed, and an increase in erosion from Lake Iroquois sediments being deposited via stream transport (Hengstum et al., 2007). The increased input of fine-grained sediments led to a decrease in water clarity, as well as contributed to overall wetland habitat loss in Frenchman’s Bay.

Waste disposal and toxic effluents have also historically been an issue in wetlands, particularly those in urban catchments, which further decrease water quality and make the wetlands inhospitable to species seeking a protected area to spawn or breed and feed (Vymazal, 2012). An example of how coastal wetlands have been used for waste disposal is in Ashbridge’s Bay, Toronto, where sewage was disposed of for decades, to the point where it needed to be dredged to alleviate some of the growing health concerns (Bonnell, 2009). Furthermore, road salt runoff is emerging as a major issue in urban and suburban wetlands (Meriano et al., 2009), as it contributes to elevated conductivity values to levels that are toxic for many native wetland flora and fauna (Herbert et al., 2015).

Invasive species are a major threat to Great Lakes coastal wetlands (Chow-Fraser, 1998; Morrice et al., 2008), particularly *Phragmites australis* (the common reed) and *Cyprinus carpio* (invasive common carp), due to their adaptability and higher tolerance for poor water quality, allowing them to easily outcompete native species (Chow-Fraser, 1998). *P. australis* in particular can spread quickly and is more tolerant of variable environmental conditions than the native reeds typically found in coastal wetlands (Tulbure and Johnston, 2010). *C. carpio* have been found to have no direct effect on zooplankton communities, however their impact on water turbidity, nutrient loads, and the native plant communities can indirectly influence the zooplankton communities. Research suggests that when turbidity is greater than 20 NTU, there are less than 5% of species of submergent plants (Lougheed et al., 1998). Most remaining wetlands in the Great Lakes region are considered degraded or, at the very least altered to some degree, and these wetlands are dominated by silt and turbidity-tolerant fish species like *C. carpio* (Jude and Pappas, 1992). Further obervations of the impacts of the common carp suggest that the invasive fish influence water quality in a similar manner to a lake or wetland undergoing cultural eutrophication, and that they can increase water column nutrient concentrations, suspended solids, and chlorophyll *a*¸ while decreasing dissolved oxygen concentrations, submerged macrophyte density, and photic depth (Badiou and Goldsborough, 2010).

As a result of the myriad of stressors impacting Great Lakes coastal wetlands, various local and regional monitoring and restoration projects have been initiated primarily since the early 2000s. In the highly developed watersheds of southern Ontario, this includes the Durham Region Coastal Wetland Monitoring Project, which began in 2002 and focusses on assessing the changes induced by human activities, as well as the overall heatlh of wetlands, from Pickering to Clarington (Central Lake Ontario Conservation Authority, 2013). It is carried out in partnership with the Central Lake Ontario Conservation Authority, Toronto and Region Conservation Authority, Ganaraska Region Conservation Authority, and Environment Canada. As our understanding of ecological dynamics and health of coastal wetlands improves, it will generate new opportunities for further remediation.

Remediation activities are partially guided by the Great Lakes Coastal Wetland Monitoring Program that began in 2010 (United States Environmental Protection Agency, 2017). It is classified as a binational monitoring program between Canada and the United States, which aims to report and better understand the present status of wetlands falling within the Laurentian

Great Lakes, both by type of wetland and the lake it is located along, as well as temporal trends (United States Environmental Protection Agency, 2017). The program accomplishes this by assessing the biological, physical, and chemical conditions of the wetlands by use of fish, birds, amphibians, vegetation, macroinvertebrates, and water quality indices (United States Environmental Protection Agency, 2019). The results of this monitoring program are used to better prioritize remediation, protection, and future monitoring by resource managers and decision-makers (United States Environmental Protection Agency, 2019).

The Coastal Wetland Monitoring Program provides data on the current health of many at-risk coastal wetlands. However, it does not offer a long-term perspective on the ecological changes that have occurred in the past, which continue to influence the status of wetlands today. Understanding past changes and ecosystem dynamics is necessary to better understand ecosystem resilience in response to multiple interacting anthropogenic stressors. A temporal perspective assists in determining if ecosystem health is in decline from its previous state, what the former conditions were, and if any recovery has occurred in response to remediation initiatives (Smol, 1992; Willis and Birks, 2006). An analysis of the long-term changes is also beneficial in determining if remediation goals should aim to achieve the wetland's past conditions or anticipate a new ecosystem structure (Willis and Birks, 2006), while a spatial perspective helps to provide context to present-day conditions in response to current and recent anthropogenic stressors. Great Lakes coastal wetlands have been impacted by humans since early European settlement, and there are no monitoring programs that extend this far back. In absence of long-term monitoring, lake sediment cores can be used as an indirect method to reconstruct the missing long-term perspective for evaluating past ecosystem conditions and long-term ecological change (Willis and Birks, 2006).

1.2 Paleolimnology

The field of paleolimnology analyzes dated sediment cores from depositional basins in aquatic ecosystems to study the physical, chemical and biological characteristics of a lake or wetland through time (Saulnier-Talbot, 2016). Following the Law of Superposition, where sediment layers on the surface are younger than those layers below (Watson, 1997), paleolimnology allows for the layers of sediment to essentially be used as pages in a history book, and the physical, chemical and biological characteristics analyzed tell us what was happening during the time that the sediment was deposited. Water turbulence moves sediment from shallower areas to the deeper zones within a lake, known as sediment focusing (Blais and Kalff, 1995), which in most cases, allows for a single sediment core extracted from the deepest basin in aquatic ecosystems to be representative of changes occurring in the waterbody through time. Paleolimnological studies are also highly reproducible (Heggen et al., 2012), meaning that the results from a single core collected from the deepest basin of a waterbody can be replicated when examining additional sediment cores collected from the same location.

Sediment cores can be sectioned into 0.25-1 cm intervals to provide a high temporal resolution (~2-10 years) on ecosystem changes over centennial to millennial timescales, and recent (last ~200 years) sediments can be dated using ^{210}Pb radioisotopes to determine the approximate age of the sediment (Appleby, 2001). There are two models that can be applied to derive an age-depth model from ^{210}Pb activities: the Constant Rate of Supply (CRS) model, and the Constant Initital Concentration (CIC) model. The CRS model assumes that the influx of ^{210}Pb from the surrounding environment is constant despite changing sedimentation rates (Lubis, 2006), and therefore this model is adopted when the amounts of sedimentary inputs are expected to have varied throughout time, i.e. sedimentation rate has not been constant through time (de

Souza et al., 2012). The CIC model assumes that both the sedimentation rate and ^{210}Pb inputs have remained constant throughout time (Lubis, 2006; de Souza et al., 2012). For sediments older than ~200 years, age-depth models can be reconstructed based on ^{14}C dating (Lougheed et al., 2017).

Two primary coring methods are used to collect sediment records: gravity cores, which are mainly used for surface sediments dating back ~300 years, and piston cores, which are used for much longer (millennial) time spans. Gravity cores (Glew et al., 2001) rely on a weight at the bottom of the corer to enable the tube to sink through the mud, and the top of the core is then plugged to allow the sediment to be pulled to the surface (much like the action of capping a straw in water with your finger). However, if the sediment is very sandy or dense, it can be difficult for the corer to sink into the sediment. In these cases, an additional weight with rope attached, known as hammer action, can be used to essentially hammer the corer and tube further into the mud (Glew et al., 2001). Gravity cores can be collected from the surface of the water in a boat, as they do not require a stable platform to collect. Piston cores (Glew et al., 2001), in contrast, require a stable surface, generally lake ice or four floation rafts to create a stable platform for sediment coring. These corers work by being manually pushed into the sediment from the stable platform, as opposed to using gravity and weights like the gravity corer.

There are various physical, chemical, and biological proxies that are used to reconstruct temporal changes in aquatic ecosystems. Diatoms, which are a group of silicious algae, are effective indicators of pH, nutrients, and temperature and light regimes in waterbodies (Battarbee et al., 2001). Chironomids (larval midges) are useful for reconstructing air temperatures, salinity, lake productivity, and hypolimnetic anoxia (Walker, 2001). Pollen and plant macrofossils are used for inferring environmental, climatic, and vegetation changes throughout time (Birks, 2002), and macroscopic charcoal are used to reconstruct fire regimes (Enache and Cumming, 2006). Some physical proxies include sediment grain size, type, and magnetism, which are effective indicators of past climate, and human activity (United States Geological Survey, 2020). Examples of chemical proxies include stable isotopes and biomarkers, which can be useful for reconstructing past climate and temperature fluctuations (United States Geological Survey, 2020). My thesis primarily uses Cladocera (Crustacea: Branchiopoda) as a biological proxy for ecological change in Great Lakes coastal wetlands. Cladocera occupy an intermediate position in aquatic food webs (Jeppesen et al., 2001), making them sensitive to both bottom-up and top-down ecosystem controls. Great Lakes coastal wetlands have well-documented changes in aquatic macrophytes and fish communities in some areas, making Cladocera a useful tool for assessing how these changes have influenced historical and current wetland composition. Importantly, the study of cladoceran subfossils to reconstruct changes in coastal wetlands through time can also link with current monitoring efforts, as the zooplankton index for wetland quality is one of the key water quality indicators used in the Coastal Wetland Monitoring Program (Lougheed and Chow-Fraser, 2002).

1.2.1 Cladoceran subfossils as indicators of ecosystem change

Cladocera are a taxonomic Order of microinvertebrates that are an important component within the larger crustacean zooplankton community in lakes, but also include species living in littoral and benthic habitats. Their exoskeletons are made of chemically inert chitin, and as such, cladoceran subfossil remains are well represented in lake sediments, and are useful biological indicators of ecosystem change (Ejsmont-Karabin and Karabin, 2013). Cladocera graze on phytoplankton, helping to regulate primary productivity (Nevalainen et al., 2018; Tolotti et al., 2016). They are also an important prey source for larger aquatic macroinvertebrates and planktivorous fish, and it was found that the removal of large filter-feeding herbivorous Cladocera via planktivorous fish predation can lead to worsened environmental conditions within a eutrophic lake (de Bernardi et al., 1987). The Order Cladocera is made up of more than 600 species globally (Korovchinsky, 1996), and each species lives in specific microhabitats, which includes pelagic, bottom substrate, and attached to vegetation (Nevalainen and Luoto, 2016). For example, there are many species that live associated with macrophytes, such as *Pleuroxus* spp. (Korhola and Rautio, 2001), and *Leydigia* spp. are mud-dwellers that tend to be found beneath aquatic macroyphytes (Korhola and Rautio, 2001). There are also many open water species, particularly *Bosmina* spp., which are strictly planktonic and can migrate vertically throughout the water column (Makinol et al., 1996). *Daphnia* spp. tend to be found in open water as well, however due to their large size making them a desirable food source for visual-feeding predators, they have been known to associate with macrophytes as a predator-avoidance strategy (Lougheed et al., 2004). Cladocera also have specific feeding guilds, where they can be divided into three main groups: filterers, scraper-detritivores, and predators (Nevalainen and Luoto, 2016). Some examples of filter-feeders include *Bosmina*, *Daphnia*, and *Sida crystallina*, and a well-known predatory cladoceran is *Leptodora*, which preys upon smaller Cladocera (Nevalainen and Luoto, 2016).

Cladocera preserve well in sediments due to their chemically inert chitinous exoskeletons. Cladocera disarticulate upon death, and can be identified by a variety of body parts including headshields, carapaces, postabdominal claws, postabdomens, mandibles, and ephippia (a resting stage); however, some taxa preserve better than others. Chydorids and bosminids are well represented in sediments by their headshields and carapaces, whereas *Daphnia* headshields/carapaces do not preserve, and instead are identifiable from their small postabdominal claws and ephippia, due to variable levels of chitin hydration among the different taxonomic groups (Korhola and Rautio, 2001). Multiple studies have been done that compare the subfossil Cladocera assemblage to Cladocera assemblages collected through contemporary sampling, in order to understand the potential impacts of preservation bias on Cladocera-based paleolimnological reconstructions. Differential preservation can contribute to an underrepresentation of *Daphnia, Diaphanosoma, Ceriodaphnia, Limnosida,* and *Leptodora* in the sediment record compared to the living populations, while *Bosmina* and *Chyordus* can be overrepresented (Korhola and Rautio, 2001). Despite this, however, subfossil Cladocera still provide a reliable snapshot of changes in cladoceran communities through time (Frey, 1960; Cotton, 1985; Korhola and Rautio, 2001). In fact, subfossil records are often more accurate representations of the cladoceran community than contemporary sampling of zooplankton, since sediments incorporate seasonal fluctuations in cladoceran populations that are often missed in contemporary sampling that typically occurs only once or twice a year (Korhola and Rautio, 2001). Furthermore, littoral cladocerans are often more easily identifiable from their subfossil remains, since diagnostic features like headpore arrangement are more visible in subfossils (Frey, 1960).

Cladocera are sensitive to changes in the abiotic and biotic environment, and as such, are useful paleoecological indicators for studying trophic changes, water chemistry alterations, and habitat changes (Korhola and Rautio, 2001). In Pocket Lake, (Yellowknife, Northwest Territories, Canada), Cladocera were more sensitive to arsenic contamination from historic gold mining activities than chironomids or diatoms (Thienpont et al., 2016). Cladocera have also been used to assess nutrient enrichment in 68 lakes with varying levels of total phosphorus (TP), as a weak relationship between TP and Cladocera was observed (Lotter et al., 1998). Lake eutrophication has been associated with increases in *Chydorus sphaericus* abundances, as well as the proportion of planktonic versus littoral/benthic cladocerans, and the overall abundance of cladocerans (Manca et al., 2007). They have also been used to investigate lake acidification and associated declines in lakewater calcium. Near-extirpations of calcium-rich *Daphnia* have been observed in softwater boreal lakes with declining calcium levels, with corresponding increases in *Bosmina* and *Holopedium* that have minimal calcium requirements (Jeziorski et al., 2008; Jeziorski et al., 2015; Korosi et al., 2012), as well as increases in *Bosmina* when *Daphnia* populations have declined. Lakes near Sudbury, Ontario exposed to intense acidification and metal contamination from mining and smelting activities in the 1960s showed pronounced changes in Cladocera assemblages that tracked industrial acitivity (Labaj et al., 2015).

Cladoceran remains have also been used to reconstruct changes in food web structure. *Bosmina* size structure in particular has been shown to be a promising paleoecological indicator for assessing historic changes in predator-prey dynamics (Korosi et al., 2013). *Bosmina* respond to the presence of different predators (e.g. fish versus invertebrates) by altering their morphology, including changes in the length and shape of appendages (antennules and mucro lengths), and overall body size. This process is known as cyclomorphosis, whereby cyclical or seasonal alterations occur in the phenotype of an organism like *Bosmina* through generational succession in response to changes in the dominant predators. It occurs in small aquatic invertebrates like Cladocera who produce several generations annually and reproduce primarily via parthenogenesis (Korhola and Rautio, 2001). Parthenogenesis is the process in which an egg is hatched without fertilization (Korhola and Rautio, 2001) unless environmental conditions become more degraded or overcrowding occurs, when they switch to sexual reproduction

(Korhola and Rautio, 2001). When predators consume *Bosmina* with predation-susceptible body shapes and sizes, the less-desirable (to predators) individuals are left to dominate and reproduce. For example, copepods prey on *Bosmina* with shorter mucro and antennules because they are easier to manipulate for access to the soft inner carapace (Kerfoot 1975; Post et al. 1995). It has been observed that in aquatic ecosystems where copepod predation is high, *Bosmina* tend to have longer mucros and antennules as a predator-deterrence strategy, and sometimes curved antennules as a defence mechanism (Kerfoot 1975; Post et al. 1995). Copepods also predominantly feed on smaller-bodied *Bosmina* (Brooks and Dodson, 1965; Nilssen, 1987); when copepod predation is high, larger-bodied *Bosmina* are more abundant.

Changes to the food web, therefore, can impact cladoceran species assemblage and *Bosmina* size structure (Korosi et al., 2012). Fish are visual feeding predators that typically select for larger prey, which can lead to reductions in overall body size of the Cladocera community (Brooks and Dodson, 1965; Nilssen, 1978). For example, there were observed decreases in *Bosmina* size following the introduction of planktivorous ice fish into lakes in China (Shi et al., 2016). Alaskan lakes with native salmon populations had larger *Bosmina* with larger appendages at high salmon densities, suggesting strong copepod predation with less juvenile salmon predation; however, in a lake with an introduced sockeye salmon population, *Bosmina* morphologies were observed to be smallest during periods of high salmon densities within the lake, suggesting less copepod predation on the *Bosmina* (Sweetman and Finney, 2003). As many of the negative impacts occurring in Great Lakes coastal wetlands are related to changes in fish communities, and fish are a primary management concern for wetlands, Cladocera also have the potential to be very effective paleoecological indicators in these systems.

1.2.2 Paleolimnology and lake/wetland management

Paleolimnology is a useful tool in determining "reference" conditions of an aquatic ecosystem, as well as the range of natural variability. Using sediment records to reconstruct past conditions provides researchers with the ability to determine what a particular ecosystem would have looked like "pre-impact," or prior to significant stressors, as well as the fluctuations that occur periodically as a result of natural processes. While primarily developed for use in lake ecosystems and management, paleolimnological inferences of "reference" conditions have also been successfully applied in wetlands for management purposes. Paleolimnology can be successfully applied and used to determine how and when humans began to impact a waterbody, in order to better inform present and future environmental decision making (Bennion et al., 2011). This information can provide researchers with better insight into whether their remediation goals are actually feasible, or if the ecosystem is too degraded for a full recovery, as well as whether returning a lake or wetland to its "natural" state is even suitable given surrounding watershed alterations.

Paleolimnology has been used to study nutrient enrichment in a eutrophic Baptiste Lake, (Alberta, Canada) resulting from anthropogenic activities, to better manage degraded aquatic ecosystems (Adams et al., 2014). Water quality measures were conducted from the early 1980s to the mid-2000s, and a sediment core was collected to study any changes occurring in the last ~150 years. The results of this study provided greater insight into the usefulness of nitrogen as an indicator of limiting primary production and a predictor of Chlorophyll *a*. Prior to this study, phosphorus was considered the most important limiting nutrient to primary production. Furthermore, this analysis offered a greater understanding of the timing of nutrient fluctuations and that, despite Baptiste Lake already being a eutrophic lake for at least the last ~150 years, anthropogenic impacts have likely increased nutrient loading over the past ~10 years. This research further highlights the importance and applicability of paleolimnology to managing and remediating degraded waterbodies and understanding their reference conditions.

Trophic dynamics before and after human disturbance have also been studied for management purposes (Kattel et al., 2014). A sediment core was collected from the deepest point of Kings Billabong wetland in Australia, and analyzed subfossil cladocerans, chironomids and diatoms, as well as stable isotopes of carbon and nitrogen. The results found that there was a significant transition in the Kings Billabong ecosystem following nearby river regulation. Furthermore, it was observed that sediment accumulation and rates increased, and that there were widespread disturbances of catchments caused by humans. Following these disturbances, it was noted that the littoral and benthic assemblages of subfossil cladocerans and diatoms decreased significantly. A historical perspective provided the opportunity to determine the cause of lake ecosystem changes from pre-impact to post-impact, and similarly highlighted the need for an understanding of reference conditions to inform the future directions of management.

In the Great Lakes watershed, paleolimnological approaches were applied in a wetland to inform ongoing efforts of wetland remediation in the Saint Lawrence River region, where deforestation for agricultural purposes and water level regulations for hydro-power, flood prevention, and shipping impacts are widespread (Boxem et al., 2017). These anthropogenic stressors led to an abundance of invasive cattails since the late 19th century. The invasion of alien cattail species has been known to trigger shifts in the plant communities, with a loss of native plant and animal species, and an overall decline in species diversity (Boxem et al., 2017). The examination of plant macrofossils in wetland sediments over the past ~600 years provided a greater understanding of the pre-invasion wetland plant community in the Jones Creek wetland,

and tracked the spread of invasive cattails and the resulting impacts on the structure and function of the wetland ecosystem. The results of this study suggested that Jones Creek was previously a sedge wetland prior to the predominant cattail monoculture, and that water quality has decreased since anthropogenic impacts became more widespread (Boxem et al., 2017). The paleolimnological perspective allowed management recommendations to take into consideration why cattails were able to become the dominant plant within the wetlands, and provided insights on how to better regulate them, such as the potential to reintroduce natural variation in water levels to drown out cattail seeds and promote the reintroduction of many native plant species reliant on highly variable water levels (Boxem et al., 2017).

1.3 Thesis organization and rationale for study

Great Lakes coastal wetlands in southern Ontario are becoming increasingly exposed to anthropogenic stressors as urban sprawl becomes more widespread. They are key components to the greater Laurentian Great Lakes ecosystem and watershed, and as such, it is necessary to understand the environmental changes occurring within them. Evidence suggests that wetlands which have undergone Remedial Action Plans since the early 1980s have since experienced improved biodiversity in fish species (Seilheimer et al., 2011), indicating that remediation strategies can be effective when implemented appropriately. This thesis focuses on three coastal wetlands in southern Ontario (Cootes Paradise, Jordan Harbour, and McLaughlin Bay), where studies indicate the most compromised wetlands are found due to high population densities and associated anthropogenic stressors (Seilheimer et al., 2011). The wetlands differ in hydrological connectivity to Lake Ontario, history of environmental stressors, and implemented and/or planned remediation efforts.

This thesis focuses on ecosystem change over the last ~100 years in Great Lakes coastal wetlands. Because of this, gravity cores using a Glew gravity corer and UWITEC gravity corer were taken and sectioned into slabs of 0.25 – 1 cm intervals for high resolution representing 2 – 10 years per interval. ^{210}Pb dating was conducted to determine the approximate age of each sediment interval with supplementary ^{137}Cs dating, and I used the Constant Rate of Supply (CRS) model because aquatic ecosystems falling within the catchments of highly developed watersheds are known for having highly variable sedimentation rates. The Constant Initial Concentration (CIC) model and the Constant Flux, Constant Sedimentation (CFCS) model both assume constant sedimentation rates. Cladocera were used as the primary paleoecological indicator, but diatoms were also used in McLaughlin Bay to compare against Cladocera-based paleoecological inferences. The thesis is formatted as two independent manuscripts (Chapter 2 and Chapter 3), conceptually connected through a general introduction and literature review (Chapter 1), and overall conclusions regarding the application of paleolimnological techniques for understanding ecological change in Great Lakes coastal wetlands (Chapter 4). Chapter 2 analyzes Cladocera and diatom remains in a sediment core from McLaughlin Bay (Oshawa, Ontario), in the context of a proposed remediation plan to remove the barrier beach isolating the marsh from Lake Ontario, in order to improve water quality and biodiversity. Chapter 3 analyzes two highly degraded river-mouth marshes (Cootes Paradise, Hamilton; Jordan Harbour, Lincoln) to assess ecological responses to multiple interacting anthropogenic stressors over the last several hundred years, and provide a long-term context for past and ongoing restoration efforts.

The primary goal of this thesis is to assess the application of integrating paleolimnological data into ongoing efforts to understand and manage Great Lakes coastal wetlands, by determining if paleoecological data can be used to evaluate ecosystem change in

Great Lakes coastal wetlands in a way that can be used to inform ecosystem restoration. Specifically, I aim to achieve this goal by answering the following research questions: Can we obtain high-resolution stratigraphic sediment records for the last ~100 years? How have Cladocera responded to coastal wetland degradation in southern Ontario? Does *Bosmina* size structure respond to known fluctuations in fish community structure?

