CLADOCERAN BODY SIZE AS A METRIC OF ENVIRONMENTAL CHANGE IN NEW BRUNSWICK LAKES

BY

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A thesis submitted to the
Department of Geography and Environment
Mount Allison University
in partial fulfillment of the requirements for the
Bachelor of Environmental Science degree with Honours
April 19th, 2021

Abstract

Cladocerans are a widespread order of zooplankton frequently used as bioindicators of aquatic conditions and environmental change. Shifts in cladoceran body size, at both the taxon and assemblage level, can occur as a response to environmental stressors. A standardized body size metric was developed for cladoceran taxa commonly observed in lakes of eastern Canada to provide an assessment method for understanding body size responses to environmental change. The metric was then applied to dated sediment records of five New Brunswick lakes that experienced severe DDT contamination in the mid-20th century. Body sizes of *Bosmina* sp. and *Daphnia* sp. remains were also measured from two lakes to determine whether size trends based on the metric occurred at a taxon level. A decline in body sizes in two lakes initially associated with historical DDT inputs has remained to present, and suggests that past and contemporary stressors, such as climate change, may structure both assemblage composition and body sizes of cladocerans. The shift towards smaller body sizes during the DDT impact period was reflected at both the assemblage (via the size metric) and taxon levels. The agreement in the metric and measurements of cladoceran remains from the sediment record suggests that the metric accurately tracks trends in average cladoceran body size through time. Overall, the body size metric provides a tool to further assess the impact of environmental stressors on cladocerans and determine potential implications for aquatic food webs.

Acknowledgements

I would like to deeply thank Dr. Joshua Kurek, especially for taking me on as an honours student during his year on sabbatical. A deep thank you as well to Dr. Andrew Labaj for co-supervising my honours project, who put a great deal of effort into carefully editing my first writing drafts. I have learned a lot over the year concerning lake systems and the role of Cladocera (capitalized), and all individual cladocerans (lowercase). I feel so lucky to have worked at ECAB LAB at Mount Allison University. Thank you to fellow ECABers for the great company and fun conversations over the year which made microscope work more exciting: Alex DiLonardo, Meghan Fraser, Jacob Demers, and Julia Campbell. This thesis work has been invaluable in my academic evolution as a student and I am very appreciative of this opportunity and of all who have helped me learn and grow.

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1.0 Introduction

1.1 Overview of the Cladocera

The Cladocera are an order of crustacean zooplankton with a global distribution of over 600 species – the majority of which inhabit freshwater systems such as lakes, ponds, and wetlands (Forró et al. 2008). These small organisms (ranging from 0.2—6 mm in size) are generally herbivorous (with a few predatory taxa), feeding on phytoplankton by filter-feeding or grazing (Forró et al. 2008; Dodson et al. 2010). Cladocera are keystone zooplankton taxa that occupy a mid-trophic level position in the aquatic food web, situated between the primary producers and predatory invertebrates and planktivorous fish (Brooks and Dodson 1965). Their intermediate position in the food web makes them susceptible to both "bottom-up" and "top-down" controls, giving them the ability to reflect changes in the aquatic food web (Wojtal-Frankiewicz 2012). Two of the most common cladoceran taxa in Atlantic Canada, *Daphnia* sp. and *Bosmina* sp., are both pelagic taxa, inhabiting the open-water zone of lakes. *Daphnia* sp. is the larger-bodied of the two taxa, and is an efficient filter-feeder, capable of competitively excluding smaller-bodied *Bosmina* sp. from algal food resources (Brooks & Dodson, 1965). Compared to *Bosmina* sp., *Daphnia* sp. generally have lower resiliency to stressors, such as temperature increases, toxicants, and toxic algal blooms (Havens and Hanazato 1993; Adamczuk 2016).

Cladocera are commonly used indicators for studying applied lake management issues, as they are abundant, have rapid generation times, and respond quickly to environmental conditions. Cladoceran subfossil remains are well-preserved in lake sediments, enabling identification at the species or genus level, making them useful for studying the impacts of long-term environmental change (Korhola and Rautio 2001; Frolova 2018). Shifts in assemblage compositions may provide insight into changing environmental conditions and food web dynamics. Additionally, knowing the tolerances and ecological optima of individual taxa to environmental stressors, such as chemical toxicants, can reveal environmental changes when assemblage compositions shift.

1.2 Cladoceran Body Size

Due to the phenotypic plasticity and life cycle of cladocerans, shifts in body size can be indicative of environmental changes in lakes through time. Body size is important, as it can influence the rate of metabolism, feeding, and growth, as well as size at maturity and the number of offspring produced (Moore and Folt 1993; Hart and Bychek 2011; Smirnov 2014). Changes in body size can occur as a response to environmental stress. Under stressful conditions, there is often a shift towards smaller body size both intra- and inter-specifically (Daufresne et al. 2009; Hart and Bychek 2011). Well-known stressors that may influence zooplankton body sizes include predation pressure (Brooks and Dodson 1965), toxicants and water chemistry (Hanazato 2001; Rogalski et al. 2017) and changes in thermal stratification (Hart and Bychek 2011; Nevalainen et al. 2014).

Under stressful conditions, a smaller body size often has lower metabolic requirements. Respiration rates can increase under stressful conditions which in turn increases metabolic demands, making smaller sizes favoured (Baird et al. 1990). Similar to other crustaceans, cladocerans molt and regrow their carapace in order to expand in size as they grow and mature. Under stressful conditions, there often is a shift of energy allocated towards maturation and reproduction rather than individual growth, where energy is required for both somatic growth and carapace growth (Hart and Bychek 2011). Reducing the number of molts until maturity also favors smaller size at maturity, as well as smaller brood chambers, which in turn contain fewer offspring.

Chemical stressors are known to alter fitness of larger-bodied cladocerans. Exposure to toxic levels of contaminants can lead to smaller body size, as well as reduce survival rates and fecundity of cladocerans (Moore and Folt 1993). Key behaviours, such as filtering rates and respiration, are also reduced under the presence of certain toxicants, which in turn lower growth rates (Moore and Folt 1993; Hart and Bychek 2011). The degradation rate of toxicants factors into the recovery time of the cladoceran community after the initial exposure, where recovery is delayed dependent on the persistence of the chemical in the environment (Hanazato and Yasuno 1990). With the introduction of toxicant stressors, larger, more sensitive taxa are often reduced in abundance while smaller cladocerans consequently increase (Moore and Folt 1993). Reduced body size may present a paradox to cladoceran survival under toxicant stress, as smaller body sizes have an increased surface area to volume ratio of the body, and in turn increase potential chemical uptake from ambient exposure (Hart and Bychek 2011).

Cladoceran body size is also influenced by temperature, with smaller body sizes often associated with warmer water temperatures. Climate change, and the associated physical shifts taking place in the epilimnion of lakes, will almost certainly affect Cladocera given that many abundant taxa are pelagic. For example, increased air temperatures result in earlier lake ice-off and ice-on periods, with implications for the synchronization of cladoceran population dynamics and phytoplankton abundances (Winder and Schindler 2004; Wojtal-Frankiewicz 2012). Smaller cladocerans not only show resilience to warmer temperatures, but better competitive abilities during phytoplankton blooms — a phenomena forecasted to become more frequent as climate change progresses (Rühland et al. 2008; Nevalainen et al. 2014). Larger cladocerans such as *Daphnia* sp. tend to exhibit lower resilience to higher temperatures and experience reduced fitness as respiration and molting demands increase (Moore and Folt, 1993). Warmer temperatures accelerate development and favor more frequent molting, which is energetically taxing (Moore and Folt 1993). Cladoceran mean body size and population dynamics, including timing of *Daphnia* sp. seasonal peak abundances, demonstrate latitudinal patterns as temperature and climate indirectly and directly influence ambient lake conditions (Gillooly and Dodson 2000; Beaver et al. 2020). Temperature-induced shifts in aquatic species composition, growth, and reproduction affect fitness at the various biological scales of the assemblage, population, and individual (Daufresne et al. 2009). Typically, there is a decrease in mean body size, and decrease in the dominance of larger species such as *Daphnia* sp. (Beaver et al. 2020). Smaller species account for a greater proportion of the assemblage as climate change affects lake thermal properties. Within populations, the proportion of juveniles increases due to the reduced lifespan of larger individuals in warmer conditions, and the physiological response causes a shift towards offspring production at a smaller size (Daufresne et al. 2009; Hart and Bychek 2011; Smirnov 2014).

Predation and food availability also affect zooplankton body size. Increased predation pressure by visual predators (such as planktivorous fish) favours cladocerans with smaller bodies, where smaller sizes may reduce visibility by fish predators. When predatory invertebrates are present, cladoceran populations may undergo a shift to larger sizes, where larger-sized cladocerans may be more difficult to catch, as invertebrate predators use tactile means to grasp prey, and are often limited in the size of prey they can ingest (Hart and Bychek 2011). Climate change and temperature increases can also increase metabolic requirements of fish and macroinvertebrate populations that in turn increase the predation pressures exerted on cladocerans (Adamczuck 2016).

The importance of cladoceran body size lies in its implications for overall lake food web dynamics and energy transfer of nutrients such as carbon, phosphorus, and calcium. Smaller organisms offer lower quantities of energy and nutrients to predators in comparison to larger organisms (Hart and Bychek 2011). Thus, cladoceran body size changes can affect the flow of energy to higher trophic levels. Shifts in cladoceran body sizes can also influence important lake parameters such as water quality. For example, body size can impact water clarity under assemblage-level shifts from larger to smaller cladocerans, which are less efficient at filtering-feeding (Stemberger and Miller 2003; Korosi et al. 2012).

1.3 Paleolimnology

In order to understand how lake ecosystems respond to environmental stressors, it is critical to establish baseline (pre-impact) conditions (Armstrong and Kurek 2019; Daly et al. 2019; Kurek et al. 2019). The challenge, however, is that past stressors often occurred before the initiation of monitoring programs. Paleolimnology enables the reconstruction of past environmental conditions of aquatic systems using dated lake sediment records, which act as natural archives of chemical, physical, and biological conditions within lakes as well as their watershed (Smol 2010). The utility of paleolimnological research lies in the ability to reconstruct the environment of the past and extend the study timescale to include periods “before”, “during”, and occasionally “after” a particular stressor occurred (Reavie 2019). Understanding the past environment can help put modern lake conditions into proper context and allow restoration targets to be established (Saulnier-Talbot 2016).

