

The Ecological Impacts and Effectiveness of Control Methods for
the Invasive Macrophyte *Myriophyllum spicatum* in Eastern
Ontario Lakes

by
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Abstract

In this thesis, the impacts a year after the treatments of both burlap benthic barriers and mechanical cutting of *M. spicatum* on physical, chemical, and biological components of eastern Ontario lakes were investigated. One lake was selected for studying the burlap barrier treatment, and three lakes were selected across a nutrient gradient for the mechanical harvesting treatment to observe links between total phosphorus concentrations and macrophyte-zooplankton dynamics. The impacts of the treatments on this invasive species were minimal, but there was a clear shift in zooplankton assemblage to more open-water types following the benthic barrier application. Results from both treatments displayed minimal impacts to localized abiotic factors and zooplankton assemblages. While the change in zooplankton community structure at Malcolm Lake did appear to be partially driven by the burlap barrier treatment, zooplankton communities within both types of treatments were instead mainly driven by their surrounding environmental variables and seasonal variance.

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Chapter 1.0 – Introduction

Invasive species are one of the leading causes of reductions in aquatic biodiversity, and are among the biggest threats to freshwater ecosystems (Chandra & Gerhardt, 2008; Dextrase & Mandrak, 2006; Reid et al., 2019). Invasion by non-native species of macrophytes are a worldwide phenomenon that can lead to various ecological impacts (Williamson, 1999), such as increasing habitat complexity, promoting hypoxia, decreasing the abundance of native fish and macroinvertebrates (Schultz, 2012), and ultimately causing loss of biodiversity (Hussner, 2010). The most problematic invasive macrophyte in Canadian freshwater ecosystems is *Myriophyllum spicatum* L. (common name Eurasian watermilfoil) (Newman, 2004; Smith & Barko, 1990). It is often problematic in lakes as it is able to quickly proliferate and outcompete native species of macrophytes (Boylen, 1999). It is native to Europe, Asia and northern Africa, and was introduced to the Chesapeake Bay in the United States sometime between the late 1800s and 1940 (Madsen et al., 1991). It is thought that hybridisation between the non-indigenous (*M. spicatum*) and native species (*Myriophyllum sibiricum*) has caused heterosis and local extinction of the native species through gene contamination and has resulted in the extensive spread of invasive populations of *M. spicatum* in North America (Moody et al., 2002).

Due to its resiliency and propagation mechanisms, such as plant fragmentation and transport between lakes by recreational boat traffic (Smith, 1990), *M. spicatum* has spread across nearly all of North America (Aiken, 1979). Once established, *M. spicatum* is known to decrease property values and development (Goodenberger & Klaiber, 2016;

Horsch & Lewis, 2009), and inhibit recreational activities such as swimming, fishing, and boating (Eiswerth et al., 2000). This often leads to the use of management techniques (Eiswerth et al., 2000) in an attempt to control the abundance and spread of this invasive macrophyte. However, these managements techniques only control and reduce the abundance and spread of *M. spicatum*, as it is nearly impossible to eradicate them from a lake once it is established (Nichols & Buchan, 1997).

Macrophytes, also referred to as aquatic plants, are a diverse group of photosynthetic organisms that are characterised by their emergent, floating, or submerged growth formation (Chambers et al., 2008). Rooted submerged macrophytes are found in the littoral zone of lakes and are an important component of aquatic ecosystems as they determine much of the structure and function of the littoral environment (Pandit, 1984); provide habitat for zooplankton (Stansfield, 1997), macroinvertebrates (Declerck, 2011; Keast, 1984; Sagrario, 2009) and fish (Cross, 2006; Crowder 2011; Miranda, 2000); reduce turbidity in the water column by stabilising bottom sediments (Horppila, 2005; Rooney, 2003); and absorb nutrients directly from nutrient-rich sediments, with leaf uptake of nutrients from the water column as a secondary nutrient source (Carignan, 1980; Carignan 1982). This influence on fish, zooplankton and macroinvertebrate community structure could even limit algal growth (Phillips, 2016). However, macrophytes can also be seen as a nuisance to lake and river users, as they can reduce the aesthetic quality of the water body, hinder water transport and recreational activities (Nes et al., 1999; Verhofstad et al., 2019), and impact ecosystem processes such as by creating anoxic zones when macrophytes are present in dense beds (Kagami et al., 2019). Thus,

dense stands of invasive macrophytes can often bring important environmental and economical impacts (Eiswerth et al., 2000; Smith et al., 1990).

Since the threat of invasive macrophytes has been impacting aquatic ecosystems globally (Evangelista et al., 2014; Gallardo et al., 2016; Schultz & Dibble, 2012), a variety of macrophyte management practices have emerged. Control measures include mechanical removal, the application of benthic barriers, biological controls or waterborne herbicides (Caffrey et al., 2010; Carpenter & Adams, 1978; Cuda et al., 2008; Sesin et al., 2018). Because the use of aquatic herbicides are strictly regulated in Canada (Ontario, 2014) as they are often associated with potential environmental impacts (Carpenter & Greenlee, 1981), other macrophyte management techniques such as mechanical harvesting and benthic barriers are being investigated.

Mechanical harvesting has a long history in Ontario, with some lakes within the Kawartha Lakes area removing macrophyte biomass since the early 1970s (Weinberger, 1980). It can provide many advantages such as potentially removing a source of phosphorus for internal loading and reduce plant decomposition (Carpenter & Adams, 1978). Macrophyte harvesting can also help maintain aquatic habitats as plants can regrow, and the application is site specific since the harvester only removes plants where it is operated (Gettys et al., 2009). The use of benthic barriers to shade out unwanted macrophytes has been around for several decades and in many cases has also proven to be an effective management practice (Cooke & Gorman, 1980; Mayer, 1978). Biodegradable benthic barriers have more recently gained some attention, as they do not cause environmental impacts such as oxygen depletion from rapid plant decay (Boston & Perkins, 1982; Gunnison & Barko, 1992) and are associated with lower management

costs (Caffrey et al., 2010). Burlap benthic barriers may be a preferred physical macrophyte treatment in Eastern Ontario waters, as suppressing macrophytes using mechanical removal often requires multiple harvests a year to provide adequate control (Madsen, 2000).

There are other emerging methods of physical and biological controls that have shown some promise such as water jets (Dorenbosch & Bergsma, 2014) and weevils (Cooper et al., 2017), though more research on the effectiveness and the impacts of these management practices is needed. The efficacy of both mechanical harvesting and burlap benthic barriers can vary, but are generally well known in the literature (Hussner et al., 2017; Madsen, 2000), while studies on the ecological impacts of these two management techniques are less known.

Before applying any large scale macrophyte management practice, it is important to consider the potential impacts on lake ecosystems. Managing macrophytes may unintentionally impact chemical, physical, and biological properties of these inland waters. Zooplankton communities play a crucial role in aquatic ecosystems by grazing phytoplankton, driving carbon and nutrient cycles, and acting as prey for fish (Keister et al., 2012; Yang et al., 2017). They are especially great indicators of aquatic ecosystem health, as studies have reported that zooplankton diversity and abundance are both well correlated with various abiotic factors of their environment such as temperature, pH, dissolved oxygen, nutrients, and conductivity (Jakhar, 2013). Thus, investigating zooplankton abundance and diversity in areas experiencing macrophyte control would illuminate potential environmental impacts from these practices.

Macrophyte control may even influence alternating stable states within freshwater ecosystems. Eutrophic lakes containing a regime with moderate to high nutrient levels, varying turbidity, and varying macrophyte abundance are thought to be less resilient than a oligotrophic to mesotrophic system with a regime characterized by low nutrients, abundant macrophytes, and clear waters (Valley & Drake, 2007). Technically, a regime that contains high nutrient levels and possesses a turbid algae-dominated state can also be considered a resilient system. However, this state is typically less desirable as it does not provide the same ecosystem functions and services as the alternate state (Hilt et al., 2017).

One of the reasons why a eutrophic system with varying macrophyte abundance may possess a lower ecosystem resilience is that the system lacks macrophyte abundance. Lower macrophyte abundance leads to less zooplankton refuges from fish predation (Scheffer, 2001) and increased sediment resuspension (Horppila & Nurminen, 2003, 2005). This decrease in zooplankton refuges increases the turbidity throughout the water column as sediments are more likely to be resuspended and phytoplankton abundance increases from reduced grazing pressure (Tessier & Horwitz, 1990), ultimately reducing light penetration to lower depths (Järvenpää et al., 2004). Higher nutrient levels can also stimulate phytoplankton growth (Schindler, 1977), further increasing turbidity and reducing macrophyte abundance (Jeppesen et al., 1992).

These eutrophic systems that still contain an abundant macrophyte biomass have more difficulty recovering from environmental disturbances and are more likely to experience a regime shift towards a turbid algae-dominated stable state (Holling, 1973; Scheffer et al., 2001). These algal-dominated states can imply reduced natural ecosystem

functions and services (Hilt et al., 2017) and any attempt at restoring these systems to their former state can be challenging and costly (Genkai-Kato & Carpenter, 2005). Therefore, any macrophyte management techniques that could possibly trigger a shift towards the turbid-water stable state should be applied cautiously.

Unfortunately, macrophyte management practices are more often prompted for the benefits of humans over any ecological function (Chambers et al., 1999; Van Nes et al., 2002). This type of motivation has created important gaps in our understanding of the direct impacts of macrophyte management techniques on lake ecosystems.

This thesis will investigate the impacts of two different macrophyte management techniques on aquatic ecosystems while attempting to control the most problematic invasive macrophyte in Ontario lakes. The objective is to determine the impacts of controlling dense patches of *Myriophyllum spicatum* in lakes at a small scale typical of what is being carried out by lake users and lake managers in Ontario. I will specifically examine physical, chemical, and biological components of freshwater ecosystems in lakes located in the south-eastern Ontario before and after inducing these management practices, with the goal of providing information that could help improve invasive macrophyte management planning across Canada. Another objective of this thesis is to compare lakes containing different trophic levels to test the pre-existing theory that lakes that are more nutrient rich (greater total phosphorus concentrations) are more susceptible to ecological change (Valley & Drake, 2007) following macrophyte removal.

I hypothesise that using either of these macrophyte management techniques at a smaller scale will cause noticeable changes in the lake's health indicators such as dissolved oxygen, nutrients, or zooplankton community structure. However, both

macrophyte treatments will likely only have a minimal local impact on overall aquatic ecosystems when attempting to control stands of *M. spicatum*. The thesis will also test the hypothesis that a high trophic status system dominated by macrophytes will have greater environmental changes and slower recovery from local perturbations when compared to lower trophic status systems containing greater ecosystem resilience. If this is true, a less resilient eutrophic lake containing a regime with moderate to high nutrient levels and varying turbidity will likely show greater changes in certain lake properties and demonstrate slower recovery when compared to oligotrophic and mesotrophic lakes. As a result, it is also likely that the eutrophic system will take longer to return these properties to their original state.

This thesis is structured in a way that will first help explain key components of freshwater ecosystems and provide important information on native and invasive macrophytes, as well as assess various management techniques used to control overabundance of macrophytes. It then explores the application of burlap benthic barriers on the physical, chemical, and biological properties of a single lake ecosystem a year after the application. The thesis also investigates the impacts of mechanical macrophyte harvesting on the same ecological properties and timescale on three different lakes over a gradient of total phosphorus concentrations. Three different lakes were chosen to study the impacts of mechanical removal in order to test if sites with greater phosphorus concentrations displayed greater differences in chemical, physical or biological components when compared to their natural state, and if these sites displayed slower recovery following the induced disturbance.

Chapter 2.0 – Literature review

2.1 Freshwater ecosystems

Freshwater ecosystems are hotspots of biodiversity (Dudgeon et al., 2006; Reid et al., 2019). Unfortunately, freshwater habitats are particularly vulnerable to environmental change, and are highly threatened by human utilisation (Dudgeon et al., 2006). Global anthropogenic impacts that threaten these crucial ecosystems include pollution, overexploitation, species invasion, thermal alterations, habitat destruction, species extinctions, and climate change (Dodds et al., 2013).

There is growing recognition that healthy and biologically complex aquatic ecosystems provide many valuable services and can offer many short term benefits to society (Collen et al., 2014; Wilson et al., 1999). Inland waters supply essential ecosystem services to humans by providing water for industry, transportation, recreation, drinking water, water purification, food supply, flood control, and carbon sequestration (Collen et al., 2014; Williamson et al., 2008). Inland waters can also provide long term benefits, including the sustained provision of these services, as well as ecosystem resilience to respond to any future environmental alterations or disturbances (Baron et al., 2002). While freshwater ecosystems provide these benefits to the well-being of humans and aquatic biodiversity, they only amount to approximately 0.009% of the earth's water (Dodds, 2002). The services provided from such a limited commodity is why it is critically important to use effective management and conservation action to protect freshwater ecosystems around the globe.

2.2 Macrophytes

The distribution patterns of rooted submerged macrophytes depend on various factors such as light availability, water depth, temperature, exposure to wind and waves, alkalinity, nutrient enrichment, sediment characteristics, and hydrological variations (Dar et al., 2015; Duarte & Kalff, 1990). When present, macrophytes have many interactions with various components in lake ecosystems such as interactions with their physical/chemical environment and interactions with biota (Carpenter, 1986). Because of these interactions, macrophytes are responsible for much of the structure and function within freshwater ecosystems (Jeppesen et al., 2015).

Macrophytes can decrease water flow through littoral zones of lakes (Carpenter et al., 1986) and have the ability to reduce resuspension of benthic sediments that could be reintroducing phosphorus in the water column (Dzialowski et al., 2005). Because macrophyte plant beds increase sedimentation compared to non-vegetated areas (Horppila et al., 2005), the reduced turbidity allows better light penetration through the water column, which increases light availability for macrophytes and allows for deeper macrophyte growth (Canfield et al., 1985; Patricia et al., 1985). Due to this sedimentation effect, it is said that macrophyte beds can act as sediment traps and also as sinks of phosphorus, organic matter and trace metals, since they remove them from the open water (Rooney et al., 2003). This is particularly important, since resuspended sediments can increase algal biomass in lake ecosystems due to increased nutrient availability and/or meroplankton establishment (Dzialowski et al., 2008).

Dense macrophyte beds can bring about periods of anoxia near the sediment surface at night when plants are not photosynthesising, providing another mechanism for anoxia

in lakes (Jeppesen et al., 1998). Macrophyte beds are also an important factor in determining the diversity and abundance of zooplanktons (Figure 1) in lakes, as they can act as habitat structure for zooplankton communities (Phillips et al., 2016). Various zooplankton species abundances have been shown to be positively correlated with macrophyte characteristics such as species number, plant biomass and plant type (Choi et al., 2014). Kuczyńska-Kippen et al. (2009) also demonstrated that increases in macrophyte cover and biomass in a small water body following decreases in water levels throughout the summer contributed to an increase in littoral species abundance of zooplankton. The reason for this increase is that submerged macrophytes can provide an excellent refuge for zooplankton against fish predation. Even at a high predation pressure from zooplanktivorous fish, macrophytes beds still provided a refuge effect for *Daphnia* and other cladocerans (Stansfield et al., 1997) (Schriver et al., 1995).

This refuge effect is also improved by horizontal diurnal migration (HDM), where zooplankton use the macrophyte's habitat structure as a day-time refuge when fish predation risks are high and feed in the open water at night when the risk is lower (Burks et al., 2002; Davies, 1985). The zooplankton protection effect from these aquatic plants has even been considered one of the most important mechanisms in stabilizing the clear-water state of freshwater ecosystems (Blindow et al., 2014), since Cladocera are capable of exerting a high grazing pressure on algae (Stansfield et al., 1997; Timms & Moss, 1984).

The structural complexity of mixed macrophyte stands can act as complex habitat that influences the composition of benthic and littoral macroinvertebrate biomass and abundance (Blindow et al., 2014; McAbendroth et al., 2005). This allows macrophytes to

host a diverse and abundant macroinvertebrate assemblage within their plant structure, some of which may prey on zooplankton entering littoral zones for refuge. This structural complexity suggests that the refuge effect offered to zooplankton by macrophyte beds can be controlled by the predatory macroinvertebrate assemblage within the macrophyte beds, especially if the beds are inhabited by macroinvertebrates such as *Buena sp.* (backswimmer), *Palaemonetes argentinus* (grass shrimp) and *Cyanallagma interruptum* (damselfly) (Sagrario et al., 2009). Some freshwater ecosystems with macrophyte beds providing habitat structures in littoral areas could therefore lose their refuge effect for zooplankton, if they also host a high abundance and diversity of littoral macroinvertebrate predators (Mamani et al., 2019).

2.3.0 Ecological Stressors in Freshwater Systems

Lakes are exposed to various stressors leading to direct or indirect impacts to their ecosystem. Globally, there are many threats to freshwater systems that are caused by human activities such as eutrophication, invasive species, salinization, contaminants, microplastics, changing climate and the cumulative effect of these stressors (Kaushal et al., 2018; Reid et al., 2019). The ecosystem changes associated to these stressors can have global consequences for well-being, including the loss and alteration of goods and services such as fisheries, clean drinking water, climate stabilisation, and cultural and recreational uses (Pejchar & Mooney, 2009).

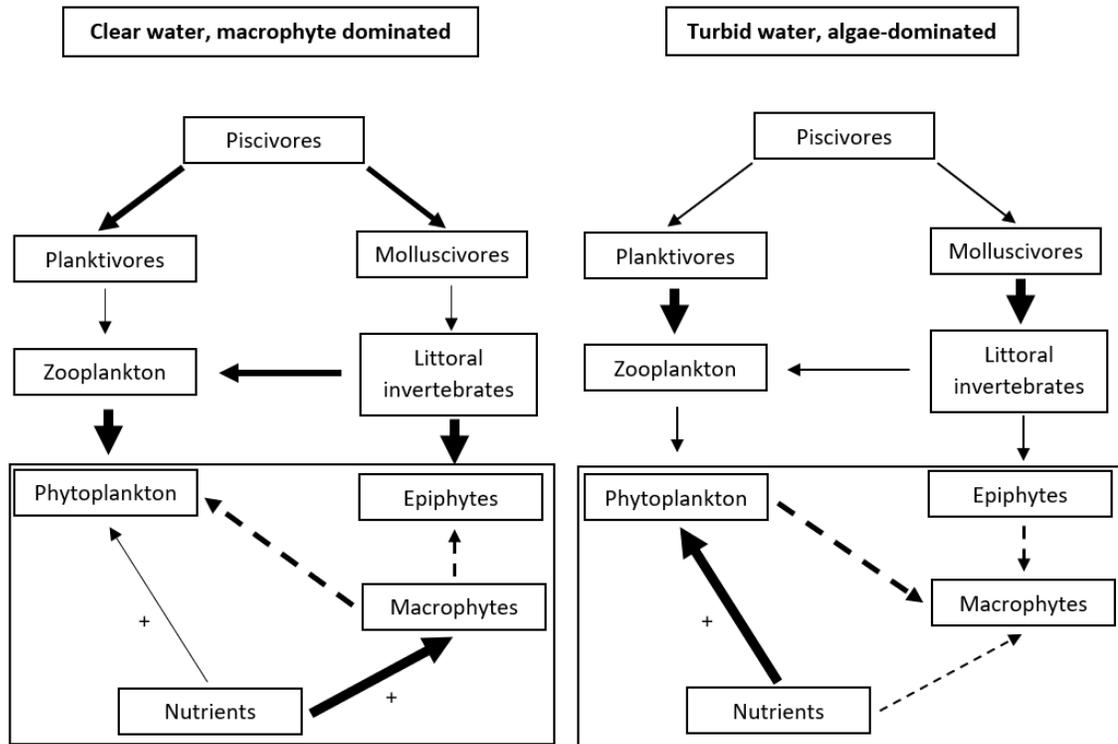


Figure 1. Diagram summarising interactions in clear water, macrophyte-dominated lakes (on the left), and diagram summarizing interactions in turbid-water algal-dominated lakes (on the right). Arrows show the direction of the pressure/effect, and their sizes show the magnitude. The boxed in bottom half of each diagram focuses on bottom-up effects related to nutrients, while the top half of each diagram shows top-down effects. Dashed arrows show other effects including allelopathy, nutrient sequestering, shading, and physiological constraints. Arrows containing symbols (+) indicate positive effects, while the remaining arrows represent a negative effect or pressure. For simplicity feedback effects of macrophytes on piscivores and littoral invertebrates via provision of refugia are not shown. Figure inspired by Phillips et al. (2016).