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Chapter 2: A paleolimnological approach for evaluating a proposed restoration strategy in a highly degraded Lake Ontario coastal wetland

2.1 Abstract

Laurentian Great Lakes coastal wetlands have been impacted by multiple interacting anthropogenic stressors since European settlement in ~1850. As a result, restoration and management strategies have been developed and altered throughout the past several decades to address environmental impacts and improve water quality. In McLaughlin Bay (Oshawa, Ontario, Canada), significant ecological changes have occurred over the recent past as a direct result of human actions. Alterations of the watershed have been widespread, and have led to increased sedimentation rates and subsequent shoreline degradation, the introduction of invasive species, and decreases in water quality have been noted primarily due to shoreline degradation and large influxes of de-icing salts from nearby roadways. A potential remediation plan to remove a barrier beach separating McLaughlin Bay from Lake Ontario has been proposed to mitigate increased conductivity levels related to winter road salt application. The barrier has been breached at two confirmed occasions in the past (1954 and 2005). Therefore, we used a paleolimnological approach to assess whether water quality improved following these known historical breaches, to provide long-term ecological evidence for evaluating the potential success of the proposed remediation plan. We collected a ~30 cm sediment core from McLaughlin Bay, which incorporated the last ~70 years, and used diatom (siliceous algae) and cladoceran (microscopic crustaceans) subfossil remains as ecological indicators. Cladocera showed an improvement in littoral species diversity around the times of breaches that was not apparent in the diatoms. Diatoms have exhibited a steady decrease in species diversity since ~2005, and the assemblage was primarily composed of *Fragilaria* taxa tolerant of high conductivity and low light. The paleolimnological assessment of McLaughlin Bay indicates that periodic breaches of

the barrier beach had only short-term benefits on water quality, and that water quality degradation has been most pronounced over the last decade.

2.2 Introduction

Great Lakes coastal wetlands are integral components to the overall Laurentian Great Lakes ecosystem as transition zones between upland ecosystems and the Great Lakes (Shantz, 2018). Coastal wetlands provide many essential ecosystem services, such as filtering watershed run-off before it enters into the Great Lakes, and also the provision of critical habitat and breeding grounds for many waterfowl, amphibians, and fish (Sierszen et al., 2012). Urban development has resulted in significant changes to habitat and water quality in Great Lakes coastal wetlands, particularly in southern Ontario where watershed development has been most intensive (Nature Conservancy Canada, 2018). An emerging water quality issue of concern is the salinization of urban waterbodies, not just coastal wetlands, as a result of road salt application in the winter (Dugan et al., 2017). Urbanization has resulted in a growth of impervious surfaces such as asphalt, tarmac and cement, which promotes water moving across the surface as opposed to percolating through to the ground water, carrying road salt applied surficially to nearby bodies of water. With the increase in impervious surfaces over the last couple decades, there has been a corresponding increase in the need for road de-icing salts in the winter.

Road salt poses a major threat to the ecology and overall health of Great Lakes coastal wetlands. For example, it has been observed that road de-icing salts can be lethal for chironomid (benthic macroinvertebrate) populations in wetlands located in catchments near major roads (Lob and Silver, 2012). Amphibian embryonic and larval survival in vernal pools in the United States noticeably declined as conductivity increased from 25 to 500 $\mu S\ cm^{-1}$, and again to 3000 $\mu S\ cm^{-1}$

(Karraker et al., 2008). Road salt can also alter the biogeochemistry of wetlands by decreasing pH, and increasing manganese, iron, and exchangeable cations, stimulating the growth of microbial mats (Kim and Koretsky, 2013). With a reduction in water quality related to conductivity increases, the marshes become a hostile environment for many native aquatic macrophyte species, creating conditions favourable for the establishment of salt-tolerant invasive species such as *Phragmites australis* (Lissner and Schierup, 1997; Vasquez et al., 2005).

In this study, we focus on McLaughlin Bay, a barrier beach wetland located along the north-central shore of Lake Ontario (Oshawa, Ontario), where salinization from road salt application is a major concern. McLaughlin Bay has a large catchment:wetland area ratio, and is proximate to major roadways, including Canada's busiest highway, the 401. McLaughlin Bay is isolated from Lake Ontario by a barrier beach, which prevents mixing with Lake Ontario surface waters, thus minimizing any potential dilution effects of road salt run-off to the marsh. Over the past several years, conductivity in McLaughlin Bay has averaged ~1150 μS cm^{-1}, compared to nearby Cranberry Marsh that is primarily rain-fed, and has conductivity levels of ~375 μS cm^{-1} (Central Lake Ontario Conservation Authority, 2010). Frenchman's Bay, a nearby wetland ~25 km from McLaughlin Bay, received ~850 tonnes of chloride (~1400 tonnes of salt) from nearby Pine Creek alone during the 2004 – 2005 road salting season, and receives ~3700 tonnes from the entire watershed each year (Meriano et al., 2009). McLaughlin Bay, due to its location within a highly developed watershed, likely receives amounts of chloride comparable to Frenchman's Bay, with little opportunity for dilution from Lake Ontario. McLaughlin Bay has also experienced ecological challenges associated with increased turbidity and reduced macrophyte coverage resulting from past shoreline alterations (Ontario Parks, 2019) and invasive common carp (Ontario Ministry of Natural Resources, 2011). Throughout the recorded history of the marsh, there have been two confirmed breaches of the barrier beach that separates McLaughlin Bay from Lake Ontario (1954 and 2005) however, it is unknown whether the periodic, temporary breaches provided any long-term ecological benefit to the marsh. McLaughlin Bay's shoreline has undergone natural alterations over the past ~200 years, and there has been some suggestion that shifts in the barrier beach over the past few decades have resulted in decreased connectivity between the marsh and Lake Ontario (Central Lake Ontario Conservation Authority, 2013). Because of this, the Central Lake Ontario Conservation Authority has proposed that the barrier beach be altered, either partially, fully, or periodically removed, to allow for more frequent mixing between McLaughlin Bay and Lake Ontario (Central Lake Ontario Conservation Authority, 2013).

The objectives of this study are to (1) infer long-term ecological and water quality changes in McLaughlin Bay post-1850, and (2) examine how the ecosystem responded to the two known historic breaches of the barrier beach in 1954 and 2005, in order to provide a long-term context that can be used to evaluate restoration strategies being proposed for McLaughlin Bay. We used diatoms (siliceous algae of the phylum Bacillariophyta) and cladocerans (crustacean zooplankton) as paleoecological indicators. Diatoms are known to be effective bioindicators of conductivity (Belore et al., 2002; Cumming et al., 1995), and cladocerans can be effective bioindicators of changes in lake/wetland trophic structure (Davidson et al., 2011). We hypothesize that diatom and cladoceran subfossil assemblage changes will reflect water quality degradation over the last two decades indicative of salinization in response to increased road salt application and decreased hydrological connectivity to Lake Ontario. We also hypothesize that known breaches of the barrier beach in 1954 and 2005 will have coincided with temporary improvements in the ecological condition of McLaughlin Bay. The long-term perspective provided by our paleolimnological approach will provide new insights into the ecological impacts of road salt run-off into hydrologically-isolated Great Lakes coastal wetlands, and will help inform decision-making on a proposed remediation strategy to alter the barrier beach to allow for mixing with Lake Ontario waters for long-term ecological improvement.

2.3 Site description

McLaughlin Bay is a barrier beach wetland located in Oshawa, Ontario, on the north-central shore of Lake Ontario (Figure 1). It has been classified as highly degraded by the Central Lake Ontario Conservation Authority (2010). The marsh has a surface area of 44-ha, and water depths fluctuate from 1-3 m seasonally and in response to variation in rainfall (Central Lake Ontario Conservation Authority, 2013). It is presently closed off to Lake Ontario by a barrier beach, although historical air photos suggest that it has been connected to the lake periodically in the past through breaches caused by excessive runoff and storms (Central Lake Ontario Conservation Authority, 2013). The General Motors office is situated behind the marsh, and is suspected to be a major source of road salt run-off into the marsh, as snow removal is disposed of in a parking lot that backs onto McLaughlin Bay. Conductivity levels in McLaughlin Bay have averaged ~1150 μS cm^{-1} over the past ~10 years, significantly higher than the ~300 μS cm^{-1} that the United States Environmental Protection Agency (2011) deems to be associated with a healthy aquatic ecosystem. McLaughlin Bay is an important transition zone and migratory stopover for species traveling between Darlington Provincial Park to the east, and Oshawa Second Marsh to the west, and as such, the improvement in water quality and ecological health is imperative.

Historically, aquatic vegetation cover in McLaughlin Bay has been variable within the marsh; however, a decreasing trend in macrophyte cover was observed based on sampling conducted through the Durham Region Coastal Wetland Monitoring Project from 2002 – 2009 (Central Lake Ontario Conservation Authority, 2010). Shoreline restoration has been one of the primary focuses for restoring the marsh, which has thus far consisted of reintroducing native plant species along the shoreline to promote a reduction of sediment inputs, with visible improvements in overall vegetative cover along the marsh edge (Ontario Parks, 2019). Common carp (*Cyprinus carpio)* and the invasive common reed (*Phragmites australis*) have had negative impacts on aquatic vegetation cover, with carp tearing much of the submerged and emergent vegetation out, and *P. australis* displacing native species and outcompeting them for water and nutrients (Brisson et al., 2010; Ontario Ministry of Natural Resources, 2011). Carp have also increased the turbidity of the water through the process of "mudding" (Vilizzi et al., 2014). Due to the lack of aquatic vegetation in the wetland, the sediment often remains suspended for long periods of time, minimizing light penetration and reducing macrophyte growth (Vilizzi et al., 2014). There is low aquatic and semi-aquatic wildlife diversity in McLaughlin Bay due to a lack of suitable habitat, poor water quality, and high salinity (Central Lake Ontario Conservation Authority, 2010).

McLaughlin Bay has been a part of the Durham Region Coastal Wetland Monitoring Project since 2002 (Central Lake Ontario Conservation Authority, 2016), which provides a context for debating appropriate remediation plans for the marsh. For example, water level drawdowns, ongoing shoreline restoration, mechanisms to exclude invasive carp, and removal of the invasive *Phragmites australis* have all been proposed as remediation and restoration strategies (Central Lake Ontario Conservation Authority, 2013). Opening up McLaughlin Bay to

Lake Ontario through modifications of the barrier beach is also being discussed as a strategy to mitigate road salt run-off and salinization of the marsh (Central Lake Ontario Conservation Authority, 2013).

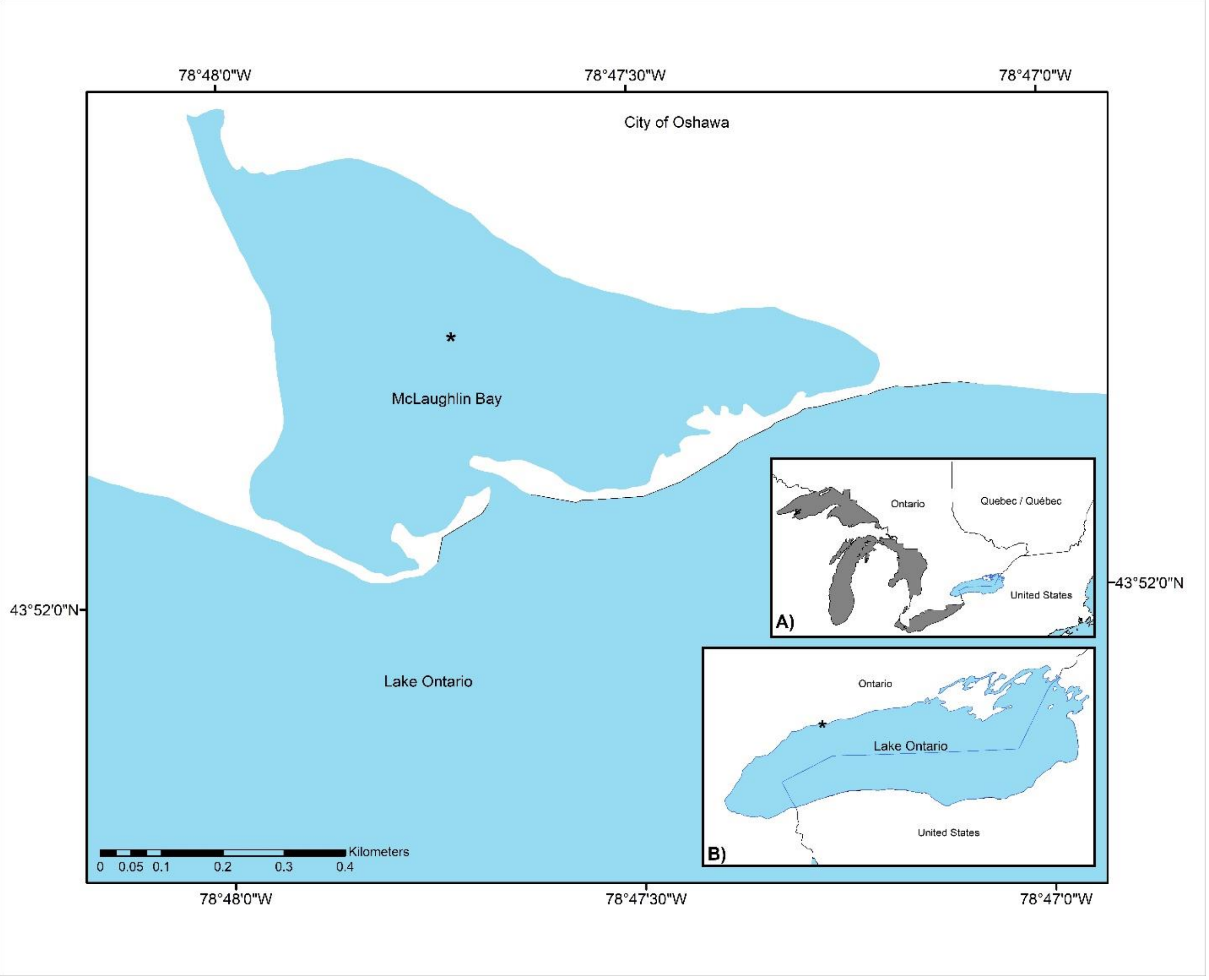

Figure 1. Map of McLaughlin Bay, with the coring location identified by an *, A) Location of Lake Ontario in relation to the other Laurentian Great Lakes, and B) Location of McLaughlin Bay (Oshawa, Ontario, Canada) along the north-central shore of Lake Ontario (Canada) identified by an *.

2.4 Methods

2.4.1 Field sampling

A Glew gravity corer (Glew, 1991) was used to collect duplicate sediment cores (25-30 cm in length) from the centre of McLaughlin Bay (Figure 1) in August 2017, which were then sectioned on site using a Glew vertical extruder (Glew, 1988). The cores were sectioned into 0.25 cm intervals for the first 10 cm, and 0.5 cm intervals from 10 cm until the bottom of the core.

2.4.2 Laboratory methods

A chronology for the sediment core collected from McLaughlin Bay was established using the ^{210}Pb method by gamma spectrometry (Appleby et al., 1986), supplemented by ^{137}Cs (Jaakkola et al., 1983), on freeze-dried and homogenized sediments at the Paleoecological Environmental Assessment and Research Lab at Queen's University (Kingston, Ontario, Canada). An age-depth model was developed using the constant rate of supply (CRS) model (Appleby and Oldfield, 1978).

Cladoceran subfossils were isolated from the sediment matrix and plated onto microscope slides using standard methods described in Korosi and Smol (2012a). Approximately 1 g of wet sediment was heated in a 10% potassium hydroxide (KOH) digestion for 30 minutes, and then passed through a 36-µm mesh sieve to filter out unwanted particles. Two drops of safranin glycerol were added to stain the cladoceran remains, and a couple drops of 80% ethanol solution was added as a preservative. Slides were prepared by pipetting 50-µl aliquots onto the slide, and allowing them to air-dry for ~24 hours in a covered microscope slide box to minimize dust contamination. For samples with low cladoceran abundances, slides received two 50-µl aliquots, and were allowed to dry in between aliquot additions. Once the samples were completely dry, slide covers were mounted using glycerin jelly. Cladoceran remains were identified to the species, species complex, or genus level under a microscope using Korosi and Smol (2012a, 2012b) as taxonomic guides. A minimum of 70 individuals were counted per interval (Kurek et al., 2010). *Bosmina* subfossil remains (headshields, carapaces) were measured using an Amscope B690C-PL microscope calibrated for measurements in µm, following methods described in Korosi et al. (2010), and a minimum of 35 headshields and carapaces (70 remains total) were measured per interval (Brahney et al., 2010).

Diatoms were isolated from the sediment matrix using approximately 0.5g of wet sediment per interval. Organic matter was removed via a hydrogen peroxide digestion (Battarbee et al., 2001). Prior to digestion, each sample was treated using a 10% hydrochloric acid (HCl) solution for 24 hours and rinsed with deionized water over multiple days until it reached neutral pH. The vials were filled with a 30% H_2O_2 solution and partially submerged in a water bath at 70°C, and rinsed again with deionized water over a period of 7 days. Final slurries were plated onto coverslips at four different dilutions, and dried at room temperature inside a covered microscope slide box to avoid contamination from air particulates. The coverslips were then mounted onto microscopes slides using Naphrax® and analyzed using an Amscope B690C-PL microscope with microscope slides viewed under an oil immersion lens (1000x magnification). A minimum of 400 diatom valves were counted per interval (Battarbee et al., 2001), to a maximum of 502 valves. Diatom identification was based on Krammer and Lange-Bertalot (1986–1991).

2.4.3 Data analysis

Species diversity changes throughout the stratigraphies were inferred using Hill's N2, calculated in R Studio version 1.1.463 using the Rioja and Vegan packages (Oksanen et al., 2012). A stratigraphically-constrained incremental sum of squares (CONISS) cluster analysis (Grimm, 1987) was done to determine periods of significant change in Cladocera and diatom assemblages through time, and the number of significant clusters was determined through use of the "rioja: Analysis of Quaternary Science Data" package version 0.9-9 using a broken-stick model of random distribution (Bennett, 1996).

2.5 Results & discussion

2.5.1 Sediment core chronology

A general decline in ^{210}Pb isotopic activity was observed in the sediment core, and background ^{210}Pb was reached at 30 cm (Figure 2). A small peak in ^{137}Cs isotopic activity (indicating ~1963) was evident at ~2 cm core depth, which does not correspond with the dates provided by the CRS age model; however ^{137}Cs activities were very low, and there is only one point elevated at 2-cm. Based on the CRS age model, the core dates back to 1963 ± 14 years at core depth of 20-cm, below which errors on the age-depth model increase substantially. A major breach in the barrier beach that hydrologically isolates McLaughlin Bay from Lake Ontario occurred in the mid-1950s, and thus the top 20-cm of the sediment core likely represents the accumulation of sediment since this event.

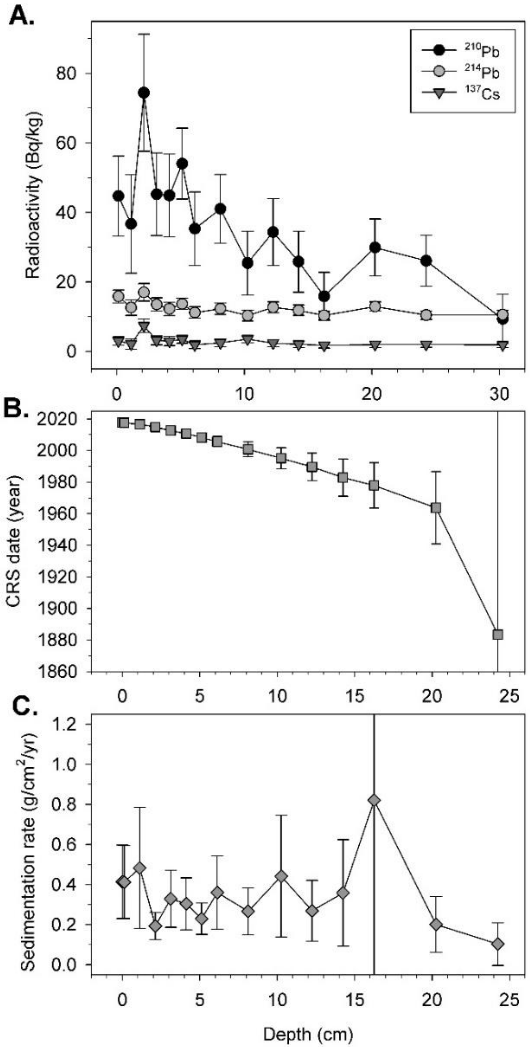

Figure 2. Results of ^{210}Pb dating for McLaughlin Bay. A) Radioisotopic activities for ^{210}Pb, ^{214}Pb and ^{137}Cs; B) sediment core depth versus date, based on the constant rate of supply (CRS) model; and C) Inferred sedimentation rate based on ^{210}Pb CRS model.

2.5.2 Subfossil Cladocera analysis

Bosmina are the dominant taxa observed within the sediment record in McLaughlin Bay, ranging in relative abundance from ~75 – 90% (Figure 3). *Chydorus brevilabris* are the second most abundant species, with relative abundances ranging from ~1 – 9%. *Alona guttata* are observed in most intervals throughout the record, ranging from ~1 – 8% relative abundance, with an increase from 8 cm (~2000) to 6 cm (~2005) depth. *Alona circumfimbriata* are absent from 15

– 18 cm depth in the core, and range from ~1 – 8% relative abundance in the earliest and latest part of the sediment record (20 – 30 cm and 0 – 14 cm respectively), and sporadic occurrences of ~1 – 3% relative abundance are observed for *Alona affinis*. Other littoral species were observed at varying abundances throughout the core. *Camptocercus* spp. were observed sporadically throughout the sediment record at ~1 – 3% relative abundance and were absent from the earliest part of the sediment record (25 – 30 cm). *Leydigia* spp. were primarily found in the top 7 cm (~2005 – 2015). *Alona costata,* and *Graptolebris testudinaria* were observed at ~1 – 3% relative abundances; however, they are not observed in more than 4 and 2 intervals respectively. *Pleuroxus* spp. were observed in low abundances below 20-cm (~1963) and are not observed in any more recent intervals.

Bosmina carapace lengths ranged from ~195 – 225 µm; mucro lengths ranged from ~22 – 30 µm; and antennule lengths ranged from ~75 – 95 µm. *Bosmina* appendage lengths are useful indicators of predation due to their observable responses to different fish and invertebrate predators (Korosi et al., 2013). Antennule and mucro lengths are known to increase in size in response to copepod predation as a natural defence mechanism making it more difficult for copepods to grasp the *Bosmina* (Kerfoot, 1975; Post et al., 1995). The longer appendages also make it difficult for the copepods to maneuver the *Bosmina* in a way that allows them to access the soft, inner carapace to consume, increasing the probability of escape from the copepod attacks (Kerfoot, 1975, 1977). Larger bodied *Eubosmina* (500 – 700 µm) have also been observed to experience very little appendage changes in lakes with both high and low copepod predation, likely due to the larger bodies being difficult to grasp compared to smaller bodied *Bosmina* species (Sprules et al., 1984). No notable trends in *Bosmina* size measures were evident in McLaughlin Bay through time that would indicate *Bosmina* populations have responded to past food web shifts.

The overall Cladocera assemblage in McLaughin Bay is generally stable throughout the entire sediment core. However, at 20 cm (1963 ±14), there was a peak in the Hill's N2 diversity, an increase in *Chydorus brevilabris* relative abundance, as well as the reappearance of *Alona affinis*, *Alona costata*, and *Pleuroxus* spp., and the appearance of *Ilyocryptus* for the first time (Figure 3).These changes in the assemblage at 20-cm may be related to the 1954 barrier beach breach when an influx of Lake Ontario water flushed into the Bay. The hydrological isolation of McLaughlin Bay due to the barrier beach increases its susceptibility to water quality degradation due to contaminant run-off from the watershed. The barrier beach breach and subsequent mixing of water would allow for dilution of contaminants in the marsh and improve water quality for pollution-intolerant species to increase in numbers. Increases in littoral cladocerans in particular may be indicating temporary increased growth of aquatic macrophytes following the barrier beach breach.

In ~2005 (core depth 6 cm), there was an additional documented barrier beach breach in McLaughlin Bay (Wittnebel, 2016), but no corresponding increase in cladoceran Hill's N2 diversity. Shortly after this event, between core depths 5-2 cm (~2008 to 2014), *Leydigia* spp. increased. *Leydigia* are sediment-associated taxa (Kinder et al., 2019; Galka et al., 2014) that are found with submerged aquatic macrophyte beds (Engel, 1985). This suggests that water quality improvements following the 2005 breach were not as prominent as the 1954 breach, and no consistent, long-term ecological improvements occurred following either breach event.