Zooplankton are frequently used bioindicators in paleolimnological research, with the Cladocera being particularly common, as they leave well-preserved, abundant, and identifiable remains in the sediments, unlike rotifers and copepods (Korhola and Rautio 2001; Labaj et al. 2021). Thus, analyzing cladoceran remains in the sediment allows assemblage compositions to be determined through time, providing insight into how assemblages or specific taxa respond to environmental stressors. In addition to reconstructing species composition, cladoceran remains, most commonly from the taxa *Bosmina* sp. and *Daphnia* sp., can be measured to provide empirical measures of body size through time. *Bosmina* sp. carapace remains can be measured to represent estimates of total body lengths (Korosi et al. 2010). *Daphnia* sp. experience greater decomposition so that post-abdominal claws are often best available for measurements. Although only a fraction of the living organism, post-abdominal claw lengths are correlated with total body length and carapace length body, so that larger claws typically reflect larger individuals (Hrbáček 1969; Manca and Comoli 1996). Post-abdominal claw lengths can therefore be used a proxy for *Daphnia* body size (Korosi et al. 2010). Common taxa are known to undergo body size changes in response to environmental changes and stressors (Moore and Folt 1993; Hart and Bychek 2011), thus body size measurements can provide further insight into changing environmental conditions over time (Korosi et al. 2008; Labaj et al. 2016; Daly et al. 2019).

1.4 Rationale

The difficulty with studying body sizes is that measuring cladoceran remains preserved in sediments is time and labour intensive. A primary goal of this thesis was to develop and test a metric to track changes in overall cladoceran assemblage body size. This allows for an efficient means of reconstructing body size changes through time. Kurek et al. (2019) revealed that legacy contaminants of DDTs in dated sediments of north-central New Brunswick lakes were among some of the highest observed in North America. This pesticide, which is toxic to many aquatic biota, is not water-soluble and can bioaccumulate and biomagnify. Concurrent with historical DDT inputs during the ~1950s-1970, cladoceran assemblages in several New Brunswick lakes experienced a shift in dominance from *Daphnia* sp. to *Bosmina* sp., which has remained to present.

This thesis examined cladoceran assemblages from five DDT-impacted lakes to answer the following research questions: 1) Do cladoceran body sizes, as tracked by the body size metric, change in response to environmental stress? If so, how have sizes changed through time? and 2) Are the trends in cladoceran size predicted by the body size metric reflected in empirical measurements of *Bosmina* sp. and *Daphnia* sp. body sizes? I predict that body size will decrease under environmental stress, so that smaller sizes will be observed under the chemical stress of severe DDT inputs and, if in recent years, potentially indicate body size responses to the indirect effects of climate change on the epilimnion of lakes. Furthermore, the interrelatedness of size changes at the individual and assemblage levels suggests that similar trends should be observed between empirical measurements from the sediment record, and metric results assessing size of the assemblage. This research will enhance the understanding of cladoceran body size changes in response to environmental stress, as well as assess the efficacy of a new method to quantify cladoceran body size from sedimentary records.

2.0 Methods

2.1 Study Sites

Cladoceran body sizes were analyzed in dated sediment samples from five north-central New Brunswick lakes — Sinclair (47.0540°, -66.5778°), California (47.4460°, -66.1587°), Goodwin (47.2606°, -66.3482°, Upsalquitch (47.4790°, -66.4999°), and Middle Peaked Mountain (46.7366°, -66.5199°) lakes (Figure 1). Throughout the mid-20th century (~1950-1970s), Dichlorodiphenyltrichloroethane (DDT) was applied aerially across much of New Brunswick to control spruce budworm outbreaks. At least 5.7 million kg of DDTs were applied within the first decade of use, with concentrations varying across the province (Figure 1). As a result of historical DDT use, study lakes contain among the greatest sedimentary concentrations of DDTs and their breakdown products in North America (Kurek et al. 2019). Since 1997, only Goodwin Lake has shown sedimentary DDT levels below levels that are known to cause adverse effects on biota (CCME 1999).

Surface area is relatively similar between all lakes, with Upsalquitch Lake having the greatest surface area (69 ha), followed by Sinclair (46 ha), California (45 ha), Goodwin (20 ha), and Middle Peaked Mountain (15 ha). These study sites are remote headwater lakes with no major stream inputs and are situated in the Northern Uplands and Highlands ecoregions of New Brunswick. The coniferous forest vegetation is dominated by balsam fir, black and white spruce, with some white pine. Hardwood species include white and yellow birch, red maple, and American mountain ash. All lakes are dimictic and oligotrophic, with total phosphorous ranging from 4 – 10 $\mu g\ L^{-1}$ (Kurek et al. 2019). The pH levels of these lakes were close to neutral, with a range of 7.1 – 7.5. Calcium levels ranged from 2 mg L^{-1} (Sinclair, California) and 3 mg L^{-1} (Middle Peaked Mountain, Goodwin) to 11 mg L^{-1} (Upsalquitch).

All lakes are home to Brook Trout (*Salvelinus fontinalis*) and minnow populations. Brook Trout populations are managed for recreational angling in California, Goodwin, and Middle Peaked Mountain lakes. No fish stocking was recorded for any of the study lakes, except for Middle Peaked Mountain, where 2000 juvenile Brook Trout were stocked in 2000. Upsalquitch Lake is also home to a "suspected" small population of Arctic Char (*Salvelinus alpinus*). Yellow Perch (*Perca flavescens*) were also found in Sinclair Lake.

2.2 Cladoceran Body Size Metric

An exhaustive literature review was conducted to determine the body size lengths of cladoceran taxa commonly found in eastern North American lakes (Table 1). Studies and measures of female cladocerans reported from North America were prioritized. When values from North American populations could not be determined, values from the Northern Hemisphere were prioritized. The metrics presented in the literature included ranges, medians, means, and "other" given values (i.e., metric not specified by author). When more than one value was presented (e.g., range and mean), the mean was used. Medians were prioritized over ranges, which were prioritized over "other" given values. The term "mean of means" describes a value where multiple references provided body size length values as a mean, and so a mean was taken from several means. "Global mean" describes a mean taken from different types of body size values. When ranges were provided in the literature, the midpoint value was used in the global mean calculation.

A weighted-average function, weighting the sizes of cladoceran taxa by their percentage relative abundance in each sediment interval, was then used to estimate the overall body size of a cladoceran assemblage for a given sediment interval. To determine how cladoceran assemblage body size changed in relation to historical DDT inputs and contemporary climate change, the body size metric was then applied to previously-identified cladoceran assemblages in dated sediment cores of five New Brunswick lakes reported in Kurek et al. (2019). The sediment records of these lakes extend from the early-1900s to present based on ^{210}Pb dating results and were divided into three periods: pre-impact (<1950), impact (1950-1970), and post-impact

(>1970), based on known records of historical DDT use in the region (Figure 1) as well as sedimentary measures of DDT concentrations reported in Kurek et al. (2019).

2.3 Cladoceran Size Measurements

Photomicrography of *Bosmina* sp. and *Daphnia* sp. remains were carried out with a Zeiss compound microscope using brightfield illumination at between 200–400× magnification. Cladoceran body size measurements were taken using image measurement tools in Zeiss Zen Blue software following methods outlined in Korosi et al. (2011). Identification of *Bosmina* sp., *Daphnia pulex* complex, and *Daphnia longispina* complex followed descriptions given in Korosi and Smol (2012). Cladoceran size measurements were obtained from two of the five northern NB lakes: Goodwin Lake, which showed the lowest DDT concentrations in modern years, and Upsalquitch Lake which showed one of the highest concentrations. For each lake, subfossil remains from three sediment intervals were measured per each pre-defined time period (i.e., pre-impact, impact, post-impact). Sediment intervals corresponding to similar ages were sampled between lakes when possible to maintain similar temporal comparisons. To ensure that size measurements were representative, a target of at least 30 carapaces (*Bosmina*) and 30 post-abdominal claws (*D. pulex* or *D. longispina* complexes) were measured from each sediment interval. Only one *D. longispina* complex remain was found in Upsalquitch lake in the 0–0.5 cm depth interval (corresponding to ~2016). In Goodwin Lake, only one *D. pulex* complex remain was found in the 12–12.5 cm interval (~1937). These intervals were excluded from the graphs but were included in the statistical analyses.

2.4 Statistical Analyses

All statistical analyses were carried out using the R software environment (R Core Team 2021). A one-way analysis of variance (ANOVA) was used to determine whether there was a significant difference ($p < 0.05$) between body sizes of the pre-impact, impact, and post-impact periods for the cladoceran size metric and empirical measures of body size. ANOVA reveals whether there are significant differences between groups. When ANOVA identified a significant difference between groups, a Tukey's Honest Significant Difference test was used to determine the periods that differed significantly. The assumptions of both tests are that: 1) data groups are independent from each other and demonstrate random sampling, 2) each data group is normally distributed, and 3) the data groups have common variance. Details of statistical results are listed in the appendices.

3.0 Results

3.1 Cladoceran Body Size Metric

Upsalquitch Lake showed a decrease in assemblage body size coincident with DDT inputs, while California Lake assemblage body size started to decreased post-DDT application (Figure 2a). Significant differences were observed in the assemblage body sizes between periods in California ($p < 0.01$, $F = 11.47$, *d.f.* = 2) and Upsalquitch lakes ($p < 0.01$, $F = 23.46$, *d.f.* = 2). The assemblage body sizes at both these lakes in the post-impact period were significantly ($p < 0.01$) smaller than the pre-impact and impact periods (Figure 2a). The body size of the cladoceran assemblage in both Upsalquitch and California lakes were stable during the pre-impact period (until ~1950). Upsalquitch assemblage body size decreased from an average of 1.04 mm at 12–12.5 cm (~1946) to 0.74 mm at 8.5–9 cm (~1973). Upsalquitch Lake body size remained stable, within the range of 0.68–0.79 mm over a 20-year period (~1973–1993), after which it decreased further to reach the present average value of 0.46 mm. The average body size of the assemblage in California Lake varied within the range of 1.15–1.28 mm between the years ~1889 and 1979. Size then declined to the smallest in the record (0.67 mm) observed at the depth interval of 7–7.5 cm (~1993). California Lake showed an increase in body size to an average of 0.94 mm in the most recent sediment interval at 0-0.5 cm.