2.3.1 Invasive Species

While poorly defined and often subjective (Colautti et al., 2004), invasive species are generally referred to as organisms that are not native to a specific area, that negatively alter the new environment in which they are introduced (Davis et al., 2000). Once introduced, invasive species can threaten ecosystems as they can alter essential ecosystem functions and degrade environments. If not properly managed, invasive aquatic species can impact ecosystem services from lakes, leading to a range of environmental, economic, and social repercussions.

There are many ways that invasive species can be introduced to Canadian lakes, including international shipping, ballast water discharge from shipping traffic along the Great Lakes, the construction of canals, movement of recreational boats, aquarium plants, and fish releases (Bailey, 2015; Dextrase & Mandrak, 2006; Rahel & Olden, 2008). Once introduced to freshwater ecosystems in Canada, invasive species can trigger ecological changes through various mechanisms. For instance, predation triggering top-down changes, species grazing by herbivores that may prompt bottom-up cascades, and competition between species within the same trophic level are three major mechanisms that can propel negative impacts in freshwater ecosystems (Gallardo et al., 2016). Invasive species can also fundamentally change aquatic ecosystems by introducing genetic alteration such as hybridization, and introducing diseases and parasites which ultimately cause a negative impact on native species driving a decline in ecosystem biodiversity (Dextrase & Mandrak, 2006; Michalek et al., 2016).

In Canadian lakes, invasive species are the second most prevalent threat to the biological diversity of fishes and molluscs (Horsch & Lewis, 2009), affecting approximately 60% of species listed as Extinct, Extirpated, Endangered, and Threatened under the Committee on Status of Species at Risk in Canada (COSEWIC).

Invading lower trophic level species such as invasive macrophytes can eventually overgrow and outcompete native macrophytes and substantially change the hydrology, sedimentation, clarity and nutrient state of lakes, reducing habitat availability for other species positioned higher in the trophic web such as invertebrates and fish (Gallardo et al., 2016; Gérard & Triest, 2018). An example of this is reduced native plant species after the introduction of the invasive macrophyte species *Myriophyllum spicatum* in North

American lakes. Studies have shown that this invasive aquatic plant can negatively impact the density and diversity of native aquatic plant communities, decreasing species richness and habitat complexity throughout the system (Abernethy et al., 1996; Boylen et al., 1999; Kowalczewski & Ozimek, 1993).

Industries, governments, and consumers are all impacted by the costs imposed by invasive species. A main driver of these excessive costs are direct operating costs from maintenance-type actions, such as cleaning, fixing damaged infrastructure, or management techniques to control the invasive species impacting the industry (Rosaen, Grover, & Spencer, 2012). There are also economical costs when the invasive species directly affects productivity or product demand of certain industries. For example, tourism locations and hospitality industries, such as resorts, could have reduced customers because invasive aquatic plants can often prompt changes in scenery and water use (Nentwig W., 2007; Rosaen et al., 2012). Water treatment plants or energy generation plants could also be reduced in output levels and productivity, especially if their pipeline systems are clogged by invasive species, such as zebra mussels or *M. spicatum* (Eiswerth et al., 2000; Pimentel, 2005). Moreover, according to Horsch & Lewis, (2009), lakes in North America that were invaded by *M. spicatum* experienced a 13% mean reduction in property values. Even new parcels of land on and near lakes that were invaded *M. spicatum* are less likely to be developed for housing (Goodenberger & Klaiber, 2016).

2.3.2 Nutrient Enrichment

Another common issue among lake ecosystems is the overabundance of nutrients, primarily phosphorus. Anthropogenic activities such as agricultural, industry and sewage disposal that discharges into aquatic ecosystems can contribute to high levels of nutrients

in lakes (Chrislock et al., 2013), creating ideal conditions for algae and aquatic plants to grow. Phosphorus is a limiting nutrient that is very important for controlling primary production in most freshwater ecosystems, and it is been nearly 50 years since the discovery that enrichment by phosphorus is the leading cause of eutrophication in lakes (Schindler, 1974, 1977). High concentrations of phosphorus in aquatic environments can contribute to excessive algal and/or macrophyte growth, and often triggers eutrophication within the ecosystem. This could in turn generate water quality issues such as algal blooms, increased turbidity, and anoxia (Baldwin, 2013; Chrislock et al., 2013).

Since nutrient enrichment affects all primary production in lakes, increased nutrient concentrations in lakes can lead to an increase and over-abundance of macrophytes (Chambers, 1987; EPA, 2021). The proliferation of macrophytes caused by nutrient enrichment may interfere with ecosystem processes and recreational activities (Othman et al., 2015) or lead to various consequences such as clogging waterways and degrading habitat for other aquatic species (Shitao et al., 2012). In fact, studies have suggested that overabundant macrophytes can even trigger localized oxygen depletion within their plant beds, due to reduced water mixing and light penetration (Turner et al., 2010). This can lead to other impacts such as inducing nutrient release from the sediments and producing anoxic conditions within the plant beds that can affect various species of fish and invertebrates (Vilas et al., 2017).

On the other hand, increased turbidity caused by algal blooms within the water column can limit light from penetrating to lower parts of the lake which can have a shading effect on other phototrophs, including macrophytes (Dokulil & Teubner, 2011; Sand-Jensen & Søndergeerd, 1981; Søndergaard et al., 1992). This reduction in

macrophyte biomass due to shading from algae can produce a positive feedback in shallow lake ecosystems because a reduction of plant cover can promote greater turbidity and nutrient availability in the water column (Jeppesen et al., 1998; Takamura et al., 2003).

Furthermore, the excessive algal growth that is produced after nutrient enrichment of waterbodies can occasionally be considered harmful algal blooms (HABs). HABs are blooms of cyanobacteria that have the capability of producing biotoxins such as neurotoxins like microcystins (Miller et al., 2010). These blooms are notorious for their harmful effects on aquatic animal and human health, as they are often associated to mass mortality of aquatic organisms (Landsberg, 2010). Due to their superior competitive abilities, species that often develop into HABs and dominate nutrient-rich lakes include *Dolichospermum*, *Microcystis*, and *Planktothrix* (Chrislock et al., 2013).

Algae do eventually die off; however, the abundant microbial decomposition of the deceased phytoplankton can lead to more water quality issues. When large masses of dead algae are decomposed, dissolved oxygen levels are severely depleted and hypoxic or anoxic conditions are created within the ecosystem (Karim et al., 2002). These depleted oxygen areas are often called “dead zones” and lack enough oxygen to support most organisms (Chrislock et al., 2013). It is for these reasons that nutrient loading is often taken into consideration during lake management practices, as it poses a large threat to species composition, biodiversity, and ecosystem services (Dokulil & Teubner, 2011).

2.4 Ecosystem Resilience and Alternate Stable States

It has been proposed that ecosystems can have more than one stable state and possess a level of resilience that dictates their recovery to their original stable state after an environmental disturbance (Holling, 1973). It is suggested that shallow lake ecosystems can occur in two main stable states (or equilibriums) that contain their own stable structure, functions, processes, and feedbacks (Angeler et al., 2014). One of these states consists of clear waters, low nutrient levels, and moderate macrophyte coverage (oligotrophic or mesotrophic lakes), while the other state consists of high nutrients, turbid water, lower macrophyte cover, and greater algal abundance (eutrophic or hypereutrophic lakes) (Scheffer & Carpenter, 2003) (Scheffer, 1990, 2001). Generally, the algae dominated ecosystem state supports a lower biodiversity of aquatic organisms and reduced natural ecosystem functions and services and possesses a lower value for human use (Hilt et al., 2017). In particular, the dominance in cyanobacteria in these high nutrient states is associated with lower phytoplankton community diversity, which can directly lower zooplankton abundance and diversity due to reduced resources and decreased niche partitioning (Bockwoldt et al., 2017). The zooplankton community structure within these systems typically remains as small-bodied individuals (e.g. *Bosmina*, or copepods) that lack algae grazing pressure (Jeppesen et al., 2000; Lampert, 1987).

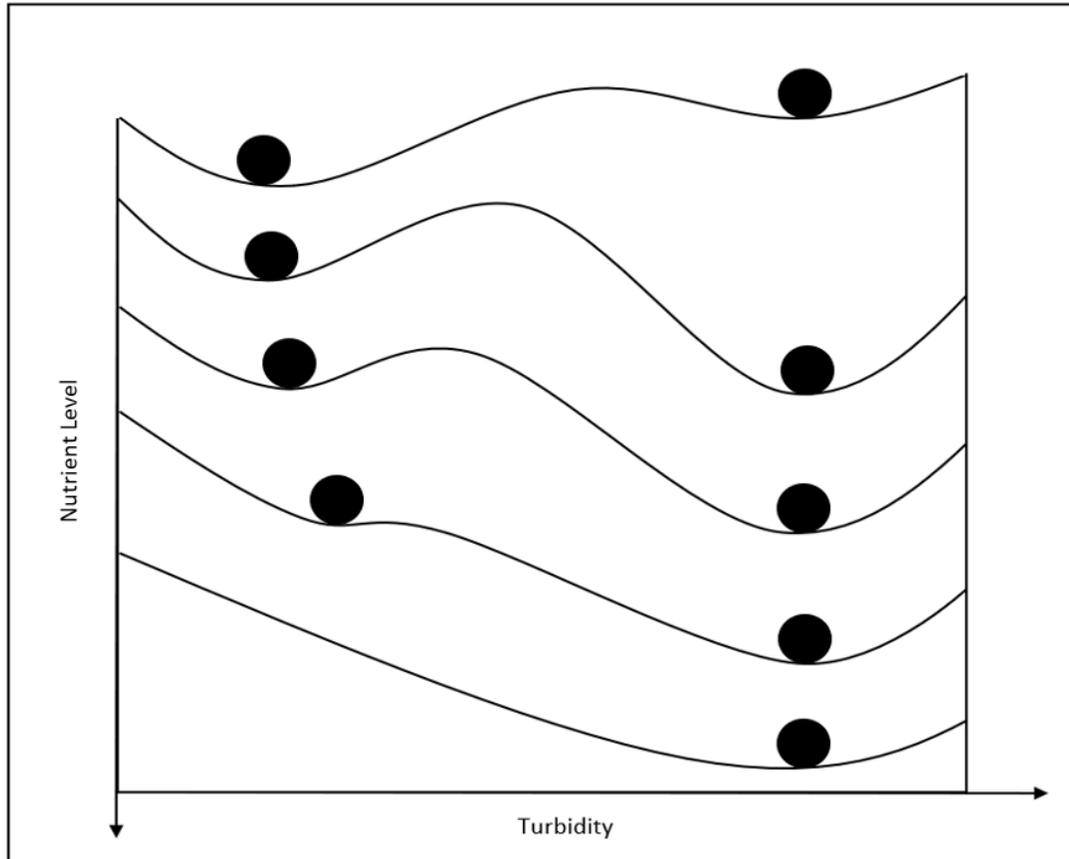


Figure 2. 'Marble-in-a-cup' representation of stability at five different nutrient levels. The minimums within the cups correspond to stable states, while the maximums represent unstable tipping points. Figure inspired by Scheffer's (1990) simple 'minimal' modelling.

When a lake ecosystem experiences environmental stressors or disturbance, it can undergo a regime shift where the ecosystem undergoes a non-linear change between these two alternate stable states (Carpenter et al., 2011). During the transition from one stable state to another, the drastic change in ecosystem structure and function can often be triggered after reaching a certain threshold or tipping point. Once this point is crossed, it means that the lakes adaptive capacity has been exhausted and the ecosystem abruptly transitions to the new stable state (Angeler et al., 2014) (Figure 2).

A system reaching past the threshold of the stability landscape is more likely to occur in ecosystems that possess a low ecological resilience, especially if an

environmental disturbance occurs. To put it simply, ecological resilience is the amount of disturbance that a system can tolerate without triggering a regime shift from one stable state to another (Walker et al., 2004). It is essentially the capacity of a system to absorb disturbances or environmental changes and maintain its current ecological structure and function by adapting to these disturbances through various mechanisms (Folke et al., 2004). Ecological resilience is therefore greater with more species diversity since it provides a diverse structure and function to the ecosystem state. While the loss of one species would be redundant within a highly diverse system, any species loss from environmental stressors or disturbance within a system lacking diversity would decrease ecological resilience of the system, making the lake more vulnerable to ecological collapse towards an algae dominated state (Peterson, 1998).

These regime shifts towards an algae dominated state have been linked to environmental stressors such as invasive species, nutrient enrichment, and macrophyte habitat destruction in shallow lake ecosystems (Matsuzaki et al., 2009; Morris et al., 2003; Phillips et al., 2016; Scheffer, 1990; Søndergaard et al., 1992). Ecosystems with low or decreasing ecological resilience are more easily tipped towards the algae dominated stable state. These systems with low resilience are theorized to have slower recovery rates, especially as the system approaches the a regime shift (Dakos et al., 2015). The system recovery rate back to its original state ultimately depends on its initial ecological resilience, dispersal rate of organisms, and the scale of environmental disturbance (van de Leemput et al., 2018). Generally, the more a shallow lake ecosystem is nearing its tipping point to an algal dominated state, the slower its recovery will be. This phenomenon is known as critical slowing down and is believed to indicate a loss in

ecosystem resilience and is thought to be a good indication of when an ecosystem is nearing its tipping point (Van Nes & Scheffer, 2007).

2.5 *Myriophyllum spicatum* L.

Myriophyllum spicatum L. is a well-documented invasive macrophyte that was introduced to North America in the late 1800's (Painter & McCabe, 1988). It is a perennial with very fine dissected leaves that typically grows in waters 1-4 m deep, but it can also be found in waters up to 10m deep (Nichols & Shaw, 1986; Smith et al., 1990). Due to its ability to proliferate quickly and dominate the littoral zones of the lakes it colonizes, it has spread across North America (Aiken et al., 1979; Wood & Netherland, 2017). It does, however, need adequate light to thrive, since it is not a shade tolerant macrophyte (Abernethy et al., 1996; Madsen et al., 1991). *M. spicatum* is notorious for being difficult to control and manage since it can easily spread through vegetative reproduction from small plant fragments released into the water from auto fragmentation or human disturbances such as motor entanglement (Gettys et al., 2009). These fragments can also be easily spread by transportation of boats and other equipment, between lakes (Eiswerth et al., 2000). *M. spicatum* can outcompete native vegetation by forming dense canopies and are often associated with declines of both native macrophyte and macroinvertebrate biodiversity (Boylen et al., 1999; Wilson & Ricciardi, 2009). Their presence could even impair the ability of some fish species to spawn in the littoral zone, resulting in a decline in fish populations (Boylen et al., 1999; Keast, 1984).

M. spicatum can produce adverse effects to water quality and ecosystem services of lakes. Once dense canopy covers are established, they can reduce the abundance and

species richness of native plant species by shading them out (Boylen et al., 1999). According to other studies, *M. spicatum* does support macroinvertebrate communities; however, they differ from its native counterparts and other native macrophytes (Wilson & Ricciardi, 2009). Thus, if this invasive plant outcompetes other native macrophytes, it could alter macroinvertebrate community structure and composition and have indirect effects on other species in the aquatic food web of the lake (Madsen, 1991). Since *M. spicatum* beds have been shown to harbor fewer benthic macroinvertebrate species (Wilson & Ricciardi, 2009) such as isopods, midges (chironomids), caddisflies (trichoptera) and mayflies (ephemeropteran nymphs), these shallow littoral zones generally host a different species composition of fish due to the change in prey availability (Keast, 1984). The decrease in native macrophyte species richness and abundance could therefore create food shortages for certain fish since dense covers of *M. spicatum* will host less diverse invertebrates. Large beds of this invasive macrophyte are also known to hinder swimming of pelagic fish and impair piscivores feeding and growth from reduced foraging success, which can lead to poor growth and lower fish fecundity (Engel, 1995; Savino & Stein, 1982). Not only does this indirectly affect recreational fishing for humans, but beds of *M. spicatum* can also decrease the quantity and quality of other recreational activities such as boating and swimming (Eiswerth et al., 2000). It can even have economic repercussions such as decreased property value, reduced productivity of industries, and increased cost in infrastructure maintenance (Goodenberger & Klaiber, 2016; Rosaen et al., 2012). All these impacts, especially those directly affecting humans, often provoke lake associations and/or management experts to take action to control this invasive macrophyte.

Although *M. spicatum* generally causes adverse effects in the lakes it invades, some lakes may benefit from its presence. *M. spicatum* can provide structure and habitat in algal dominated lakes that could not otherwise support other native plant beds (Engel, 1995), supporting greater numbers of taxa and abundance of macroinvertebrates when compared to open water habitats (Pardue & Webb, 1985). This might be explained by the fact that *M. spicatum* can support more epiphyte and periphyton biomass due to its leaf morphology (Cattaneo & Kalff, 1980), which in turn provides a better food source for macroinvertebrates and fish. Some studies suggest that *M. spicatum* appears to have little impact on invertebrate taxa richness and diversity compared to native macrophyte communities, and when integrated with them, appears to provide useful habitat to fish and invertebrates (Leguizamon, 2017). In fact, many of the benefits from native macrophytes that encourage a macrophyte-dominated state can also be found in lakes invaded by *M. spicatum*. Generally speaking, plant beds of the invasive plant could provide the same zooplankton refuge effect as other macrophyte species, encouraging phosphorus sedimentation and nutrient sequestration (Carignan, 1982; Carignan & Kalff, 1980; Nakai et al., 2000; Stansfield et al., 1997; Zhu et al., 2010). However, many of these benefits may be lessened if *M. spicatum* is present in large dense beds. In large canopies, *M. spicatum* can create anoxic conditions within its beds, especially near the sediment surface, limiting benthic invertebrates and possible even causing fish deaths during plant decay in the fall (Engel, 1995; Keast, 1984).

2.6.0 Macrophyte control

Although macrophytes have an important role in the functioning of healthy aquatic ecosystems, the potential large scale negative impacts of dense, monocultures of invasive macrophytes taking over lakes has stimulated research focused on the development of management techniques for effective aquatic plant control. These management techniques are more often developed for the benefits of humans (Van Nes et al., 2002), since large dense beds of macrophytes of any species can be seen as a nuisance. This is especially the case when invasive macrophytes interfere with recreational activities (Bickel & Closs, 2009), reduce the aesthetic quality of the water body, or hinder other economic services. All the management practices can be divided into physical, biological, and chemical control methods. Each practice can greatly differ in the scale, costs, and effectiveness of the application, and each can have their own advantages and disadvantages, often making it difficult to determine the correct management option for a given lake (Hussner et al., 2017).