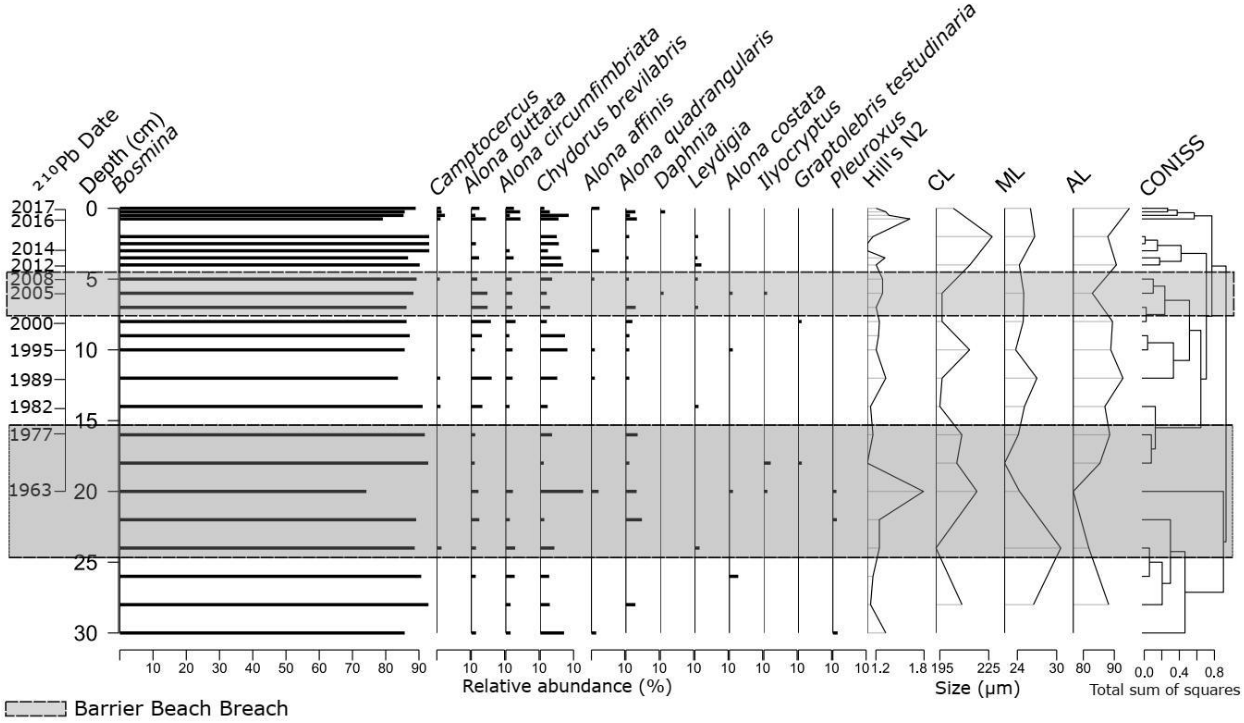

Figure 3. Stratigraphic profile showing changes in Cladocera subfossil assemblages, Hill's N2 diversity, and *Bosmina* size measures (CL = carapace length; ML = mucro length; AL = antennules length). Approximate dates are based on the ^{210}Pb Constant Rate of Supply (CRS) model, and are shown on the left. Results of the constrained incremental sum of squares (CONISS) cluster analysis are also shown on the right, with no significant breaks based on the broken stick model. The sediment intervals corresponding to the known barrier beach breach events (2005 and 1954), incorporating errors based on the CRS age-depth model, are indicated by grey shading.

2.5.3 Subfossil diatom analysis

The profiles of the diatom taxa have been relatively stable throughout time in McLaughlin Bay (Figure 4). Small-benthic *Fragilaria (sensu lato)* are the dominant diatom taxa observed throughout the sediment record, ranging from ~35 – 65% relative abundance, comprised primarily of *F. construens var. venter* and *F. pinnata*, as well as *F. construens, F. construens var. binodis and F. construens var. subsalina* in smaller abundances. Lower abundances of small-benthic *Fragilaria* are seen in the earliest part of the sediment record, with an increase at 18-cm depth (~1970) followed by a decline to 7-cm (~2003) and then increase again to 1-cm (~2016). *Gyrosigma acuminatum* are the second most abundant species throughout the core, ranging from ~3 – 12% relative abundance, with the greatest abundance at 9-cm (~1997). Large-benthic *Navicula* species are also observed throughout the core, ranging from ~4 – 11% relative abundance. This group is comprised primarily of *N. cryptocephala, N. cryptofallax, N. cryptotenella, N. riparia,* and *N. trivialis* in McLaughlin Bay. *Cyclotella* (mostly *C. meneghiniana* and *C. ocellata*) are also observed throughout the sediment record, with relative abundances ranging from ~1 – 9%. *Fragilaria berolinensis* are observed in very low abundances of ~1 – 2% until 8-cm (post-2000), when they increase to ~5% relative abundance. *Amphora* spp. range from ~4 – 10%, and *Cocconeis* spp. range in relative abundance from 2 – 5%.

Many of the species observed throughout the McLaughlin Bay core are known to be tolerant of elevated conductivity/salinity. Small-benthic *Fragilaria* have been found over a wide salinity gradient, ranging from freshwater to brackish water (Gasse et al., 1995; Van Dam and Mertens, 1993; Veres et al., 1995). *Gyrosigma acuminatum* are predominantly observed in aquatic ecosystems with low or moderate electrolyte content (Sterrenburg, 1995; Trigueros and Orive, 2001), and *G. acuminatum* are also a known low-to-moderate pollution-tolerant species (Mangadze et al., 2017), having been observed in aquatic habitats with high conductivity along with large benthic forms of *Navicula sensu lato* (Zgrundo et al., 2017). *Fragilaria berolinensis, F. brevistriata,* and *F. capucina* (the latter two were the dominant species in the long-planktonic groups) have all been reported in high salinity aquatic environments (Bahls, 2012; Gasse et al., 1995; Kashima et al., 1997; Tuchman et al., 1984; Van Dam and Mertens, 1993; Veres et al., 1995). *Cyclotella meneghiniana,* one of the dominant species in the *Cyclotella* group within McLaughlin Bay, have been observed primarily in shallow ecosystems, and tolerant of nutrient-rich waters (Lowe and Kheiri, 2015).

Small-benthic *Fragilaria* species fluctuate throughout the sediment core, however at 20-cm (~1963 ± 14), the estimated time of the earliest recorded barrier beach breach, the species increase in relative abundance, and continue to increase until 18-cm (~1970). Small-benthic *Fragilaria* are known for their tolerance of high salinity (Gasse et al., 1995; Van Dam and Mertens, 1993; Veres et al., 1995), in addition to their competitive colonization during low light conditions (Smol and Stoermer, 2010) such as high turbidity. During this time, species observed in high salinity environments like *Gyrosigma acuminatum, Fragilaria berlolinensis, F. brevistriata,* and *F. capucina* decrease, suggesting a temporary decrease of conductivity within the marsh as salt-tolerant species increase again within ~10 years.

In 2005 during the most recent barrier beach breach, there is another decrease in Hill’s N2 diversity and a similar trend to the 1954 breach exhibited in the assemblage. Salt and low light-tolerant small-benthic *Fragilaria* species once again increase in relative abundance during and after the breach, while salt-tolerant species like *Fragilaria berolinesis* and *F. brevistriata* decrease during 2005 and increase shortly after, and species observed in both freshwater and brackish water like *Gyrosigma acuminatum* (Chaput, 2014) also decrease during this time, which prior to the breach, had a higher relative abundance. However, it is suspected that because of the lack of signal during the 2005 breach and continued decline following, the diatoms are not responding to the barrier beach breach, but rather to a general decline in water quality post-2000.

While shoreline reconstruction that began in 2010 has visibly increased vegetative cover in McLaughlin Bay, diatom assemblages still indicate a trend of water quality degradation. This is reflected in the diatom assemblage with an increase in low light-tolerant small-benthic *Fragilaria*, and a decrease in saline-tolerant species such as *Gyrosigma acuminatum, Fragilaria berlolinensis, F. brevistriata,* and *F. capucina* during known breaches, yet no visible improvements in diatom diversity during these times. Thus, in contrast to subfossil Cladocera assemblage changes, diatom assemblages do not indicate subsequent improvements in water quality following barrier beach breaches, but rather that they have experienced declines in diversity following barrier beach breaches. This is most likely due to general decreases in water quality within the bay, and continued water quality degradation since the mid-2000s as development has intensified in the surrounding watershed.

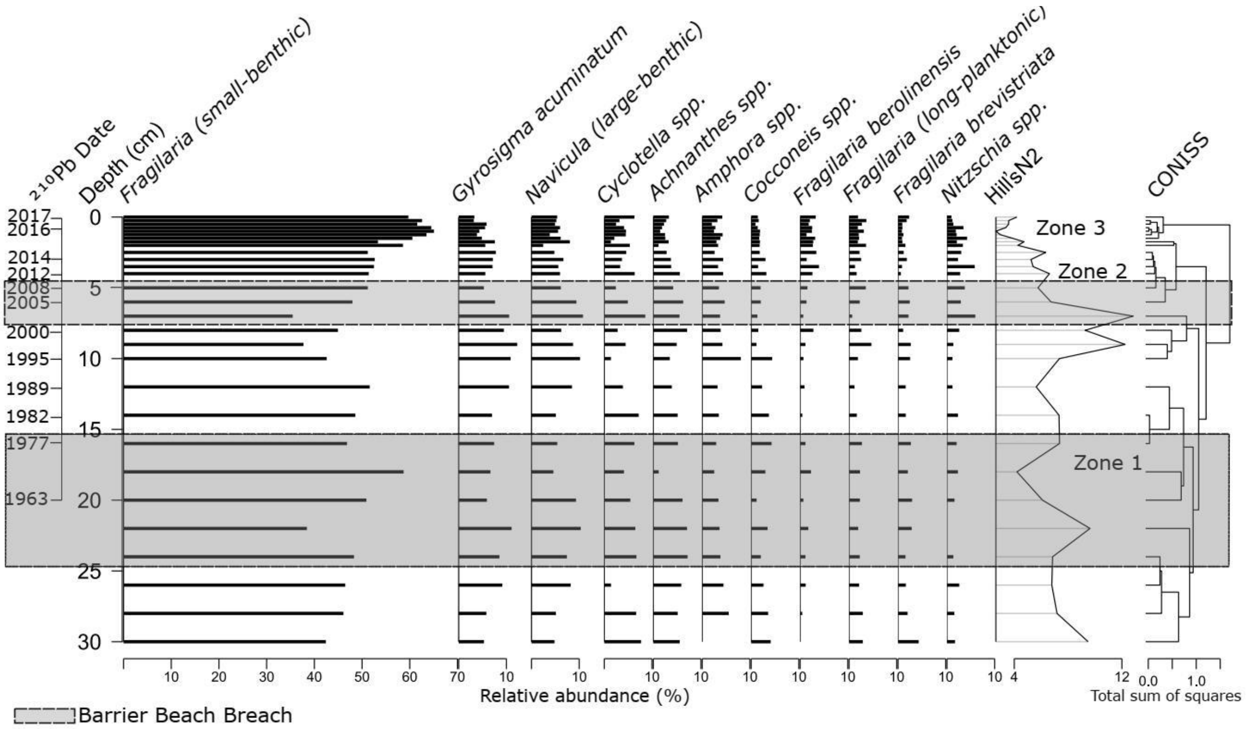

Figure 4. Stratigraphic profile showing changes in diatom assemblages, and Hill's N2 diversity. The Constrained incremental sum of squares (CONISS) cluster analysis results are shown on the right. Approximate dates based on the ^{210}Pb Constant Rate of Supply (CRS) model are shown on the left. The sediment intervals corresponding to the known barrier beach breach events (2005 and 1954), incorporating errors based on the CRS age-depth model, are indicated by grey shading.

2.6 Conclusions

We hypothesized that diatom and cladoceran subfossil assemblages will reflect water quality degradation over the last few decades indicative of salinization, and that known breaches of the barrier beach in 1954 and 2005 will have coincided with temporary improvements in ecological condition of the marsh. Diatoms showed clear evidence of a decline in water quality since the early 2000s, evident with increases in *Fragilaria* taxa and a decrease in Hills N2 diversity, but the Cladocera assemblage did not exhibit recent decreases in species diversity or assemblage changes indicative of water quality degradation. In contrast, Cladocera exhibited increases in epiphytic taxa corresponding to known barrier beach breaches, particularly in 1954, that indicate temporary ecological improvements, but this was not evident in the diatom assemblages. While barrier beach breaches can be beneficial to improving ecosystem health by bringing in water from the lake to dilute contaminant buildup within the marsh, the Cladocera and diatom subfossil assemblages indicate that the short-term breaches do not have a long-term ecological impact, and cannot mitigate water quality degradation in the marsh without substantive efforts to reduce pollutant loadings to McLaughlin Bay.

Great Lakes coastal wetlands are dynamic aquatic ecosystems that require individualized remediation plans based on the hydrology, ecology, and history of each wetland. A paleolimnological approach can provide the historical context on the ecological impacts of past and current stressors, and thus provide insights on potential remediation protocols.

2.7 Acknowledgements

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Chapter 3: Reconstructing long-term ecological change in two highly degraded Lake Ontario (Canada) coastal wetlands

3.1 Abstract

Coastal wetlands are integral to the ecosystem health of the Laurentian Great Lakes (North America) watershed. Multiple anthropogenic stressors have been impacting coastal wetlands since European settlement in ~1850, and remain a fundamental concern for wetland health as watershed development intensifies. We used paleolimnological techniques to explore temporal ecosystem dynamics in two highly degraded Lake Ontario coastal wetlands located in southern Ontario, Canada, using Cladocera (Branchiopoda, Crustacea) subfossil remains as paleoecological indicators. In Cootes Paradise Marsh (Hamilton, Ontario), cladoceran assemblage changes exhibited a shift in dominance from *Chydorus* to *Bosmina*, observed at the turn of the twentieth century. This shift is likely tracking the loss of aquatic macrophytes, and corresponds to the postulated timing of the arrival of invasive carp. Despite recent efforts to exclude carp from the wetland, very little ecological recovery is evident from the subfossil Cladocera assemblage. No *Daphnia* remains were observed in our sediment core from Cootes Paradise, in contrast to previous studies on contemporary zooplankton communities that reported a large *Daphnia* population in the western end of the marsh in the 1940s. This could indicate that our sediment core was predominantly recording ecological changes constrained to the eastern end of Cootes Paradise Marsh. In Jordan Harbour (Lincoln, Ontario), *Bosmina* were dominant throughout the sediment record, and increases in littoral cladocerans were observed in the most recent sediments, particularly the appearance of periphytic *Pleuroxus* taxa after ~2008. This suggests that some recovery of aquatic macrophyte communities has occurred in response to shoreline remediation efforts. *Bosmina* size structure exhibited only minimal changes in both wetlands, despite known large-scale changes in fish community structure. Overall, our study

provides perspectives on the benefits and limitations of paleolimnology for documenting long-term ecological change in the Laurentian Great Lakes coastal wetlands.

3.2 Introduction

The Laurentian Great Lakes are transboundary glacial lakes located in Ontario, Canada, and the midwestern United States of America, and include Lake Superior, Lake Michigan, Lake Huron, Lake Erie, and Lake Ontario (United States Environmental Protection Agency 2019a). The Laurentian Great Lakes hold an estimated ~20% of the world's freshwater supply, and have economic, agricultural, and cultural significance, where many people use them for recreational purposes like bird watching, canoeing, kayaking, photography, and fishing (United States Environmental Protection Agency 2019b).

Great Lakes coastal wetlands are classified as wetlands located along the Canadian and American shores of the Laurentian Great Lakes. There are three main types of coastal wetlands, categorized as lacustrine, barrier beach, or drowned river-mouth wetlands, and wetlands in southern Ontario are primarily classified as barrier beach or drowned river-mouth (Albert et al. 2005). Great Lakes coastal wetlands are integral to the health of the Laurentian Great Lakes watershed, where they provide multiple ecosystem services including water filtration, habitat for many important and rare wetland species, shoreline erosion prevention, flood prevention, and recreation (Sierszen et al. 2012). Laurentian Great Lakes coastal wetlands have long been subjected to multiple anthropogenic stressors, especially in heavily populated areas where development of areas surrounding the watershed have been extensive. This includes increased sedimentation rates due to shoreline and watershed alterations (Chow-Fraser 2006; Morrice et al. 2008), the introduction of invasive alien species (Chow-Fraser 1998; Morrice et al. 2008), toxic effluents (Van Dongen 2019), and increased road salt inputs resulting in elevated chloride concentrations of ecotoxicological significance (Meriano et al. 2009).

The Great Lakes Coastal Wetland Monitoring Program is a binational collaboration that has conducted routine monitoring of major coastal wetlands across the entire Laurentian Great Lakes since 2010 using a comprehensive and standardized approach (United States Environmental Protection Agency 2010). Independent long-term monitoring programs have also been implemented across the Laurentian Great Lakes at a local-scale by various conservation authorities. Current monitoring programs have been used to establish the degradation status of coastal wetlands using various biotic and water quality indices (Chow-Fraser 2006; Host et al. 2019; Uzarski et al. 2017), as a foundation for assessing future changes in ecological conditions. Great Lakes coastal wetlands have been significantly impacted by anthropogenic stressors since the time of European settlement, and thus recent long-term monitoring programs do not extend far enough back in time to capture the full trajectory of ecological change due to anthropogenic impacts. The application of paleolimnological approaches in Great Lakes coastal wetlands can be beneficial for extending recent monitoring records back in time, providing an historical context for evaluating contemporary changes.

The application of paleolimnological approaches in wetlands is challenging due to shallow waters that are susceptible to sediment mixing (Maynard and Wilcox 1997). Extensive macrophyte growth can contribute further to sediment mixing (Eyles et al. 2003; Nigel and Rughooputh 2010; Rooney et al. 2003). As well, wetlands located within highly developed watersheds are also subject to substantial erosion and high siltation (Lee et al. 2006), resulting in low ^{210}Pb activity and subsequent challenges in establishing sediment chronologies (Ahn 2018). Despite these challenges, paleolimnological approaches have been successfully applied in many instances to assess wetland ecological change and ecological resilience in the face of multiple interacting stressors (Gell et al. 2012; Lintern et al. 2016a). For example, metal(oid) contamination in Australian wetlands was evaluated using paleolimnological methods in order to identify the mechanisms, history, and source of contamination, to inform future management and monitoring, and to establish remediation plans based on established pre-impact baselines that were site-specific (Lintern et al. 2016a; 2016b).

In this study, we used cladoceran subfossil remains as paleoecological indicators to reconstruct long-term ecosystem change over the last ~120 years in two highly degraded coastal wetlands in southern Ontario, Canada. Cootes Paradise (Hamilton, Ontario) and Jordan Harbour (Lincoln, Ontario) are considered ecologically significant habitat for many rare wetland species, including the Northern Water Snake (*Nerodia sipedon*), Blanding's Turtle (*Emydoidea blandingii*), Eastern Spiny Softshell (*Apalone spinifera*) and Common Musk Turtle (*Sternotherus odoratus*. Cootes Paradise has been extensively monitored and studied for water quality changes, emergent and submerged vegetation, fish community composition, and wildlife and waterfowl usage dating as far back as 1874, when the marsh was designated a fish sanctuary (City of Hamilton 2017). The provision of fish spawning habitat is one of the most important ecosystem services provided by Great Lakes coastal wetlands. Since the late 1800s, when the common carp (*Cyprinus carpio*) invaded, native fish communities have been negatively impacted, with planktivorous fish commonly replacing piscivorous species (Kim et al. 2016). Therefore, we analyzed *Bosmina* size changes in addition to species assemblage changes, to infer food-web responses to known changes in fish communities (Korosi et al. 2013). This study provides insight into the changes occurring in Cootes Paradise and Jordan Harbour over the past ~120 years to better understand how anthropogenic stressors have impacted two ecologically significant Great

Lakes coastal wetlands, and to evaluate the benefits and limitations associated with applying paleolimnology to Great Lakes coastal wetland ecosystems.

3.3 Study site descriptions

3.3.1 Cootes Paradise

Cootes Paradise is the westernmost and largest coastal wetland (riverine marsh) within the Lake Ontario basin, located in Hamilton, Ontario (Figure 1; Table 1). It is designated as a Class 1 Provincially Significant Wetland because it is home to hundreds of rare plant species and endangered wildlife, including birds, fish and amphibian species (City of Hamilton 2017). It is also listed as an “Area of Natural and Scientific Interest” and an “Environmentally Sensitive Area” in Hamilton (City of Hamilton 2017). Intensive watershed development has resulted in substantial run-off of nutrients and sediments into Cootes Paradise, resulting in an increase in turbidity and a subsequent decline in macrophyte coverage, which contributed further to turbidity issues, reaching 30.8 NTU (Croft and Chow-Fraser 2009). It is classified as hypereutrophic and highly degraded (Kim et al. 2016). Cootes Paradise is part of the Great Lakes Coastal Wetland Monitoring Program, which aims to improve the overall health of the wetland for both public enjoyment and species living within and migrating to the marsh (United States Environmental Protection Agency 2010).

Cootes Paradise has been of interest to researchers and nature enthusiasts alike for well over one hundred years, and thus recorded and anecdotal information on water quality and changes to the vegetation and fish communities are readily available. Since the 1800s, the aquatic vegetation in Cootes Paradise has changed substantially. Prior to significant human interference, Cootes Paradise was a hunting ground lush with waterfowl and wild animals such as otters, beavers, mink and muskrats (Lord 1993). Historical maps have suggested that the entire marsh basin, from Dundas to Burlington Heights, was completely filled with emergent vegetation prior to 1850 (Painter et al. 1989). In the early 1900s, the marsh vegetation began to recede westward, which left a large area of open water at the eastern margin adjacent to Burlington Heights (Lord 1993). The emergent vegetative cover in Cootes Paradise extended over 90% of the surface area at the turn of the twentieth century and decreased to less than 15% by the 1990s (Chow-Fraser et al. 1998; Chow-Fraser 2005).

The vegetation decline in Cootes Paradise resulted in a shift in the fish community, as Cootes Paradise transitioned from a warm-water fishery primarily comprised of Northern Pike (*Esox Lucius*) and Largemouth Bass (*Micropterus salmoides*) to one dominated primarily by planktivorous and benthivorous species like Brown Bullheads (*Ameiurus nebulosus*), the invasive Common Carp (*Cyprinus carpio*), and Alewife (*Alosa pseudoharengus*) (Kim et al. 2016). The decline in the percent cover of the submerged macrophyte community also coincided with a reduction in water clarity and an increase in nutrient influx from pollution via wastewater and storm water effluent (Chow-Fraser et al. 1998). The Dundas Waste Water treatment plant was upgraded to a tertiary treatment facility in 1978, and since then, a decrease in phosphorus levels in Cootes Paradise has occurred (Chow-Fraser et al. 1998). The water quality in Cootes Paradise has been improving in recent years, yet the biotic community has not recovered (Kim et al. 2016), and the marsh is still dominated by pollution-tolerant species like the Common Carp (Thomasen and Chow-Fraser 2012). Cootes Paradise is proving difficult to restore, and research suggests that high turbidity has prevented emergent vegetation from re-establishing (Thomasen and Chow-Fraser 2012). Recently, a sewage gate was left open for 4 years spanning from 2014 – 2018, during which an estimated 24-billion litres of sewage flowed into Chedoke Creek which runs directly into Cootes Paradise (Van Dongen 2019).

3.3.2 Jordan Harbour

Jordan Harbour is a large riverine marsh located along the southern shore of Lake Ontario (Figure 1; Table 1). Historically, it is not a well-studied wetland compared to Cootes Paradise, with only limited historical records and prior scientific research available. However, Jordan Harbour is classified as a "Provincially Significant Wetland" as well as an "Area of Natural and Scientific Interest" because of the rare wetland species found within it (Niagara Peninsula Conservation Authority 2015). It has been subject to significant nutrient runoff, as well as increasingly high sedimentation rates, as a result of nearby agricultural practices such as farming, vineyards, and orchards (Croft and Chow-Fraser 2009). Invasive carp have been spreading around the Great Lakes region since they were first introduced to Lake Ontario in the late 1800s in Wilmot Creek, and quickly spread to many other wetlands around the Lake Ontario shoreline (Royal Botanical Gardens 1998). Carp pose a significant ecological threat to native plant species due to their tendency to increase turbidity through "mudding," as well as the potential for outcompeting native fish species (Ontario's Invading Species Awareness Program 2019). Large carp also decrease submerged and emergent aquatic vegetation cover through their ability to tear rooted plants out (King and Hunt 1967), which further contributes to an increase in turbidity.