Goodwin, Middle Peaked Mountain, and Sinclair lakes all showed only minor changes in body sizes in comparison to California and Upsalquitch lakes. Body sizes within these three lakes in recent years were also similar to their pre-impact values (Figure 2b). Sinclair and Middle Peaked Mountain lakes did not experience any significant changes in assemblage body size over the sediment record. Significant differences in body size were observed in Goodwin Lake ($p < 0.01$, $F = 6.92$, *d.f.* = 2). A significant ($p < 0.01$) difference in assemblage body size occurred between the pre-impact and impact periods in Goodwin Lake, (Figure 2b; Table 3.0). The pre-impact average size in Goodwin Lake averaged 1.01 mm. Goodwin lake sizes increased to an average of 1.19 mm at 8.5–9 cm (~1951), then decreased to 1.10 mm at 6.5–7 cm (~1967). Post-impact assemblage size averaged 1.03 mm, with an assemblage size of 1.18 mm in the most recent depth interval 0–0.5 cm (~2015), demonstrating a size increase of 0.21 mm from the second-most recent depth interval (2-2.5 cm, ~2005).

3.2 Empirical Measures of Bosmina sp. and Daphnia sp. Body Sizes from the Sediment Record

3.2.1 Goodwin Lake

Bosmina sp. carapace and *D. pulex* complex post-abdominal claw lengths were variable and there were no significant differences in size between periods (Figures 3 and 4). *Daphnia longispina* complex were significantly ($p < 0.01$) larger in the pre-impact period (mean = 155.1 μm) than both the impact and post-impact periods (mean = 136.9 μm and 139.3 μm, respectively) (Figure 5).

3.2.2 Upsalquitch Lake

No remains of *D. pulex* complex were recovered from Upsalquitch Lake. *Bosmina* sp. carapace lengths were variable in each of the periods; however, lengths were significantly ($p < 0.01$) shorter in the post-impact period (mean = 262.7 μm), compared to the pre-impact (mean = 388.3 μm) and impact (mean = 367.9 μm) periods (Figure 6). *Daphnia longispina* complex post-abdominal claw lengths exhibited minor variation within the three periods, with average claw length significantly ($p < 0.01$) greater in the post-impact period (131.5 μm) compared to the pre-impact (118.5 μm) and impact (121.4 μm) periods (Figure 7).

4.0 Discussion

4.1 Do Cladoceran Body Sizes Tracked by the Metric Reflect Environmental Stress?

Trends in cladoceran body sizes showed lake-specific differences to regional stressors known to impact cladoceran assemblages in low-nutrient lake ecosystems in north-central New Brunswick (Armstrong and Kurek 2019; Daly et al. 2019; Kurek et al. 2019). Body sizes were stable through the ~1900s to present in three lakes (Sinclair, Goodwin, and Middle Peaked Mountain), while two lakes (California and Upsalquitch) experienced significant declines in body size coincident with or following the application of DDT to the watershed, as beginning of these declines differ by several decades. The cladoceran assemblages of California and Upsalquitch lakes shifted to smaller *Bosmina* sp. becoming the dominant taxon during the post-impact period, co-occurring with a decline in *D. longispina* complex*,* with a near extirpation of this larger-bodied pelagic taxon observed at Upsalquitch Lake (Kurek et al. 2019). The shift towards smaller body sizes in the post-impact periods at Upsalquitch and California lakes may have been caused, at least in part, by the application of DDT to the watershed. Following this, the emergence of recent environmental stressors, namely climate change, may further favor cladocerans of smaller body size (Daufrense et al. 2009). All lakes studied had sedimentary DDT and related breakdown product concentrations that exceeded probable effects levels (PELs) in the post-impact period (Kurek et al. 2019). This highlights the need to assess individual lake systems with respect to understanding why two of the five study lakes exhibited significant size differences after ~1970 compared to previous time periods.

In general, smaller-bodied cladocerans exhibit greater tolerance to toxicants than their larger-bodied counterparts (Havens and Hanazato 1993). During events of chemical stress, like DDT inputs to freshwaters, small-bodied organisms are favoured, and there is often a trend for smaller taxa to replace larger-bodied taxa in terms of assemblage dominance (Havens and Hanazato 1993). *Bosmina* sp. exhibit better initial tolerance and ability to recover from direct effects of pesticides in comparison to *Daphnia* sp. (Havens and Hanazato 1993). A shift towards smaller body size under chemical stress has reproductive implications, as smaller adults have less physical space to store eggs. In turn, fewer eggs are produced and there can be an overall decline in the number of offspring (Hanazato 2001). This reproductive response may lead to reduced offspring and greater mortality, resulting in lower abundances of the next generation. Under sub-optimal environmental conditions, this cycle disadvantages larger-bodied organisms, such as *Daphnia* sp., while favouring smaller organisms with better fitness under stress. A shift to increased *Bosmina* sp. abundance did occur in periods of presumably higher DDT exposure, with this shift being primarily responsible for driving the size metric. This was particularly evident at Upsalquitch and California lakes (Kurek et al. 2019).

The continued reduction in body sizes in the most recent sediments of Upsalquitch and California lakes is likely a result of climate change, which has impacted other New Brunswick lake assemblages resulting in greater *Bosmina* sp. abundance, fewer *Daphnia* sp. and a decreased assemblage body size in recent decades relative to the mid-20th century (Daly et al. 2019). Climate change, as a intensifying stressor, is a driver towards smaller body sizes as a result of earlier ice-off and ice-on dates, elongated growing seasons, and more pronounced thermal stratification (Armstrong and Kurek, 2019; Daly et al. 2019; Beaver et al. 2020). Summer and spring air temperatures in Northern New Brunswick have increased significantly since the 1920s and correlate with decreases in regional cladoceran assemblage richness and body size (Daly et al. 2019). Time of ice-off dates have also shifted over the last century, occurring ~15 days earlier across New England and southeastern NB (Patterson and Swindles 2014). Earlier ice-off dates influence lake properties crucial to phytoplankton, such as water-column mixing and light availability, causing many northern hemisphere lakes to experience climate-induced shifts in diatom assemblages (Rühland et al. 2008). A mismatch in spring diatom blooms and peak zooplankton abundances may also occur with earlier ice-off, which can affect individual and assemblage sizes of cladocerans as nutrient levels and competitive pressure change throughout the growing season (Winder and Schindler 2004; Rühland et al. 2008). Furthermore, changes in climate and water temperature at regional scales influence *Daphnia* sp. seasonal dynamics, where periods of minimum abundance are correlated with water temperatures greater than 20^{o}C in temperate lakes (Gillooly and Dodson 2000). Not only does the peak abundance of *Daphnia* sp. populations no longer align with the peak abundance of the food they need, but earlier-emerging *Bosmina* sp. now experience longer timespans before *Daphnia* sp. emerge and exert competitive dominance over available nutrients (Winder and Schindler 2004; Korosi et al. 2012). The delay in the start of the growing season and hatching of *Daphnia* sp. means emerging daphniids face the competitive pressure from an increased population of smaller taxa in conditions where larger organisms already demonstrate reduced fitness (Winder and Schindler 2004). As such, warmer climates and earlier ice-off dates have created an opportunity where *Bosmina* sp., a species with greater resiliency and better competitive ability in changing limnological conditions, have overtaken *Daphnia* sp. as the dominant cladoceran taxon in many lake ecosystems, especially those low in nutrients (Nevalainen et al. 2014; Armstrong and Kurek 2019). Climate change also intensifies the strength and lengthens the period of thermal stratification (Nevalainen et al. 2014; Thienpont et al. 2015) which may apply to the case of California and Upsalquitch lakes, which were the two deepest lakes sampled in this study (Kurek et al. 2019). Prolonged stratification can create low oxygen conditions of the colder hypolimnion, which makes this habitat intolerable for *Daphnia* sp. who often use deeper water for refuge from competition and predation (Tessier and Welser 1991).

Other factors, namely land use change and predation pressure, may be important to consider, however there is no direct monitoring data available to assess such stressors. California and Upsalquitch lakes are both situated closest to small-scale mining operations, where gold and silver was extracted near California Lake during 1989-1992, and an open-pit mining operation began in 1997 near Upsalquitch Lake (Leybourne et al. 2003). Whether there has been atmospheric, surface runoff, or groundwater movement of mining-related contaminants that *Daphnia* sp. exhibit sensitivity to (Rogalski et al. 2017), is unknown for these lakes. The correlation between the timing of cladoceran size decline events and mining activities suggests the possibility that local mining was at least partially responsible for the decline of cladoceran sizes over recent years exists. Predation pressure can also influence cladoceran sizes especially in cases of trophic cascade events (Brooks and Dodson 1965; Jeppesen et al. 2007). Despite the potential for a shift in predation to accompany fish introductions (Gliwicz and Boavida 1996), the body size of the cladoceran assemblage in Middle Peaked Mountain did not appear to change as a result of stocking, suggesting the metric trends mainly reflect other environmental stressors.

4.2 Do Cladoceran Empirical Measurements Verify Metric Trends?

The trends in the measured sizes of *Bosmina* sp. and *Daphnia* sp. generally aligned with the results of the assemblage metric, suggesting that the metric accurately tracks changes in body size for common pelagic taxa. At Goodwin Lake, there is a decline in the abundance of *D. longispina* complex in the post-impact period corresponding with an increased abundance of *D. pulex* complex and to a lesser degree *Bosmina*. *D. longispina* complex was significantly larger in the pre-impact period compared to following time periods, which likely indicates greater environmental stress, beginning with DDT applications to the watershed and continuing cumulative stress with climate change in recent decades. However, this body size decline does not reflect change at the assemblage level due to the increased presence of *D. pulex* complex, which emphasizes the different changes occurring within and between taxa that are noticeable at each biological scale.