2.6.1 Benthic Barriers

Benthic barriers are a physical management practice that has been used and tested for some time, but that has not received widespread management or scientific attention. The main idea of this control method is to cover the plants with a layer of material to inhibit their growth (Caffrey et al., 2010). There have been many different materials used with this method such as spreading sediments of sand, silt, or clay as well as covering macrophytes with synthetic or organic sheets/screens/mats (Cooke, 1980). Some of these materials have had more success in controlling macrophytes than others, while others have run into some complications. For instance, using sediment materials is often not an

effective long-term solution since new plants can establish on top of the new layer with relative ease (Madsen, 2000). Synthetic or plastic mats appear to have had a little more success in macrophyte management by successfully shading out the plants while the barrier maintained structural integrity (Mayer, 1978). The problem with synthetic sheeting/mats (e.g. long-lasting plastic, polyethylene sheeting etc.) is that these applications cause plant death, but also limit the exchange of dissolved gases between the water column and the sediments, possibly reducing dissolved oxygen levels to near-zero underneath the barrier due to plant decomposition (Boston & Perkins, 1982). This could in turn reduce macroinvertebrate density, in some cases up to 90%, and alter their community composition (Hussner et al., 2017; Ussery et al., 1997). Gases produced from the plant decomposition underneath the synthetic barriers could also cause the barriers to lift, affecting long-term stability and function (Caffrey et al., 2010; Madsen, 2000).

Biodegradable mats (Figure 3A), such as burlap made from jute, have been used as an effective benthic barrier. Unlike plastic mats, burlap mats are permeable and allow gas and water exchange at the water-sediment interphase, and some native plants can even grow through the mats (Caffrey et al., 2010; Hofstra & Clayton, 2012). However, the biodegradation of burlap also means that these mats may need to be replaced more frequently to sustain macrophyte control (Jones & Cooke, 1984). Shading with mats is also not species-specific and is limited to small scale management, since using it over widespread areas could be cost prohibitive (Hussner et al., 2017; Madsen, 2000). Lastly, burlap mats will still cause plant decay and potential oxygen depletion near the sediments, though this outcome can be mitigated by simply deploying the burlap mats during cooler periods of the year (spring or fall), when the standing crop and

decomposition rates of macrophytes are low (Gunnison & Barko, 1992). Overall, burlap benthic barriers can provide a cost-effective and a temporary site-specific solution for invasive macrophyte control (Hussner et al., 2017). In Canada, this management practice is not commonly used as you need federal or provincial permission to apply this macrophyte control method across Canadian inland waters since they do not fall under the provincial rules for removing invasive macrophytes in Ontario (Simkovic, 2020). But since this practice can be a cost-effective solution in controlling invasive macrophytes at smaller scales, the practice could help lake users and managers across Canada develop better plans for managing *M. spicatum* invasions, especially if treatments have little to no impacts to inland waters.

2.6.2 Mechanical harvesting

Physical control measures are among the oldest, and still to this day, the most widely used method for macrophyte management (Murphy, 1988). Mechanical harvesting is a commonly used practice that can be especially effective at cutting large areas containing macrophytes (Figure 3B). Mechanical harvesting provides advantages such as little plant decomposition, maintains some plant habitat, and is site specific (Gettys et al., 2009). It has even been suggested that macrophyte harvesting can be a useful tool for improving growth rates and size structures of desirable fish (Olson et al., 1998). However, mechanical harvesting can carry certain risks as it involves the spread and distribution of the targeted macrophyte species (e.g. creating plant fragments), could instigate a regime shift to an algae-dominated state from excess mowing in lakes containing high nutrient levels according to models from Kuiper et al. (2017), and can often cause unintentional by-catch of fish, turtles, amphibians and invertebrates that get

entangled in the harvested vegetation (Booms, 1999; Haller et al., 1980). The spread and distribution of the targeted macrophyte species can especially occur during the harvest of *M. spicatum* since the plant is known for its capability of easily proliferating from small plant fragments (Eiswerth et al., 2000).

The maximum cutting depth of most harvesters is about 2m water depth, which means that in instances where macrophytes reach lower depths, parts of the plants can remain rooted and intact (Barrett et al., 1999). It is for this reason that one mechanical harvest of a macrophyte bed is usually only a temporary solution as macrophytes tend to regrow after the technique is applied. However, harvesting 2-3 times a year in at the same harvested plot can provide annual control for most macrophyte species, possibly even reducing growth the following year (Madsen, 2000).

Mechanical harvesting has a relatively affordable cost at smaller scale treatments, can provide a temporary site-specific solution for macrophyte control, and has also been effective at temporarily suppressing *M. spicatum* (Madsen, 2000). The main reason this practice was chosen for this thesis was because it is the control method most commonly used by the Parks Canada, the federal agency applying macrophyte management in the Rideau Waterway.



Figure 3. (A) Burlap benthic barriers being applied on a large stand of *M. spicatum* L. at Malcolm Lake, Ontario. (B) Mechanical harvesting in action at Otter Lake, Ontario, using an aquatic weed harvester.

2.6.3 Other Management Practices

Another type of macrophyte management practices is biological controls. This category of macrophyte control uses a specific organism to reduce the abundance and reproductive success of another (in this case, targeted macrophyte) in aquatic ecosystems (Heimpel & Mills, 2017). Popular biological controls used for managing overabundant macrophytes include introducing herbivores such as grass carp (*Ctenopharyngodon idella*), or macroinvertebrates such as weevils (*Eurychiopsis lecontei*). The grass carp is one of the most controversial biological control agents for submersed aquatic plants, since there are concerns about its negative impact on water quality and non-target species due to potentially altering entire freshwater ecosystems (Cuda et al., 2008). They have a

strong influence on both macrophyte and macroinvertebrate diversity and abundance through their selective plant grazing (Mammoser, 2013), and if not properly controlled, the disappearance of macrophytes could lead to an algal dominated state in more eutrophic lakes (Krupska et al., 2012). There have also been tests on the application of weevils as an effective biological control of *M. spicatum* (Cooper et al., 2017; Frew, 2016; Havel et al., 2017). The milfoil weevil is native to North America and it has been shown to have better survival on *M. spicatum* than other native species, and preferentially attacked and grazed the invasive macrophyte (Cuda et al., 2008). While weevils showed some initial promise as a biological control of *M. spicatum*, this practice can be costly and is not an ineffective long-term solution (Cooper et al., 2017).

Another category of macrophyte control that has been given attention is chemical control methods, more specifically, herbicides. Since this treatment method often yields more effective results that produce longer-lasting control (Barrett et al., 1999), herbicides are one of the most widely used means of controlling nuisance growth of macrophytes world wide (Carpenter & Greenlee, 1981). Often due to fear and uncertainty of their effects, the public does not perceive herbicides as a viable macrophyte management practice (Schmidt, 1984). However, this is a misinterpretation of the management practice, since evidence shows that if used correctly, following the proper guidelines, herbicides are no more damaging for the environment than alternate methods (Barrett et al., 1999). Studies have shown that when comparing the impacts of herbicides to other forms of macrophyte control (e.g. mechanical harvesting), in most cases herbicides do not appear to be any worse, and in some cases are less damaging to the environment and more cost effective (Breckels & Kilgour, 2018; Hussner et al., 2017; Rydell, 2018).

Herbicides can also be to some extent species-specific. For instance, 2,4-D (2,4-dichlorophenoxyacetic acid) can target non-native broadleaved plants such as *M. spicatum*, while avoiding harm to other species such as plants from the *Potamogeton* genus (Helsel et al., 1996; Netherland & Willey, 2017).

Nevertheless, the efficiency of this treatment can also be a cause for concern, since it can produce longer lasting control that alters aquatic environments for a longer period of time (Barrett et al., 1999). Their effectiveness against macrophytes can cause large masses of plant death and rapid decay, ultimately causing deoxygenation of affected areas of the lake and potential mortality of various aquatic organisms (Carpenter & Greenlee, 1981). Certain surfactants used to increase herbicide efficiency could possibly be more toxic than the herbicides themselves and should also be considered during application (Breckels & Kilgour, 2018). Furthermore, the decaying macrophytes could also release stored nutrients (e.g. phosphorus) that may promote rapid regrowth of resistant macrophytes (Sesin et al., 2018) or accelerate lake eutrophication due to an increase in algal abundance (Carpenter & Adams, 1978). Though, similar to applications of benthic barriers, these consequences could be mitigated by dosing the herbicide during cooler periods of the year (Barrett et al., 1999; Gunnison & Barko, 1992).

While some of these management techniques can have their advantages, they all pose various drawbacks such lack of previous research, higher treatment costs, scaling limits or environmental risks. Though the main reasoning behind avoiding these practices and studying the impacts of burlap barrier applications and mechanical harvesting in eastern Ontario waters was due to their known use in Ontario and the Rideau Canal system (e.g., due to strict regulations for aquatic herbicides in Canada) (Ontario, 2014).

2.6.0 Study Region

Eastern Ontario is a region within the Canadian province of Ontario located between the Ottawa River and the St. Lawrence River. This eastern-most part of the province shares boundaries with the Canadian province of Quebec, and New York State, United States of America. The area is generally a flat plain consisting of farmland, but is scattered with large areas of forests, lakes, and wetlands. The region between the cities of Ottawa and Kingston includes three main drainage basins (Mississippi, Rideau and Cataraqui watersheds) that feature many lakes and rivers. Within this area, a 202 km long canal system named the Rideau Canal National Historic Site of Canada connects Ottawa to Kingston through a series of locks and structures that connect sections of the Rideau and Cataraqui River, and several lakes (Government of Canada, 2021b).

Within the Cataraqui Watershed, the canal passes through multiple lakes including Colonel By Lake, Cranberry Lake, Whitefish Lake, Sand Lake, Opinicon Lake, Indian Lake, and Newboro Lake. At Newboro Lake, the canal then crosses the border between the Rideau and Cataraqui watersheds and enters Upper Rideau Lake. From here, the waterway continues through the Rideau watershed incorporating the Upper, Big, and Lower Rideau lakes. From the city of Smiths Falls, the canal follows the Rideau River all the way to Ottawa where it eventually separates from the Rideau River once it reaches Hog's Back Falls. The canal continues north, passes through Dow's Lake, and eventually connects to the Ottawa River. The interconnected series of lakes, rivers, and artificial canals acts as an important economic, environmental and recreational resource utilized by boaters, businesses, residents, and vacationers (Government of Canada, 2021c).

The creation of the Rideau Canal has increased connectivity between eastern Ontario waters by linking many lakes and rivers throughout both the Rideau and Cataraqui watersheds. While the canal served as an alternate shipping route between Montreal and Kingston during fears of an American invasion (Government of Canada, 2021b), today it is mainly used and navigated by pleasure crafts. By increasing interconnectivity between these inland waters, the waterway now connects previously disconnected habitats, allowing both native and invasive species to pass through the locks and dams and enter new habitats. This, as well as increased traffic between lakes from recreational boats, increases the potential spread of invasive species throughout aquatic ecosystems in eastern Ontario.

Currently, the Invasive Species Act enforces rules in Ontario to help prevent and control the spread of invasive species by prohibiting and restricting a number of invasive species that pose a risk to Ontario's natural environment (Ontario, 2021), as well as the federal *Fisheries Act* and its associated Aquatic Invasive Species Regulations that apply to federal waters, including the Rideau Canal. There are also multiple ongoing initiatives, projects, and programs that have similar objectives for the Rideau waterway.

Within the Rideau Canal, Parks Canada actively harvests excessive macrophyte growth along the navigation channel using its own mechanical harvesters and other equipment. They currently operate an aquatic plant removal program at 20 different locations along the canal (Government of Canada, 2021a).

If any individual or group is interested in removing aquatic plants (native or invasive) in Ontario, they may remove invasive aquatic plants such as *M. spicatum* by mechanical means or by hand without a permit if they follow all the rules for removing aquatic

invasive plants in accordance with the Regulations prescribed in O. Reg. 239/13 under the Public Lands Act. If any of these rules cannot be met, individuals or groups will need a work permit issued by the Ontario Ministry of Natural Resources and Forestry. If the area falls within federal lands or waterbodies, such as the Rideau Canal National Historic Site, authorization is required from Parks Canada for any plant removal activity in these waters. Fisheries and Oceans Canada might also be involved in the authorization, if the area is important fish habitat or is listed as critical habitat for species at risk under the federal *Species at Risk Act* (Simkovic, 2020).

Parks Canada does issue permits for waterfront owners who wish to remove any aquatic vegetation within the Rideau Canal waterways in front of their property up to a maximum of a 300 m² area (10 m x 30 m). To obtain this permit to remove macrophytes, a waterfront owner must strictly follow certain rules and conditions, including only using mechanical devices or removing plants by hand (Ontario, 2017). For herbicides, their use is strictly prohibited within the Rideau Canal. However, their use for preventing the introduction or spread of, or for controlling or eradicating invasive aquatic plants may be authorized by Fisheries and Oceans Canada, Parks Canada, or the Ontario Ministry of Natural Resources and Forestry according to the federal Aquatic Invasive Species Regulations, though you would also need the approval from the Ontario Ministry of the Environment, Conservation and Parks (MECP) (Simkovic, 2020).

Macrophyte control methods such as placing benthic barriers do not fall under the provincial rules for removing invasive or native macrophytes in Ontario. Projects on Provincial Crown Land require the Ministry of Natural Resources and Forestry approval

under the Public Lands Act. Benthic barriers are also not permitted by Parks Canada in the Rideau Canal.

This thesis will investigate the effectiveness and impacts of two control techniques for *M. spicatum*. Part of my thesis will focus on a lake within the Mississippi watershed (Malcolm Lake) to study the physical, chemical, and biological impacts of applying burlap benthic barriers to large beds of *M. spicatum* and will help determine if this management practice is a viable solution for invasive aquatic plant management in Ontario waters. The other project in my thesis will focus on three lakes within the Rideau waterway with different trophic levels and will study the physical, chemical, and biological impacts of applying Parks Canada's most common technique (mechanical harvesting) for controlling excessive macrophyte growth in the Rideau Canal. Studying the impacts from both macrophyte control methods will help develop improved management strategies for invasive macrophyte growth in Canadian waters.

Chapter 3.0 - Methods

3.1.0 Study Sites

3.1.1. Malcolm Lake

Malcolm Lake was chosen to study the impacts of covering large *M. spicatum* patches by the application of benthic barriers. Malcolm Lake is located on the Canadian shield in Frontenac County, Ontario, Canada (Figure 4) within the township of North Frontenac and is situated approximately 1.2 km South of the town to of Ardoch (Figure 4). Malcolm Lake is a shallow (mean depth = 2.2 m), oligotrophic to mesotrophic lake (TP = 0.004 - 0.0195 mg/L) with a total surface area of 2.2 km² (Malcolm/Ardoch Lake Stewardship Committee, 2016). The lake is at an elevation of 253 meters above sea level and supports a warm water fishery. As of 2015, there were 107 properties taking up approximately 75% of the shoreline, with the remainder of the shoreline mostly being vacant property or crown land (Malcolm/Ardoch Lake Stewardship Committee, 2016). To control the water levels for recreational use, a concrete dam was built by the Ministry of Natural Resources and Forestry on crown land at the outlet of the lake in 1969; this dam was eventually taken over by the Mississippi Valley Conservation Authority (MVCA) in 1986 (Malcolm/Ardoch Lake Stewardship Committee, 2016). Since then, MVCA also periodically conducts water quality monitoring of the lake through their watershed watch program. A volunteer-based lake association named Malcolm and Ardoch Lakes Landowners Association (MALLA) also helps manage the lake with a stewardship plan. The sampling location for this thesis is situated in and around one of

the larger *M. spicatum* patches in the lake, located along the northeastern shore (44°55'12"N 76°53'21"W) (Figure 4C).

The Lakes Stewardship Committee began tracking the growth of invasive macrophytes in 2015 and has attempted to address the *M. spicatum* invasion in Malcolm Lake due to growing fears of its rapid spread. This has led the Stewardship Committee to develop a Eurasian Watermilfoil Management Plan. The goal of the plan is to provide long-term management and control of *M. spicatum* growth, obtaining government funding, preventing further spread, monitoring existing populations, and implementing management/control of *M. spicatum* (MALLA, 2018). Thus, part of the thesis was conducted with MALLA to help the association reach their management plan goals by implementing a burlap benthic barrier regime on *M. spicatum*. This helped MALLA determine if burlap benthic barriers are an effective treatment for *M. spicatum* management within Malcolm Lake, which could in turn help lake users and managers across much of Canada better develop management plans for *M. spicatum* using this treatment.

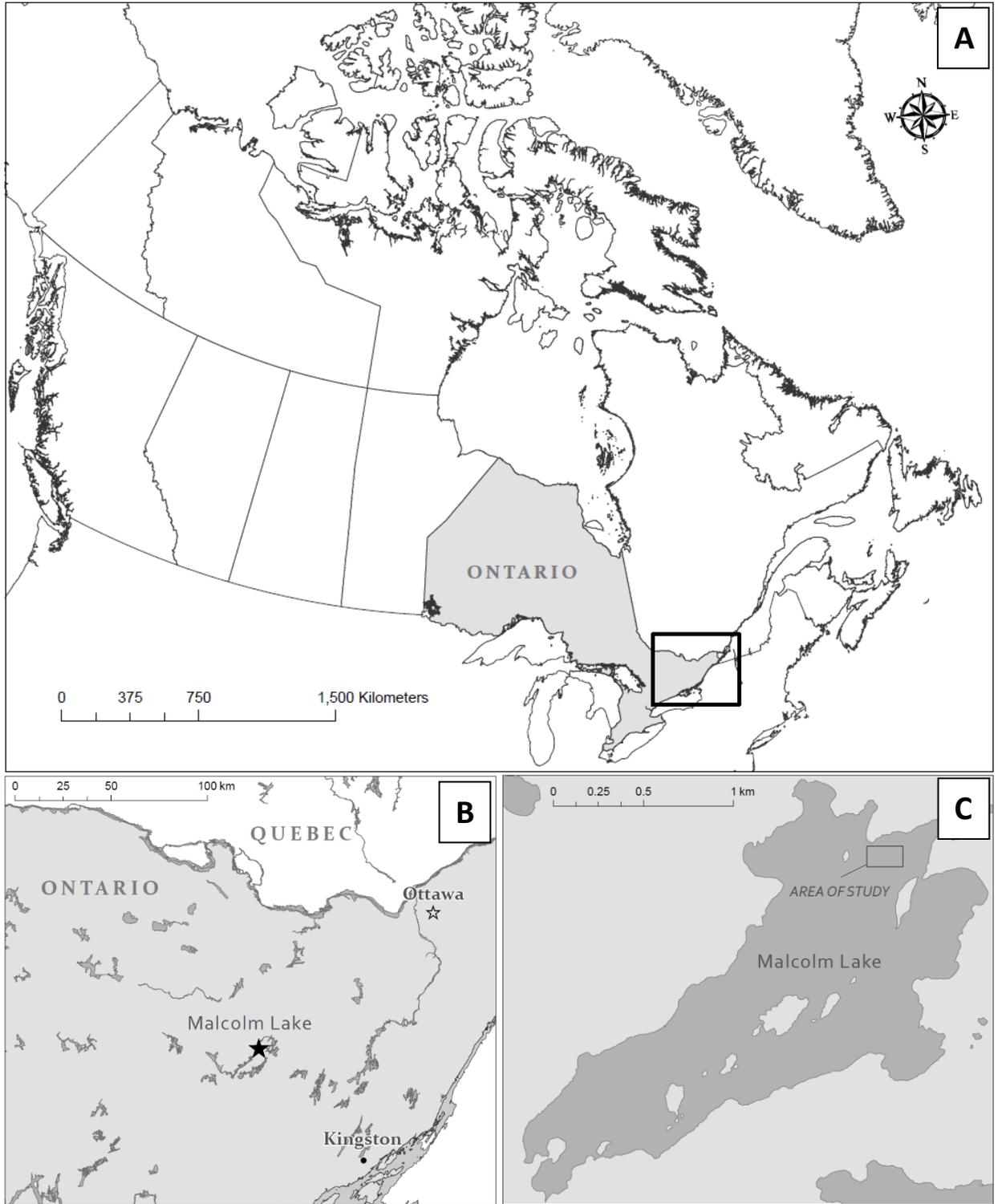


Figure 4. Map showing the location of Malcolm Lake (44°55'N, 76°55'W) within Canada (A) and Ontario (B). The study site was located in a single large stand of *M. spicatum* in the northeastern corner of the lake (C).