The marsh is characterized by very dense submerged and floating plant growth in some areas, which minimizes sediment re-suspension via wind and wave action, while other sections have reportedly high turbidity values compared to what are considered pristine conditions (Croft and Chow-Fraser 2009). Jordan Harbour had a higher turbidity of 8.7 NTU compared to some wetlands in Georgian Bay characterized as being in pristine condition for the region, which have turbidity values ranging from 0.4-1.54 NTU (Croft and Chow-Fraser 2009). To combat these ongoing anthropogenic stressors, Jordan Harbour is a part of the Coastal Wetland Monitoring

Program. Currently, the only noted restoration strategy is shoreline improvement planned by the Niagara Peninsula Conservation Authority as a way to improve accessibility (Niagara Peninsula Conservation Authority 2015).

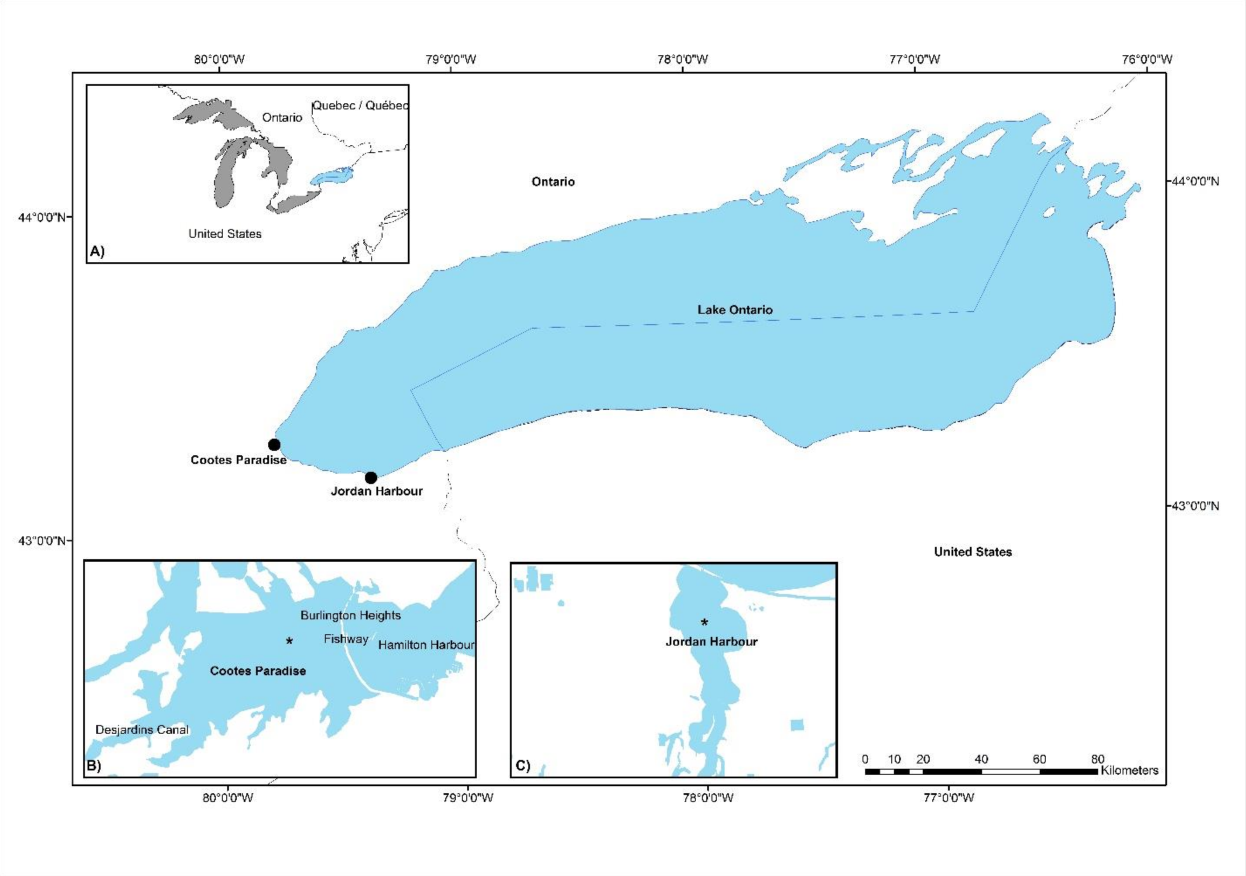

Figure 1. Map showing the locations of Cootes Paradise (Hamilton, Ontario) and Jordan Harbour (Lincoln, Ontario) along the Canadian shores of Lake Ontario. A) Lake Ontario in relation to the other Laurentian Great Lakes, B) Map of Cootes Paradise, with the coring location indicated by an *, and C) Map of Jordan Harbour, with the coring location indicated by an *.

Table 1. A summary of primary concerns, physical characteristics, ecological conditions, and remediation plans within Cootes Paradise and Jordan Harbour

	Cootes Paradise	***Jordan Harbour***
Location and Jurisdiction	Hamilton, Ontario Royal Botanical Gardens	Lincoln, Ontario Niagara Peninsula Conservation Authority
Anthropogenic Stressors of Primary Concern	High nutrient runoff Invasive carp High turbidity[1] Water regulation Pollution and toxic effluents	High nutrient runoff High sedimentation Invasive carp High turbidity[2]
Wetland Type	Drowned river-mouth	Drowned river-mouth
Ecological Status	Degraded[3]	Degraded[4]
Remediation Plans	Improved vegetation amount and diversity Removal and exclusion of invasive carp Shoreline restoration Improved water quality[5]	Shoreline restoration[4]

[1] Chow-Fraser et al. 1998

[2] Croft and Chow-Fraser 2009

[3] Kim et al. 2016

[4] Niagara Peninsula Conservation Authority 2015

[5] Thomasen and Chow-Fraser 2012

3.4 Methods

3.4.1 Field sampling

A UWITEC gravity corer with hammer action was used to collect sediment cores (lengths ~50-60 cm) from Jordan Harbour and Cootes Paradise in May and June of 2018, respectively, and cores were sectioned onsite into 0.5- and 1.0-cm intervals. Areas of high erosional inputs were avoided when collecting the sediment cores by consulting run-off maps to determine where major entry points into to the wetlands from the surrounding watershed were located (Royal Botanical Gardens 1998). We also avoided areas that have been dredged, such as the Desjardins Canal in Cootes Paradise.

3.4.2 Laboratory methods

Chronologies for the sediment cores were established using the ^{210}Pb method by gamma spectrometry (Appleby et al. 2001). ^{210}Pb dating was done at the Paleoecological Environmental Assessment and Research Laboratory at Queen's University (Kingston, Ontario) using the Constant Rate of Supply (CRS) model (Appleby 2001). Cladoceran subfossils were isolated from the sediment matrix using a 10% potassium hydroxide (KOH) solution to deflocculate sediment subsamples, after which samples were rinsed with deionized water, safranin glycerol was added to stain the cladoceran remains, and a couple drops of 80% ethanol was added as a preservative (Korhola and Rautio 2001). Slides were prepared using a pipette set to 50 µl per aliquot. Slides for Cootes Paradise received single aliquots, while slides prepared for Jordan Harbour received double aliquots (samples were dried between aliquots) due to lower cladoceran abundances. Cladoceran remains were identified under a light microscope at 200-400x magnification using standard methodologies (Korosi and Smol 2012a; 2012b). The number of individuals for each species were determined based on the most abundant body part identified, and a minimum of 70 individuals were counted per interval (Kurek et al. 2010). *Bosmina* size structure (carapace, mucro, and antennules lengths) was analyzed in the sediment core from Cootes Paradise using an Amscope B690C-PL microscope calibrated to measure size in micrometers, following methods outlined in Korosi et al. (2010). *Bosmina* size structure was only analyzed in the top 5-cm and bottom 5-cm in Jordan Harbour, due to low cladoceran abundances.

3.4.3 Statistical methods

Stratigraphic changes in species diversity were inferred using Hill's N2, calculated in R Studio version 1.1.463 using the Rioja and Vegan packages (Oksanen et al. 2012). A stratigraphically-constrained incremental sum of squares (CONISS) cluster analysis (Grimm 1987) was conducted to identify periods of significant change in Cladocera assemblages through time, and the number of significant clusters was determined by comparison with a broken-stick model of random distribution (Bennett 1996) using the "rioja: Analysis of Quaternary Science Data" package version 0.9-9. A Shapiro-Wilk test for normality was done on the *Bosmina* size measures of the antennule, mucro, and carapace lengths for the top and bottom 5 cm of each wetland in R Studio version 1.1.463. Size measures with normal distributions had a student's paired t-test done to determine if the size changes from bottom to top were significant. Size measures that were not normally distributed had a Wilcoxon Signed-Rank Test done to determine if the size changes from bottom to top were significant.

3.5 Results

3.5.1 Cootes Paradise

3.5.1.1 ^{210}Pb dating

A general non-monotonic decline in ^{210}Pb isotopic activity was observed downcore, indicating a stratigraphic record with a variable sedimentation rate was obtained (Figure 2). The oldest date obtained from the CRS chronology was ~1900 at 40 cm. ^{137}Cs isotopic activity was stable throughout the core, and no peaks were observed.

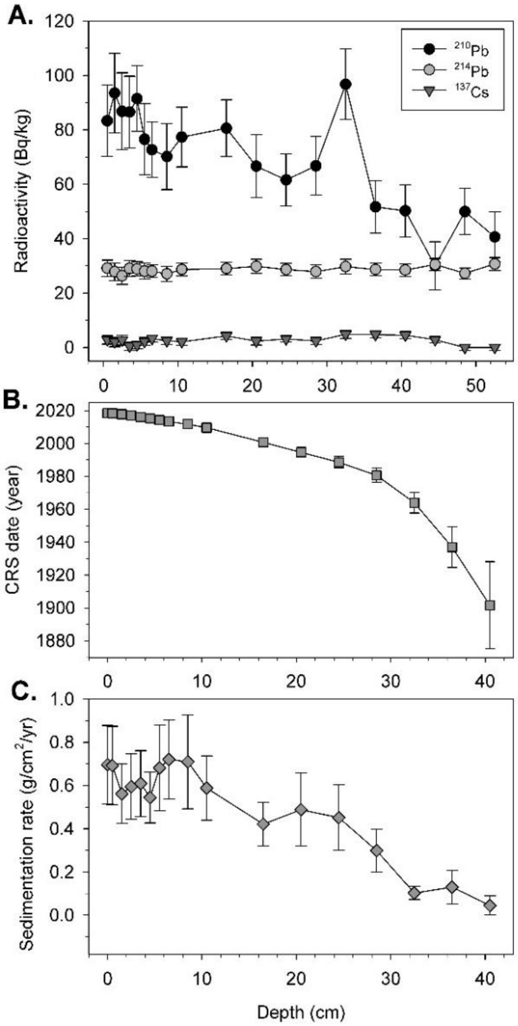

Figure 2. Results of ^{210}Pb dating for Cootes Paradise: A) radioisotopic activities for ^{210}Pb, ^{214}Pb and ^{137}Cs with depth; B) sediment core depth versus date based on the constant rate of supply (CRS) model; and C) ^{210}Pb-inferred sedimentation rate with depth based on the CRS model.

3.5.1.2 Cladocera assemblage

The main change in cladoceran assemblage observed in the Cootes Paradise sediment core was a shift in dominance from *Chydorus brevilabris* to *Bosmina* spp. at the bottom of the core (core depth 50 cm), above which the relative abundances of taxa were generally stable (Figure 3). In the earliest part of the sediment record, *Bosmina* were present at relative abundances of <30%, and showed an increasing trend above 50-cm (pre-1900), stabilizing at

~80-90% relative abundance above 45-cm to the surface. *Chydorus brevilabris* comprise ~50% relative abundance in the earliest part of the sedimentary record, decreasing to ~28% relative abundance at 50-cm. At 45-cm, *C. brevilabris* were found at ~8% relative abundance, and from 40 – 0 cm (~1900 – 2018), relative abundance ranged from ~1 – 5%. Additional species observed at low abundances (<5%) at 55-cm core depth included *Alona guttata*, *Alona circumfimbriata*, *Alona costata*, *Alona excisa, Camptocercus rectirostris, Sida crystallina, Graptolebris testudinaria*, and *Pleuroxus* spp. *A. guttata* appeared sporadically throughout the sediment record in abundances ranging from ~1 – 7%. *Pleuroxus* spp. were only observed in the earliest part of the sediment record (1-3%) and disappeared above a core depth of 40-cm (~1900). Hill's N2 indicates that cladoceran species diversity decreased by two-fold from 55-cm to 45-cm, above which it remained relatively stable.

3.5.1.3 *Bosmina* size structure

Bosmina carapace lengths ranged from ~183 – 225 μm, mucro lengths ranged from ~20 – 29 μm, and antennule lengths ranged from ~74 – 102 μm (Figure 3). Increases in mean length were observed for all size measures between 55 – 45 cm depth, corresponding with the decline in Hill's N2 diversity. Above 45-cm, carapace, mucro, and antennule sizes experienced only minor fluctuations.

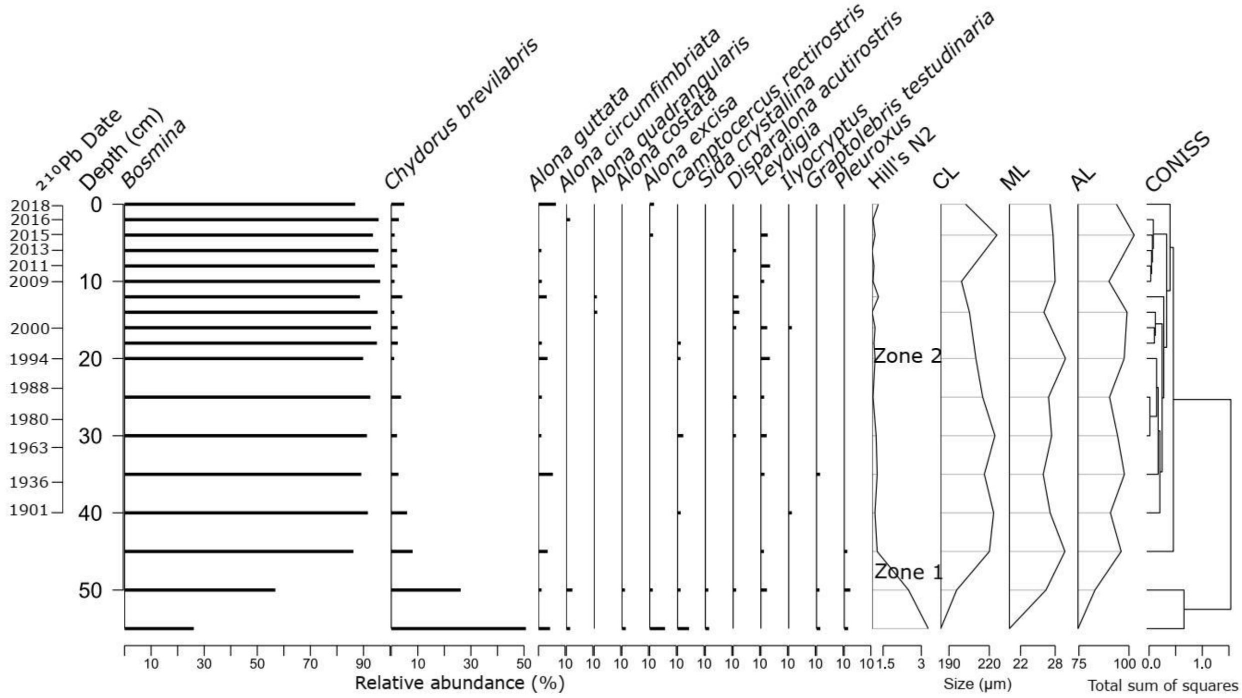

Figure 3. Stratigraphic diagram showing changes in subfossil Cladocera assemblages, Hill's N2 diversity, and *Bosmina* size measures in Cootes Paradise marsh (Hamilton, Ontario, Canada) Dates based on the Constant Rate of Supply model, and are shown on the left. Results of the constrained incremental sum of squares (CONISS) cluster analysis and broken stick zones are also shown. CL = carapace length; ML = mucro length; AL = antennules length.

3.5.2 Jordan Harbour

3.5.2.1 ^{210}Pb dating

Sediment mixing or an increased sedimentation rate is evident in the top ~10 cm of the sediment core, where the level of ^{210}Pb detected fluctuates and no declining trend is evident (Figure 4). A steady decline in ^{210}Pb isotopic activity was observed between 15-35 cm, when the ^{210}Pb approaches background. The earliest date provided by the CRS model is ~1900, at 30-cm. There was a discrepancy between the ^{210}Pb CRS age model and the ^{137}Cs, where a peak in ^{137}Cs isotopic activity (indicating ~1963) was evident at ~40 cm core depth. The ^{137}Cs age model indicates a date of ~1945 at 30-cm, compared to ~1900 based on the CRS model.

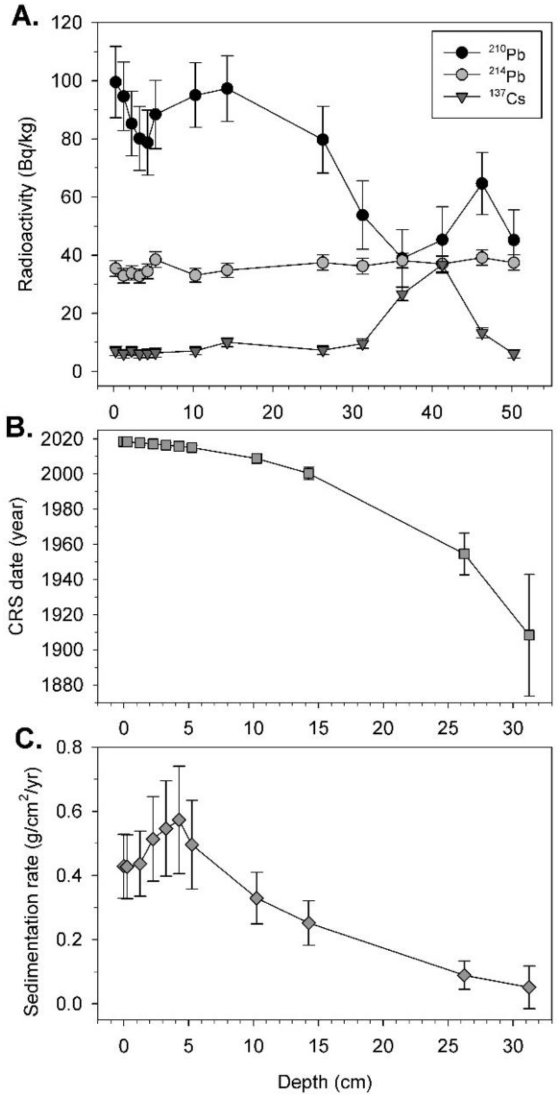

Figure 4. Results of ^{210}Pb dating for Jordan Harbour (Lincoln, Ontario, Canada):of A) radioisotopic activities for ^{210}Pb, ^{214}Pb and ^{137}Cs with depth; B) sediment core depth versus date based on the constant rate of supply (CRS) model, and C) ^{210}Pb-inferred sedimentation rate with depth based on the CRS model.

3.5.2.2 Cladocera assemblage

Bosmina were the dominant taxa in Jordan Harbour throughout the sediment record, ranging between ~65 – 90% in relative abundance. *Chydorus brevilabris* were the next most abundant taxon in Jordan Harbour, ranging in relative abundances from 4 – 20% (Figure 4). *C. brevilabris* exhibited a slight increase from ~10% to 15 – 20% abundance above a core depth of

10-cm, with a corresponding decrease in *Bosmina* spp. from ~80 to 70% abundance. The highest relative abundance of *Bosmina* (and lowest relative abundance of *C. brevilabris*) was observed in the surface interval.

Several changes were observed in the littoral and benthic cladocerans (Figure 4). *Alona guttata* were found in the bottom three intervals (40 – 50 cm depth) at relative abundances of 2 – 5%, but were not observed again until a core depth of 18-cm, when relative abundances ranged from 2 – 7% between 18-and 0-cm core depth. A similar trend was observed for *Alona quadrangularis*. *Alona circumfimbriata* were only observed in the sediment core between 35 – 25 cm depth, when *A. guttata* and *A. quadrangularis* were absent, at relative abundances of <5%. *Pleuroxus* spp. appeared in the sediment at core depth of 16-cm (~2008) and ranged from ~2-8% relative abundance between 16-and 0cm core depth. The Hill's N2 diversity index indicates that the lowest diversity occurs at 20-cm (~2000) and the highest diversity occurs at 4-cm (~2017).

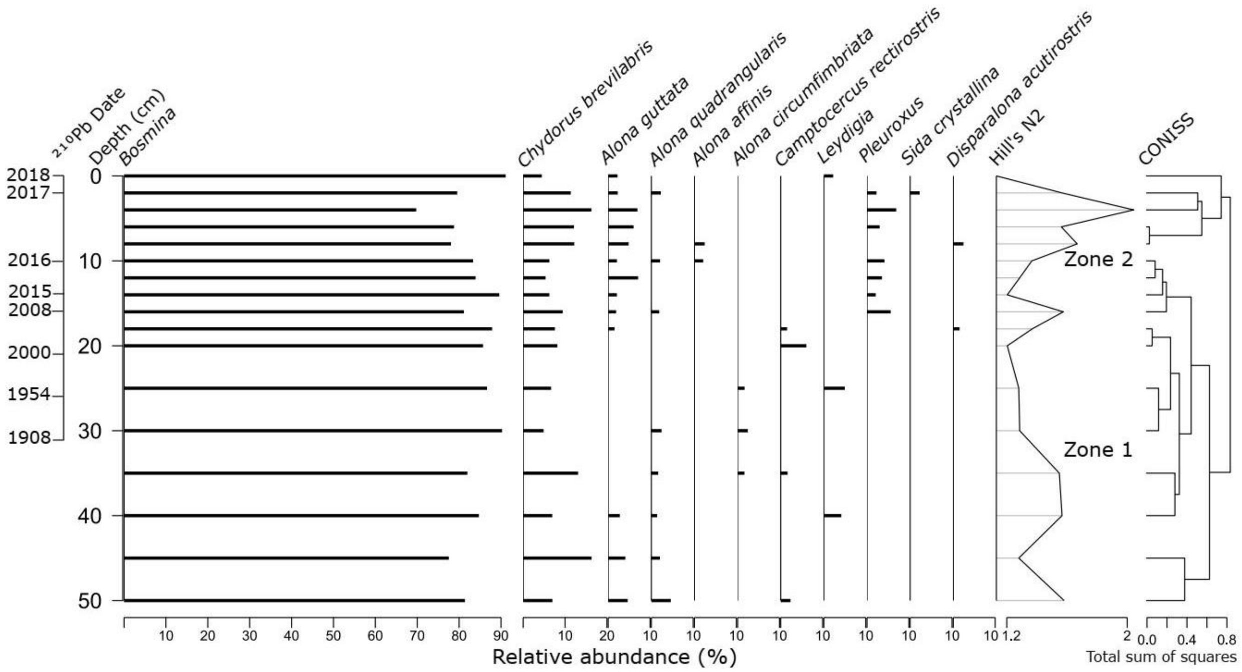

Figure 5. Stratigraphic diagram showing changes in subfossil Cladocera assemblages and Hill's N2 diversity in Jordan Harbour (Lincoln, Ontario, Canada). Dates based on the Constant Rate of Supply model are shown on the left. Results of the constrained incremental sum of squares (CONISS) cluster analysis and broken stick zones are also shown.