The cladoceran assemblage in Upsalquitch Lake has shifted to *Bosmina* sp. becoming the dominant taxon in post-impact conditions, co-occurring with a near extirpation of *D. longispina* complex (Kurek et al. 2019). Upsalquitch Lake measurements of both *D. longispina* complex and *Bosmina* sp. in the post-impact period differ significantly from earlier periods, where *D. longispina* complex are larger and *Bosmina* sp. are smaller in recent times. Not only are the smaller-bodied taxa more dominant, but individuals of *Bosmina* sp. are also smaller. *Daphnia longispina* complex postabdominal claw sizes increase in the post-impact period of Upsalquitch Lake, which contrasts the general theory that *Daphnia* sp. exhibit a shift to smaller sizes under stress (Moore and Folt 1993; Wojtal-Frankiewicz 2012). Within current assemblages where *Bosmina* sp., a less efficient grazer, are most abundant, there may be greater food availability for remaining *Daphnia* sp., which may allow the few *Daphnia* sp. remaining to grow larger under less inter-specific competition. The timing of a decline in *Bosmina* sp. carapace lengths of Upsalquitch Lake over the post-impact period has co-occurred with an assemblage-level shift to smaller sizes that has persisted to the present. Thus, a shift towards smaller sizes has occurred at the taxon level as well as the assemblage level under presumably prolonged conditions of environmental stress.

4.3 Implications

Cladocerans are a critical link in aquatic food webs, and changes in their assemblages can influence multiple trophic levels. Understanding body size changes of cladoceran assemblages in lakes is thus important for the comprehension of lake dynamics, in addition to using the metric to possibly identify events of environmental stress. Filtering rates increase exponentially with size (Stemberger and Miller 2003), and thus cladoceran body size can be an indirect factor in controlling lake water clarity. In a study across 59 northeastern U.S. lakes, cladoceran body length was found to be an important predictor for water transparency, total phosphorus, DOC, and chlorophyll *a* concentrations (Stemberger and Miller 2003). The larger *Daphnia* sp. are faster at filtering particles and feed on a wider range of phytoplankton than *Bosmina* sp., so that a shift to *Bosmina* sp. dominance decreases the top-down grazing pressure exerted by the assemblage (Korosi et al. 2012). Shifts in the trophic dynamics of cladocerans and algae are exacerbated by climate warming, such that algal bloom events are more likely to occur under *Bosmina* sp. dominated assemblages (Korosi et al. 2012; Daly et al. 2019). Furthermore, *Daphnia* sp. abundance can decline with increases in cyanobacteria, as they experience greater sensitivity to toxic algal populations than *Bosmina* sp., which leaves more opportunities for increased abundance of smaller cladocerans like *Bosmina* sp. (Dupuis and Hann 2009; Jiang et al. 2014; Daly et al. 2019). Besides the extirpation of populations at Upsalquitch, *Daphnia* sp. in the study lakes may face further population declines, and in turn assemblage size declines, as their hatching time is asynchronous with times of highest food availability, adding the stressor of food availability to thermal and chemical stressors experienced (Winder and Schindler 2004). Especially with the increase of algal blooms in oligotrophic lakes taking place in the Northern Hemisphere, including remote New Brunswick lakes, nutrient levels are important to consider in assessing the effects of body size changes on lake resiliency for the future (Armstrong and Kurek 2019; Daly et al. 2019).

Not only are smaller zooplantkton less intense grazers, but the positive relationship existing between size and nutrition value means smaller cladocerans are also less nutritious prey. Larger, efficient grazers like *Daphnia* sp. provide a higher quality food source, which transfer more carbon, calcium, and phosphorus through the food web than smaller-bodied *Bosmina* sp. (Persson et al. 2007). As *Daphnia* sp. are less abundant during early algal blooms due to later hatch times, less energy is transferred to higher trophic levels when *Bosmina* sp. are the most available prey (Winder and Schindler 2004; Adamczuck 2016). Smaller organisms also possess a higher metabolic cost, relative to hunting success, so assemblages with lower potential energy can in turn affect the standing stock of the secondary trophic level (Persson et al. 2007). The dominance of smaller cladoceran species—which themselves are getting smaller—may affect organisms further up the food web.

5.0 Conclusion

The goal of this thesis was to develop a simple metric that can rapidly assess cladoceran body size changes through time based on sedimentary abundance data. Along with measuring the sizes of individual cladoceran remains in sediments over distinct time periods, lakes with known environmental stress events were chosen to further test the validity of the metric in response to severe DDT inputs reflected in dated sediment records. The agreement between trends in measured sizes and metric trends confirm that this approach offers promise as a method of rapid size assessment in cladoceran assemblages through time. The shift from *Daphnia* sp. to *Bosmina* sp. dominance corresponds to shifts in overall assemblage size, which has implications for nutrient flow and energy transfer to higher trophic levels. Two of the five lakes have significantly smaller cladocerans at the scales of both the individual and assemblage levels in the post-impact periods (> 1970), while minor changes in three other lakes indicates that these sites potentially avoided the cumulative effects of multiple stressors due to differences in lake resiliency. Overall, the metric shows potential for recognizing trends in cladoceran body sizes over a series of time, thus serving as a useful tool for future studies of aquatic environments.

Figures and Tables

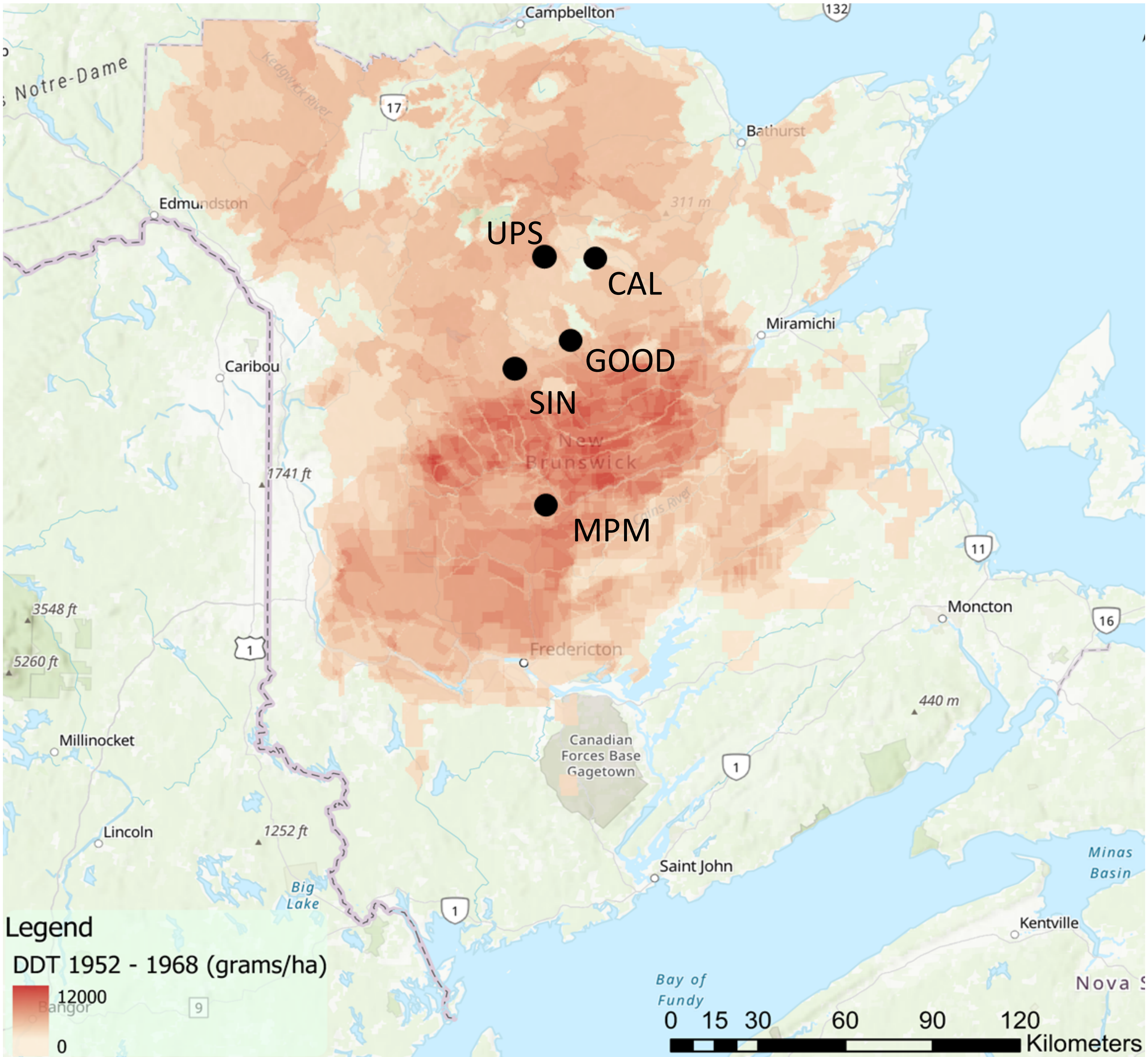

Figure 1. DDT concentrations applied across New Brunswick between 1952-1968 (Map courtesy of New Brunswick Natural Resource & Energy Department). Black points indicate lakes studied: Upsalquitch (UPS), California (CAL), Goodwin (GOOD), Sinclair (SIN), and Middle Peaked Mountain (MPM) lakes.