3.1.2. Rideau Lakes

Three lakes located in both the North-flowing Rideau River watershed and the South-flowing Cataraqui River Watershed were selected to investigate the impacts of mechanical macrophyte harvesting on *M. spicatum* beds. Large macrophyte beds nearshore were first confirmed on site for each lake. Based on historical data collected by local conservation authorities (Laird, 2017; Rideau Valley Conservation Authority, 2015; Township of Rideau Lakes, 2021), the studied lakes were selected along a TP gradient. The first lake, Wolfe Lake, is oligotrophic (TP < 0.012 mg/L), Otter Lake is mesotrophic (TP = 0.012-0.024 mg/L), and Dog Lake is eutrophic (TP = 0.024-0.096 mg/L) (Carlson & Simpson, 1996) (Figure 5).

Wolfe Lake is located about 5 km west from the town of Westport, within the Municipality of South Frontenac, Ontario, Canada (44°41'49"N; 76°29'17"W). The underlying geology of the lake is dominated by sandstone and gravel (Hollingsworth et al. 2012), and the lake has a total surface area of 9.5 km². The lake has an elevation of 135 m above sea level, a maximum depth of 31.1 m, and a mean depth of 10.3 m. Managed by the RVCA and the Wolfe Lake (Westport) Association, the lake also contains suitable habitat conditions for warm water fish species (Angler's Atlas, 2021) and contains an average TP concentration of 0.011 mg/L (Rideau Valley Conservation Authority, 2014). According to RVCA (2014), zebra mussels have colonized the lake and may have brought clearer waters after 2004 – 2006.

Otter Lake is 5.3 km long, with a surface area of 6.9 km², a maximum depth of 36 m, and a mean depth of 10.0 m. It has an elevation of 124 m and is located about 17 km South-West of the town of Smiths Falls (44°41'49"N; 76°29'17"W). According to the

Otter Lake Landownership Association (OLLA) (2021) and RVCA (2015), the lake is positioned in an area where a limestone plain intersects with the pre-Cambrian Shield, making the underlying geology sandstone, dolostone, igneous, and high-grade metamorphic rocks. Managed by the RVCA and the Stewardship plan from OLLA, Otter Lake contains suitable habitat conditions for warm water fish species and, according to past monitoring data, the lake is mesotrophic most years (OLLA, 2020), with an average TP concentration of 0.013 mg/L (Citizens Statewide Lake Assessment Program, 2018). Otter Lake contains very clear waters, which is likely partially attributed to the establishment of zebra mussels that have been reported since 2001 (RVCA, 2015).

Dog Lake is about 20km North-East of Kingston situated between Battersea and Brewer Mills in the Frontenac County, Ontario, Canada (44°26'01"N; 76°21'30"W). Dog Lake has a surface area of 9.6 km², a maximum depth of 50.0 m (Ecottagefilms, 2021), and a mean depth of 5.8 m (Catarqui Region Conservation Authority, 2017). The underlying geology of the region is granite and sandstone based (Hollingsworth et al., 2012), and the lake hosts a high diversity of fish species especially with its bathymetry-containing portions of deep-water depths. This bathymetry allows the lake to contain suitable habitat conditions for warm and cold-water species such as North American Catfish, sunfish, Pikes, Suckers, trout, and more (Catarqui Region Conservation Authority, 2017). According to Catarqui Region Conservation Authority (2017), the lake contains an average Secchi disc depth that is less than 3 m, as well as an average TP concentration of 0.036 mg/L. The lake is managed by the Catarqui River Conservation Authority (CRCA) and the Dog Lake Association. There have been reports of an establishment of zebra mussels, as well as large establishments of *M. spicatum*, which are

now the predominated growth of aquatic vegetation within the lake (Catarraqui Region Conservation Authority, 2017; Wall, 2021).

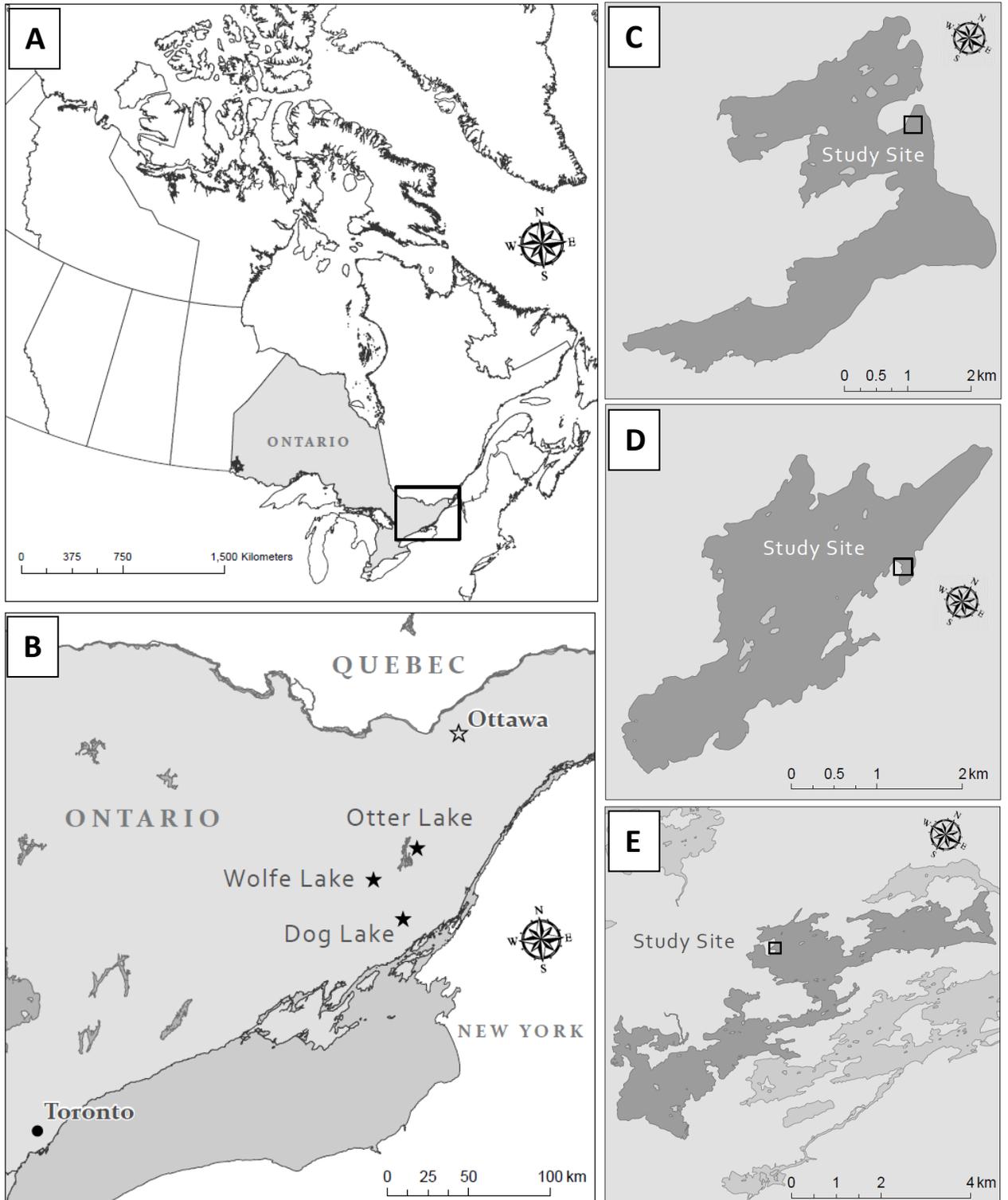


Figure 5. Map showing the location of Wolfe Lake (44°55'N, 76°55'W), Otter Lake (44°55'N, 76°55'W), and Dog Lake (44°55'N, 76°55'W) within North America (A) and Ontario (B). The study site for each lake (C Wolfe, D Otter and E Dog) was located in a single large stand of *M. spicatum* where the water depth was approximately 2 m.

3.2.0 Materials and Methods

3.2.1 Study Design

To detect any potential environmental impacts of both macrophyte management techniques, I used a before-after-control-impact (BACI) study design (E. P. Smith, 2006). The design required the selection of a ‘control’ site and an ‘impact’ site near each other within a study area of interest. The control site for all the lakes was designated ‘milfoil’, while the impact site for Malcolm Lake was designated ‘burlap’, and the impact sites for the Rideau Lakes were designated ‘harvested’. Since the goal of my thesis was to determine the ecological impacts of controlling large dense patches of *M. spicatum*, the lake’s single largest dense stand of the invasive species, at a water depth of approximately 2 m, was first chosen for all the lakes. Measurements of physical, chemical, and biological parameters were then collected ‘before’ and ‘after’ the barrier application date of June 20th, 2019, for the burlap barriers, and September 7th for the mechanical removal treatment.

With the help of volunteers from MALLA, the benthic barrier treatment consisted of laying down a 46 m x 2.5 m burlap barrier over the selected macrophyte patch using a dispensing barge and personal watercraft. A third sample site designated ‘bare’ was also selected in an adjacent macrophyte-free area of similar depth (2 m) for Malcolm Lake to compare a bare sediment patch to the burlap site. Due to the timing of the burlap application by the lake association, only one set of samples prior to the treatment could be obtained (June 8th, 2019). Five sampling periods after the treatment date were obtained: three sampling periods for 2019 (July 3rd, August 1st, and September 14th), and two for the fall of 2020 (September 11th and September 20th).

As for the macrophyte harvesting treatments for the remaining lakes, one 5 m by 20 m strip parallel to shore of each lake was harvested using Rideau Aquatic Weed Control's weed harvester barge. Plant biomass was cut at a depth of around 2 m, as recommended by the operator of the harvester, and the cuttings were removed from the lakes for offshore composting, using the conveyor/sickle bar mower of the harvester barge. Two sampling periods were undertaken 'before' the treatment date in 2019 (July 9-10th and August 7-8th), two 'after' during the same year (September 12-13th and October 3-4th), and two 'after' during the fall of 2020 (September 15-16th, and September 23rd-October 5th).

While the treatment plot sizes used for both types of macrophyte treatments in this study are relatively small, they are representative of a typical treatment size that would be carried out by lake users and lake managers to help control *M. spicatum* in Ontario. Due to strict regulations on macrophyte control for inland waters, experimenting on treatment plots of this size is in truth more representative of macrophyte management that can generally be conducted within the province. It is for these reasons that the small treatment plot sizes were chosen and performed during this study.

3.2.2 Field sampling

In 2019, field sampling occurred monthly from June to September for Malcolm Lake, and July to October for the three Rideau Lakes. Originally, this sampling schedule would have remained similar the following year (2020); however, due to COVID-19 restrictions, 2020 sampling could only resume during the fall. All samples were collected using a 14-foot aluminum boat and took place between 10 am and 2 pm. In all cases for each of the sites ('milfoil', 'burlap', and 'bare' for Malcolm Lake, and 'milfoil' and

'harvested' for the Rideau Lakes), water samples were collected at a water depth of 0.5 m using triple-rinsed 4 L plastic containers and a Van Dorn sampler. At each site, triplicate zooplankton samples were collected at a water depth of 0.5 m, by collecting 24 L of water using a Van Dorn and filtering that water through an 80 μm Wisconsin Plankton Sampler. The filtrate was transferred to triple-rinsed 250 mL plastic containers, and 40 mL of 95% ethanol was added to preserve the samples (Black & Dodson, 2003).

Temperature ($^{\circ}\text{C}$), dissolved oxygen (DO) (%), pH, and conductivity (μS) were also measured just below the water surface, and also at 1 m and 2 m (near the sediment surface) water depths using a YSI ProPlus Multiparameter Instrument in triplicate for each site. Only YSI measurements taken near the sediment surface were used during statistical analysis, because while water exchanges with the pelagic area or adjacent sites (harvested, milfoil and bare) may still occur, comparing measurements at this depth is a closer representation of changes in environmental properties within the plant beds after the induced treatment. Samples were placed in a cooler containing ice packs, and then transferred to a refrigerator for short-term storage upon return.

Macrophyte density in both 'milfoil and 'burlap' sites for the lakes within the Rideau and Cataraqui River watersheds were estimated and recorded two weeks after the treatments, and during the fall of 2020 for all lakes. These estimations were done using an Echosounder side scan mosaics where the Percent Volume of Inhabited of the plants (PVI) was measured before and after the mechanical harvesting treatments. More specifically, the estimations were performed using Echosounder sweeps from a Lowrance Elite 7 Fishfinder/GPS, and using ReefMaster (ReefMaster Software Ltd., 2021) and Fiji (Schindelin et al., 2012) software to analyse the data. Echosounder scans were not

conducted at Malcolm Lake, since the burlap benthic barrier treatment completely covered the plants, bringing the PVI from approximately 100% to 0%. Macrophyte species composition at every lake was also recorded by observing the plant beds and identifying the species present using video recordings that were taken within the plant beds and treated sites. Before the treatments, each site was almost completely covered in *M. spicatum* with only the occasional native broad-leaved *Potamogeton* species appearing within the dense stands.

3.2.3 Laboratory analysis

Water samples were sent to the City of Ottawa laboratory facilities at the Robert O. Pickard Environmental Centre to determine Total Kjeldahl Nitrogen (TN) and Total Phosphorus (TP) measurements from acid digested samples using a Skalar segmented flow autoanalyzer.

Using an 80-micron mesh, zooplankton samples were filtered into 20 mL scintillation vials using a wash bottle containing distilled water. The samples were then transferred into a Bogorov counting chamber to enumerate and identify zooplankton families using a Leica S9i Digital Stereo Microscope. Identification was to family level where possible, using Haney's (2013) image-based key of north American zooplankton. Since the intended purpose of the zooplankton samples were to only monitor and study their generalized responses, a higher taxonomic resolution was not selected, preventing the need for further species-level identification requirements (Bailey et al., 2001; Bozzuto & Blanckenhorn, 2017; Olsgard & Somerfield, 2000).

3.2.4 Statistical analysis

All statistical analyses were performed in R version 3.6.1 (R Core Team, 2020). Analysis of similarities (ANOSIM) were performed using the ‘vegan’ package (Oksanen et al., 2019) on all zooplankton taxa data to test whether there was a significant difference in zooplankton communities between milfoil and burlap or milfoil and harvested sites after the macrophyte treatments for all the lakes.

A Principal Component Analysis (PCA) ordinations was also performed on the zooplankton data using the same ‘vegan’ package, which was first Hellinger transformed to reduce the impact of dominant taxa and to give lower weight to rare families (Legendre & Gallagher, 2001). This specific data transformation was chosen for these analyses as it is appropriate for species community data and is often preferred over other data transformations (Vaughan & Ormerod, 2005). Zooplankton families with low abundances are often removed during methods based on Euclidean distances such as PCAs, because the many zeros introduced by including these rare individuals can be troublesome for the analysis (Blanchet et al., 2014). Therefore, taxa containing less than three occurrences throughout the datasets were not included in the ordinations to further reduce any undue influence on the analyses (Legendre & Gallagher, 2001; Legendre & Legendre, 1998; Rao, 1995). For data from Malcolm Lake, two different PCAs were conducted: (1) data from 2020 only; (2) all collected data (from 2019 and 2020). This allowed me to compare differences in any longer-term impacts that did not develop immediately, and the overall impact on zooplankton community structure a year after the treatment.

Redundancy analyses (RDA) were performed on the zooplankton and abiotic data for the Rideau Lakes to assess which variables (Temperature, pH, TP, TN, DO and Conductivity) were most important in structuring zooplankton assemblages using the *vegan* package in R (Oksanen et al., 2019). Since this analysis is also a linear ordination method, taxa containing less than three appearances throughout the datasets were not included in the ordinations. The zooplankton data was similarly Hellinger transformed, and a multicollinearity test was used by verifying the variance inflation factors (VIF) of the explanatory variables. The multicollinearity test was applied to determine which collinear explanatory environmental variables should be removed from the analysis (Kassambara, 2018). During the Redundancy analysis, only the explanatory variables that showed statistical significance were kept and presented in the results.

Using RDA ordinations, the variation in the zooplankton community data matrix was then partitioned by three explanatory variables (environment, treatment, and time) using the partitioning function from the same ‘*vegan*’ package in R. The environmental variation grouped all the explanatory environmental variables that were considered statistically significant during the first set of RDAs. The spatial variation composed of treated vs. non-treated sites, while the temporal variation simply compared sampling dates. This test was used to determine the fraction of variance in zooplankton community structure that could be explained by each individual partitioned explanatory variance as well as the overlapping variance explained by the explanatory variables (Borcard et al., 1992).

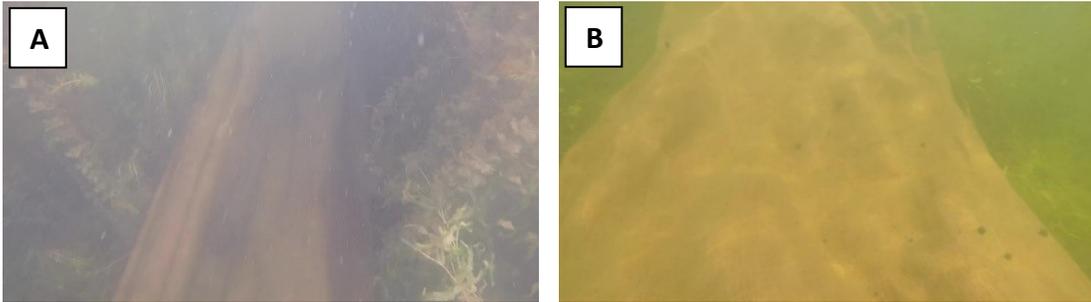
Chapter 4.0 - Results

4.1.0 Malcolm Lake

At the end of the summer in 2019, the burlap was intact and was covered in a thin layer of periphyton (Figure 6A, B). During the 2020 sampling season, the benthic barriers were mostly intact; however, in some areas the burlap material showed substantial degradation (Figure 6C). Macrophytes were also observed growing through the burlap in some parts of the barriers. Some areas within the barrier where macrophytes started to grow through contained up to 30-40% plant cover but these plants had grown no more than half the height and density of the surrounding *Myriophyllum* patches.