3.5.3 “Top-Bottom” comparison of Bosmina *size in Cootes Paradise and Jordan Harbour*

Boxplots were generated to compare size measures between the top (0-5 cm) and bottom (50-55 and 45-50 cm) sediments in Cootes Paradise and Jordan Harbour respectively, in order to visualize changes in the overall distribution of *Bosmina* size structure between present-day and ~1900. In Cootes Paradise, there was considerable overlap in size structure for all measures (carapace, mucro, and antennules lengths), but a small increase in the median antennule length in the most recent sediments is evident (Figure 6). The median *Bosmina* carapace lengths are comparable between top and bottom intervals at ~190 μm. The upper whisker extends to ~300 μm and the lower whisker reaches ~130 μm, with an interquartile range of ~175 – 240 μm. Cootes Paradise has one outlier in the top interval. In the bottom interval, the upper whisker extends to ~250 μm and the lower whisker reaches ~125 μm. There is an interquartile range of ~175 – 210 μm, and no outliers. The median *Bosmina* mucro lengths are comparable between the top and bottom at ~27 μm and ~24 μm respectively. In the top interval, the upper whisker reaches to ~ 41 μm and the lower whisker extends to ~12 μm, with an interquartile range of ~24 – 32 μm. There are two outliers, one above the upper whisker and one below the lower whisker. In the bottom interval, the upper whisker is comparable to the top interval, extending to ~40 μm. The lower whisker reaches ~11 μm, and there is an interquartile range of ~17 – 27 μm. There is one outlier. The median *Bosmina* antennule lengths varied between the top and bottom intervals, with lengths of ~95 and ~85 μm respectively. The upper whisker in the top interval extends to ~130 μm and the lower whisker reaches ~63 μm. The interquartile range is from ~85 – 110 μm, with two outliers, one above the upper whisker, and one below the lower whisker. The bottom interval upper whisker extends to ~125 μm, the lower whisker reaches ~50 μm and the interquartile range is ~65 – 95 μm. There are no outliers in the bottom interval. Shapiro-Wilk tests to determine if the data were normally distributed were run. The carapace top size measures were not normally distributed while the bottom size measures were normally distributed, and p=0.3 for the Wilcoxon Signed-Rank Test, indicating that the size changes were not statistically significant. The mucro top and bottom size measures were both normally distributed, and p=0.07 for the student's t-test, indicating that the size changes were not statistically significant. The antennule top and bottom size measures both had normal distributions, and p=0.001 for the student's t-test, indicating that the size changes were statistically significant.

In Jordan Harbour, overlap in all size measures is evident, although there is a decrease in the median carapace length and an increase in the median antennule length in the top interval (Figure 6). The median carapace lengths vary between the top and bottom with values of ~174 and ~ 208 μm respectively. The upper whisker of the top interval reaches ~250 μm, the bottom whisker extends to ~140 μm, and the interquartile range is from ~155 – 205 μm. There are two outliers. The upper whisker of the bottom interval reaches to ~250 μm, the lower whisker extends to ~150 μm, and the interquartile range is ~190 – 225 μm. There are no outliers. The median mucro lengths for *Bosmina* are comparable between the top and bottom intervals at ~23 and ~24 μm respectively. The upper whiskers are also comparable, the top interval extending to ~37 μm and the bottom interval extending to ~36 μm. In the top interval, the lower whisker reaches ~10 μm, the interquartile range is ~18 – 27 μm, and there are three outliers. In the bottom interval, the lower whisker extends to ~15 μm, the interquartile range is ~20 – 28 μm, and there are four outliers. The median antennule lengths for *Bosmina* vary between the top and bottom intervals, measuring ~99 and ~85 μm respectively. The upper whisker in the top interval extends to ~130 μm, the lower whisker reaches ~55 μm, and the interquartile range is ~85 – 110 μm. The upper whisker in the bottom interval reaches ~110 μm, the lower whisker extends to

~50 µm, and the interquartile range is ~75 – 95 µm. There is one outlier. Shapiro-Wilk tests were also run to determine if the top and bottom interval size measures were normally distributed. For the carapace lengths, the top interval was not normally distributed while the bottom interval was normally distributed, and p=0.01 for the Wilcoxon Signed-Rank Test, indicating that the size changes were statistically significant. Both the top and bottom intervals were not normally distributed for the mucro size measures, and p=0.3 for the Wilcoxon Signed-Rank Test. The antennule top and bottom intervals were normally distributed, and p=0.02 for the student's paired t-test, indicating that the size changes were statistically significant.

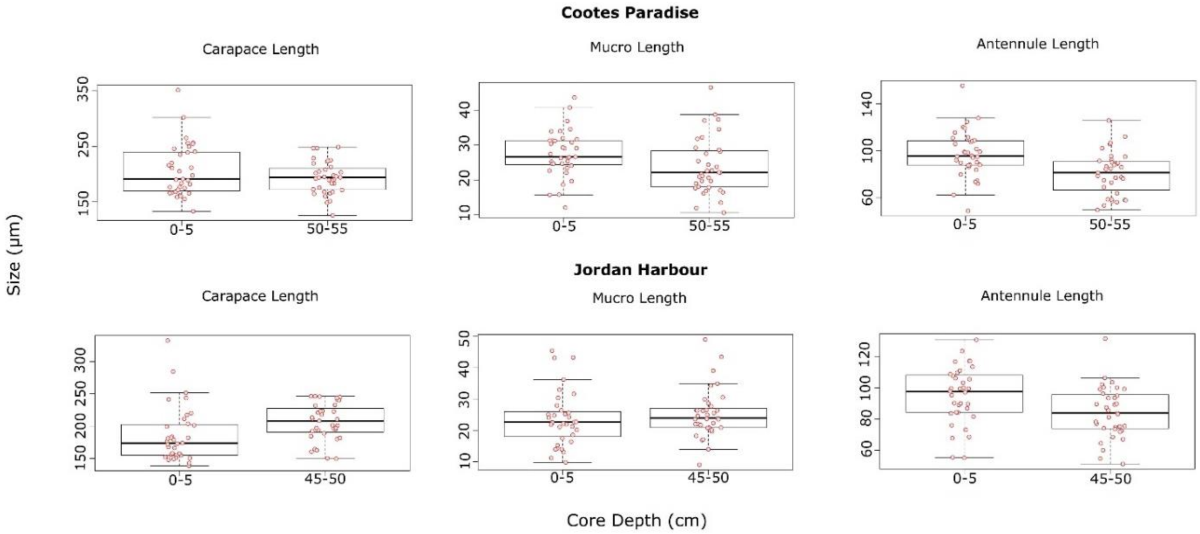

Figure 6. Boxplots comparing the distribution of subfossil *Bosmina* size measures between the top 5 cm (recent) and bottom 5 cm (~1900) sediments in Cootes Paradise and Jordan Harbour.

3.6 Discussion

3.6.1 Cootes Paradise

Bosmina and *Chydorus brevilabris* were the dominant cladocerans throughout the last ~120 years in Cootes Paradise based on subfossil remains. We did not observe any *Daphnia* subfossil remains, in contrast to previous investigations into extant zooplankton communities over the past ~60+ years that recorded prominant *Daphnia* populations (Chow-Fraser et al. 1998). *Daphnia* were particularly abundant in the vegetated areas of the western portion of the marsh until the 1940s, after which *Daphnia* declined rapidly into the 1970s due to increased turbidity and were replaced by smaller, planktonic *Bosmina* taxa (Chow-Fraser et al. 1998). Further research has showed that during years with colder springs, *Daphnia* were able to repopulate many areas of the marsh, especially as carp exclusion allowed for some regrowth of aquatic macrophytes in areas where *Daphnia* has inhabited in the past (Lougheed et al. 2004).

The discrepancies between the monitoring and paleolimnogical records are likely due to *Daphnia* associating with emergent aquatic macrophytes in Cootes Paradise, a predator-avoidance strategy, as *Daphnia* were not found in many of the deeper, open-water areas of the marsh (Lougheed et al. 2004). We collected our sediment core from the deeper basin to the west of Burlington Heights, at the eastern end of the marsh. The eastern part of the marsh (where we collected our sediment core) was the first to lose its emergent vegetation, and the vegetation gradually receded westward until very little remained (Lord 1993). From this, we infer that our sediment core is spatially-constrained to recording the ecological history of the eastern portion of Cootes Paradise (described below), and does not incorporate ecological changes in the western part of the marsh.

A shift in cladoceran species assemblage from dominance by *Chydorus brevilabris* to *Bosmina* occurred ~1900. This corresponds to the estimated time that invasive carp arrived in Cootes Paradise. Invasive carp were first reported in Wilmot Creek Marsh in the Durham region in the late 1800s, and rapidly expanded to other Great Lakes coastal wetlands (Royal Botanical Gardens 1998). It is estimated that carp was the dominant fish species in Cootes Paradise by the

1930s, outcompeting native piscivorous fish species and altering the predation dynamics of the wetland (Royal Botanical Gardens 1998). The introduction of carp in Great Lakes coastal wetlands results in increases in turbidity as a result of mudding and the uprooting of submerged and emergent aquatic vegetation. *Chydorus* species are generally found in macrophyte-covered areas (Tremel et al. 2000), whereas *Bosmina* are true planktonic taxa (Korosi et al. 2013; Matveev 1991). The decrease in *Chydorus brevilabris* at 45-cm depth and subsequent increase in *Bosmina* is likely tracking a loss in emergent aquatic vegetation in the area our core was extracted from.

The *Bosmina* sizes (carapace and appendage lengths) we recorded in the sediment core from Cootes Paradise are among the smallest values reported in the paleolimnological literature (e.g. Korosi et al. 2010; Sakamoto and Hanazoto 2008; Sprules et al. 1984), and median carapace and appendage lengths are not appreciably different between top and bottom sediments (with the exception of a slight increase in antennule lengths). An increase in *Bosmina* appendage lengths typically indicates greater predation on *Bosmina* by predatory invertebrates such as copepods (Korosi et al. 2013), and the slight increase in antennule lengths may be an indication of this. Carp are a planktivorous fish species that tend to feed on larger species of zooplankton (Florian et al. 2016). The lack of appreciable change in *Bosmina* size measures indicate that the documented changes to fish community structure in Cootes Paradise (Kim et al. 2016) did not have a substantial impact on the size structure of *Bosmina* populations.

The Cootes Paradise fishway was opened in 1996 (Royal Botanical Gardens 1998), and while there is evidence that carp populations have decreased since that time (Lougheed et al. 2004), there are no corresponding changes in cladoceran assemblages or *Bosmina* size structure. This suggests that, despite the carp reductions and other efforts at improving water quality, there has not been any extensive ecological recovery in Cootes Paradise Cladocera, corroborating recent monitoring records that hysteresis has delayed or prevented ecological recovery in the marsh (Vincent 2017).

3.6.2 Jordan Harbour

Due to the relatively limited historical data and scientific research on Jordan Harbour, the timeline of land-use changes and subsequent environmental impacts is less well characterized compared to Cootes Paradise. Thus, our paleolimnological study provides some of the only insight into long-term ecological change in Jordan Harbour since ~1900. Similar to Cootes Paradise, *Bosmina* were the dominant cladoceran in Jordan Harbour throughout the sediment record. The main changes observed in the cladoceran assemblage of Jordan Harbour occur for the littoral and benthic taxa. A decrease in *Chydorus brevilabris* was observed at 35-cm, (pre-1900) corresponding to the disappearance of *Alona guttata* subfossil remains from the sediment record. These species are known to live in association with macrophytes (Korhola and Rautio 2001), and thus this change indicates a decrease in aquatic vegetation, similar to observations in Cootes Paradise that may also be attributed to increases in turbidity.

In contrast with Cootes Paradise, *Chydorus brevilabris* and *Alona guttata* increase again above a sediment core depth of 10-cm (~2016) and 18-cm (~2005) respectively, suggesting some ecological recovery has occurred. *Pleuroxus* spp., another group of cladocerans that live associated with vegetation (Korhola and Rautio 2001), appear for the first time in the sediment record at a core depth of 16-cm (~2008). These changes suggest that macrophyte coverage in the marsh has increased over the last ~15 years. In 2005, part of the marsh was acquired by the Niagara Peninsula Conservation Authority, who have been in the process of implementing shoreline restoration plans (Rosts 2010). Our paleoecological results indicate that Jordan

Harbour has demonstrated some ecological recovery, which suggests that shoreline restoration plans have the potential to be successful in improving water quality.

A "top-bottom" comparison of *Bosmina* size structure in Jordan Harbour showed that, similar to Cootes Paradise, there is considerable overlap in size structure skewed towards smaller individuals, and a slight increase in antennule length was observed between top and bottom intervals. In contrast to Cootes Paradise, there was a slight decrease in carapace length in the surface sediments. *Bosmina* are subject to predation simultaneously by both visual, size-selective fish and gape-limited invertebrates. A smaller carapace length (a correlate of total body size) may indicate an increase in the strength of predation by planktivorous fish, which select for larger individuals, while an increase in antennule length may be a response to copepod predation (Korosi et al. 2013). Long, curved antennules shield vulnerable swimming antennae from damage during an attack by small grasping copepod predators, increasing the likelihood that the bosminid will survive the attack (Kerfoot 1975; Post et al. 1995). Due to low abundances of cladocerans in Jordan Harbour, and a high proportion of fragmented remains, we were unable to measure *Bosmina* size structure downcore to establish the trajectory and timing of changes in size measures. Overall, the dominance of small-bodied Bosmina with short appendage lengths indicates that fish planktivory in Jordan Harbour and Cootes Paradise has been a dominant control on *Bosmina* populations throughout the last ~120 years.

3.7 Conclusions

In Cootes Paradise, a shift in dominance from *Chydorus brevilabris* to *Bosmina* spp. occurred in the early 1900s, corresponding with the suspected timing of invasion of carp and the subsequent decrease in submerged vegetation. Despite recent restoration efforts to exclude carp from the wetland, the paleolimnological record suggests that very little, if any, recovery has occurred in Cootes Paradise Marsh. In Jordan Harbour, a decrease in *C. brevilabris* and *Alona guttata* in the early part of the sediment record (pre-1900) indicates a loss of littoral habitat similar to Cootes Paradise Marsh; however, ecological recovery is evident after ~2008, when *C. brevilabris* and *A. guttata* increase in relative abundance and *Pleuroxus* spp., another group of littoral cladocerans, appear in the sediment record. Overall, this study shows that, despite challenges with sediment mixing and high sedimentation rates, paleolimnological methods can be useful for documenting long-term ecological change in Laurentian Great Lakes coastal wetlands, extending recent monitoring records further back in time. However, a lack of strong sediment focusing in large, relatively uniformly shallow systems, may result in sediment cores reconstructing localized changes rather than integrating changes across the entire wetland. This was evident in the sediment core from Cootes Paradise, which did not capture well-documented historical changes in *Daphnia* populations. With these caveats in mind, future paleolimnological investigations in Great Lakes coastal wetlands still offer an opportunity to document the range of natural variability, the success of previous mitigation efforts, as well as assess how various land-use changes in the surrounding watersheds have altered ecological conditions in the wetlands.

3.8 Acknowledgements

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Chapter 4: General conclusions

4.1 Thesis summary and conclusions

Anthropogenic stressors are impacting aquatic ecosystems at unprecedented rates compared to previous times in history, and Great Lakes coastal wetlands in highly developed watersheds of southern Ontario are frequently the recipients of the bulk of the changes. However, remediating these wetlands has proven difficult due to a lack of long-term monitoring data, as the range of natural variability, reference conditions, and the ecological changes coastal wetlands have undergone are poorly understood in many areas. The primary goal of this thesis was to assess the usefulness of integrating paleolimnological methods into ongoing management efforts in Laurentian Great Lakes coastal wetlands of southern Ontario. I aimed to determine if paleoecological data can be used to evaluate long term ecosystem change in Great Lakes coastal wetlands, as well as how paleolimnological data can be used to inform present and future ecosystem restoration strategies in Great Lakes coastal wetlands. The research presented in this thesis aimed to achieve this goal by answering three main research questions regarding the application of paleolimnology in Great Lakes coastal wetlands, explored in two different studies described in Chapters 2 and 3.

4.1.1 Can we obtain high-resolution stratigraphic sediment records for the last ~100 years?

Long cores were collected from each of the three wetlands using a gravity corer, with the goal of establishing suitable chronologies for the last ~100 years to gain a greater understanding of the changes that have occurred in Great Lakes coastal wetlands in southern Ontario during the last century. As a whole, I was able to achieve this goal, as the cores collected and dated spanned

~70 – 120 years in all three wetlands. I used the Constant Rate of Supply (CRS) model because sedimentation rates have varied over the last ~100 years, and successfully established age-depth models based on exponential decay of ^{210}Pb in all three wetlands. Temporal resolution of sediment intervals was low due to some sediment mixing that is inevitable in shallow systems in urbanized watersheds, likely due to increased sediment loads and/or sediment resuspension. This also meant that meant that the gravity cores needed to be long (~50 cm) in order to capture at least the last ~100 years. The obtainment of stratigraphic sediment records in Cootes Paradise, Jordan Harbour, and McLaughlin Bay was possible due to the hydrological nature of these wetlands, which allows some sediment focussing to occur. While it was feasible for the wetlands I chose to study, paleolimnolgoical approaches are not possible in all Great Lakes coastal wetlands.

4.1.2 How have Cladocera responded to coastal wetland degradation in southern Ontario?

Overall, Cladoceran assemblages in all three degraded wetlands were dominated by pelagic *Bosmina* species, and secondarily by *Chydorus brevilabris*. Diversity in all of the wetlands was low throughout the stratigraphies, particularly during known periods of degradation in the early 1900s. One major stressor common to all three wetlands throughout the last ~50-100 years was the introduction of invasive carp, and invasive carp introduction was the most likely cause of ecological changes in Cladocera in Cootes Paradise, mainly the transition from a *Chydorus*-dominated to a *Bosmina*-dominated ecosystem. Despite efforts towards carp exclusion from Cootes Paradise, Cladocera have not shown pronounced recovery, most likely due to multiple interacting stressors contributing to the continued degraded status of the wetland. Littoral cladocerans, although a small component of the cladoceran assemblage (<10%), were most sensitive to changing ecosystem conditions in all three wetlands studied. Littoral cladocerans are most likely responding to the prevalence of aquatic macrophytes in the wetlands; the loss of littoral cladocerans track the loss of aquatic macrophytes, and conversely, their reappearance in Jordan Harbour most likely indicates some recovery in aquatic macrophyte communities due to shoreline restoration efforts.

4.1.3 Does Bosmina *size structure respond to known fluctuations in fish community structure?*

Bosmina size measurements were conducted for all three wetlands, to assess whether they tracked to known fluctuations in fish communities. *Bosmina* size measures in all 3 wetlands were smaller than what has been reported in previous studies (Korosi et al., 2010; Sakamoto and Hanazoto, 2008; Sprules et al., 1984), and surprisingly did not exhibit any strong responses to known historical shifts in fish assemblages, where records indicate that shifts from piscivorous to planktivorous fish populations have occurred (Kim et al., 2016). Great Lakes coastal wetlands provide nursery habitat for fish fry, and zooplankton planktivory has likely always been a strong pressure on *Bosmina* throughout the history of the Great Lakes coastal wetlands. The unfortunate consequence of this is that *Bosmina* size structure may not be a sensitive measure of changes in trophic status in these wetlands.

4.2 Management implications

Paleolimnological records offer a unique perspective on Great Lakes coastal wetlands that have undergone remediation in the past several years, and can provide insight into methods that are both effective and ineffective. This thesis demonstrates the intricacy of coastal wetlands, particularly those which are influenced by multiple interacting anthropogenic stressors over a long period of time. Although many coastal wetlands have undergone management plans to improve the overall health and quality by managing authorities like the Royal Botanical Gardens (Cootes Paradise), Central Lake Ontario Conservation Authority (McLaughlin Bay), and Niagara Peninsula Conservation Authority (Jordan Harbour), the paleolimnological records indicate that due to the interacting nature of the stressors, future directions need to address the multiple major issues simultaneously.

In Cootes Paradise, carp exclusions to reduce turbidity and improve some aspects of water quality to allow native wetland vegetation to recolonize have had little impact on the Cladocera assemblages, corroborating contemporary observations that Cootes Paradise continues to exhibit substantial hysteresis. Recently, local news outlets reported that 24-billion liters of raw sewage was released into Cootes Paradise between 2014 and 2018, further confounding recovery efforts. In Jordan Harbour, there was evidence of recovery with the return of littoral cladocerans following some shoreline reconstruction, suggesting that one of the more major issues was turbidity resulting from shoreline degradation. In McLaughlin Bay, barrier beach breaches had positive, but short-lived, impacts on the Cladocera community that were not evident in the diatoms. Recent decreases in diatom species diversity (not evident in the Cladocera) may be linked to increases in chloride concentrations from road salt that could potentially be partially alleviated by a permanent opening to Lake Ontario that would allow flushing of wetland waters combined with sustained efforts to reduce salt loadings to the marsh.

Extensive efforts have been made to establish consistent water quality and ecological indicators to be used in bi-national Great Lakes coastal wetland monitoring programs (United States Environmental Protection Agency, 2018). However, widespread ecological changes have been occurring in coastal wetlands since well before monitoring programs began. Consequently, the development of paleolimnological approaches to complement ongoing wetland monitoring could provide a historical context for wetland management, at least for open-water wetlands that have stratigraphic sediment accumulation. My thesis demonstrated that paleolimnology can indeed document long-term ecological trends in Great Lakes coastal wetlands, with some important caveats in mind. For example, diatoms and Cladocera can give very different interpretations on ecological change, as was evident in McLaughlin Bay. The use of multiple ecological indicators could provide a broader picture on the history of ecological change within the wetlands. Overall, it is evident from my research that anthropogenic stressors are indeed influencing the ecology of wetlands, with major ecological changes occurring in the early 20th century, and that paleolimnology can provide a useful tool to complement ongoing bi-national wetland monitoring protocols and future management.

4.3 Future directions

Collectively, the research presented in this thesis highlights the importance of understanding the history of Great Lakes coastal wetlands for effective and appropriate remediation measures, especially in areas where monitoring is limited or nonexistent, using three highly degraded wetlands in southern Ontario as case studies. The potential for future work on applying and improving the use of paleolimnology in Great Lakes coastal wetlands is vast, and could include large-scale surface “calibration” sets, analyzing paleoecological indicators across large spatial gradients in wetland degradation health status, size, catchment features, type, and location. Ideally, this would be connected to the datasets generated through the many current and past Great Lakes coastal wetland monitoring programs, and paleoecological indicators would complement the indicators used by the water quality monitoring program. This information could be used to determine which paleoecological indicators are ideal for different purposes, for example tracking changes in certain water quality parameters such as salinity, turbidity, and total phosphorus. New “calibration sets” specific to Great Lakes coastal wetlands could then support more down-core paleolimnological studies on wetlands across degradation and land-use gradients, further supporting local and bi-national management and remediation programs throughout the Laurentian Great Lakes region. Anthropogenic stressors affect wetlands throughout the entire Laurentian Great Lakes region, and paleolimnology is a useful tool in determining how and why certain responses occur in a given marsh, and how we as humans can combat the effects of our actions moving into the future.