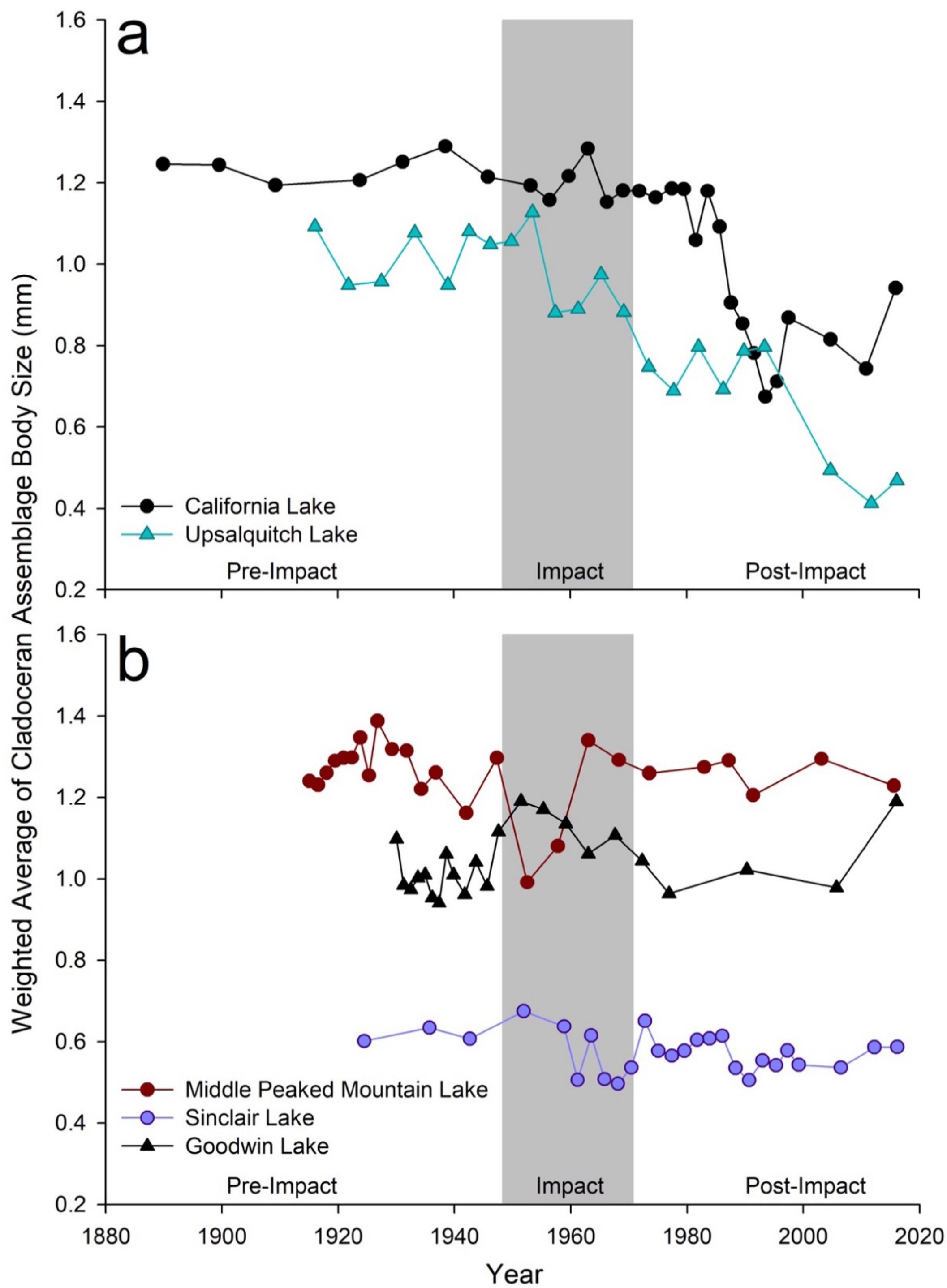

Figure 2. Body sizes (weighted average values calculated by the size metric) of five New Brunswick lakes across time. a) Lakes exhibiting significant ($p < 0.05$) changes between pre-impact and post-impact periods. b) Lakes showing no significant changes in the assemblage sizes of the pre-impact and post-impact periods.

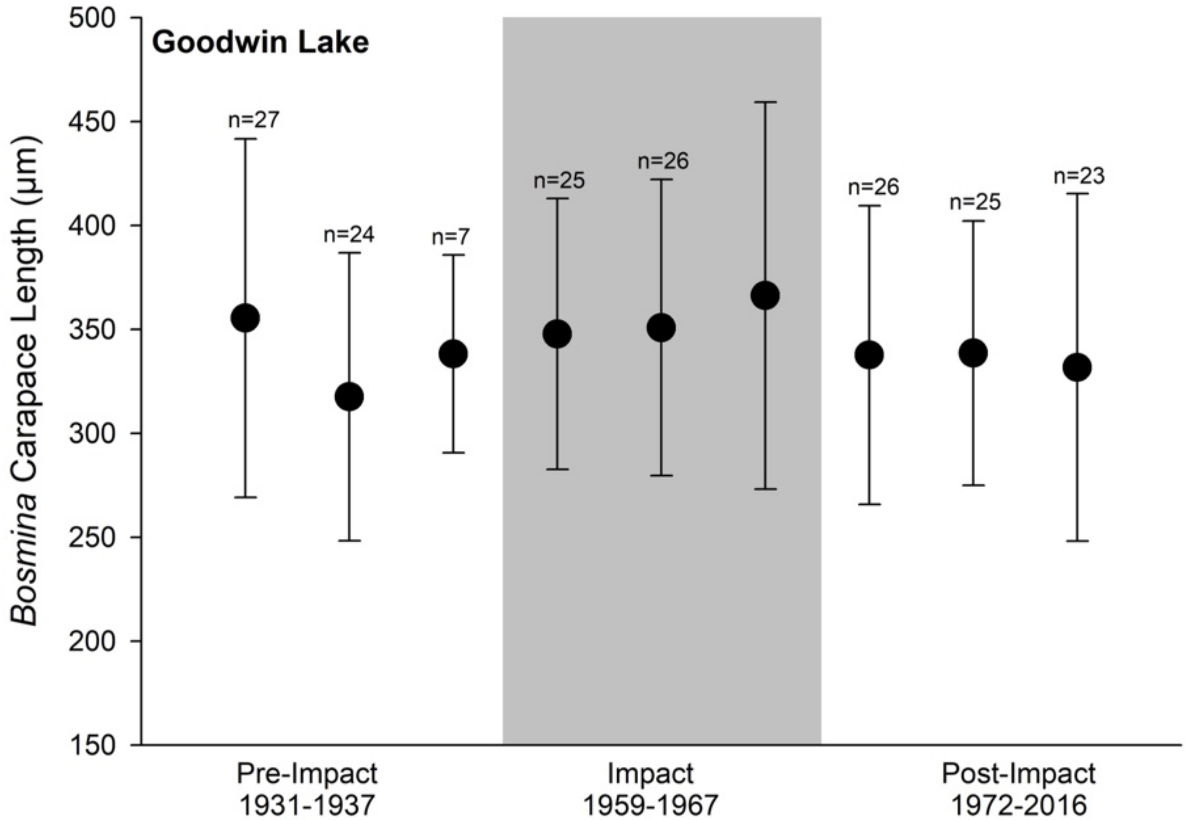

Figure 3. *Bosmina* sp. carapace lengths at Goodwin Lake. Points represent means, and bars represent standard deviation. Numbers above samples indicate the number of individuals measured when the minimum sample size threshold (30) was not met.

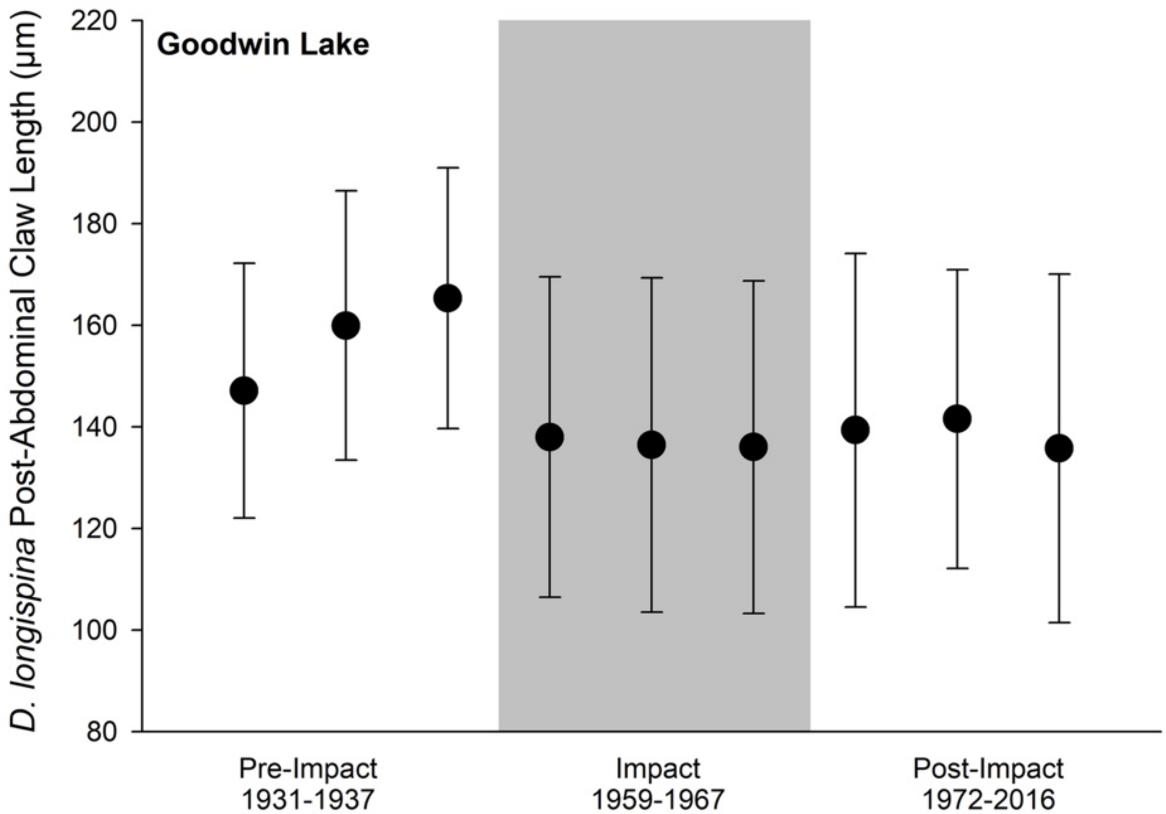

Figure 4. *Daphnia longispina* complex post-abdominal claw lengths at Goodwin Lake. Points represent means, and bars represent standard deviation.