Approximately 60% of the macrophyte stems that had grown through the barrier belonged to the native macrophyte from the *Potamogeton* genus (Figure 5E, F), while 35% of stems were *M. spicatum* (Figure 6D). In the sections where the burlap suffered major degradation, the ground was no longer covered by the barrier and appeared to mostly be colonized by a grass-like aquatic plant in the Isoetaceae family (likely *Isoetes lacustris* L.) (Figure 6C).

Summer 2019



Fall 2020

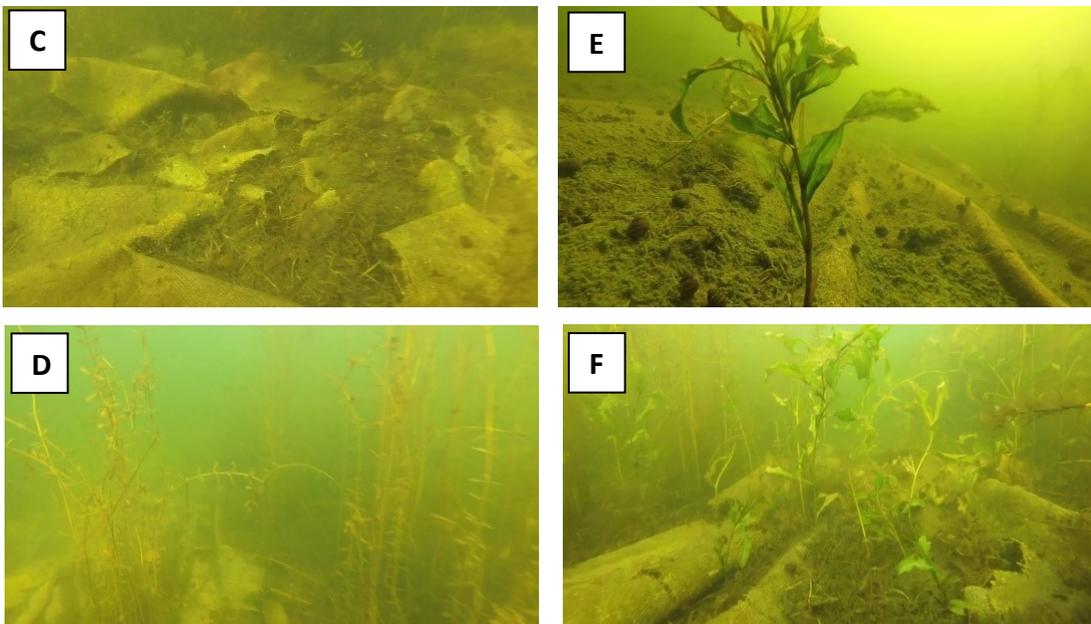


Figure 6. The condition of the burlap benthic barriers during the summer of 2019 (A, B) and the fall of 2020 (C, D, E, F)

For all the data collected from 2019 to 2020, there were no clear trends for any of the abiotic variables when comparing the three sites (milfoil, burlap, and bare) after the applying the treatment (Figure 7), except for the Shannon index, which displayed greater zooplankton family diversity at the milfoil site (Figure 7H). Certain individual zooplankton taxa also demonstrated some affinity towards the milfoil site such as Chydoridae (Figure 8A), Sididae (Figure 8C), and Ostracoda (Figure 8E).

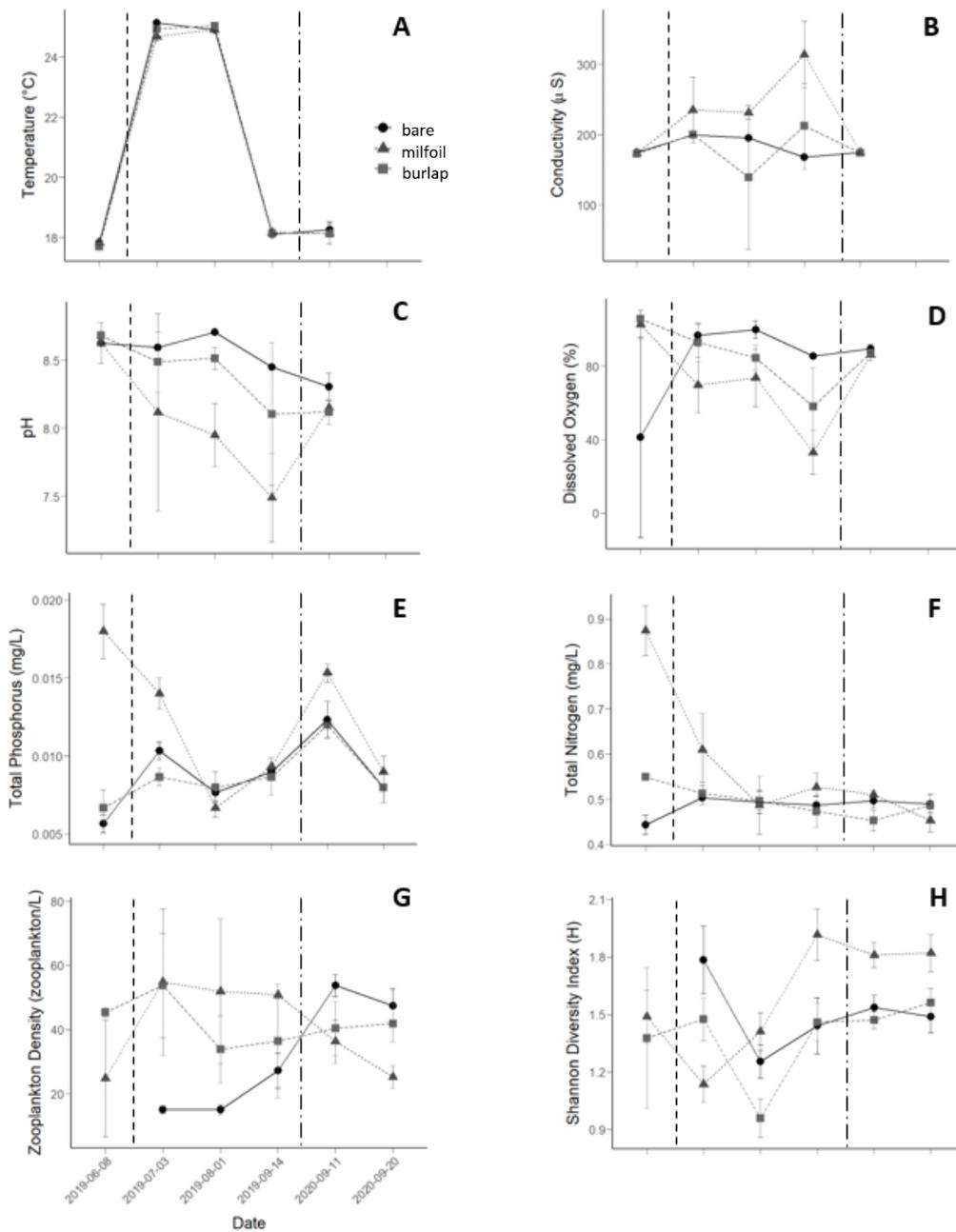


Figure 7. The physical (A temperature), chemical (B conductivity, C pH, D dissolved oxygen, E total phosphorus, F total Kjeldahl nitrogen), and biological (G zooplankton density, I Shannon index) parameters measured by sampling time (June 8th, July 3rd, August 1st and September 14th 2019, September 11th and 20th 2020). The first dotted vertical line represents the treatment date (June 20th, 2019), and the second dotted vertical line represents the division between 2019 and 2020. Black circles represent samples taken from the milfoil site (within the macrophyte stands), dark grey triangles represent samples taken from the burlap site (within the macrophyte bed ‘trenches’ formed by burlap mats) and light gray squares represent samples taken from the bare site (open water region adjacent to macrophyte bed). The points represent mean values (n=3) and the error bars standard error.

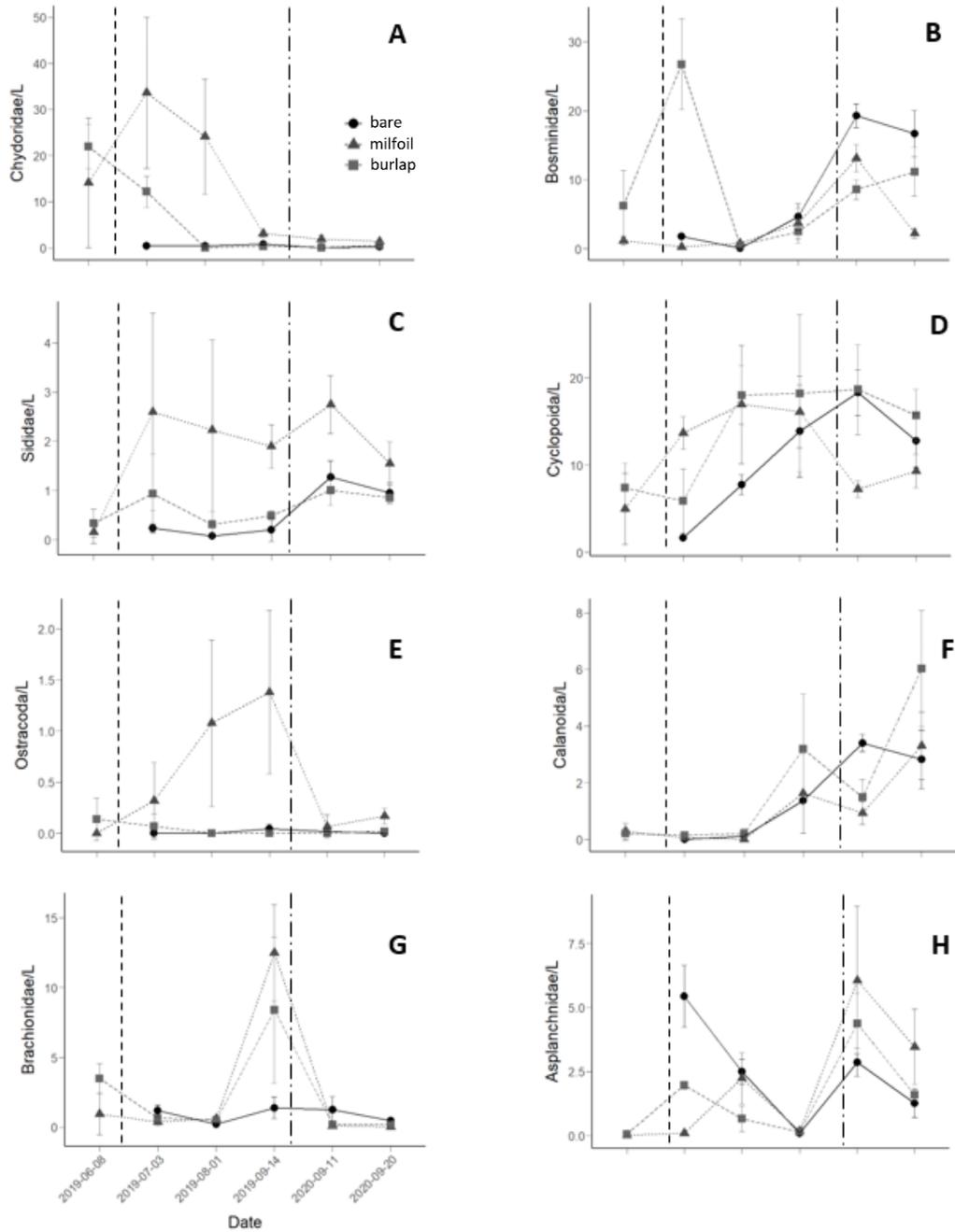


Figure 8. The individual zooplankton taxa measured per Liter of sample by sampling time (June 8th, July 3rd, August 1st and September 14th, 2019, September 11th and 20th 2020). The first dotted vertical line represents the treatment date (June 20th, 2019), and the second dotted vertical line represents the division between 2019 and 2020. Black circles represent samples taken from the milfoil site (within the macrophyte stands), dark grey triangles represent samples taken from the burlap site (within the macrophyte bed ‘trenches’ formed by burlap mats) and light gray squares represent samples taken from the bare site (open water region adjacent to macrophyte bed). The points represent mean values (n=3) and the error bars standard error.

4.1.1 Principal Component Analysis

In the zooplankton ordination by Principal Component Analysis that only includes data from 2020 (Figure 9. A.), the first principal component axis explains 45.4% of the observed variance, while the second axis explained 24.3%. The ordination clearly presented distinct clustering among the different sites. The burlap and bare sites had distinct ellipses, yet were grouped near one another along the first principal component (PC1) of the ordination, while the milfoil site contained a wider clustering along PC2. Most families of zooplankton also appeared to tend towards one of these site aggregations. Many large Cladoceran families such as Chydoridae, Sididae, *Daphnia* and Macrothricidae, as well as other zooplankton groups such as Ostracoda and Asplanchnidae, were assembling within the milfoil site's ellipse. The burlap site appeared to be mainly attracting copepods as both Cyclopoida and copepod nauplii were mostly grouping near its ellipse, with Calanoida also grouping nearby as well. The bare sediment site appeared to be mostly attracting Brachionidae rotifers and may have also explained a portion of *Bosmina*'s variance, as it is near the ellipse. However, the species' variance for *Bosmina* also appeared to be influenced by other external factors other than sampling site.

The PCA that includes all data from 2019 and 2020, the first principal component axis explained 29.1% of the observed variance, while the second axis explained 15.3%. The data clustering in this ordination was less distinct than the first PCA; however, there was still general grouping of the zooplankton data according to their sampling site. The burlap and bare sediment sites no longer contained a distinct separation between the two sites since much of their data points overlapped with one another. The milfoil site was yet again a larger cluster of data with an ellipse that remained separate from the bare

sediment site. However, the ellipses of the burlap and milfoil sites now appeared to slightly overlap with each other. The same large Cladoceran families as well as Ostracoda once again primarily assembled inside the control site ellipse, with the addition of Hydrachnida and Chironomidae. Both overlapping impact and bare sediment sites looked as if they were mainly driven by Cyclopoida and copepod nauplii, with the addition of Asplanchnidae. Brachionidae. Polyphemidae and Conochilidae that fell within the impact site. Bosminidae and Calanoida were the only two families that fell outside of any ellipses. This may again suggest that the variance of *Bosmina* and Calanoida assemblages were being more strongly influenced by other external factors.

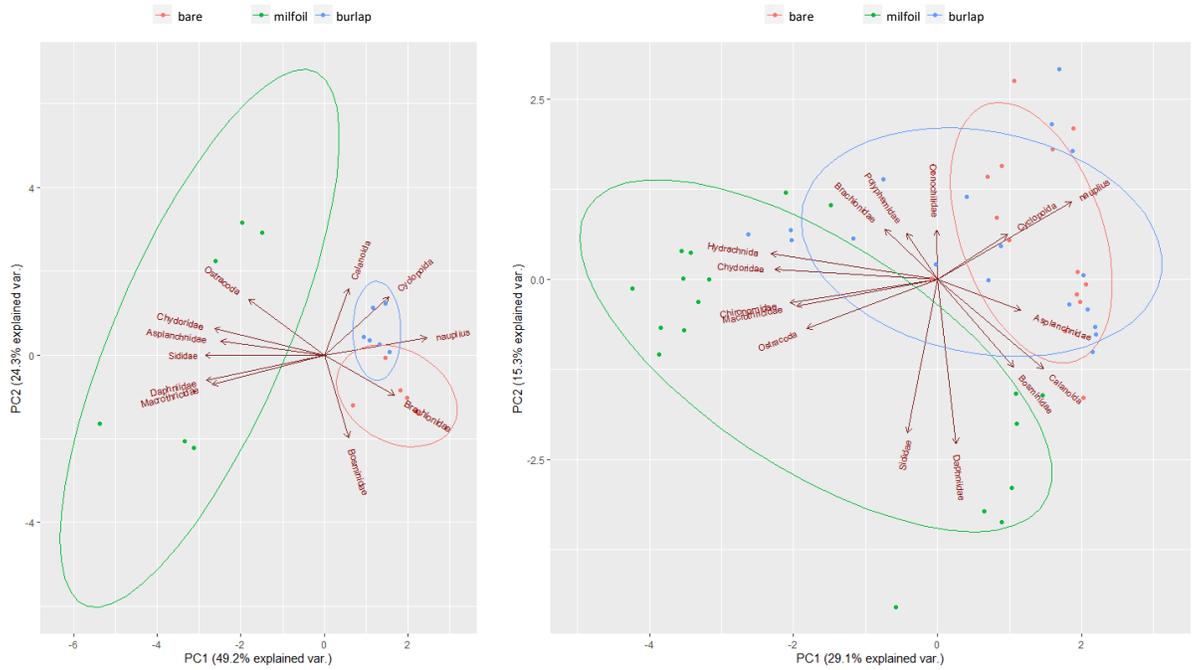


Figure 9. Principal Component Analysis (PCA) diagrams of the zooplankton taxa per Liter of sample data for Malcolm Lake. Left diagram represents data from 2020 only, while the right diagram represents all collected data (from 2019 and 2020). Taxa (identified to Family, where possible) were Hellinger transformed, and any containing less than 3 appearances throughout the dataset were dropped. Individual taxa are represented by arrows, red points represent ordinated ‘bare’ sites, green points represent ordinated ‘milfoil’ sites and blue points represent ordinated ‘burlap’ sites. Likewise, 68% confidence interval ellipses are drawn from the centroids of both groups, again using blue for ‘burlap’ and red for ‘milfoil.’ Percent variance explained for the first two PC axes are indicated in each plot.

4.1.2 ANOSIM

The Analysis of similarities (ANOSIM) on all the zooplankton taxa data for Malcolm Lake indicated there was a significant ($p=0.0002$) difference in zooplankton assemblages between both 'control' and 'impact' sites after the burlap barrier treatment. However, the ANOSIM R statistic that compared the mean of ranked dissimilarities between groups to the mean of ranked dissimilarities within groups was 0.1122, suggesting that while there was a significant difference between the 'control' and 'impact' groups after the burlap barrier treatment, the groups remained similar with their zooplankton assemblage.

4.1.3 Variation Partitioning

For Malcolm Lake, 51% of the overall zooplankton variation could be explained by three explanatory matrices. Of this variation, the environmental structure represented 34% of the overall zooplankton variation (Figure 10). This is the fraction of zooplankton variation that can be explained by the remaining significant environmental variables (*i.e.*, temperature, TP, TN, DO, conductivity), and includes shared variation with both spatial and temporal structures. This overall environmental variation within the zooplankton data had some overlapping variation that was shared by the temporal data representing 11% of the overall variability in zooplankton assemblages. Zooplankton variation explained by the environmental variables also overlapped with the variation that could be explained by the treatment data; however, this was a small portion of the total variation (2%). The environmental variables alone explained the largest single (independent of other explanatory variables) portion of known variance within the zooplankton family data (18%) (Figure 10).