4.4 References

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Appendix A: Raw data

A1. Raw diatom counts for McLaughlin Bay (chapter 2)

Interval (cm)	*Achnanthes acres*	*Achnanthes aff. engelbrechtii* (*)	*Achnanthes clevei*	*Achnanthes conspicua* (*)	*Achnanthes delicatula*	*Achnanthes exigua*	*Achnanthes implexiformis*	*Achnanthes lanceolata* (*)	*Achnanthes laterostrata*	*Achnanthes meniculus*	*Achnanthes minutissima* (*)	*Achnanthes oblongella*	*Achnanthes rupestoides*	*Achnanthes subatomoides*	*Achnanthes submolesta*	*Achnanthes subsalsa*	*Achnanthes suchlandtii*
0-0.25	0	1	0	5	1	0	0	2	0	0	6	0	0	0	0	0	0
0.25-0.5	0	0	0	4	1	3	0	5	0	0	3	0	0	0	0	0	0
0.5-0.75	0	0	0	5	1	2	0	3	0	0	2	0	0	0	0	0	0
0.75-1	0	1	0	2	0	1	1	3	0	0	2	0	0	0	0	0	0
1-1.25	0	1	0	4	0	2	0	0	0	0	2	0	0	0	0	0	0
1.25-1.5	0	0	0	4	1	2	0	0	0	0	7	0	0	0	0	0	0
1.5-1.75	0	0	0	10	1	0	0	1	0	0	1	0	0	0	0	0	0
1.75-2	1	3	0	5	1	0	0	3	0	0	3	0	1	1	0	0	0
2-2.25	1	1	0	4	1	3	0	0	0	0	0	0	0	0	0	0	0
2.5-2.75	0	2	1	8	0	1	2	2	0	0	2	0	0	0	0	0	0
3-3.25	0	3	0	13	0	3	0	0	0	0	0	2	0	0	0	0	0
3.5-3.75	0	3	0	7	0	1	0	8	1	0	0	0	0	0	0	0	0
4-4.25	0	6	0	16	0	4	0	4	0	0	2	0	0	0	2	0	0
5-5.25	0	1	0	9	0	0	0	4	0	0	5	0	0	0	0	0	0
6-6.25	0	12	0	11	0	1	0	5	0	0	0	0	0	0	0	0	0
7-7.25	1	2	0	13	0	0	0	7	1	0	3	0	0	0	1	0	0
8-8.25	0	13	0	14	0	0	0	5	0	0	1	0	0	0	0	0	0
9-9.25	0	5	2	11	0	2	0	6	0	1	0	0	0	0	0	0	0
10-10.5	0	4	1	10	0	2	0	0	0	0	1	0	0	0	0	0	0
12-12.5	0	2	0	10	0	1	0	2	0	0	3	0	0	0	0	0	0
14-14.5	1	2	0	12	0	4	1	4	0	0	4	0	0	0	0	0	0
16-16.5	1	3	0	12	0	1	1	8	0	0	0	0	0	1	0	0	0
18-18.5	2	0	1	3	0	2	0	1	0	0	1	0	0	0	0	0	0
20-20.5	0	2	0	12	1	0	0	5	0	0	8	0	0	0	0	0	0
22-22.5	0	5	0	11	0	1	0	15	0	0	6	0	0	0	0	0	0
24-24.5	0	3	0	12	0	0	1	7	0	0	10	0	0	0	0	1	1
26-26.5	0	3	0	11	0	4	0	3	0	0	8	1	0	0	0	0	0
28-28.5	1	3	0	12	1	3	1	3	0	0	5	2	0	0	0	0	0
30-30.5	0	1	0	11	1	0	1	6	0	0	5	2	0	0	0	0	0

Raw diatom counts for McLaughlin Bay (cont'd)

Interval (cm)	*Amphora fogediana*	*Amphora inariensis*	*Amphora libyca* (*)	*Amphora ovalis*	*Amphora pediculus* (*)	*Anomoeoneis vitrea*	*Aulacoisera ambigua*	*Aulacoisera crenulata*	*Aulacoisera granulata*	*Aulacoisera islandica*	*Caloneis amphisbaena*	*Caloneis bacillum*	*Caloneis schumanniana*	*Cocconeis placentula* (*)	*Cocconeis neodiminuta*
0-0.25	0	0	10	2	6	0	0	0	2	0	1	0	0	5	0
0.25-0.5	0	1	7	0	6	0	0	0	4	0	0	0	0	5	1
0.5-0.75	0	0	5	3	1	0	0	0	0	0	0	0	0	4	0
0.75-1	0	0	6	0	6	0	0	0	4	0	0	0	0	6	0
1-1.25	0	2	6	0	4	0	0	1	2	0	1	0	0	9	0
1.25-1.5	0	0	12	1	6	0	0	0	5	0	0	0	0	8	0
1.5-1.75	0	2	8	2	6	0	0	0	2	0	0	0	0	8	0
1.75-2	3	2	9	2	0	0	0	0	2	1	1	0	0	8	0
2-2.25	1	2	3	1	9	0	0	0	0	0	0	0	0	8	0
2.5-2.75	0	0	7	1	5	0	1	0	0	0	1	0	0	9	0
3-3.25	0	2	7	2	8	0	0	0	0	0	0	0	1	13	0
3.5-3.75	0	0	11	2	3	0	0	0	0	0	0	0	0	6	0
4-4.25	0	4	6	2	10	0	0	0	0	0	0	0	0	16	0
5-5.25	0	1	8	1	6	0	0	1	2	0	2	0	0	9	0
6-6.25	0	0	9	2	10	0	0	0	0	0	0	0	0	9	0
7-7.25	1	0	10	0	7	1	0	0	0	0	3	0	0	6	0
8-8.25	0	0	9	2	7	0	0	4	0	0	0	0	0	6	0
9-9.25	0	0	9	5	5	0	0	2	0	0	1	0	0	5	0
10-10.5	0	4	17	4	10	0	0	0	0	0	0	0	0	19	0
12-12.5	0	5	5	0	4	0	0	1	0	2	0	0	0	10	0
14-14.5	0	2	7	1	5	0	0	0	0	0	2	0	0	15	0
16-16.5	0	2	6	1	4	0	0	0	0	1	1	0	0	16	0
18-18.5	0	0	7	1	3	1	0	0	0	0	1	0	0	13	0
20-20.5	0	0	11	0	4	0	0	0	0	0	0	0	0	5	0
22-22.5	0	2	16	1	0	0	0	0	1	0	0	0	0	18	0
24-24.5	0	0	8	2	7	0	0	0	0	0	0	0	0	8	0
26-26.5	0	0	12	1	6	0	0	0	1	0	0	0	0	11	0
28-28.5	0	0	11	2	12	0	0	0	0	0	0	1	0	16	0
30-30.5	0	0	0	0	0	0	2	0	0	0	0	0	0	17	0

Raw diatom counts for McLaughlin Bay (cont'd)

Interval (cm)	*Cocconeis neothumensis*	*Cyclotella bodanica*	*Cyclotella glabriuscula*	*Cyclotella meneghiniana* (*)	*Cyclotella ocellata* (*)	*Cyclotella pseudostelligera*	*Cyclotella stelligera* (*)	*Cyclotella striata*	*Cyclotella styriaca*	*Cymatopleura elliptica*	*Cymatopleura solea*	*Cymbella caespitosa*	*Cymbella cesatii*	*Cymbella cistula*	*Cymbella cymbiformis*	*Cymbella descripta*	*Cymbella ehrenbergii*
0-0.25	1	0	0	10	13	1	3	0	1	0	2	0	0	3	0	0	0
0.25-0.5	1	0	0	5	8	1	0	0	0	0	2	1	0	1	0	0	0
0.5-0.75	0	0	0	2	7	1	1	0	0	0	1	1	1	6	0	0	0
0.75-1	0	0	0	6	7	2	3	0	0	0	0	0	0	0	0	0	0
1-1.25	0	0	1	2	14	2	4	0	0	0	0	0	0	0	0	0	0
1.25-1.5	0	0	0	6	8	3	2	1	0	0	4	0	0	0	0	0	2
1.5-1.75	0	1	0	3	6	0	1	0	0	0	0	0	0	2	0	0	0
1.75-2	0	0	0	1	4	0	1	0	0	0	3	2	0	6	0	0	0
2-2.25	0	0	0	6	13	4	1	0	0	0	1	0	0	1	0	0	0
2.5-2.75	0	0	0	5	15	0	3	0	0	0	3	0	1	8	0	0	0
3-3.25	0	1	0	4	8	0	4	0	0	0	1	0	0	4	0	2	0
3.5-3.75	1	0	0	5	8	0	3	0	0	0	2	0	0	2	0	2	0
4-4.25	0	0	0	4	21	0	7	0	0	0	0	0	0	6	0	0	0
5-5.25	0	0	0	5	5	0	1	0	0	0	2	1	0	6	0	0	0
6-6.25	0	0	0	6	15	0	1	0	0	0	2	0	0	2	0	0	0
7-7.25	0	0	0	3	27	0	5	4	0	0	1	3	0	2	0	1	0
8-8.25	1	0	0	2	6	0	5	0	0	0	1	2	0	1	0	0	0
9-9.25	0	0	0	8	7	0	4	1	0	0	1	0	0	6	0	0	0
10-10.5	0	0	0	3	0	0	2	1	0	1	1	0	0	6	0	0	0
12-12.5	0	0	0	8	7	0	2	0	0	0	0	0	0	1	0	0	0
14-14.5	1	0	0	11	12	0	8	0	0	0	2	2	0	1	0	0	0
16-16.5	3	0	0	8	9	2	8	1	0	0	0	0	0	6	0	0	0
18-18.5	0	0	0	2	7	0	9	0	0	0	3	0	0	0	0	0	0
20-20.5	0	0	0	8	14	0	2	0	0	0	0	2	0	1	0	0	0
22-22.5	0	0	0	15	8	4	7	0	0	0	0	0	0	5	0	0	0
24-24.5	1	0	0	9	16	3	2	0	0	0	0	0	0	2	0	0	0
26-26.5	0	0	0	2	4	0	0	0	0	0	0	0	0	6	0	0	0
28-28.5	0	0	0	10	9	1	8	2	0	0	0	2	0	3	0	0	0
30-30.5	0	0	0	5	4	3	18	2	0	0	0	2	0	4	1	0	0

Raw diatom counts for McLaughlin Bay (cont'd)

Interval (cm)	*Cymbella falaisensis*	*Cymbella helvetica*	*Cymbella hybrida*	*Cymbella microcephala*	*Cymbella muelleri*	*Cymbella silesiaca*	*Cymbella subcuspidata*	*Denticula kuetzingii*	*Diatoma moniliformis*	*Diploneis subconstricta*	*Epithemia adnata*	*Epithemia sorex*	*Epithemia turgidia*	*Eunotia arcus*	*Eunotia bilunaris*	*Eunotia circumborealis*	*Eunotia formica*
0-0.25	0	1	0	0	0	1	0	0	0	0	2	1	0	0	1	0	0
0.25-0.5	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0
0.5-0.75	0	0	0	0	0	1	0	0	0	0	4	3	1	0	0	0	0
0.75-1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
1-1.25	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0
1.25-1.5	0	0	0	0	0	0	0	0	0	0	3	2	0	0	0	0	0
1.5-1.75	0	0	0	0	0	1	0	0	1	0	2	1	1	0	0	0	0
1.75-2	0	0	0	0	0	0	0	0	0	0	7	1	1	0	1	0	0
2-2.25	0	0	0	1	0	2	0	0	0	0	4	0	0	0	0	0	0
2.5-2.75	1	0	0	0	0	1	0	1	1	0	3	0	2	0	0	0	0
3-3.25	0	0	0	0	0	0	0	0	0	0	7	2	0	0	0	0	0
3.5-3.75	0	0	0	0	0	0	0	2	0	0	0	3	0	0	1	2	0
4-4.25	0	0	0	0	1	0	0	2	0	0	1	2	0	0	0	1	0
5-5.25	0	0	0	0	0	1	0	0	0	0	5	0	0	0	1	0	0
6-6.25	0	0	0	1	0	0	0	0	0	0	1	1	0	0	0	0	0
7-7.25	4	0	0	0	2	1	0	0	0	0	2	2	0	0	1	0	0
8-8.25	0	0	0	2	5	0	0	1	0	0	3	5	0	0	0	0	0
9-9.25	4	0	1	0	2	1	1	0	0	0	2	3	0	0	0	0	0
10-10.5	0	0	0	0	0	0	1	0	0	1	7	2	0	0	0	0	0
12-12.5	0	0	0	0	0	0	0	0	0	0	7	1	0	0	0	0	0
14-14.5	0	0	0	0	0	0	0	0	0	0	3	0	0	0	1	1	0
16-16.5	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0
18-18.5	0	0	0	0	0	1	1	0	0	0	5	2	0	0	0	1	0
20-20.5	0	0	0	0	1	0	1	0	0	0	2	1	0	0	1	0	0
22-22.5	0	0	0	0	0	1	0	0	0	0	5	2	1	0	2	0	0
24-24.5	0	0	0	0	0	1	0	0	0	0	6	0	0	0	0	0	0
26-26.5	0	0	0	0	0	0	0	0	0	0	7	2	0	0	0	1	0
28-28.5	0	0	0	0	0	0	0	0	0	0	5	1	0	0	0	0	0
30-30.5	0	0	0	0	0	2	1	0	0	0	15	0	0	1	1	0	0

Raw diatom counts for McLaughlin Bay (cont'd)

Interval (cm)	*Flagilaria berolinensis (*)*	*Flagilaria biceps (*)*	*Flagilaria brevistriata (*)*	*Flagilaria capucina (*)*	*Flagilaria construens (*)*	*Flagilaria construens var. binodis*	*Flagilaria construens var. venter (*)*	*Flagilaria construens var. subsalina*	*Flagilaria dilatata*	*Flagilaria exigua*	*Flagilaria fasciculata*	*Flagilaria neoproducta*	*Flagilaria nitzschioides*	*Flagilaria parasitica*	*Flagilaria pinnata (*)*	*Flagilaria subsalina*	*Flagilaria tenera*
0-0.25	14	1	10	7	14	1	200	3	1	0	1	0	0	0	37	0	1
0.25-0.5	9	2	8	14	13	1	230	2	0	0	0	1	0	0	30	0	0
0.5-0.75	8	1	4	11	8	1	230	0	0	0	0	1	0	0	35	0	3
0.75-1	11	1	4	7	13	0	234	2	0	0	0	0	1	0	31	0	0
1-1.25	12	2	4	13	8	1	290	2	0	0	2	0	0	0	14	0	2
1.25-1.5	6	1	5	8	13	0	250	2	0	0	0	1	0	0	17	0	1
1.5-1.75	15	1	4	9	12	0	237	2	1	0	0	0	0	2	36	0	0
1.75-2	12	1	4	7	12	0	190	1	1	0	1	0	0	0	29	0	0
2-2.25	12	1	7	15	18	0	210	5	1	0	0	1	0	0	30	0	3
2.5-2.75	17	3	6	8	21	0	180	6	0	0	0	0	1	0	49	0	2
3-3.25	6	3	8	8	14	1	179	1	0	0	0	0	0	0	33	0	0
3.5-3.75	19	4	4	2	19	1	200	1	0	0	0	0	0	0	31	0	3
4-4.25	13	3	3	5	18	3	184	2	0	0	0	4	0	1	51	0	2
5-5.25	7	3	10	13	18	0	180	1	0	0	0	0	0	0	34	0	1
6-6.25	6	1	11	9	20	1	157	6	0	0	0	0	0	2	30	0	1
7-7.25	3	2	10	1	22	2	90	0	1	0	0	1	0	0	47	0	0
8-8.25	13	1	5	11	14	0	134	8	0	0	0	2	0	1	53	0	0
9-9.25	5	12	12	9	27	0	106	5	0	0	0	0	0	0	31	0	0
10-10.5	3	2	11	6	19	0	144	5	0	0	0	0	0	0	16	0	1
12-12.5	4	2	7	3	22	1	168	7	0	0	0	3	0	0	27	0	0
14-14.5	2	3	7	4	22	1	142	2	0	0	0	4	0	0	42	0	0
16-16.5	5	2	12	10	20	1	151	6	0	0	0	1	0	0	30	0	0
18-18.5	10	4	9	6	15	0	205	1	0	0	0	0	0	0	35	0	2
20-20.5	3	2	13	8	26	0	164	6	0	0	0	2	0	0	28	0	0
22-22.5	9	2	15	8	22	0	141	2	0	0	0	2	0	0	36	0	0
24-24.5	5	3	7	7	21	1	156	0	0	1	0	2	0	0	37	0	1
26-26.5	5	2	7	7	8	0	144	3	0	0	0	0	0	0	43	0	0
28-28.5	2	2	9	11	19	0	153	4	0	0	0	1	0	1	31	0	0
30-30.5	0	2	18	10	24	3	120	8	1	1	0	1	0	3	21	0	0

Raw diatom counts for McLaughlin Bay (cont'd)

Interval (cm)	*Flagilaria ulna*	*Frustulia cuspidata*	*Gomphonema acuminatum*	*Gomphonema aquaemineralis*	*Gomphonema augustum*	*Gomphonema auritum*	*Gomphonema gracile*	*Gomphonema hebridense*	*Gomphonema minitum (*)*	*Gomphonema parvalum*	*Gomphonema sarcophagus*	*Gomphonema truncatum*	*Gyrosigma acuminatum (*)*	*Mastogloia smithii*	*Meridion circulare*	*Navicula abiskoensis*	*Navicula absoluta*
0-0.25	0	0	0	0	1	0	1	0	0	0	0	1	14	0	0	0	0
0.25-0.5	0	0	1	0	0	1	0	0	5	2	1	0	14	0	0	0	0
0.5-0.75	0	0	1	0	0	0	0	0	3	2	0	1	26	1	0	0	0
0.75-1	0	0	1	0	0	0	1	0	1	0	0	0	24	0	0	0	0
1-1.25	0	0	0	1	0	0	0	0	0	1	0	1	21	0	0	0	0
1.25-1.5	0	0	0	0	0	0	0	1	1	0	0	1	17	0	0	0	0
1.5-1.75	0	0	0	0	0	0	0	0	4	0	0	0	23	0	1	0	0
1.75-2	0	0	0	0	0	0	0	0	2	0	0	0	33	0	0	0	0
2-2.25	0	0	0	0	0	0	0	0	1	0	0	0	25	0	0	0	0
2.5-2.75	0	0	0	0	0	1	4	0	2	0	0	1	39	0	0	1	0
3-3.25	0	1	0	0	0	2	0	0	1	0	0	0	31	0	1	0	1
3.5-3.75	0	1	3	0	1	1	4	0	1	1	1	0	34	0	0	0	1
4-4.25	0	0	0	0	0	2	3	1	2	0	0	2	28	1	0	0	0
5-5.25	0	0	0	0	0	0	0	2	6	0	0	1	24	0	2	0	0
6-6.25	0	0	1	0	2	0	2	0	0	1	0	1	34	0	0	0	0
7-7.25	0	1	1	0	0	1	2	0	1	2	0	0	48	0	1	0	0
8-8.25	0	0	0	0	1	0	2	0	5	0	0	1	44	0	0	0	3
9-9.25	1	0	1	0	0	1	1	2	4	3	0	3	55	0	0	0	0
10-10.5	0	0	1	0	0	2	0	0	4	1	0	0	47	0	1	0	2
12-12.5	0	0	0	0	1	0	2	0	4	0	0	0	46	0	0	0	2
14-14.5	0	0	0	0	0	1	1	0	4	2	0	1	30	0	1	0	1
16-16.5	0	0	0	0	0	0	3	0	10	1	0	2	33	0	0	0	0
18-18.5	0	0	0	0	0	0	0	0	1	0	0	0	29	0	1	1	0
20-20.5	0	0	0	0	0	0	5	0	2	2	0	0	26	0	0	0	0
22-22.5	0	0	0	0	1	0	4	0	5	1	0	2	58	0	0	0	0
24-24.5	0	0	1	0	0	0	0	0	1	0	0	1	38	0	1	0	0
26-26.5	0	0	3	0	1	0	3	0	5	0	0	1	39	0	0	0	0
28-28.5	0	0	0	0	0	0	3	0	3	1	0	1	26	0	0	0	0
30-30.5	0	0	1	0	0	1	3	0	11	1	0	4	22	0	0	1	0

Raw diatom counts for McLaughlin Bay (cont'd)

Interval (cm)	*Navicula capitatoradiata*	*Navicula capitata*	*Navicula constans*	*Navicula crypocephala*	*Navicula cryptofallax*	*Navicula cryptotenella (*)*	*Navicula cuspidata*	*Navicula dealpina*	*Navicula elginensis*	*Navicula explanata*	*Navicula gastrum*	*Navicula integra*	*Navicula laevissima*	*Navicula margalithii*	*Navicula microcari*	*Navicula minima*	*Navicula peregrina*
0-0.25	0	0	1	0	5	3	0	0	0	0	0	0	2	0	0	0	0
0.25-0.5	0	0	0	0	2	1	2	2	0	0	1	0	0	0	1	0	0
0.5-0.75	0	4	0	0	2	4	2	0	0	0	0	0	0	0	0	0	1
0.75-1	0	2	0	0	4	3	2	1	0	0	0	0	0	0	0	0	0
1-1.25	0	0	0	1	4	4	1	1	0	0	0	0	0	0	1	0	0
1.25-1.5	0	3	0	0	1	3	2	2	0	0	0	0	3	0	0	0	0
1.5-1.75	0	2	0	0	5	5	2	3	0	1	0	0	1	0	0	1	0
1.75-2	0	1	0	0	10	4	3	1	0	1	0	0	2	0	2	0	0
2-2.25	0	4	1	1	4	2	1	1	0	0	0	0	1	0	0	0	0
2.5-2.75	3	2	0	1	3	4	3	2	0	0	0	0	2	0	3	2	0
3-3.25	2	0	1	0	5	7	1	1	0	0	0	0	1	0	0	0	0
3.5-3.75	1	3	0	0	5	5	2	1	0	0	0	0	1	0	0	0	0
4-4.25	0	1	0	5	8	6	1	2	0	0	0	0	0	0	0	0	0
5-5.25	0	3	0	3	1	7	5	5	0	0	0	0	0	0	0	0	0
6-6.25	1	0	0	2	8	8	4	4	0	1	0	0	1	0	0	0	0
7-7.25	0	3	0	0	6	10	1	2	0	0	0	0	3	0	0	1	0
8-8.25	0	4	0	0	8	4	1	6	0	0	0	0	1	0	0	0	0
9-9.25	0	3	0	1	8	10	2	2	0	2	0	1	0	0	0	0	0
10-10.5	0	0	0	5	1	0	1	6	0	0	0	0	1	0	0	0	0
12-12.5	0	1	0	0	5	2	5	6	0	0	0	0	1	0	0	0	0
14-14.5	0	2	0	0	4	3	1	4	0	0	0	0	2	0	0	0	0
16-16.5	0	5	0	0	2	4	2	3	0	0	0	0	2	0	0	0	0
18-18.5	1	0	1	0	0	2	2	4	0	0	0	0	1	0	0	0	0
20-20.5	0	3	2	0	3	8	3	2	0	0	0	0	3	0	0	0	0
22-22.5	0	1	2	0	6	10	1	2	0	1	0	0	0	0	0	0	0
24-24.5	0	2	2	0	3	3	3	3	0	0	0	0	0	0	0	0	0
26-26.5	0	3	2	2	0	2	1	5	0	0	0	0	0	1	1	0	0
28-28.5	0	3	1	0	1	2	2	6	0	2	0	0	3	0	0	0	0
30-30.5	0	6	0	1	0	0	0	4	1	0	1	0	2	0	0	0	0

Raw diatom counts for McLaughlin Bay (cont'd)

Interval (cm)	*Navicula placentula*	*Navicula pseudoventralis*	*Navicula pupula*	*Navicula pygmaea*	*Navicula radiosa*	*Navicula reinhardtii*	*Navicula riparia*	*Navicula schoenfeldii*	*Navicula scutelloides*	*Navicula splendicula*	*Navicula trivialis* (*)	*Navicula tuscula*	*Neidium affine*	*Neidium ampliatum*	*Neidium bergii*	*Nitzschia angustata*	*Nitzschia constructa*	*Nitzschia gracilis*
0-0.25	0	0	0	0	0	0	0	0	0	0	15	0	0	1	2	3	0	0
0.25-0.5	1	1	0	0	1	1	0	1	0	2	20	0	0	1	0	0	0	1
0.5-0.75	1	1	0	0	1	2	2	0	0	0	14	0	0	1	2	0	0	1
0.75-1	0	0	0	0	0	1	2	1	0	0	17	0	0	0	0	4	0	1
1-1.25	0	0	1	0	1	0	2	0	0	0	16	0	0	0	1	1	0	0
1.25-1.5	1	0	0	0	0	0	3	0	0	0	10	0	0	1	0	0	0	0
1.5-1.75	0	0	0	0	0	0	2	1	0	0	17	0	0	1	2	3	0	1
1.75-2	0	0	1	0	1	1	0	0	0	0	21	0	0	0	0	0	0	2
2-2.25	0	0	4	0	1	0	4	0	0	2	0	0	0	0	1	1	0	1
2.5-2.75	1	2	1	0	2	0	0	0	0	0	16	0	0	1	0	1	1	4
3-3.25	0	0	2	1	1	0	0	0	0	0	17	0	0	0	0	2	0	0
3.5-3.75	0	1	2	0	0	0	2	0	0	0	15	1	0	0	0	4	3	4
4-4.25	0	0	1	0	3	0	1	0	0	0	10	0	0	0	0	0	0	2
5-5.25	1	3	0	0	2	0	4	0	0	0	13	0	1	1	1	4	0	2
6-6.25	0	0	0	1	0	1	1	0	0	0	23	0	0	0	0	2	0	2
7-7.25	0	2	3	0	2	0	1	2	0	0	32	0	0	0	0	1	1	5
8-8.25	2	6	0	0	2	0	3	0	0	0	14	0	0	1	0	0	0	0
9-9.25	0	1	2	0	1	0	0	1	0	0	20	0	0	1	2	0	0	1
10-10.5	0	2	0	0	0	0	2	0	0	0	36	0	0	0	0	3	0	0
12-12.5	0	1	0	2	0	0	0	0	0	0	30	0	0	1	2	0	0	0
14-14.5	0	0	0	0	0	0	0	0	0	0	15	0	0	1	0	0	0	1
16-16.5	0	2	0	0	0	2	0	0	0	0	18	0	0	1	3	0	0	4
18-18.5	0	2	0	0	1	1	1	0	0	0	17	0	0	0	1	3	0	0
20-20.5	1	3	1	0	0	0	1	1	0	0	29	0	0	0	0	1	0	1
22-22.5	1	2	0	1	0	1	1	1	0	0	37	0	0	3	2	0	0	0
24-24.5	0	1	0	0	0	0	1	1	0	0	26	0	0	2	2	2	0	0
26-26.5	1	1	1	0	0	0	0	0	1	0	31	0	0	0	0	2	0	0
28-28.5	0	1	1	0	2	2	0	0	0	0	20	0	0	1	2	1	1	1
30-30.5	1	0	1	0	1	1	0	0	0	0	19	0	0	1	0	1	0	0