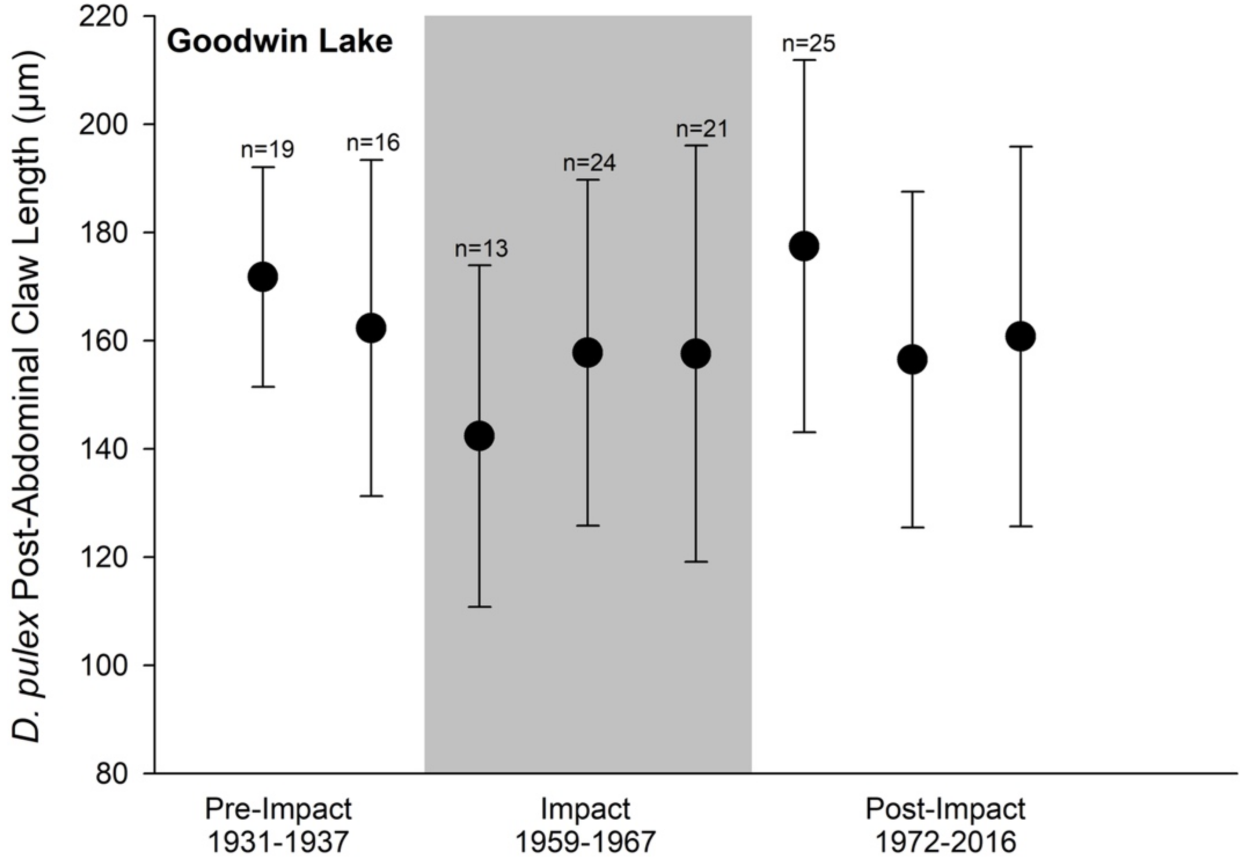

Figure 5. *Daphnia pulex* complex post-abdominal claw lengths at Goodwin Lake. Points represent means, and bars represent standard deviation. Numbers above samples indicate the number of individuals measured when the minimum sample size threshold (30) was not met.

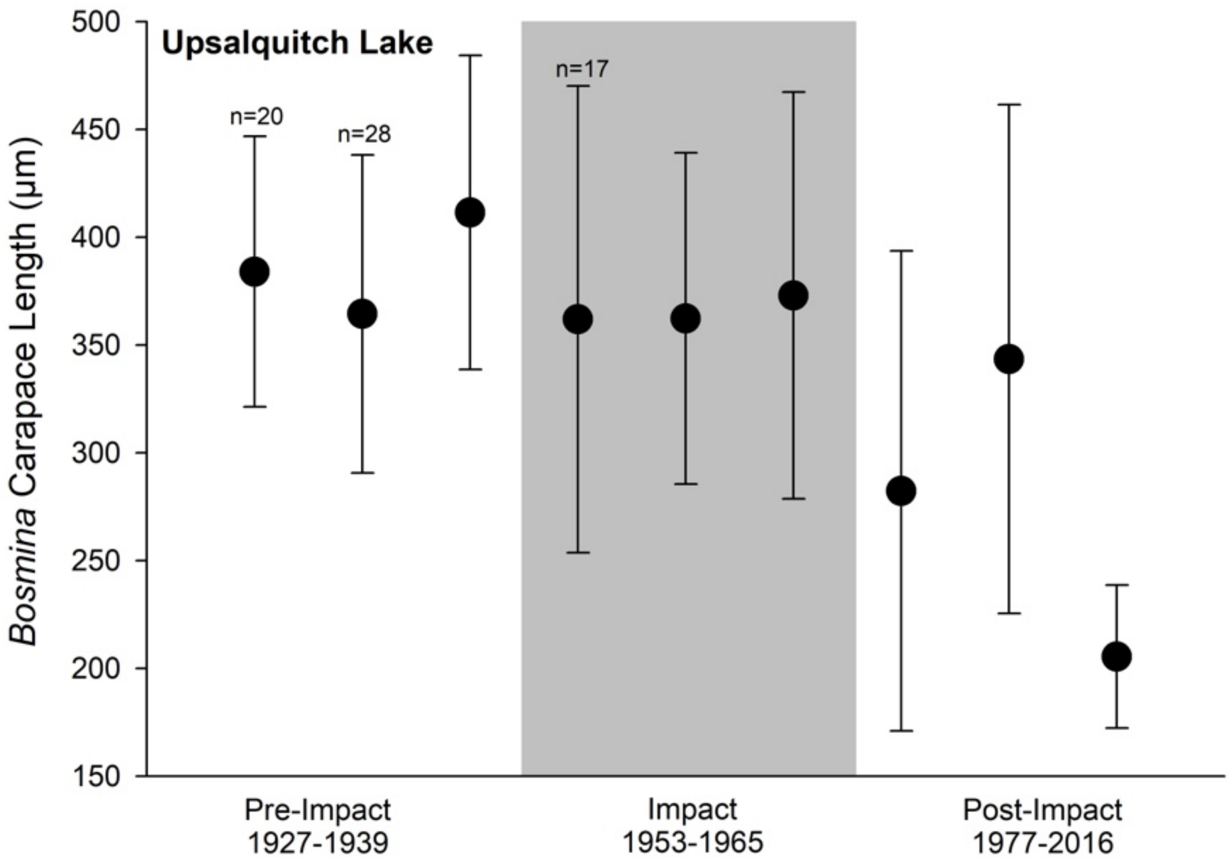

Figure 6. *Bosmina* sp. carapace lengths at Upsalquitch Lake. Points represent means, and bars represent standard deviation. Numbers above samples indicate the number of individuals measured when the minimum sample size threshold (30) was not met.

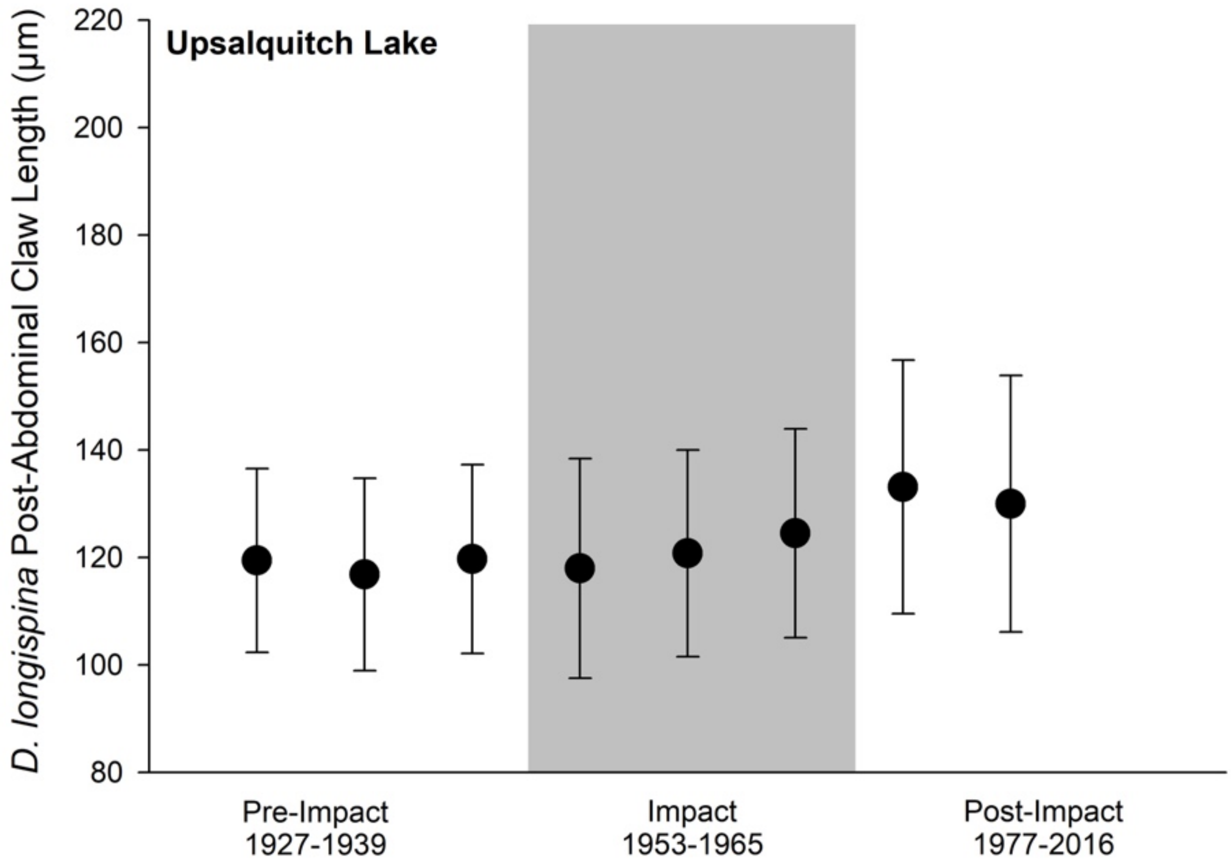

Figure 7. *Daphnia longispina* complex post-abdominal claw lengths at Upsalquitch Lake. Points represent means, and bars represent standard deviation.

Table 1. Summary of information used to compile the cladoceran body size metric. Values represent an average body size for female organisms of the given taxon. "Type of metric" describes the type of value that was given in the literature. The sources of the data contributing to the metric are included, as well as their respective study locations. *Daphnia* sp. were grouped into larger respective species complexes of either *Daphnia longispina* complex (DLC) or *Daphnia pulex* complex (DPC), and the average value for all taxa within the complex was then applied to the relative abundance data of study lakes.