The temporal variation had an overall explanatory power of about 22%. While the portion of independent temporal variation was quite small (4%), the 11% overlapping variation with the environmental structure was the largest shared variation between the three explanatory variables, suggesting that this was temporal changes in the environmental variables that can occur with seasonality for example. For the spatial variation, the partitioning analysis indicated that the overall spatial variation had an explanatory power of 19%, indicating that the macrophyte treatment had an influence over the zooplankton composition. Ten percent of the overall zooplankton variation could be explained by the macrophyte treatment alone, with small overlapping explanatory variation between the two other explanatory variables, indicating that the burlap treatment had some impact on zooplankton assemblages (Figure 10).

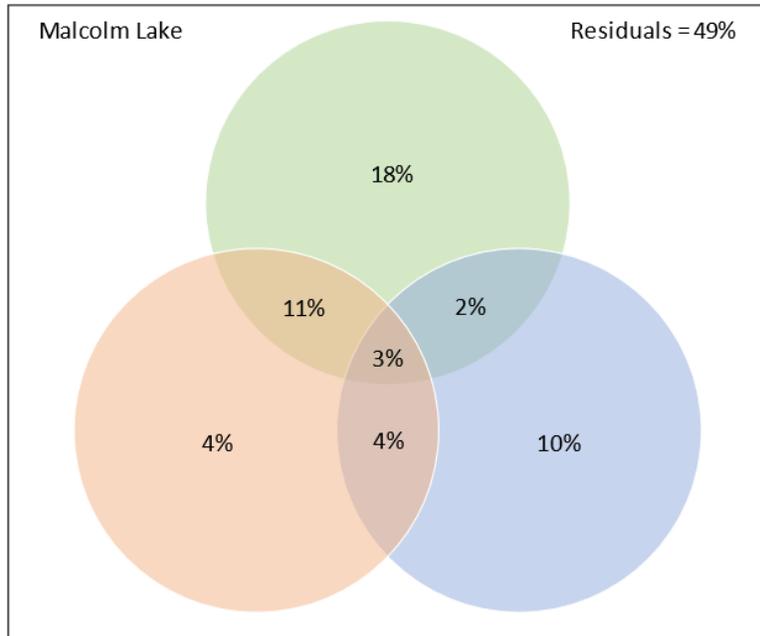


Figure 10. Venn diagram representing the variance partitioning of the zooplankton taxa data matrix. The data was partitioned into three explanatory variables: Environmental variables (green circle at the top), Date (pink circle to the left), and Treatment (blue circle to the right). The statistically significant environmental explanatory variables represent the overall environmental variation, the treatment (spatial) explanatory variable represent the variation explained by treated vs. non-treated sites, and the Date (temporal) explanatory variable represents the variation explained by seasonal change.

4.2.0 Rideau Lakes

The areas treated with the mechanical harvester were not completely devoid of plants following cutting. Some plants were pushed aside near the peripheral areas of the swathe, though these plants eventually bounced back. During the final months of sampling in 2019, the treated areas began showing some regrowth, which is unsurprising since *M. spicatum* is known to grow rapidly once present in a lake (Boylen et al., 1999). Still, the Percent Volume Inhabited (PVI) was reduced by 23%, 33% and 43% for Wolfe, Otter and Dog Lakes respectively, indicating that there were less plants at the impact sites

a month after their treatment. However, by the following year (Fall of 2020), the invasive macrophyte beds had almost completely regrown at all study sites.

For all the abiotic variables that were measured and collected from 2019 to 2020, none of lakes demonstrated any noticeable or significant difference between the control and impact sites after the mechanical treatments (Figure 11, 12, and 13). Zooplankton density and diversity also did not display any noticeable differences between both control and impacts sites. Additionally, ordinations by Principal Component Analysis for all three lakes did not clearly demonstrate any differences in zooplankton community structure when compared to the milfoil and burlap sites (Figure 14). However, the analyses did appear to show that the Bosminidae, Daphniidae, Chydoridae, and Calanoida all played a large part in explaining zooplankton community structure.

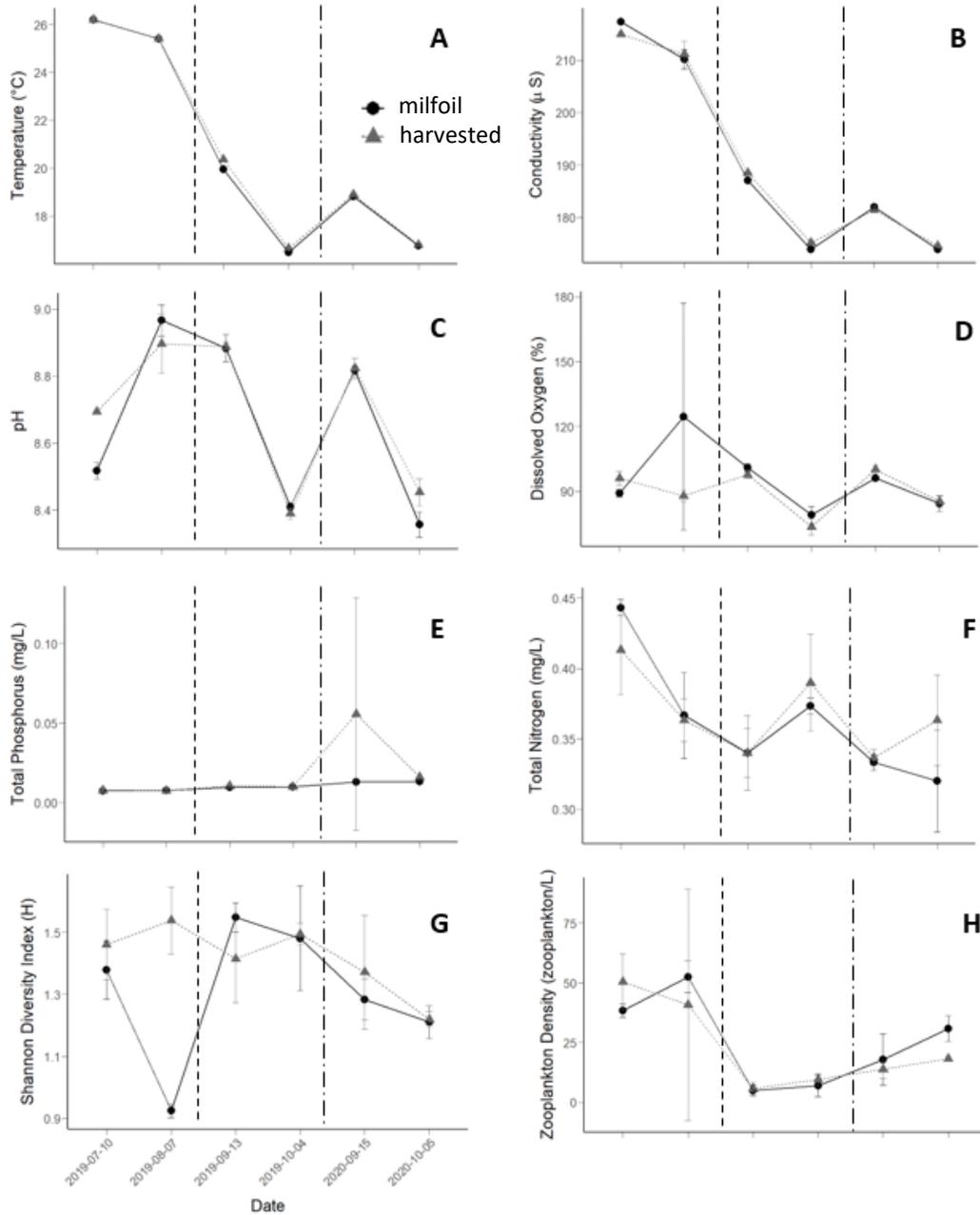


Figure 11. The physical (A temperature), chemical (B pH, C conductivity, D dissolved oxygen, E total phosphorus, F total Kjeldahl nitrogen), and biological (G zooplankton density, I Shannon index) parameters measured by sampling time (July 10th, August 7th, September 13th and October 4th 2019, September 15th and October 5th 2020) at Wolfe Lake. The first dotted vertical line represents the treatment date (September 7th, 2019), and the second dotted vertical line represents the division between 2019 and 2020. Black circles represent samples taken from the milfoil site (within the macrophyte stands), dark grey triangles represent samples taken from the burlap site (within the macrophyte bed ‘trenches’ formed by burlap mats) and light gray squares represent samples taken from the bare site (open water region adjacent to macrophyte bed). The points represent mean values (n=3) and the error bars standard error.

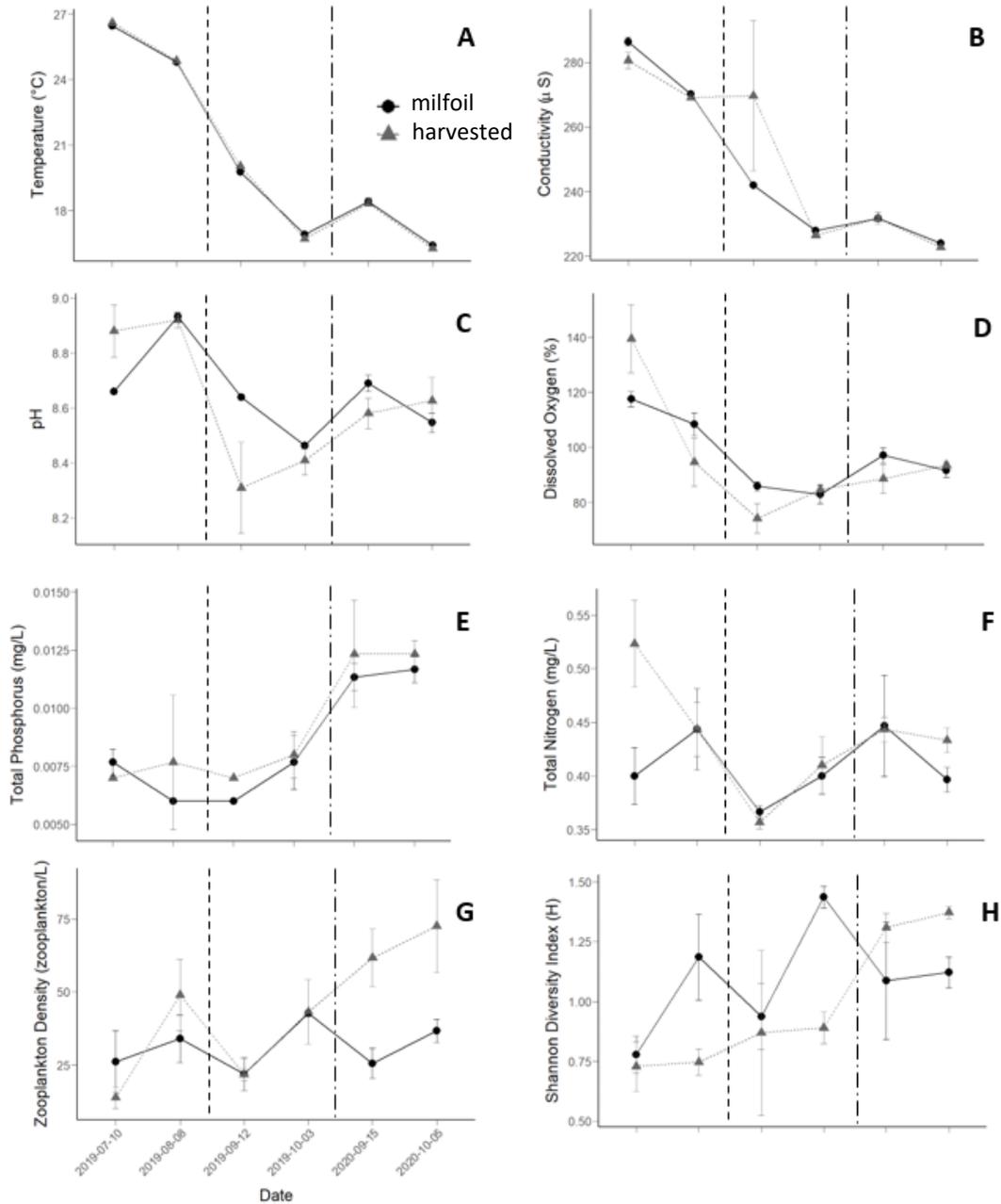


Figure 12. The physical (A temperature), chemical (B pH, C conductivity, D dissolved oxygen, E total phosphorus, F total Kjeldahl nitrogen), and biological (G zooplankton density, I Shannon index) parameters measured by sampling time (July 10th, August 8th, September 12th and October 3rd, 2019, September 15th and October 5th, 2020) at Otter Lake. The first dotted vertical line represents the treatment date (September 7th, 2019), and the second dotted vertical line represents the division between 2019 and 2020. Black circles represent samples taken from the milfoil site (within the macrophyte stands), dark grey triangles represent samples taken from the burlap site (within the macrophyte bed ‘trenches’ formed by burlap mats) and light gray squares represent samples taken from the bare site (open water region adjacent to macrophyte bed). The points represent mean values (n=3) and the error bars standard error.

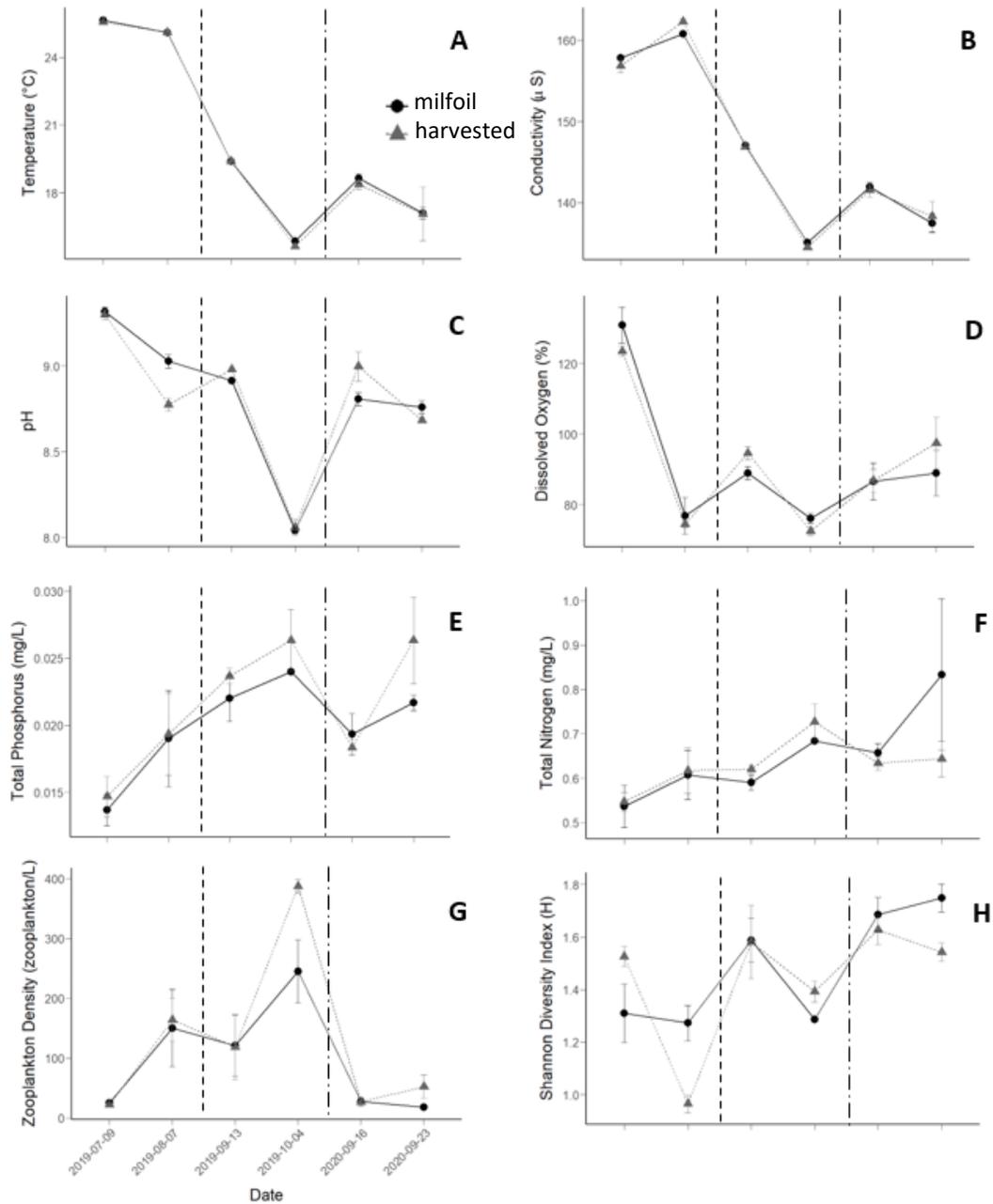


Figure 13. The physical (A temperature), chemical (B pH, C conductivity, D dissolved oxygen, E total phosphorus, F total Kjeldahl nitrogen), and biological (G zooplankton density, I Shannon index) parameters measured by sampling time (July 9th, August 7th, September 13th and October 4th 2019, September 16th and 23rd 2020) at Dog Lake. The first dotted vertical line represents the treatment date (September 7th, 2019), and the second dotted vertical line represents the division between 2019 and 2020. Black circles represent samples taken from the milfoil site (within the macrophyte stands), dark grey triangles represent samples taken from the burlap site (within the macrophyte bed ‘trenches’ formed by burlap mats) and light gray squares represent samples taken from the bare site (open water region adjacent to macrophyte bed). The points represent mean values (n=3) and the error bars standard error.

4.2.1 PCA

For the Wolfe Lake zooplankton ordination (Figure 14A), there are three significant PC axes which explained 27.8%, 20.8% and 16.1% of the observed variance. While the analysis had three significant principal components, only the first two components explaining the most variance were presented in the ordination. Both ellipses were closely centered to the origin, were large and circular, with considerably overlap. The taxa with the greatest loadings on the first PC axis were the Copepoda nauplii (0.4058) and the Brachionidae (-0.4058), whereas the taxa with the highest loadings on the second PC axis were the Bosminidae (-0.4744) and the Calanoida (-0.3854).

For the Otter Lake zooplankton ordination (Figure 14B), there was only one significant PC axis (PC1) explaining 26.7% of the observed variation. While the second PC axis was almost considered significant, it only explained 16.0% of the observed variance. The ellipse representing samples from the control sites was very large and circular and was closely centered to the origin. The ellipse representing the impact site was rather oval and small in size, was less centered to the origin, and appeared to have a slight alignment with the first PC axis. Yet, both ellipses were almost entirely overlapping. The taxa with the highest loadings on the first PC axis were Sididae the (-0.4199), the Calanoida (-0.4021) and the Daphniidae (-0.4009), whereas the taxa with the highest loadings on the second PC axis were the Chydoridae (0.5447) and the Macrothricidae (0.4625).

In the case of the Dog Lake zooplankton ordination (Figure 14C), there were two significant PC axes which explain 24.5% and 22.0% of the observed variance. Both ellipses were closely centered to the origin, while the axis of the control site ellipse had a

small alignment with the first PC axis and the axis of the impact ellipse was similarly aligned with the second PC axis. However, the ellipses were fairly large and circular and were considerably overlapped. The taxa with the highest loadings on the first PC axis were the Daphniidae (0.4681) and the Ostracoda (0.4206), whereas the taxa with the highest loadings on the second PC axis were the Bosminidae (-0.4848) and the Chydoridae (-0.3785).