Raw diatom counts for McLaughlin Bay (cont'd)

Interval (cm)	*Nitzschia levidensis*	*Nitzschia palea*	*Nitzschia perminuta (*)*	*Nitzschia recta*	*Nitzschia sigmoidea*	*Pinnularia gibba*	*Pinnularia maior*	*Pinnularia stomatophora*	*Pinnularia streptoraphe*	*Pinnularia viridis (*)*	*Rhopalodia gibba*	*Suriella brebissonii*	*Suriella capronii*	*Suriella tenera*	*Stauroneis acuta*	*Stauroneis anceps*	*Stauroneis phoenicenteron*
0-0.25	0	0	1	0	0	0	0	0	1	3	1	0	0	0	0	0	0
0.25-0.5	0	0	3	1	0	0	0	0	0	0	1	0	0	0	0	0	0
0.5-0.75	0	0	0	3	2	0	0	0	0	5	1	0	0	1	0	0	0
0.75-1	0	3	5	2	0	0	0	0	0	1	0	1	1	0	0	0	0
1-1.25	0	3	2	3	0	0	0	0	0	3	0	0	0	1	0	0	0
1.25-1.5	0	2	5	2	0	0	0	0	1	1	1	0	0	0	0	0	0
1.5-1.75	0	5	9	2	0	0	0	0	0	0	1	0	1	0	0	0	0
1.75-2	0	1	2	4	0	0	0	0	0	4	4	0	1	0	0	0	0
2-2.25	0	1	5	5	2	0	1	0	0	4	2	0	0	1	0	0	0
2.5-2.75	0	3	1	5	0	0	0	0	0	1	3	0	2	0	0	1	0
3-3.25	0	0	3	5	0	0	0	0	0	3	0	1	1	1	0	0	0
3.5-3.75	0	3	10	4	0	0	0	0	1	4	0	0	1	0	0	0	1
4-4.25	0	2	6	4	0	0	0	0	0	2	0	0	1	0	0	0	0
5-5.25	0	3	6	2	0	0	0	0	0	7	1	0	0	0	0	0	0
6-6.25	0	3	3	2	1	0	0	0	0	2	1	0	0	0	0	0	1
7-7.25	0	3	13	3	1	0	0	0	0	3	1	0	0	0	0	0	2
8-8.25	0	0	10	2	0	0	0	0	1	2	0	0	0	2	0	0	0
9-9.25	0	1	3	1	0	0	0	0	0	8	1	0	0	0	1	0	1
10-10.5	0	0	2	0	0	0	0	2	2	2	1	0	0	0	0	0	2
12-12.5	0	2	3	0	0	0	0	0	0	2	0	0	0	0	0	0	0
14-14.5	0	5	3	1	0	0	0	0	0	10	1	0	0	0	0	0	0
16-16.5	0	1	4	0	0	0	0	0	0	0	3	0	0	0	0	0	0
18-18.5	0	0	3	2	2	0	0	0	0	3	1	0	0	0	0	0	1
20-20.5	0	2	1	2	0	0	0	0	0	4	1	0	0	0	0	0	0
22-22.5	0	0	0	0	0	0	0	0	0	8	3	0	0	0	0	0	1
24-24.5	0	0	4	0	0	1	0	0	0	2	1	0	1	0	0	0	0
26-26.5	0	0	8	1	0	0	0	0	0	7	2	0	0	0	0	0	0
28-28.5	0	1	2	1	0	0	0	0	0	5	3	0	0	0	0	0	0
30-30.5	1	0	4	1	0	0	0	0	0	4	1	0	0	0	0	0	0

Raw diatom counts for McLaughlin Bay (cont'd)

Interval (cm)	*Stephanodiscus neoastraea*	*Stephanodiscus rotula*	*Tabellaria flocculosa*	*Unknown Flagilaria spp1*	*Unknown Flagilaria spp2 (long)*	*Unknown Gomphonema spp1*	*Unknown Gomphonema spp2*	*Unknown spp1 (hold in the middle)*
0-0.25	0	0	0	1	0	0	0	0
0.25-0.5	0	0	2	0	0	0	0	1
0.5-0.75	0	1	0	2	0	0	0	1
0.75-1	0	0	1	0	0	0	0	0
1-1.25	0	0	2	0	0	1	0	0
1.25-1.5	0	0	1	1	0	0	0	0
1.5-1.75	0	0	2	0	0	0	0	0
1.75-2	0	0	3	0	0	0	0	0
2-2.25	0	0	1	1	0	0	1	0
2.5-2.75	1	0	1	1	0	0	0	0
3-3.25	0	0	1	0	0	0	0	0
3.5-3.75	0	0	1	0	0	0	0	0
4-4.25	0	0	2	0	0	0	0	0
5-5.25	0	0	2	0	0	0	0	0
6-6.25	0	0	0	0	1	0	0	0
7-7.25	0	0	2	0	0	0	0	1
8-8.25	0	0	3	0	0	0	0	0
9-9.25	0	0	1	0	0	0	0	0
10-10.5	0	0	2	0	0	0	0	0
12-12.5	0	0	1	1	0	0	0	1
14-14.5	0	0	5	0	0	0	0	0
16-16.5	0	0	1	1	0	0	0	0
18-18.5	0	0	1	0	0	0	0	0
20-20.5	0	0	0	0	0	0	0	0
22-22.5	0	0	5	0	0	0	0	0
24-24.5	0	0	2	0	0	0	0	0
26-26.5	0	0	0	0	0	0	0	0
28-28.5	0	0	2	0	0	0	1	1
30-30.5	0	0	3	0	0	0	0	0

A2. Raw Cladocera counts for McLaughlin Bay (chapter 2)

Species	Remain	0-0.25	0.25-0.5	0.5-0.75	0.75-1	2-2.25	2.5-2.75	3-3.25
Bosmina spp.	Headshield	61	56	73	68	80	53	74
	Carapace	73	56	58	72	94	67	81
Camptocercus	Headshield		1	2	1			
	Postabdomen	1						
Alona guttata	Headshield	2			4		1	
Alona circumfimbriata	Headshield	1	3	1	4			1
Chydorus brevilabris	Headshield	1	2	4	1	2	4	
	Shell	1	2	13	9	5		2
Eurycerus spp.	Headshield	1						
Acroperus harpae	Postabdomen	1						
Alona affinis	Headshield							1
	Postabdomen							1
Alona quadrangularis	Headshield		2		2			
	Postabdomen			1	2	1		
Sida crystallina	Exopodite sedments		1					
Daphnia longispina	PA claw							
Daphnia spp.	ephippium		1					
Disparalona acutirostris	Headshield				2			
Leydigia spp.	Headshield				2	1		
	Postabdomen							
Anchistropus minor	Carapace							1
Alona rustica	Headshield							
Paralona pigra	Headshield							
Alona costata	Headshield							
Ilyocryptus	Marginal setae							
Graptolebris testudinaria	Headshield							
Alonella nana	Headshield							
	Postabdomen	1						
Pleuroxus straminius	Headshield	1						
Pleuroxus spp.	Shell							
	Postabdomen							
Chydorus sphaericus	Headshield							
Alonella excisa	Headshield							

Raw Cladocera counts for McLaughlin Bay (cont'd)

Species	Remain	3.5-3.75	4-4.25	5-5.25	6-6.25	7-7.25	8-8.25
Bosmina spp.	Headshield	98	81	90	86	86	66
	Carapace	111	92	100	91	88	88
Camptocercus	Headshield			1			
	Postabdomen						
Alona guttata	Headshield			2	5	5	6
Alona circumfimbriata	Headshield	3		1	2	2	3
Chydorus brevilabris	Headshield	3	1	2		1	2
	Shell	7	6	3	2	3	2
Eurycerus spp.	Headshield						
Acroperus harpae	Postabdomen						
Alona affinis	Headshield			1			
	Postabdomen			1			
Alona quadrangularis	Headshield			1		2	
	Postabdomen	1				1	2
Sida crystallina	Exopodite sedments						
Daphnia longispina	PA claw				1		
Daphnia spp.	ephippium						
Disparalona acutirostris	Headshield						
Leydigia spp.	Headshield					1	
	Postabdomen	1	2	1			
Anchistropus minor	Carapace						
Alona rustica	Headshield	1					
Paralona pigra	Headshield		1				
Alona costata	Headshield				1		
Ilyocryptus	Marginal setae				1		
Graptolebris testudinaria	Headshield						1
Alonella nana	Headshield						
	Postabdomen						
Pleuroxus straminius	Headshield						
Pleuroxus spp.	Shell						
	Postabdomen						
Chydorus sphaericus	Headshield						
Alonella excisa	Headshield						

Raw Cladocera counts for McLaughlin Bay (cont'd)

Species	Remain	9-9.25	10-10.5	12-12.5	14-14.5	16-16.5	18-18.5
Bosmina spp.	Headshield	55	69	74	60	62	59
	Carapace	82	83	82	82	78	90
Camptocercus	Headshield			1	1		
	Postabdomen						
Alona guttata	Headshield	3	1	6	3	1	1
Alona circumfimbriata	Headshield	1	2	2	1		
Chydorus brevilabris	Headshield		1	1	2		
	Shell	7	6	5	1	2	1
Eurycerus spp.	Headshield						
Acroperus harpae	Postabdomen						
Alona affinis	Headshield		1	1			
	Postabdomen						
Alona quadrangularis	Headshield		1	1		1	
	Postabdomen	1				2	1
Sida crystallina	Exopodite sedments						
Daphnia longispina	PA claw						
Daphnia spp.	ephippium						
Disparalona acutirostris	Headshield						
Leydigia spp.	Headshield						
	Postabdomen				1	1	
Anchistropus minor	Carapace						
Alona rustica	Headshield						
Paralona pigra	Headshield						
Alona costata	Headshield		1				
Ilyocryptus	Marginal setae						2
Graptolebris testudinaria	Headshield						1
Alonella nana	Headshield						1
	Postabdomen						
Pleuroxus straminius	Headshield						
Pleuroxus spp.	Shell						
	Postabdomen						
Chydorus sphaericus	Headshield						
Alonella excisa	Headshield						

Raw Cladocera counts for McLaughlin Bay (cont'd)

Species	Remain	20-20.5	22-22.5	24-24.5	26-26.5	28-28.5	30-30.5
Bosmina spp.	Headshield	53	66	51	37	27	37
	Carapace	69	74	63	68	63	59
Camptocercus	Headshield			1			
	Postabdomen						
Alona guttata	Headshield	2	2	1	1		1
Alona circumfimbriata	Headshield	2	1	2	2	1	1
Chydorus brevilabris	Headshield	4	1	1	1		1
	Shell	8	3	2	2	2	3
Eurycerus spp.	Headshield						
Acroperus harpae	Postabdomen						
Alona affinis	Headshield	2				1	
	Postabdomen						1
Alona quadrangularis	Headshield	2					
	Postabdomen	3	4			1	
Sida crystallina	Exopodite sedments						
Daphnia longispina	PA claw						
Daphnia spp.	ephippium						
Disparalona acutirostris	Headshield						
Leydigia spp.	Headshield			1			
	Postabdomen						
Anchistropus minor	Carapace						
Alona rustica	Headshield						
Paralona pigra	Headshield						
Alona costata	Headshield	1			2		
Ilyocryptus	Marginal setae	1					
Graptolebris testudinaria	Headshield						
Alonella nana	Headshield						
	Postabdomen						
Pleuroxus straminius	Headshield	1					
Pleuroxus spp.	Shell		1				
	Postabdomen						1
Chydorus sphaericus	Headshield		1				
Alonella excisa	Headshield	2					

A3. Raw Cladocera counts for Cootes Paradise (Chapter 3)

Species	Remain	0-1	2-3	4-5	6-7	8-9
Bosmina spp.	Headshield	46	62	73	88	82
	Carapace	53	66	55	58	58
	Postabdomen	2				
Camptocercus rectirostris	Headshield					
	Postabdomen					
Alona guttata	Headshield	4			1	
	Postabdomen					
Alona circumfimbriata	Headshield		1			
Chydorus brevilabris	Headshield	2	1		2	1
	Shell	4	2	1	1	3
Eurycerus spp.	Headshield					
	Postabdomen					
Acroperus harpae	Postabdomen					
Alona affinis	Postabdomen					
Alona quadrangularis	Headshield					
	Postabdomen	1		2		3
Sida crystallina	Exopodite segments					
	PA claw					
Daphnia longispina	PA claw	1		1		
Disparalona acutirostris	Headshield				1	
	Shell					
Leydigia spp.	Postabdomen	1		2		3
Alona costata	Headshield					
Ilyocryptus	Marginal setae					
Graptolebris testudinaria	Shell					
Pleuroxus spp.	Shell					
	Postabdomen					
Chydorus sphaericus	Headshield					
Alonella excisa	Headshield			1		
	Shell					
	Postabdomen	1				
Kurzia latissima	Headshield					
Alona intermedia	Postabdomen					

Raw Cladocera counts for Cootes Paradise (cont'd)

Species	Remain	10-11	12-13	14-15	16-17	18-19
Bosmina spp.	Headshield	78	80	82	78	77
	Carapace	58	86	56	59	54
	Postabdomen					
Camptocercus rectirostris	Headshield					1
	Postabdomen					
Alona guttata	Headshield	1	3			1
	Postabdomen					
Alona circumfimbriata	Headshield					
Chydorus brevilabris	Headshield	1	4		1	
	Shell	1		1	2	2
Eurycerus spp.	Headshield					
	Postabdomen					
Acroperus harpae	Postabdomen					
Alona affinis	Postabdomen		1			
Alona quadrangularis	Headshield			1		
	Postabdomen	1	1		2	
Sida crystallina	Exopodite segments					
	PA claw					
Daphnia longispina	PA claw					
Disparalona acutirostris	Headshield		2	2	1	
	Shell					
Leydigia spp.	Postabdomen	1			2	
Alona costata	Headshield					
Ilyocryptus	Marginal setae				1	
Graptolebris testudinaria	Shell					
Pleuroxus spp.	Shell					
	Postabdomen					
Chydorus sphaericus	Headshield					
Alonella excisa	Headshield					
	Shell					
	Postabdomen					
Kurzia latissima	Headshield					
Alona intermedia	Postabdomen	1				

Raw Cladocera counts for Cootes Paradise (cont'd)

Species	Remain	20-21	25-26	30-31	35-36	40-41
Bosmina spp.	Headshield	80	74	84	66	77
	Carapace	64	75	69	43	76
	Postabdomen					
Camptocercus rectirostris	Headshield	1				1
	Postabdomen					
Alona guttata	Headshield	3	1	1	4	
	Postabdomen					
Alona circumfimbriata	Headshield					
Chydorus brevilabris	Headshield		2	2	2	5
	Shell	2	3	1		1
Eurycerus spp.	Headshield					
	Postabdomen	1				
Acroperus harpae	Postabdomen					
Alona affinis	Postabdomen					
Alona quadrangularis	Headshield					
	Postabdomen	3	1	2	1	
Sida crystallina	Exopodite segments					
	PA claw					
Daphnia longispina	PA claw					
Disparalona acutirostris	Headshield					
	Shell		1	1		
Leydigia spp.	Postabdomen	3	1	2	1	0
Alona costata	Headshield					
Ilyocryptus	Marginal setae					
Graptolebris testudinaria	Shell				1	
Pleuroxus spp.	Shell					
	Postabdomen					
Chydorus sphaericus	Headshield					
Alonella excisa	Headshield					
	Shell					
	Postabdomen					
Kurzia latissima	Headshield					
Alona intermedia	Postabdomen					

Raw Cladocera counts for Cootes Paradise (cont'd)

Species	Remain	45-46	50-51	55-56
Bosmina spp.	Headshield	67	40	12
	Carapace	75	50	18
	Postabdomen			
Camptocercus rectirostris	Headshield		1	2
	Postabdomen			1
Alona guttata	Headshield	3		3
	Postabdomen		1	
Alona circumfimbriata	Headshield		2	1
Chydorus brevilabris	Headshield	4	23	31
	Shell	7	19	35
Eurycerus spp.	Headshield			
	Postabdomen			
Acroperus harpae	Postabdomen		1	
Alona affinis	Postabdomen			
Alona quadrangularis	Headshield			
	Postabdomen	1	2	
Sida crystallina	Exopodite segments		1	
	PA claw			1
Daphnia longispina	PA claw			
Disparalona acutirostris	Headshield			
	Shell		1	
Leydigia spp.	Postabdomen	1	2	0
Alona costata	Headshield		1	1
Ilyocryptus	Marginal setae			
Graptolebris testudinaria	Shell		1	1
Pleuroxus spp.	Shell		1	1
	Postabdomen		1	
Chydorus sphaericus	Headshield			1
Alonella excisa	Headshield		1	
	Shell			4
	Postabdomen			
Kurzia latissima	Headshield		2	
Alona intermedia	Postabdomen			

A4. Raw Cladocera counts for Jordan Harbour (chapter 3)

Species	Remain	0-0.5	2-2.5	4-4.5	6-6.5	8-8.5	10-10.5
Bosmina spp.	Headshield	36	27	30	26	32	40
	Carapace	41	35	22	20	31	27
Chydorus brevilabris	Headshield					1	
	Shell	2	5	7	4	5	3
Alona guttata	Headshield	1	1	3	2	2	1
Alona quadrangularis	Headshield						1
	Postabdomen		1				
Alona affinis	Headshield						
	Postabdomen					1	1
Alona circumfimbriata	Headshield						
	Postabdomen						
Camptocercus spp.	Headshield						
Leydigia spp.	Postabdomen	1					
Pleuroxus spp.	Headshield		1		1		
	Shell			3			2
Sida crystallina	PA claw		1				
Disparalona acutirostris	Headshield						
	Shell					1	

Raw Cladocera counts for Jordan Harbour (cont'd)

Species	Remain	12-12.5	14-14.5	16-16.5	18-18.5	20-20.5	25-25.5
Bosmina spp.	Headshield	25	31	35	53	27	48
	Carapace	37	43	43	58	42	52
Chydorus brevilabris	Headshield	1			3	1	
	Shell	2	3	5	5	4	4
Alona guttata	Headshield	4	1	1	1		
Alona quadrangularis	Headshield						
	Postabdomen			1			
Alona affinis	Headshield						
	Postabdomen						
Alona circumfimbriata	Headshield						
	Postabdomen						1
Camptocercus spp.	Headshield				1	3	
Leydigia spp.	Postabdomen						3
Pleuroxus spp.	Headshield	1	1				
	Shell	2					
Sida crystallina	PA claw						
Disparalona acutirostris	Headshield				1		
	Shell						

Raw Cladocera counts for Jordan Harbour (cont'd)

Species	Remain	30-30.5	35-35.5	40-40.5	45-45.5	50-50.5
Bosmina spp.	Headshield	37	46	31	33	29
	Carapace	29	50	61	38	35
Chydorus brevilabris	Headshield	2			5	3
	Shell	2	8	4	8	3
Alona guttata	Headshield			2	2	2
Alona quadrangularis	Headshield				1	
	Postabdomen	1	1	1	1	2
Alona affinis	Headshield					
	Postabdomen					
Alona circumfimbriata	Headshield	1	1			
	Postabdomen					
Camptocercus spp.	Headshield					1
Leydigia spp.	Postabdomen			3		
Pleuroxus spp.	Headshield					
	Shell					
Sida crystallina	PA claw					
Disparalona acutirostris	Headshield					
	Shell					

A5. *Bosmina* size measures for McLaughlin Bay (chapter 2)

0-0.25			2-2.25			4-4.25		
CL	ML	AL	CL	ML	AL	CL	ML	AL
161.22	9.09	77.93	218.78	39.97	90.92	275.03	32.6	103.75
161.62	20.12	93.25	217.96	24.69	63.09	216.56	12.64	72.37
187.87	49.13	121.17	161.42	35.85	80.32	193.5	24.3	80.13
205.63	18.67	112.59	266.81	13.54	113.15	218.19	30.83	77.22
290	20	127.73	275.78	23.03	66.07	187.45	18.56	74.5
176.74	23.05	75.03	205.26	23.97	77.91	172.96	22.19	105.46
168.32	19.46	105.68	312.72	27.31	46.73	258.57	35.85	99.79
240.63	28.78	95.52	183.59	24.96	65.48	180.95	22.16	96.1
317.08	29.16	98.65	187.12	26.19	92.4	129.33	14.98	63.72
187.87	26.83	105.41	297.48	24.21	100.13	138.57	8.96	104.26
166.57	32.64	133.5	271.8	51.36	69.27	203.02	24.67	79.06
171.48	13.14	72.64	191.61	19.14	70.99	208.24	18.9	103.88
176.02	35.33	98.17	238.24	31.5	130.74	186.47	19.07	96.76
174.29	23.78	61.36	213.01	25.06	59.73	159.96	22.06	69.93
187.85	26.39	146.7	254.23	31.85	74.27	192.6	29.08	80
202.34	21.75	86.84	253.42	41.18	106.82	258.3	25.45	93.63
181.68	39.63	92.81	245.42	29.46	101.28	276.25	21.45	81.66
184.04	18.59	97.95	197.02	34.01	96.11	202.94	18.92	84.14
206.11	35.04	82.54	265.79	21.83	118.16	209.44	20.81	142.68
119.2	12.6	108.45	180.87	31.21	69.03	216.62	29.06	113.36
179.54	21.34	117.79	262.76	14.63	82.21	282.17	24.51	58.37
280	30	114.15	247.55	14.53	76.33	213	17.43	132.97
176.47	13.54	52.44	197.01	16.62	105.34	168.89	24.96	76.85
156.76	26.43	97.43	268.37	18.52	115.23	293.09	39.97	59.49
159.24	17.46	108.19	184.81	29.61	99.15	238.03	22.06	112.79
304.59	18.16	107.64	222.1	27.35	78.8	246.59	21.33	77.37
199.09	21.77	111.33	263.57	29.16	98.54	278.4	26.19	102.11
188.14	18.21	65.15	215.25	22.32	116.78	241.43	21.21	113.78
170.62	48.79	73.24	169.86	25.07	51.07	204.56	37.17	101.07
244.63	28.88	93.09	253.14	23.13	111	193.47	18.46	94.11
196.83	30.12	72.81	218.69	23.45	68.49	220.45	26.61	62.97
261.71	38.63	66.3	218.02	36.15	82.78	178.57	35.49	68.47
186.8	19.97	92.18	172.68	12.23	107.76	161.79	34.09	85.68
289.55	38.28	99.23	189	32.87	115.74	232.36	23.51	114.38
254.37	35.19	55.62			74	233.02	25.13	