Taxa	*Length (mm)*	*Type of metric*	*Location*	*References*
Acantholeberis curvirostris	1.09	Midpoint of Range	Belgium	Dumont et al. (1975)
Acroperus harpae	0.57	Mean of Means	Great Lakes; Ontario [1]; Wisconsin	Kratz (2019); Pedruski (2008), Sturtevant (2006)
Alona affinis	0.77	Mean of Range Midpoints	Great Lakes; North America [2]	Lynch (1980), Sturtevant (2006),
Alona barbulata	0.39	Mean	Minnesota	Megard (1967)
Alona bicolor	0.56	Mean	Massachusetts	Frey (1965)
Alona circumfimbriata	0.34	Mean	Minnesota, New Mexico	Megard (1967)
Alona costata	0.46	Global Mean	Great Lakes; Michigan, Minnesota; Ontario [1]	Frey (1965), Pedruski (2008), Sturtevant (2006),
Alona guttata	0.27	Mean	Ontario [1]	Pedruski (2008)
Alona intermedia	0.28	Midpoint of Range	Nova Scotia	Paterson (1991)
Alona quadrangularis	0.62	Midpoint of Range	Great Lakes; Minnesota; North America [2]; Ontario [1]	Dodson & Frey (2001), Herrick (1884), Pedruski (2008), Sturtevant (2006),
Alona rectangula	0.52	Mean of Range Midpoints	Minnesota; North America [2]; USA [2]	Birge (1918), Dodson & Frey (2001), Herrick (1884)
Alona rustica	0.39	Mean of Means	Massachusetts, Washington	Frey (1965)
Alonella exigua	0.31	Global Mean	Minnesota; Ontario [1]; USA [2]	Birge (1918), Herrick (1884), Pedruski (2008)
Alonella excisa	0.30	Global Mean	Minnesota; Ontario [1]	Herrick (1884), Pedruski (2008)
Alonella nana	0.22	Global Mean	Ontario [1]; USA [2]	Birge (1918), Pedruski (2008)
Alonella pulchella	0.23	Mean	Newfoundland, Minnesota, Florida	Hann & Chengalath (1981)
Alonopsis americana	0.63	Mean	New Hampshire	Kubersky (1977)
Alonopsis elongata	0.73	Mean	Great Lakes; North America [2]; USA [2]	Birge (1918), Lynch (1980), Sturtevant (2006),
Anchistropus minor	0.35	Mean	Great Lakes; USA [2]	Birge (1918), Sturtevant (2006)
Bosmina spp.	0.38	Global Mean	Alaska; Great Lakes; Midwest USA; North America [2]; Ontario [1][3]; South/Central/North America & Caribbean; USA [2]; Wisconsin	Balcer et al. (1984), Beaver et al. (2020), Birge (1918), Canfield & Jones (1996), Culver et al. (1985), Gillooly & Dodson (2000), Kratz (2019), Lathrop (2013); Lynch (1980), Pedruski (2008), Sturtevant (2006), Sweetman & Finney (2003), Valois (2008), Weckel (1914), Zorn et al. (2020)

Eubosmina spp.	0.42	Global Mean	Great Lakes; Ontario [3]; South/Central/North America and Caribbean; USA [2]; Wisconsin	Balcer et al. (1984), Barbiero & Tuchman (2004), Beaver et al. (2020), Culver et al. (1985), Gillooly & Dodson (2000), Lathrop (2013), Sturtevant (2006), Valois (2008)
Camptocercus spp.	0.83	Mean of Range Midpoints	British Columbia, Cape Cod, Florida, Manitoba, Mississippi, New Hampshire, Ontario, Rhode Island, Texas, Washington, Wisconsin; Minnesota	Herrick (1884), Sinev (2018)
Chydorus arcticus	0.49	Midpoint of Range	Europe [1]	Bledzki & Rybak (2016)
Chydorus bicornutus	0.5	Midpoint of Range	New York	Frey (1981)
Chydorus biovatus	0.39	Mean	Montana	Frey (1985)
Chydorus gibbus	0.52	Global Mean	Europe [2]; USA [2]	Birge (1918), Bledzki & Rybak (2016)
Chydorus globosus	0.71	Midpoint of Range	Great Lakes; Minnesota; North America [2]	Herrick (1884), Lynch (1980), Sturtevant (2006)
Chydorus faviformis	0.55	Global Mean	Great Lakes; USA [2]	Birge (1918), GLERL (2006)
Chydorus latus	0.7	Mean	Great Lakes; Minnesota	Herrick (1884), Sturtevant (2006),
Chydorus linguilabris	0.28	Midpoint of Range	Florida, Nova Scotia, North Carolina	Frey (1982)
Chydorus sphaericus	0.35	Global Mean	Lake Ontario; Minnesota; North America [2]; Ontario [1]; South/Central/North America and Caribbean; USA [2]; Wisconsin	Beaver et al. (2020), Birge (1918), Culver et al. (1985), Gillooly & Dodson (2000), Herrick (1884), Lynch (1980), Lathrop (2013), Pedruski (2008)
Daphnia galeata mendotae	1.55	Global Mean	Lake Erie; North America [2]; Ontario [1][4]; USA [2]	Almond et al. (1996), Beaver et al. (2020), Culver et al. (1985), Lynch (1980), Pedruski (2008)
Daphnia lumholtzi	2.54	Global Mean	Great Lakes; North America [2]; USA [2]	Beaver et al. (2020), Lynch (1980), Sturtevant (2006)
Daphnia magna	3.58	Global Mean	Alabama; Nebraska; North America [2]	Barnett et al. (2013), Fordyce (1904), Lynch (1980), Modlin (1982)
Daphnia middendorffiana	2.56	Global Mean	British Columbia; North America [2]; South/Central/North America & Caribbean	Colbourne et al. (1997), Gillooly & Dodson (2001), Green et al. (1996), Lynch (1980)
Daphnia retrocurva	1.35	Global Mean	Great Lakes; North America [2]; Ontario [3][4]; South/Central/North America & Caribbean; USA [2]	Almond et al. (1996), Balcer et al. (1984), Beaver et al. (2020), Colbourne et al. (1997), Culver et al. (1985), Gillooly & Dodson (2001), Sturtevant (2006), Valois (2008)
Daphnia parvula	0.93	Global Mean	Great Lakes; North America [2]; South/Central/North America & Caribbean; USA [2]	Balcer et al. (1984), Beaver et al. (2020), Colbourne et al. (1997), Gillooly & Dodson (2000), Sturtevant (2006)
Daphnia longispina (DLC)	1.67	Midpoint of Range	North America [2]	Lynch (1980)
Daphnia ambigua (DLC)	0.87	Global Mean	Great Lakes; Ontario [3]; South/Central/North America & Caribbean; USA [2]	Beaver et al. (2020), Gillooly & Dodson (2000), Sturtevant (2006), Valois (2008)

Daphnia mendotae (DLC)	1.96	Mean of Range Midpoints	Great Lakes; North America [2]; Ontario [3]	Balcer et al. (1984), Colbourne et al. (1997), Valois (2008)
Daphnia dentifera (DLC)	1.42	Global Mean	Minnesota; North America [2]; South/Central/North America & Caribbean; USA [2]	Beaver et al. (2020), Colbourne et al. (1997), Gillooly & Dodson (2000), Herrick (1884)
Daphnia dubia (DLC)	1.43	Global Mean	North America [2]; Ontario [4]; South/Central/North America & Caribbean; USA [2]	Almond et al. (1996), Beaver et al. (2020), Colbourne et al. (1997), Gillooly & Dodson (2000)
Daphnia longiresmis (DLC)	1.02	Global Mean	Great Lakes; North America [2]; Ontario [3][4]; South/Central/North America & Caribbean	Almond et al. (1996), Balcer et al. (1984), Barbiero & Tuchman (2004), Colbourne et al. (1997), Gillooly & Dodson (2000), Sturtevant (2006), Valois (2008), Weckel (1914)
Daphnia pulex (DPC)	1.80	Global Mean	Alabama; British Columbia; Great Lakes; Nebraska; Nevada; North America [2]; Ontario [1][2][4]; South/Central/North America & Caribbean; USA [2]	Almond et al. (1996), Balcer et al. (1984), Beaver et al. (2020), Bowman (2008), Brandlova et al. (1972), Colbourne et al. (1997), Fordyce (1904), Gillooly & Dodson (2000), Green et al. (1997), Modlin (1982), Pedruski (2008), Sturtevant (2006)
Daphnia pulicaria (DPC)	1.74	Global Mean	Great Lakes; Nebraska; North America [2]; Ontario [3]; South/Central/North America & Caribbean; USA [2]	Balcer et al. (1984), Barnett et al. (2013), Beaver et al. (2020), Colbourne et al. (1997), Fordyce (1904), Gillooly & Dodson (2000), Sturtevant (2006), Valois (2008)
Daphnia catawba (DPC)	1.44	Global Mean	North America [2]; Ontario [3]; Pennsylvania; South/Central/North America & Caribbean; USA [2]	Barnett et al. (2013), Beaver et al. (2020), Colbourne et al. (1997), Gillooly & Dodson (2000), Tessier (1986), Valois (2008)
Daphnia minnehaha (DPC)	1.78	Global Mean	Minnesota; North America [2]; USA [2]	Beaver et al. (2020), Colbourne et al. (1997), Herrick (1884)
Diaphanosoma brachyrum	0.80	Global Mean	Great Lakes; North America [2]; Ontario [1]; South/Central/North Americas and Caribbean; USA [2]	Beaver et al (2020), Birge (1918), Gillooly & Dodson (2000), Lynch (1980), Pedruski (2008), Sturtevant (2006)
Diaphanosoma birgei	0.81	Global Mean	Great Lakes; Ontario [3]; South/Central/North America & Caribbean; USA [2]; Wisconsin	Balcer et al. (1984), Barbiero & Tuchman (2004), Beaver et al. (2020); Birge (1918), Gillooly & Dodson (2000), Kratz (2019), Lathrop (2013), Sturtevant (2006), Valois (2008)
Disparalona acutirostris	0.34	Mean of Means	Great Lakes; North Carolina; Wisconsin	Frey (1961), Kratz (2019), Sturtevant (2006)
Drepanothrix spp.	0.60	Global Mean	Great Lakes; Minnesota; USA [2]	Birge (1918), Herrick (1884), Sturtevant (2006)
Dunhevedia spp.	0.35	Global Mean	Minnesota	Herrick (1884)
Eurycercus spp.	2.39	Global Mean	Ontario [1]; North America [2]	Lynch (1980), Pedruski (2008)
Graptoleberis testudinaria	0.60	Global Mean	Great Lakes; Minnesota; USA [2]	Birge (1918), Herrick (1884), Sturtevant (2006)
Holopedium gibberum	1.20	Global Mean	British Columbia, Northwest Territories; Great Lakes; Ontario [3]; Pennsylvania; USA [2]; Wisconsin	Balcer et al. (1984), Barbiero & Tuchman (2004), Barnett et al. (2013), Birge (1918), Kratz (2019),