4.2.2 ANOSIM

Analysis of similarities (ANOSIM) on all the zooplankton taxa data for Dog and Wolfe Lake, were not significant ($p= 0.5273$ and 0.2751) and their R values were close to 0. This indicates that there was no statistically significant difference in taxa assemblages between both milfoil and harvested sites after the mechanical treatment for either of these lakes. However, Otter Lake did have significantly different zooplankton assemblages between milfoil and harvested sites ($p = 0.0146$). Though the R value of the analysis was low 0.1731, suggesting that the difference in zooplankton taxa assemblages is relatively modest.

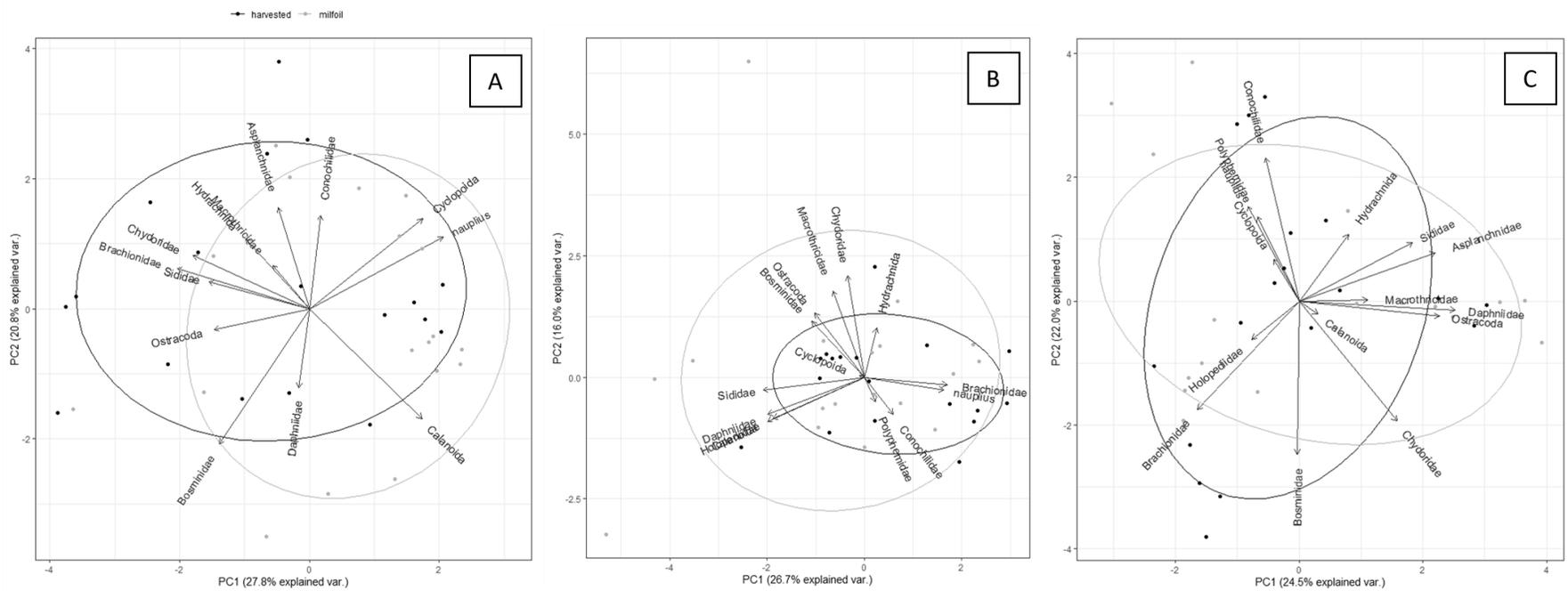


Figure 14. Principal Component Analysis (PCA) diagrams of the zooplankton taxa per Liter of sample data for Wolfe Lake (A), Otter Lake (B) and Dog Lake (C). Taxa (identified to Family, where possible) were Hellinger transformed, and any containing less than 3 appearances throughout the dataset were dropped. Individual taxa are represented by arrows, black points represent ordinated ‘harvested’ sites and grey points represent ordinated ‘milfoil’ sites. Likewise, 68% confidence interval ellipses are drawn from the centroids of both groups, again using blue for ‘burlap’ and red for ‘milfoil’. Percent variance explained for the first two PC axes are indicated in each plot.

4.2.3 RDA

Redundancy Analysis were performed for each lake to determine if any of environmental variables could explain the variation in zooplankton community structure. For Wolfe Lake, pH, Temperature and significantly ($p < 0.05$) explained some of the variance in the zooplankton assemblage, (Figure 15A) and amounted to a constrained variance of 36.76%. This ordination showed Bosminidae having a moderate negative correlation between temperature and DO. Calanoida were also exhibiting negative correlations with all three environmental variables, especially with temperature and pH.

Only temperature, TP and TN were significant in explaining the zooplankton assemblage in Otter Lake (Figure 15B). These three environmental variables explained a constrained variance of 44.79%, and the ordination suggested certain correlations with some zooplankton taxa. Bosminidae appeared to have a strong correlation with TP, and a negative correlation with temperature. Though these correlations for Bosminidae were weaker when compared to the RDA from Dog Lake. Calanoida shared similar associations to that of Bosminidae, as both taxa were closely ordinated. Copepoda nauplii appeared to be weakly correlated with Temperature, while Cyclopoida demonstrated a strong negative correlation with TN.

For Dog Lake, four explanatory variables were significant in explaining the zooplankton assemblage (Figure 15C). These explanatory variables were temperature, pH, TP and DO. When all these environmental variables were included in the RDA, the constrained variance amounted to 65.69% of the total variation in the zooplankton assemblage. Bosminidae were especially strongly correlated with TP. Bosminidae also appeared to have a strong negative correlation with temperature. Bosminidae and pH

were also strongly related. Copepoda nauplii had an association with DO, while Sididae was associated with temperature.

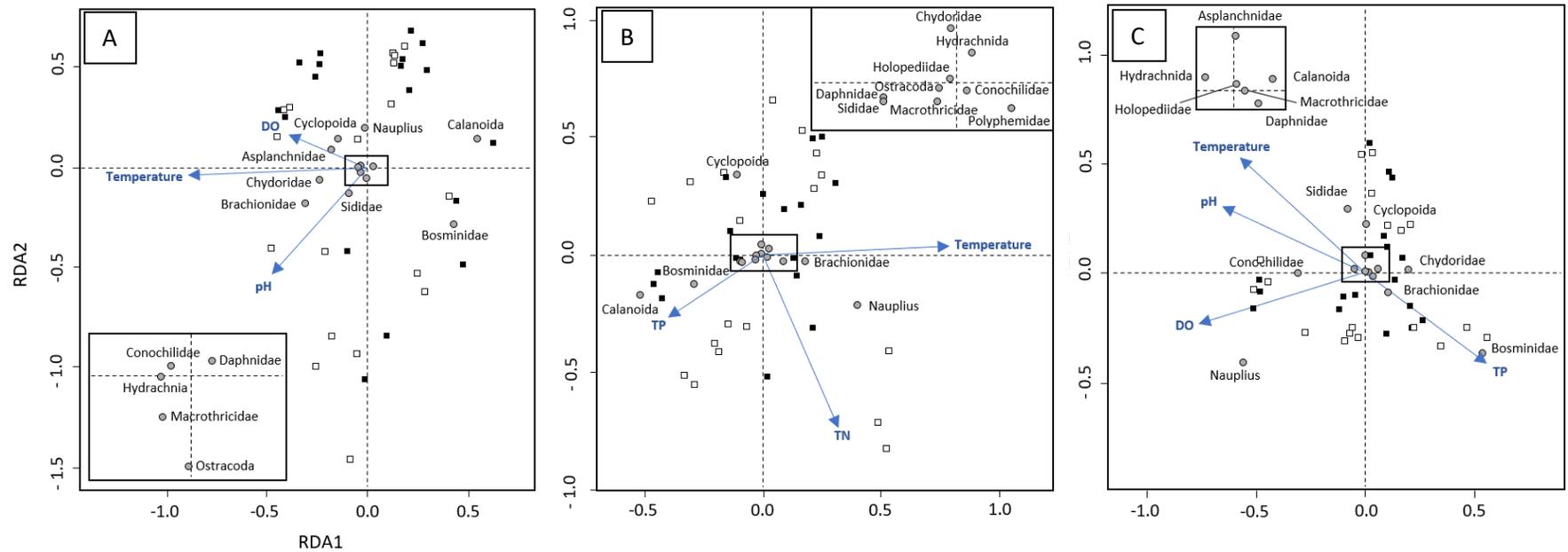


Figure 15. Redundancy Analysis (RDA) diagrams at scaling 2 of the significant environmental variables and the zooplankton taxa per Liter for Wolfe Lake (A), Otter Lake (B) and Dog Lake (C). Taxa (identified to Family, where possible) were Hellinger transformed, and any containing less than 3 appearances throughout the dataset were dropped. A multicollinearity test determined that the conductivity variable needed to be removed from each Analysis. Individual environmental variables are represented by blue arrows, while the grey circles represent individual zooplankton taxa. The squares represent the “milfoil” (black) and “harvested” (white) ordinated objects of the data matrix. The arrow length represents the strength of the correlation between the environmental variables and the zooplankton taxa. The longer the arrow length, the stronger the correlation. The angular distance between zooplankton taxa and environmental variable axes in the plot also reflects their correlations. The smaller the distance, the stronger the correlation.

4.2.4 Variation Partitioning

For Wolfe Lake, 37% of the overall variation in zooplankton assemblage could be explained by three explanatory variables. Most of the zooplankton variation was explained by the environmental variables accounting for 31% of the zooplankton variation. Twenty percent of this explanatory environmental variation overlapped with the temporal variation, while the independent explanatory environmental variation was 11%. This indicates that the environmental variable within the aquatic ecosystem explained most of the zooplankton variation, though the variation in environmental variables was mainly driven by temporal/seasonal change (20% overlap). The overall temporal explanatory variable accounted for 24% of the zooplankton variation, indicating a small independent temporal variation of 4%. The spatial structuring in the zooplankton assemblage data had the smallest explained variation at 2% of the overall family variation. This indicates that mechanical harvesting at Wolfe Lake did not have a large influence on zooplankton composition (Figure 16B).

Otter Lake had 39% of the overall zooplankton variation that could be explained by the three explanatory matrices. All this variation falls within the overall environmental explanatory variation, once again making the environmental data the largest contributor to the variation in zooplankton data. Similar to Wolfe Lake, 21% of this explanatory environmental variation overlapped with the temporal variation, while the independent explanatory environmental variation was 11%. This indicates yet again that most of the zooplankton family variation was driven by changes in environmental variables that are led by temporal/seasonal changes. The only spatial patterns that was present in the zooplankton data of this analysis was 7% explained variation that overlapped with both environmental and temporal explanatory structures, again

indicating that mechanical harvesting did not have a significant influence on zooplankton assemblage (Figure 16C).

For Dog Lake, 62% of the zooplankton species variation could be explained by the three explanatory variables. Again, the environmental variables were the largest contributor to the zooplankton species variation, since all the explained variation from the analysis falls within the overall environmental explanatory variable. Thirty-five percent of the species variation can be explained by the environmental variables (Temperature, TP, pH, DO), which were independent from any spatial or temporal structure. Both independent temporal and spatial variations had almost no explanatory power (<1%). The partitioning analysis did however present a temporal structure in the species data that is shared by the environmental data indicating that 20% of the environmental variation was driven by temporal/seasonal changes. There was also a fraction of 7% species variation that could be explained by overlapping variation between all three explanatory variables. Since the spatial patterns in the species data only accounted for a small fraction of the explained variation, mechanical harvesting also did not appear to have a large influence on zooplankton variation (Figure 16D).

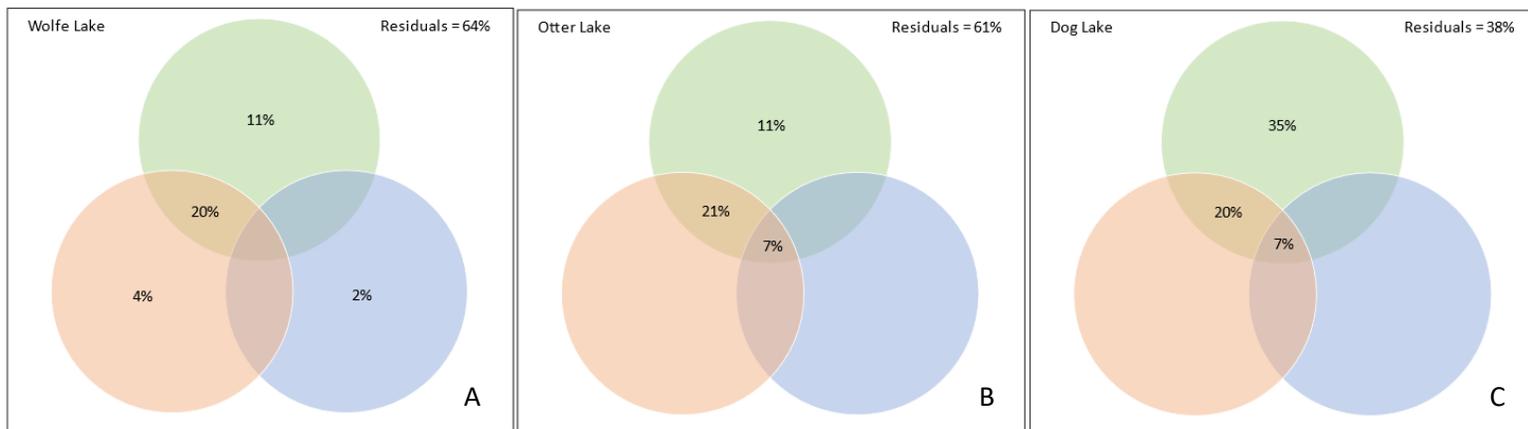


Figure 16. Venn diagrams representing the variance partitioning of the zooplankton taxa data matrix for (A) Wolfe Lake, (B) Otter Lake and (C) Dog Lake. The data was partitioned into three explanatory *variables*: Environmental variables (green circles at the top), Date (pink circles to the left), and Treatment (blue circles to the right). The statistically significant environmental explanatory variables represent the overall environmental variation, the treatment (spatial) explanatory variable represents the variation explained by treated vs. non-treated sites, and the Date (temporal) explanatory variable represents the variation explained by seasonal change. Sections with no percentages indicate an explained variance that is inferior to 1%.

Chapter 5.0 – Discussion

The burlap benthic barriers as a macrophyte control led to minor variations in the physical and chemical properties at Malcolm Lake. A year after its application, the burlap began showing major degradation in some areas and had plant growth that penetrated the barrier throughout the mat. All measured abiotic parameters between the control and impact sites did not show any significant differences pre- and post-treatment a year after the burlap application. However, results from the ANOSIM did indicate some clear changes to zooplankton community composition persisting a year after the burlap benthic barrier treatment; according to the variance partitioning, a portion of these changes (19%) is explained by the benthic barrier treatment. This impact to zooplankton community composition a year after the barrier application occurred despite the small size of the treatment in relation to the large continuous bed of the *M. spicatum*.

On the other hand, the application of mechanical harvesting had minimal localized impacts on the physical, chemical, and biological properties at all three studied lakes within the Rideau River and the Cataraqui River Watershed. All measured abiotic parameters between the control and impact sites at each lake did not have any significant differences pre- and post-treatment a year after the treatment. Only Otter Lake had a small shift in zooplankton community according to the ANOSIM. By the end of the 2019 sampling season, there were already indications of plant regrowth, and by the start of the late summer of 2020 (approximately a year after the applied treatment), most of the *M. spicatum* had grown back. This implies that employing mechanical harvesting on large dense patches of *M. spicatum* is a short-term management practice. Variance partitioning for the Rideau Lakes also indicated that the

explained variance in zooplankton community composition is mostly attributed to seasonally-induced environmental variance, and not attributed to the mechanical harvesting treatment.

5.1 Benthic Barriers

Benthic barriers have been studied and used as a potential macrophyte control since the 1970s, with polyethylene being the first type of barrier used (Born et al. 1973; Cooke et al., 1980; Nichols, 1974). Since then, there has been various types of benthic barriers that have been studied such as mesh polymeric fibers (Engel, 1983), fiberglass screens (Boston & Perkins, 1982), and more recently, biodegradable burlap mats (Caffrey et al., 2010; Wood, 2017). Burlap barriers have been gaining attention as they hold advantages over other materials, especially since they are considered more environmentally friendly than plastic material (Caffrey et al., 2010). However, one downside to biodegradable burlap mats is that it can only be used as a short-term management practice as it can only remain useful for 3-5 months before experiencing too much degradation and loses effectiveness (Jones et al., 1984).

The burlap material used in this thesis did remain intact for all of 2019 but did experience degradation by the fall of 2020. Some areas did contain plant growth, however, these areas on the burlap material never exceeded more than 40% plant cover and did not grow more than half the height of the surrounding *M. spicatum* bed that was not covered by the benthic barrier. Approximately 35% of plant stems that did begin growing through the mat were *M. spicatum*. One reason why *M. spicatum* appears to be common though the burlap is because the initial dense patch of macrophytes consisted almost entirely of *M. spicatum*. However, it has been reported that while finer-leaved species such as *M. spicatum* can eventually grow through the

weave of burlap mats (Hofstra et al., 2012), they do not grow in abundance while in the presence of other aquatic plants. Instead, native broadleaved *Potamogeton* had a considerably higher abundance (~60%) a year after the burlap application, when they were virtually absent at the site before the treatment. Studies have shown that native species of macrophytes, including native species of *Potamogeton*, can undergo relatively rapid recovery through the weave of burlap barriers when compared to targeted invasive macrophytes (Caffrey et al., 2010).

The rapid decomposition of plant material under benthic barriers can cause anoxic conditions at the sediment interface (Boston & Perkins, 1982), which in turn could support phosphorus release from the sediments to the overlying water column (Boros et al., 2011; Hupfer et al., 2008). When coupled with decaying plants directly releasing phosphorus in the water column (Carpenter, 1980; Nichols et al., 1973), the amount of phosphorus released from the sediments and decaying plant matter could lead to increased nutrient enrichment within the lake. Semi-porous biodegradable barrier, however, can allow gas exchange more easily near the sediment water interface, reducing the probability of anoxic conditions forming beneath the barriers (Caffrey et al., 2010; Eakin et al., 1995). Results appear to support this, as DO concentrations and pH levels did not differ between control and impacts sites after the barrier treatment.

Benthic barriers may also generate a blanketing effect on top of the sediments, suppressing sediment resuspension, and thus preventing increased turbidity or resuspended nutrients (Søndergaard et al., 1992). Results displayed similar values for both TP, TKN and DOC between control and impact sites post-treatment, and analyses showed no significant difference between these properties. For temperature, results also did not display any significant differences

between sites for both 2019 and 2020, which coincides with no changes in TP since higher temperatures can stimulate the liberation of sediment phosphorus (Jensen et al., 1992).

Since Malcolm Lake mostly consists of shallow waters, my results suggest that the water in the lake is well mixed and therefore it is unlikely that localized use of burlap benthic barriers would cause any impacts to the lake's nutrient dynamics. In some cases, macrophytes could induce thermal stratification in shallow lakes, potentially causing anoxic conditions to occur within macrophyte stands, since stratification could result in insufficient vertical transportation of oxygen to the bottom waters (Vilas et al., 2017). However, since there is regular mixing at Malcolm Lake, the lake likely maintains a short lake water retention time, preventing deoxygenation at the sediment water interface (Carpenter et al., 1981) and thus avoiding phosphorus release. In fact, since large dense macrophyte beds can retard waterflow through the littoral zone (Carpenter et al., 1986), burlap barriers may actually help offset this by allowing water to move more freely.