***Bosmina* size measures for McLaughlin Bay (cont'd)**

6-6.25			8-8.25			10-10.5		
CL	ML	AL	CL	ML	AL	CL	ML	AL
182.77	53.1	70.74	212.83	55.73	104.89	178.92	31.54	99.24
146.62	21.56	81.46	188.29	24.15	99.01	205.04	19.36	71.06
202.06	21.96	50.68	222.07	16.67	103.24	150.86	39.56	66.04
186.87	18.1	77.34	226.12	26.58	79.75	155.71	22.35	76.87
188.17	19.64	62.34	184.63	31.71	95.75	191.54	20.49	110.97
192.5	32.09	61.55	229.94	21.66	78.84	193.1	19.94	51.34
188.5	37.5	70.59	170.03	23.82	70.43	170.24	7.4	89.58
185.01	20.77	52.77	165.55	17.63	86.84	185.75	34.07	70.59
143.98	21.94	106.97	177.68	36.81	107.02	219.7	32.81	68.28
266.87	30.56	66.71	169.64	13.16	96.58	172.73	21.19	59.2
197.14	26.94	94.97	171.63	12.07	116.95	161.96	20.57	108.89
249.36	23.51	59.58	228.25	20.04	119.39	258.66	37.61	67
260.17	33.4	102.79	226.95	21.36	87.85	208.04	22.58	95.23
205.05	25.69	120.12	137.07	19.5	96.14	235.64	18.36	126.79
222.05	17.79	60.72	183.03	28.51	90.14	198.77	17.99	87.1
197.42	25.99	62.73	180.6	39.41	97.15	208.96	34.28	104.73
133.67	14.02	61.01	189.02	11.05	78.59	256.68	23.85	113.85
195.63	22.64	110.82	173.05	22.24	99	280.22	28.34	113.17
151.49	21.03	58.09	181.18	29.98	69.69	247.31	20.78	40.25
317.85	38.46	123.11	173.62	24.97	105.01	238.32	19.09	91.99
216.07	13.35	82.87	197.56	24.34	53.73	233.81	31.25	82.31
246.74	23.38	121.19	209.01	24.03	73.41	236.05	14.81	80.1
155.77	17.5	118.96	216.13	18.12	72.73	252.16	23.19	120.25
160.19	20.15	84.23	227.04	29.07	86.11	264.96	22.75	74.58
169.64	44.45	84.52	240.81	37.04	71.91	236.34	20.41	71.88
155.27	18.72	92.53	210.88	15.09	86.9	169.61	15.13	97.1
204.14	25.71	109.12	155.62	21.49	106.17	287.58	27.39	137.98
155.25	13.59	85.44	233.16	26.19	97.68	289.55	33.72	81.9
241.86	21.17	82.17	223.92	25.31	96.8	273.68	24.21	89.84
146.06	17.36	53.57	223.42	13.96	59.9	285.31	20.69	124.57
154.19	19.29	112.06	210.44	22.66	90.19	116.94	9.7	69.49
162.15	21.76	69.69	194.2	31.18	55.04	183.37	15.84	81.15
190.53	28.17	101.47	200.74	49.09	114.35	185.72	30.8	104.37
296.67	35.3	63.74	153.03	23.79	95.24	162.24	21.48	102.28
204.34	27.72	84.87	185.25	12.98	87.26	168.1	27.46	79.8

Bosmina size measures for McLaughlin Bay (cont'd)

12-12.5			14-14.5			16-16.5		
CL	ML	AL	CL	ML	AL	CL	ML	AL
233.82	25.54	110.01	190.13	16.24	78.81	215.64	17.99	66.82
209.68	23.51	94.35	249.39	26.73	32.07	222.11	27.68	55.5
166.27	22.84	120.71	162.76	26.87	58.24	173.39	22.03	120.89
209.91	22.06	114.2	221.25	20.77	60.96	208.4	19.66	86.18
240.59	22.94	80.85	280.61	30.06	96.44	226	37.42	81.37
195.51	16.83	83.32	188.75	29.5	72.4	204.42	36.89	95.15
164.84	17.76	118.22	214.05	22.82	95.88	171.3	25.59	96.14
233.24	18.75	122.13	220.85	15.62	80.44	178.12	34.84	100.12
245.94	22.12	101.86	249.83	21.57	62.66	262.57	30.25	97.14
198.47	29.07	69.07	208.71	15.77	123.86	216.63	24.09	48
244.35	23.44	69.31	163.33	37.75	119.91	219.9	27.3	84.23
207.83	65.43	89.52	150.74	23.94	91.57	147.05	24.83	114.52
184.46	29.59	75.66	205.8	32.23	97.73	215.49	19.37	79.81
156.88	21.3	65.12	164.47	40.31	116.88	221.66	14.94	65.18
137.06	18.22	85.74	185.93	27.13	131.85	220.52	21.42	80.11
178.64	50.32	109.46	195.83	31.67	112.94	159.98	20.25	68.63
225.09	23.79	75.77	223.74	25.97	78.69	276.63	26.25	74.72
239.82	25.34	94.72	157.07	25.41	57.35	191.72	28.33	101.58
184.82	38.08	63.83	161.89	21.4	76.71	134.4	20.72	102.36
206.28	28.74	95.07	180.18	23.36	117.42	134.78	12.78	97.2
238.87	23.1	126.72	183.86	30.13	59.5	199.7	12.3	83.04
220.36	32.1	97.7	217.83	24.96	99	239.49	21.2	89.8
193.19	40.02	76.82	172.03	29.55	86.05	242.59	42.37	107.04
195.57	19.09	84.9	141.01	26.88	65.44	265.04	17.26	99.73
170.58	22.49	123.28	160.72	22.68	93.52	294.31	23	84.53
165.78	26.16	67.18	210.95	37.12	90.97	254.28	16.83	97.27
226	23.23	72.31	238.94	28.17	55.39	170.68	22.49	84.84
205.66	17.64	86.61	229.74	24.09	37.41	155.99	20.21	99.19
126.3	26.6	79.97	200.29	15.51	83.83	125.32	13.64	85.57
163.6	52.14	80.24	181.86	27.7	120.46	251.23	27.8	63.01
238.01	35.6	84.16	174.36	23.48	125.86	155.17	16.52	96.28
126.6	23	71.07	173.95	13.59	101.38	169.53	39.84	99.06
153.92	13.34	133.91	200.44	21.28	93.5	328.63	29.16	74.08
211.24	18.99	138.77	163.22	19.87	84.44	232.6	19.46	123.57
174.03	25.34	84.89	200.49	17.8	83.59	212.57	30.38	95.73

***Bosmina* size measures for McLaughlin Bay (cont'd)**

18-18.5			20-20.5			24-24.5		
CL	ML	AL	CL	ML	AL	CL	ML	AL
166.05	23.51	67.25	238.9	15.32	103.41	212.95	22.21	80.43
229.43	16.96	106.91	193.81	32.5	103.41	28.85	164.62	90.89
253.68	23.05	89.72	211.96	37.57	81.03	212.1	19.76	107.65
191.62	12.94	71.08	162.06	29.08	59	161.85	36.42	104.58
201.78	27.91	133.88	210	18.61	77.21	196.68	31.34	71.58
150.79	21.8	112.47	230.66	21.31	48.94	195.65	39.07	127.57
136.34	21.16	125.44	148.49	22.05	95.66	152.62	24.73	120.19
199.58	19.09	82.43	275.06	25.7	79.04	193.18	13.65	78.46
209.82	23.04	63	190.56	20.01	95.02	210.46	19.91	109.86
229.11	23.86	58.19	247.47	34.13	82.76	194.79	18.63	74.1
266.57	20.9	94.33	250.44	26.14	58.82	221.43	41.06	78.68
196.21	32.54	78.24	183.71	16.24	80.8	190.6	30.47	91.72
215.1	18.83	70.03	217.46	33.56	64.41	167.44	33.28	144.17
213.34	27.35	77.62	215.59	28.15	63.84	244.99	17.03	64.57
258.78	50.49	93.43	225.02	17.63	73.05	219.7	31.82	88.03
218.29	35.01	80.32	206.28	21.42	86.93	218.87	48.67	24.2
216.88	17.76	81.58	227.16	19.21	96.12	172.99	17.28	49.46
185.75	12.98	45.09	169.74	19.91	100.17	210.28	29.08	81.83
238.53	18.88	80.74	239.42	24.1	111.13	177.65	27.12	92.38
189.8	13.64	76.36	175.31	39.48	115.59	214.4	14.25	68.76
134.53	13.36	115.59	254.93	24.52	90.66	250.89	26.35	75.25
176.66	15.89	93.9	199.85	14.41	51.27	237.84	39.89	112.15
207.77	12.95	92.67	173.85	28.96	50.71	140.08	24.58	69.71
133.15	20.34	101.88	186.4	20.94	40.94	159.24	21.81	97.02
250.4	35.26	60.47	245.09	24.34	50.28	220.43	38.43	66.17
245.71	16.4	85.62	256.69	15.97	53.59	189.51	8.22	80.28
223.87	16.26	46.28	243.86	22.59	103.49	233.38	26.92	72.5
258.11	20.92	99.61	249.55	22.63	62.65	162.53	43.98	83.21
192.74	10.7	106.22	246.13	17.54	94.62	237.5	30.02	44.76
177.3	34.49	70.65	241.64	33.5	79.85	177.86	20.52	96.93
242.23	19.29	92.97	256.03	37.92	59.63	185.69	27.28	60.96
193.88	16.81	66.92	251.37	14.39	72.23	201.56	13.79	60.3
174.62	20.69	69.03	242.34	28.73	78.86	214.8	30.2	46.07
180.52	28.74	78.79	173.28	19.62	34.27	171.28	25.15	78.93
228.73	28.17	119.62	182.05	23.55	86.87	161.57	14.43	68.61

Bosmina size measures for McLaughlin Bay (cont'd)

28-28.5		
CL	ML	AL
244.44	18.81	82.53
217.47	27.54	61.98
211.38	29.05	84.94
177.45	17.74	107.4
250.37	34.49	99.93
255.13	27.5	95.39
132.08	19.16	48.39
122.47	21.78	90.14
168.84	21.61	77.16
192.44	39.55	107.01
198.64	33.55	122.24
222.85	34.18	75.8
234.33	15.78	67.75
212.72	26.75	133.97
214.98	41.13	46.05
234.16	22.58	151.15
244.3	31.43	62.09
171.41	20.64	72.64
172.06	22.41	93.21
275.13	32.36	83.69
280.91	20.21	
203.71	42.18	
148.15	15.39	
145.7	19.08	
242.67	15.98	
245.03	27.07	
222.34	21.65	
207.55	20.21	
201.45	34.41	
212.06	18.63	
160.11	15.96	
188.36	40.22	
272.95	44	

A6. *Bosmina* size measures for Cootes Paradise (chapter 3)

0-1			4-5			10-11		
CL	ML	AL	CL	ML	AL	CL	ML	AL
162.62	26.25	111.54	254.77	33.92	105.8	188.06	24.49	96.62
176.51	34.54	74.66	197.52	33.97	95.21	251.09	40.89	110.7
166.38	24.13	115.53	245.77	43.65	72.05	246.71	18.69	97.52
256	24.71	95.71	302.29	24.69	127.83	317.21	34.41	84.5
165.42	27.43	110.89	211.71	15.83	88.05	165.71	38.42	97.79
240.56	22.65	98.33	155.09	25.35	101.62	180.79	31.05	137.48
191.2	29.04	89.86	171.71	26.51	73.59	163.72	22.72	79.6
189.9	31.98	62.46	181.87	31.66	155.41	317.01	16.47	118.05
183.55	22.07	120.1	188.97	40.75	99.09	305.5	18.91	80.98
265.28	19.67	94.22	249.54	12.11	90.24	274.57	20.69	95.37
168.2	36.89	92.02	174.66	18.67	119.32	200.19	34.94	105.59
351.52	15.74	79.84	239.1	31.23	124.72	168.51	37.68	107.99
220.03	26.32	99.26	310.61	41.75	86.48	180.54	32.21	106.49
165.78	31.31	83.84	256.08	19.88	98.7	231.45	21.55	74.25
176.44	25.27	107.59	239.8	30.22	88.15	155.03	22.63	82.73
132.93	31.3	94.89	233.42	27.52	87.63	142.88	20.76	110.74
256.58	29.31	48.86	148.7	30.12	108.86	147.84	24.02	76.25
158.71	30.78	108.2	297.6	21.83	89.32	188.98	16.63	94.79
160.3	29.1		213.47	19.73	98.89	237.18	31.2	62
205.13	24.67		262.18	24.09	96.73	173.94	32.94	82.2
216.12	23.45		354.06	27.61	67.32	177.44	30.13	73.63
235.44	30.38		151.72	25.69	95.21	150.74	16.63	80.89
210.93	26.63		161.18	27.52	120.45	202.76	16.75	66.53
			165.22	31.82	142.46	140.72	15.59	78.34
			254.28	26.58	87.6	134.29	16.54	113.98
			206.12	42.67	124.92	171.41	41.75	75.89
			322.19	25.28	94.53	186.58	42.97	92.85
			243.55	13.07	85.73	146.58	26.27	69.3
			240.14	24.49	119.33	160.41	29.63	81.89
			220.26	23.33	137.14	208.55	27.33	110.9
			239.16	24.22	81.25	323.99	50	94.67
			325.94	31.03	107.98	233.37	12.11	70.74
			144.37	27.95	120.71	172.5	42.95	65.43
			169.23	30.27	108.74	172.26	35.43	92.55
			165.46	32.32	89.71	156.62	34.3	80.99

Bosmina size measures for Cootes Paradise (cont'd)

14-15			20-21			25-26		
CL	ML	AL	CL	ML	AL	CL	ML	AL
214.46	22.19	96.17	235.94	23.92	68.86	210.07	21.09	98.93
172.45	23.21	94.05	268.29	11.59	109.61	218.17	19.74	86.45
184.18	22.89	74.17	262.53	16.7	56.65	249.47	32.63	94.51
208.76	22.7	110.36	205.25	17.99	87.5	242.97	29.97	72.59
171.48	28.18	102.28	197.44	22.46	88.34	140.11	30.94	66.95
175.25	25.83	93.25	270.71	21.33	94.15	250.84	31.85	72.16
183.14	33.08	112.51	138.19	40.27	78.51	206.13	20.93	104.55
208.17	24.09	87.61	185.12	37.76	92.89	269.52	16.24	102.07
172.31	21.21	89.28	146.23	13.21	129.43	226.67	11.61	104.87
241.31	41.8	113.93	234.98	24.65	70.23	174.01	30.76	84.7
217.35	22	90.86	160.14	30.47	127.86	169.25	10.7	81.35
226.73	4.94	60.46	245.61	39.17	108.33	204.99	27.73	79.79
126.81	30.22	78.65	150.89	27.52	77.68	179.41	27.01	74.7
156.68	30.82	103.57	239.98	30.91	91.47	199.04	36.1	125.33
149.42	29.13	91.09	200.31	29.3	76.04	186.1	28.17	104.76
205.17	28.89	110.37	216.79	31.15	183.95	211.8	14.98	79.2
275.61	23.13	133.97	300	18.91	68.78	293.26	21.14	99.27
167.24	26.8	91.68	174.94	29.75	95.16	160.79	24.67	92.23
321.88	21.45	118.54	201.56	70.12	81.42	165.07	19.24	75.73
206.92	26.31	117.5	201.23	71.39	77.3	191.29	35.74	93.9
168.3	27.95	105.48	200.49	33.73	121.67	271.99	20.58	98.99
160.05	27.25	91.41	249.85	32.3	64.62	191.53	45.58	84.6
251.05	13.35	105.82	171.73	33.74	115.72	300.58	33.49	106.42
142.4	26.12	107.9	171.52	34.21	88.58	220.39	31.79	95.47
245.38	19.52	122.81	231.22	35.89	94.16	167.52	36.75	90.64
221.74	15.42	117.82	231.03	37.35	126.34	274.23	24.03	83.49
165.26	19.46	107.73	140.53	22.22	91.31	242.31	42.79	112.15
311.63	30.48	124.28	279.04	28.78	130.35	200.22	20.81	97.55
356.36	36.42	101.2	300.34	24.84	89.51	163.33	31.17	95.99
147.42	27.51	99.03	232.25	21.95	126.28	269.96	23.29	112.28
194.61	22.59	81.82	184.72	22.94	115.87	234.48	27.32	120.44
191	32.48	67.47	174.75	38.82	154.17	158.67	36.48	81.27
167.67	27.93	67.17	136.55	38.78	79.08	269.16	21.79	60.16
249.81	30.45	104.56	182.53	10.4	64.23	185.06	28.24	65.6
224.78	46.2	91.99	228	18.28	90.16	229.45	24.56	61.16

Bosmina size measures for Cootes Paradise (cont'd)

30-31			35-36			40-41		
CL	ML	AL	CL	ML	AL	CL	ML	AL
373.11	33.72	103.64	256.66	24.96	103.45	217.22	24.76	73.98
198.09	17.18	93.94	292.19	42.8	97.68	205.71	26.25	125.35
208.06	17.18	76.22	234.08	16.79	79.4	185.44	31.56	88.1
311.23	27.95	86.82	207.38	16.54	94.48	189.74	16.24	94.94
251.12	29.68	76.55	200	25.91	83.62	183.05	29.55	65.95
256.76	35.3	116.23	121.98	23.1	68.58	215.05	24.36	114.6
230.45	23.64	62.21	295.38	33.77	89.71	301.54	15.31	66.16
171.83	19.44	107.47	268.25	36.4	98.82	211.06	22.37	124.32
291.83	29.9	68.54	210.38	20.85	121.36	211.34	14.8	78.15
239.22	19.09	89.16	274.32	38.63	124.4	196.07	18.12	117.93
152.04	23.45	113.78	297.45	34.11	104.68	304.9	9	67.98
286.59	32.05	87.55	199.15	19.73	91.17	229.87	26.48	98.19
240.87	22.84	70.46	180.82	17.81	69.22	165.46	15.94	103.63
192.67	38.2	248.59	193.13	39.22	116.46	177.07	27.54	106.9
296.88	14.73	83.36	255.73	18.18	93.23	176.21	30.98	86.04
138.51	23.57	71.27	167.4	37.4	87.84	301.82	27.95	119.03
263	24.85	114.54	165.14	17.82	103.88	176.45	37.68	78.39
359.81	39.97	83.16	176.87	26.53	124.93	211.66	15.96	44.64
168.77	22.64	72.23	302.96	29.78	92.87	334.3	32.69	100.32
179.22	24.31	118.71	215.12	32.97	56.03	265.06	27.45	117.64
291.08	35.61	79.87	154.69	23.32	128.87	211.28	22.09	111.11
206.92	20.9	119.87	214	26.99	71.54	299.23	29.68	96.97
220.02	28.05	101.65	201.66	21.78	74.15	172.72	30.76	91.64
150.13	31.93	126.11	151.49	17.12	91.03	228.63	25.56	84.6
176.62	43.7	83.37	161.19	35.83	114.27	163.15	39.79	104.37
219.68	29.69	79.82	184.83	13.64	58.51	155.01	29.14	83.23
191.47	20.29	102.79	190.1	19.44	89.2	182.84	29.46	90.1
162.07	27.04	96	205.17	20.15	134.13	171.6	38.1	92.8
201.44	36.57	77.97	377.38	17.46	114.79	363.53	28.34	50.34
233.36	15.27	96.09	182.64	22.44	105.94	151.99	22.16	96.52
231.85	31.04	110.42	231.37	19.44	80.56	303.36	43.47	62.64
170.15	31.96	67.25	192.36	22.69	112.55	271.62	40.1	86.99
249.01	35.79	84.98	249.65	27.5	108.38	272.21	25.68	109.67
179.22	18.27	68.32	179.53	36.74	123.16	237.66	29.46	43.1
153.18	32.89	62.21	180.2	29.73	111.33	170.39	40.75	99.15

Bosmina **size measures for Cootes Paradise (cont'd)**

45-46			50-51			55-56		
CL	ML	AL	CL	ML	AL	CL	ML	AL
171.13	24.11	102.63	151.15	20.96	58.65	222.69	17.88	58.13
198.48	22.62	70.12	125.45	16.49	76.99	208.06	16.19	73.06
269.2	28.08	83.15	201.75	37.33	94.96	190.06	24.39	56.31
243.62	13.32	116.14	167.88	31.8	88.83	188.91	16.83	63.97
232.73	28.37	81.86	195.06	25.41	105.64	192.06	11.96	58.67
294.08	36.28	107.06	183.55	18.99	89.84	160.04	28.18	84.91
229.74	27.07	124.29	219.38	22.17	53.52	197.58	23.78	92.33
181.92	34.24	121.41	204.71	38.73	86.04	183.16	22.12	69.07
168.71	28.17	93.86	246.76	17.65	57.91	171.35	19.74	112.12
153.28	33.44	85.66	195.86	27.58	106.76	164.58	28.68	57.2
166.3	26.43	119.56	246.81	18.1	125.7	148.47	22.64	90.22
170.93	30.11	115.05	203.27	37.01	86.73	165.32	22.75	78.69
215.55	31.13	80.63	189.38	19.75	102.21	172.56	21.67	
223.19	45.6	101.1	212.99	17.03	77.17	225	13.5	
244.31	34.67	103.79	175.54	20.61	87.98	167.79	10.64	
222.12	30.27	109.72	223.85	32.21	74.98			
178.97	36	77.45	193.99	29.21	91.41			
178.6	15.42	84.92	248.84	46.53	49.68			
263.34	31.99	92.5	228.8	21.17	84.34			
156.46	25.6	102.96	194.1	34.55	77.5			
157.21	31.12	117.07	221.66	25.38	95.43			
324.36	34.43	94.64	161.71	33.5	76.34			
322.7	25.45	85.26	198.41	14.53	81.34			
251.47	26.19	101.93	156.92	27.9	119.53			
197.47	33.07	90.59	252.27	21.81	68.63			
221.4	41.38	80.73	155.65	28.73	83.54			
192.69	32.01	115.05	234.85	18.95	85.47			
153.14	30.02	90.39	159.38	23.42	84.17			
282.08	19.71	79.78	139.02	19.52	74.61			
265.69	26.88	86.58	134.73	18.26	70.32			
197.81	26.53	71.47	138.43	34.35	65.01			
222.69	26.08	94.3	266.89	28.98	71.34			
192.07	40.69	115.11	264.78	31.82	92.22			
300.34	29.16	64.72	171.38	36.6	58.03			
259.04	33.49	101.69	184.82	27.09	103.34			

A7. *Bosmina* size measures for Jordan Harbour (chapter 3)

0-5			45-50		
CL	ML	AL	CL	ML	AL
284.61	21.93	68.56	232.64	20.89	106.5
199.16	20.15	83.7	222.65	26.49	75.29
211.22	43.3	89.11	241.32	22.32	102.36
183.45	27.91	84.42	217.94	18.23	100.43
217.12	13.04	116.95	150.12	28.66	51.22
154.65	26.4	84.42	149.37	23.89	85.6
182.12	22.16	99.58	164.36	24.96	96.39
176.22	32.94	130.96	162.47	9.01	99.26
172.81	31.63	99.64	184.3	23.68	131.74
138.4	30.38	89.75	211.4	16.87	81.22
251.55	25.6	76.03	200.79	26.25	100.28
156.42	17.47	117.37	194.97	23.55	60.02
155.09	13.96	113.66	196.21	34.28	74.19
152	25.05	109.81	188.68	34.92	64.5
147.54	24.23	67.76	244.85	39.04	87.5
167.54	22.64	86.92	201.18	49.01	99.36
180.96	22.7	90.3	208.04	20.89	103.68
178.72	22.61	90.05	246.25	24.58	89.31
219.97	22.47	97.23	193.73	23.51	74.9
157.9	25.23	99.77	179.79	22.06	54.68
171.48	21.21	81.65	210.65	27.64	95.08
166.12	18.54	55.15	239.93	26.4	93.61
173.73	36.21	117.39	196.92	29.91	89.78
157.65	24.03	97.81	196.94	21.7	72.44
332.58	11.16	55.44	243.79	30.57	67.05
203.6	45.43	123.68	246.26	17.1	77.07
201.23	43.14	107.73	223.49	13.93	83.87
181.86	14.22	95.34	160.31	25.52	89.63
150.64	15.21	108.9	192.28	19.62	73.6
148.95	13.93	111.22	220.82	43.45	93.72
150.39	21.89	106.28	245.82	20.09	75.64
151.94	24.79	103.42	231.69	26.39	71.89
241.42	16.34	100.21	223.21	21.99	83.22
142.79	21.09	73.14	213.17	26.37	68.55
243.61	9.76	105.63	181.59	20.74	78.22