				Rowe et al. (2007), Sturtevant (2006), Tessier (1986), Valois (2008)
Holopedium glacialis	0.89	Mean of Means	Northwest Territories, Ontario, Wisconsin; USA [2]	Beaver et al. (2020), Rowe et al. (2007)
Ilyocryptus spp.	0.63	Mean of Means	USA [2]	Beaver et al. (2020), Birge (1918)
Kurzia latissima	0.52	Global Mean	Ontario [1]; USA [2]	Birge (1918), Pedruski (2008)
Latona setifera	2.5	Midpoint of Range	Great Lakes; USA [2]	Balcer et al. (1984), Birge (1918), Sturtevant (2006)
Leptodora kindti	7.75	Global Mean	Great Lakes; North America [2]; Wisconsin	Balcer et al. (1984), Culver et al. (1985), Lathrop (2013), Sturtevant (2006), Weckel (1914)
Leydigia leydigi	0.73	Global Mean	British Columbia; Great Lakes	Balcer et al. (1984), Kotov (2003)
Limnosida frontosa	1.55	Mean	Europe [2]	Bledzki & Rybak (2016)
Monospilus dispar	0.47	Mean of Means	Great Lakes; USA [2]	Birge (1918), Sturtevant (2006)
Ophryoxus gracilis	1.75	Midpoint of Range	Norway [2]	Walseng (2016)
Oxyurella spp.	0.52	Mean	USA [2]	Birge (1918)
Paralona pigra	0.35	Midpoint of Range	Samples used from North America and beyond	Dumont & Smirnov (1996)
Pleuroxus aduncus	0.56	Midpoint of Range	Great Lakes; North America [2]	Lynch (1980), Sturtevant (2006)
Pleuroxus denticulatus	0.51	Global Mean	Great Lakes; Ontario [1]; USA [2]	Birge (1918), GLERL (2006), Pedruski (2008)
Pleuroxus hastatus	0.57	Global Mean	Great Lakes; Minnesota; USA [2]	Birge (1918), Herrick (1884), Sturtevant (2006)
Pleuroxus laevis	0.49	Mean	Europe [2]	Bledzki & Rybak (2016)
Pleuroxus procurvus	0.46	Global Mean	Minnesota; Ontario [1]; USA [2]	Birge (1918), Herrick (1884), Pedruski (2008)
Pleuroxus striatus	0.72	Global Mean	Great Lakes; USA [2]	Birge (1918), Sturtevant (2006)
Pleuroxus straminius	0.6	Other	Minnesota	Herrick (1884)
Polyphemus pediculus	0.92	Global Mean	Great Lakes; North America [2]; Ontario [1]; Quebec	Balcer et al. (1984), Barnett et al. (2013), Pedruski (2008), Sturtevant (2006), Weckel (1914)
Rynchotalona falcata	0.50	Global Mean	Russia & Europe [2]; USA [2]	Birge (1918), Sinev & Kotov (2014)
Sida crystallina	3.2	Mean of Range Midpoints	British Columbia; Great Lakes; North America [2]	Balcer et al. (1984), Green et al. (1997), Lynch (1980), Sturtevant (2006)
Unaptura latens	0.25	Other	Finland	Nevalainen (2008)

[1]: Frontenac County Region, Ontario
[2]: Generalized location, data collected from various regions in province/country
[3]: Canadian Shield Lakes of Ontario
[4]: Huntsville-Bancroft Region and Lake Ontario, collectively, Ontario

Appendix A

Appendix 1. Relative abundance of *Bosmina* and *D. longispina* in the cladoceran assemblages in Goodwin Lake and Upsalquitch lake through time (Kurek et al. 2019). Shaded cells represent values within the impact period, when DDT was applied to forests.

Goodwin Lake				Upsalquitch Lake		
Year	% *Bosmina* sp.	% *D. longispina* complex	% *D. pulex* complex	Year	% *Bosmina* sp.	% *D. longispina* complex
2016.0	25.7	33.0	28.4	2016.2	97.3	0.7
2005.7	35.7	23.2	19.6	2011.8	94.6	1.6
1990.3	25.2	36.9	18.9	2004.7	87.2	6.4
1976.9	33.6	36.2	7.1	1993.4	56.0	38.4
1972.3	29.2	33.3	17.5	1989.9	56.1	37.7
1967.6	22.1	55.8	8.0	1986.3	64.9	27.5
1963.0	22.3	51.5	11.7	1982.0	50.0	39.5
1959.1	21.2	56.7	2.9	1977.8	63.2	28.1
1955.3	14.0	54.4	8.8	1973.5	58.5	30.5
1951.4	19.8	61.3	4.7	1969.2	49.2	41.0
1947.6	17.9	58.1	6.0	1965.3	47.1	41.2
1945.6	21.6	50.4	4.8	1961.3	53.8	36.8
1943.7	20.9	53.6	4.5	1957.4	48.6	40.2
1941.7	25.0	46.0	4.8	1953.5	23.1	70.2
1939.8	22.9	51.1	2.3	1949.9	30.6	58.7
1938.6	22.7	47.7	4.7	1946.2	28.3	54.7
1937.4	19.6	53.6	0.9	1942.6	29.7	55.1
1936.1	22.8	46.3	5.1	1939.0	37.7	46.7
1934.9	19.7	46.2	6.0	1933.2	32.0	51.6
1933.7	13.9	53.7	3.7	1927.5	37.6	42.2
1932.5	26.1	34.8	9.6	1921.8	36.2	49.6
1931.3	16.0	47.2	2.4	1916.1	27.2	51.8
1930.0	12.5	51.9	8.7			

Appendix B: ANOVA Results

Appendix B1. ANOVA results assessing cladoceran assemblage body sizes for all study lakes.

	Degrees of Freedom	Sum of Squares	Mean of Squares	*F*	*p*
Goodwin Lake	2	0.0545	0.0272	6.9	< 0.01
Residuals	20	0.0788	0.00394		
California Lake	2	0.485	0.242	11.4	< 0.01
Residuals	26	0.550	0.0211		
Upsalquitch Lake	2	0.644	0.3221	23.4	< 0.01
Residuals	19	0.260	0.0137		
Sinclair Lake	2	0.00508	0.00254	1.1	0.34
Residuals	23	0.05295	0.00230		
MPM Lake	2	0.0331	0.0165	2.7	0.08
Residuals	22	0.132	0.00601		

Appendix B2. ANOVA results for *Bosmina* carapace lengths and *Daphnia* sp. post-abdominal claw lengths in Goodwin Lake.

	Degrees of Freedom	Sum of Squares	Mean of Squares	*F*	*p*
Bosmina	2	19027	9514	1.65	0.19
Residuals	213	1228058	5766		
Daphnia Longispina	2	44243	22122	23.6	< 0.01
Residuals	683	640226	937		
Daphnia Pulex	2	4363	2182	2.01	0.13
Residuals	209	226278	1083		

Appendix B3. ANOVA results for *Bosmina* sp. carapace lengths and *Daphnia* sp. post-abdominal claw lengths in Upsalquitch Lake.

	Degrees of Freedom	Sum of Squares	Mean of Squares	*F*	*p*
Bosmina	2	1867651	933825	99.47	< 0.01
Residuals	609	5717221	9388		
Daphnia longispina	2	13613	6807	16.91	< 0.01
Residuals	514	206942	403		

Appendix C: Tukey Test Findings for Metric Results

Appendix C1. Tukey Test results for cladoceran assemblage sizes of Goodwin Lake.

	Difference in Means	Lower Confidence Interval	Upper Confidence Interval	p_{adj}
Post-Impact vs. Impact	-0.09	-0.19	0.006	0.07
Pre-Impact vs Impact	-0.12	-0.20	-0.039	< 0.01
Pre-Impact vs Post-Impact	-0.02	-0.11	0.054	0.65

Appendix C2. Tukey Test results for cladoceran assemblage sizes of California Lake.

Period	Difference in Means	Lower Confidence Interval	Upper Confidence Interval	p_{adj}
Post-Impact vs. Impact	-0.23	-0.41	-0.06	< 0.01
Pre-Impact vs. Impact	0.03	-0.16	0.23	0.88
Pre-Impact vs Post-Impact	0.27	0.11	0.44	< 0.01

Appendix C3. Tukey Test results for cladoceran assemblage sizes of Upsalquitch Lake.

Period	Difference in Means	Lower Confidence Interval	Upper Confidence Interval	p_{adj}
Post-Impact vs. Impact	-0.29	-0.463	-0.13	< 0.01
Pre-Impact vs. Impact	0.07	-0.094	0.24	0.51
Pre-Impact vs. Post-Impact	0.37	0.227	0.51	< 0.01

Appendix D: Tukey Test Findings for Empirical Measurements from the Sediment Record

Appendix D1. Tukey Test results for *Daphnia longispina* complex in Goodwin Lake.

	Difference in Means	Lower Confidence Interval	Upper Confidence Interval	p_{adj}
Post-Impact vs. Impact	2.42	-4.34	9.19	0.67
Pre-Impact vs. Impact	18.21	11.70	24.71	0
Pre-Impact vs. Post-Impact	15.78	8.53	23.03	< 0.01

Appendix D2. Tukey Test results for *Daphnia longispina* complex in Upsalquitch Lake

	Difference in Means	Lower Confidence Interval	Upper Confidence Interval	p_{adj}
Post-Impact vs. Impact	10.05	4.61	15.48	< 0.01
Pre-Impact vs. Impact	-2.99	-7.75	1.77	0.30
Pre-Impact vs. Post-Impact	-13.04	-18.40	-7.67	< 0.01

Appendix D3. Tukey Test results for *Bosmina* sp. in Upsalquitch Lake

	Difference in Means	Lower Confidence Interval	Upper Confidence Interval	p_{adj}
Post-Impact vs. Impact	-105.23	-126.54	-83.92	< 0.01
Pre-Impact vs. Impact	20.37	-10.47	51.23	0.26
Pre-Impact vs. Post-Impact	125.60	97.64	153.56	< 0.01

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