The ANOSIM used during my analyses suggest that there was a statistically significant difference in zooplankton community structure between both 'control' and 'impact' sites after the burlap barrier treatment (significance value = 0.0002). According to the PCAs, many individual zooplankton groups also showed affinities towards certain sites. One cladoceran that was influenced by the removal of macrophytes from the benthic barrier treatment was Chydoridae. This cladoceran family is often associated with macrophytes within littoral habitats (Fairchild, 2016; Geraldine & Boavida, 2004; Lima et al., 2003), which coincides with the results displaying greater abundance within the macrophyte stands than in the burlap treatment (Figure 9).

Sididae and Macrothricidae are two other cladocerans that are associated with macrophytes as they temporarily attach to plant surfaces to filter feed (Fairchild, 2016; Lima et al., 2003). These two cladoceran families also declined in numbers within the burlap sites after treatment. However, species from the *Bosmina* family generally displayed a less obvious association with macrophytes compared to other cladocerans. While *Bosmina* did show a large increase in abundance right after the burlap application in 2019, the zooplankton family began displaying a larger presence in the control site, as well as the bare sediment site the following year. This distribution across all sites could have been anticipated as *Bosmina* can be widespread among macrophytes and pelagic zones (Geraldes & Boavida, 2004). They often display higher abundances in open waters (Lima et al., 2003; Pennak, 1966) and typically migrate towards macrophytes when pressure from fish predation increases (Jeppesen et al., 1998).

Cyclopoid copepods were the most numerous zooplankton with some samples reaching more than 20 individuals per litre of filtered water. Both the macrophyte beds and the sites treated with burlap barriers showed similar numbers of cyclopoids in 2019 but then began displaying a higher presence in the burlap barrier sites in 2020. Cyclopoids have shown high affinity to macrophyte beds in littoral areas as much of their food is located on macrophyte beds (Fairchild, 2016; Geraldes & Boavida, 2004), but smaller cyclopoids can also be more abundant in pelagic areas if there is higher presence of planktivorous predation within the lake (Jeppesen et al., 1998). Calanoida also showed similar numbers in both the macrophyte beds and in the impact trenches but had a much smaller presence than cyclopoids. Calanoida did occasionally show a slightly higher presence in the impact trenches during both years, which is in agreement with other studies where calanoids have displayed higher abundance in non-vegetated areas

(Geraldes & Boavida, 2004) as it is thought that they use this as a predation avoidance strategy from cyclopoids (Caramujo et al., 1997).

Ostracods displayed a presence in the macrophyte beds, while virtually being absent in impact trenches and bare sediment sites. This aligns with other studies claiming that non-swimming zooplankton families such as Ostracods have higher affinity for macrophytes beds, and especially higher association with plants containing a complex leaflet structure, as they directly attach to the plants to feed on periphyton (Matsuda et al., 2015; Suen & Gillett-Kaufman, 2019).

Variance partitioning indicated that a good portion of the individual zooplankton assemblage variation in my results (19%), can be explained by the burlap barrier macrophyte treatment. This supports the claim from Kuczyńska-Kippen et al. (2016), that the abundance and diversity of macrophytes have a large influence in determining zooplankton community structure, and confirms my hypothesis that the removal of *M. spicatum* using burlap barriers can impact lake health indicators such as zooplankton community structure. The variation from surrounding environmental variables also explained a large portion of zooplankton variability (34%), with a good portion of this variability overlapping with temporal changes (11%). This indicates that the zooplankton community structure is also partially explained by changes in their surrounding environmental variables, where this environmental variation is often induced by seasonal change.

5.2 Mechanical harvesting

Zooplankton community assemblage at the studied sites was mainly driven by environmental variables that were not associated with the mechanical harvesting of macrophytes. Temperature explained much of the variation in zooplankton community structure, across all three lakes, that was likely a result of the seasonality of the sampling period. This supports the theory that out of all of the environmental conditions present in a lake, temperature is thought to be one of the main drivers of intra-annual seasonal/temporal distribution of zooplankton communities (Gabaldón et al., 2019; Wolfenbarger, 1999). It has been reported that cladocerans and copepods generally have a better tolerance in environments containing temperatures below 25°C (Moore et al., 1996). My results show that calanoids are showing an affinity towards lower temperatures in Otter Lake and Wolfe Lake. This is consistent with other studies clearly demonstrating that elevated temperatures can increase mortality rates and reduce the clutch size of freshwater calanoids (Chow-Fraser & Maly, 1991; Hall & Burns, 2001). Bosminidae are also known to occur over a wide range of environmental conditions and tolerate colder environments (Kamenik et al., 2007), which may partially explain why temperature has a negative correlation with this cladoceran at all three Rideau lakes. Sididae had a small positive correlation with temperature at Dog Lake. This crustacean is known to be better adapted to warmer temperatures as it has greater fitness when compared to other cladoceran at higher temperatures (Chaparro-Herrera et al., 2010; Larsen, 1995).

According to the results from the RDAs, nutrients within Dog Lake (eutrophic) and Otter Lake (mesotrophic), especially phosphorus, are also playing an important role in explaining variance in zooplankton community structure, while nutrients at Wolfe Lake (oligotrophic) are not. Other studies have suggested that nutrients can clearly increase the abundance of

phytoplankton, and directly influence Cladocera zooplankton assemblages by altering food availability (Kamenik et al., 2007; Vanni, 2016). Thus, the lower nutrient levels at Wolfe Lake, may be why TP or TN did not explain a significant portion of zooplankton community variability within the lake. An increase in small-bodied zooplankton such as Bosminidae are also often associated with nutrient rich lakes and cyanobacterial algal blooms, while are often missing from oligotrophic lakes (Jiang et al., 2017; Kamenik et al., 2007). This is also consistent with the samples collected during this thesis, since Bosminidae were most abundant at Dog Lake and the least abundant at Wolfe Lake. Similar to Bosminidae, Calanoida had a positive correlation to TP at Otter Lake. According to Hambright et al. (2007), calanoids are a highly selective grazer that can take advantage of high phytoplankton abundances containing a broader array of species assemblages containing larger celled and grazing resistant cyanobacteria . The same study also posits that while the tissue content of cladocerans is relatively rich in phosphorus, copepods are instead typically enriched in nitrogen and tend to return lower N:P (higher phosphorus levels) to the water. This may explain why calanoids have a positive correlation with TP concentrations in Otter Lake. It may also explain why Cyclopoida demonstrated a strong negative correlation with TN at Otter Lake. Higher copepod abundance can cause TN concentrations to decrease in the water column due to copepods retaining higher N:P ratios (Hambright et al., 2007).

Dissolved oxygen (DO) is another important abiotic environmental variable that may strongly influence the distribution and community structure of zooplankton (Gabaldón et al., 2019; Kamenik et al., 2007; Wichner et al., 2018). An environment with reduced DO can directly influence zooplankton distribution or indirectly influence zooplankton community structure through trophic interactions (Hanazato, 1997). Even though the lakes in this thesis generally demonstrated no stratification or oxygen concentrations below optimum levels (80-120%

saturation) (Data Stream, 2021), results suggests that the variation in zooplankton data for Wolfe Lake and Dog Lake can be partially explained by DO. While DO at Wolfe Lake demonstrated a small contribution to the zooplankton community structure, DO explained a relatively large portion of variance in the zooplankton data at Dog Lake. The copepod nauplii at Dog Lake appears to especially have an affinity with DO.

Some studies have linked the successful reproductive development of copepods to oxygen availability (Lutz et al., 1992; McAllen & Brennan, 2009; Roman et al., 1993), suggesting that higher oxygen levels could improve hatching of copepod eggs and increase overall nauplii abundance. However, these studies only linked lower survival rates of copepods when DO levels reached anoxic to hypoxic conditions (0-25% saturation). The lowest DO levels recorded at Dog Lake were all well above these levels, with only some measurements reaching just below 80% in August and October (Figure 13D). As a result, while oxygen stress can play an important role in determining the community structure of zooplankton in lakes (Karpowicz et al., 2020), the oxygen saturation within these studied lakes is likely not directly influencing individual zooplankton species through this mechanism. Instead, it is probable that zooplankton community structure is directly influenced by another environmental variable that is also influencing DO saturation levels such as water temperature. DO saturation in lakes is directly related to water temperature, as warmer waters can hold less oxygen in solution than colder waters (Addy & Green, 1997). This is because the solubility of oxygen decreases as temperature increases, meaning that warmer water would require less dissolved oxygen to reach 100% saturation than cooler water (Wetzel, 2001). Therefore, other environmental variables such as temperature may explain why DO might have correlations with certain zooplankton species, but

is not directly causing any individual species variation or greatly influencing overall zooplankton community structure.

Another environmental factor that can be strongly associated with patterns in zooplankton community structure is pH (Gabaldón et al., 2019), as seen in Dog Lake and Wolfe Lake. Low pH levels can cause reduced zooplankton abundance, decrease biodiversity, and even trigger the loss of some species (Paturej et al., 2017). As seen in my results with Dog Lake, field studies often show an increase in relative and absolute abundance of Bosminidae as lakes become more acidic (Locke & Sprules, 2000). These taxa are generally able to remain abundant at lower pH levels since it is assumed that Bosminidae are “acid tolerant”, are able to obtain more food due to less competition, or may experience release from predation (Barmuta et al., 1990; Locke & Sprules, 2000). The results for Wolfe Lake also demonstrate a negative correlation between pH and Calanoida. According to Derry et al. (2007, 2010), Calanoida can also rapidly adapt to acidification of lake-water, allowing them to dominate zooplankton communities in acidified lakes (Malley & Chang, 1986).

However, when looking more closely at the pH measurements at each site (Figure 11C, 12C, and 13C), the values are all measured between 8-9, with some measurements even reaching above 9 at Dog Lake before the mechanical treatment. These elevated pH levels could be attributed to multiple factors such as lower precipitation (rain is generally acidic at around 5.6) (Likens et al., 1979), the minerology or soil of the surrounding landscape (e.g., carbonate-rich soils) (Reid et al., 2019), or photosynthesis from aquatic plants and/or algae (process consumes hydrogen ions) (Axelsson, 1988). High pH values from within macrophyte stands are especially often associated with macrophyte photosynthesis (Jeppesen et al., 1998). The pH concentrations at the studied lakes are by no means low or acidic and are typically within the

optimum, long-term basis pH range (6.5-9.0) for the majority of aquatic life in freshwater ecosystems (EPA, 1976). The physiological effects of high pH levels (>9) on aquatic organisms have been generally less studied than the effects of low pH levels (<6), however, some studies have concluded that environments with high pH values can cause adverse on fish due to the toxic mode of action of hydroxyl ions (OH^-) (Robertson-Bryan Inc., 2004). A study has also shown that alkaline aquatic environments can have strong adverse effects specifically on zooplankton, and at very high pH levels (>9) can inhibit their survival (Ivanova & Kazantseva, 2006).

Although pH levels at Wolfe, Otter and Dog Lake typically stayed below this threshold, some measurements did reach near and above it and did not fall below 8. This suggests that if pH is directly influencing zooplankton community structure, changes in zooplankton assemblages at these lakes are likely more associated with fluctuating levels around the upper limits of the optimum pH range, and not associated with the acid tolerance. It is possible, however, that the water pH levels from this thesis could contain correlations with certain family taxa while not directly causing their variation. For example, another environmental variable that may be influencing both pH levels and zooplankton community structure is the amount of dissolved calcium carbonate in the water. Unfortunately, this variable was not measured during this study, and therefore the explanatory variance of zooplankton community structure from presence of dissolved calcium carbonate could not be investigated.

Key stone groups of crustacean zooplankton such as Cladocera contain large differences in sensitivity to Ca concentrations and undergo cyclic moults that generally require high amounts of readily available Ca from their surrounding environments to form heavily calcified exoskeletons (Cairns & Yan, 2009; Giardini, Yan, & Heyland, 2015). These crustaceans typically acquire readily available dissolved carbonate or bicarbonate molecules during the

calcification process of their exoskeletons (Ries, 2012), indicating that several cladoceran taxa, including daphniids, bosminids and copepods, are dependant on the Ca levels from their surrounding environment (Arnott et al., 2017). The dissolution and amount of calcium carbonate into Ca^{2+} and CaCO_3 in a system dictates the lake's alkalinity or buffer capacity, and therefore directly influences its overall pH levels, as it determines the capacity of controlling water acidification (Thompson & Bonnar, 1931). Consequently, the lake's alkalinity may be an environmental variable that has a direct influence on both zooplankton community structure and pH levels and may explain why my results are displaying correlations between the zooplankton and pH data. Although, it is very likely that changes in pH are mainly driven by seasonal changes in photosynthesis from plants and phytoplankton (Leidonald et al., 2019; Tadesse et al., 2004)

It is surprising that the abiotic environmental variables did not indicate more substantial and distinctive differences between the milfoil and harvested sites, especially since these parameters are largely influenced by the presence of dense macrophyte beds in shallow lakes (Carpenter, Stephen R.; Lodge, 1986; Jeppesen et al., 1992). For instance, dense macrophyte stands can influence water temperatures by inducing thermal stratification as the plants act as a barrier to wind induced mixing (Vilas et al., 2018). This thermal stratification reduces vertical transport of gases, diminishes dissolved and particulate materials, and ultimately influences the oxygen levels at the sediment-water interface (Vilas et al., 2017). However, there may be other reasons that explain why the treatments within large dense macrophyte did not induce any change in these abiotic variables. Among them is the possibility of induced vertical mixing within shallow lakes. Due to their shallow depths, the lakes in this study may be experiencing frequent or enhanced mixing since little can prevent cooling of these waters or prevent full

column mixing from strong winds (Padisak & Reynolds, 2004). Small areas used for the treatment sites in this study (which are surrounded by areas covered by dense *M. spicatum* stands) are also likely causing the waters from both milfoil and harvested sites to amalgamate with one another. This would inhibit both milfoil and harvested sites from forming more distinct differences in their surrounding environmental variables.

Overall, results showed that mechanically harvesting small sections from dense patches of *M. spicatum* stands did not significantly impact to zooplankton communities, nor to their surrounding environmental variables. In fact, variance partitioning indicated that changes in zooplankton communities within the treated sites are mainly driven by their surrounding environmental variables. It appears that the changes in the zooplankton assemblages in both control and impact sites is mostly explained by temporal changes in environmental factors within these sites, including temperature, TP, DO and pH. Since temperature appeared to have the largest influence on zooplankton community structure, it is likely that the explained environmental variance is induced by seasonal variance. This lack of environmental impact at multiple lakes suggests that the management practice remains a viable option for managing this invasive macrophyte in eastern Ontario waters. Though, since the sections of harvested *M. spicatum* have almost completely regrown by the following year, the management practice is only effective at short-term control.

5.3 *M. spicatum* Management and Lake Ecosystem Resilience

While results showed that the application of burlap benthic barriers on small sections from dense patches of *M. spicatum* stands brought clear changes to zooplankton community

composition, these direct or indirect impacts to zooplankton communities are minimal, and the treatment had insignificant localized impacts on other physical and chemical properties at Malcolm Lake.

It is theorized that removing macrophyte biomass could diminish their inhibitory effect on algal abundance and may even trigger an algal dominated state (Hilt et al., 2017; Kuiper et al., 2017). This is likely because a eutrophic lake containing an algae dominated state and low macrophyte abundance can directly impact both zooplankton community structure and functional diversity, as the loss or lack of submerged macrophytes abundance in these systems can reduce habitat complexity (Bolduc et al., 2016). Eutrophic lakes can also have a harmful effect on efficient zooplankton grazers, such as large cladoceran like *Daphnia*, as these systems are often dominated by grazing resistant cyanobacteria colonies that can produce cyanotoxins or cause mechanical interference (Haney, 1987; Lampert, 1987). In general, there tends to be a shift in copepods from calanoids to cyclopoids as lake shift towards a eutrophic status (Dufresne, 2017; Jeppesen et al., 2000), and less efficient grazers such as copepods and *Bosmina* can dominate the zooplankton community as they are less effected by the harmful inhibitory effects of eutrophic systems (Lampert, 1987). These less efficient, small bodied zooplankton have a low grazing pressure on phytoplankton (Jeppesen et al., 2000), and thus causes eutrophic systems to be more susceptible to algal blooms. It is for these reasons my thesis predicted that using either of these macrophyte management techniques at a smaller scale would still cause notable changes in zooplankton community structure.

The reduction of zooplankton species diversity and phytoplankton grazing pressure associated with lakes containing a higher trophic status (Bockwoldt et al., 2017; Jeppesen et al., 2000) is also a reason why the thesis hypothesized that eutrophic lakes generally contain an

inferior ecosystem resilience compared to oligo-mesotrophic systems, and would therefore display greater environmental impacts, and slower recovery after a local mechanical harvesting treatment on *M. spicatum*. Nevertheless, results showed that mechanically harvesting small sections of *M. spicatum* stands did not indicate any signs of shifts in steady states at any of the three lakes. This indicates that results go against the thesis predictions, and further suggests that the commonly used management practice on small sections is likely an ideal and environmentally sound option for managing the most problematic invasive macrophyte in eastern Ontario waters.

However, during the harvesting process at the Rideau Lakes, some plant fragments and temporary increases in turbidity were observed. Not only does this indicate that the treatment could risk further spread and distribution of *M. spicatum*, mechanical harvesting could also directly or indirectly disturb the surrounding sediments and cause them to resuspend in the water column (Carpenter & Gasith, 1978; Kuiper et al., 2017). As a result, even though the studied management technique remains a good option for controlling invasive macrophytes in Ontario waters, excessive use of mechanical harvesting to control *M. spicatum* could be a cause for concern.

Furthermore, while this thesis did not present any direct early warning signs of regime shifts, past studies have shown that triggering such as shift can still occur after smaller perturbations (Kuiper et al., 2017). In fact, one study from Cooke et al. (1990) has even shown that higher levels of TP and algal blooms have occurred in areas treated by macrophyte harvesting within a eutrophic system. My thesis also could have used a larger sample size (compare more lakes and/or sites) and sampling frequency to monitor any changes in variance or slower recovery more accurately following this macrophyte treatment. Therefore, while the

results from this thesis only indicated minimal ecological effects for mechanically harvesting *M. spicatum*, one should still be cautious when using this management technique for controlling dense patches of the invasive species in eastern Ontarian lakes, especially if treatments are conducted on a larger scale, or if multiple control efforts are to be taken simultaneously, as the environmental impacts of such treatments are not yet well known.

Chapter 6.0 – Conclusion

The primary objectives of this thesis were to investigate the application of large burlap benthic barriers and mechanical macrophyte harvesting on the physical, chemical, and biological properties of various sites within lakes in eastern Ontario. For both types of treatments, we examined the changes in properties the year of, and the year after the induced disturbance to determine any short or long-term changes. Three different lakes were also chosen for the mechanical harvesting method in attempt to record greater impacts and/or slower recovery rates of the disturbed macrophytes in lakes containing higher nutrient contents.

The findings from this thesis suggest that the overall impacts of applying burlap benthic barriers as a macrophyte control on localized abiotic factors were minimal. The barriers even began showing major degradation and substantial plant growth in some areas only a year after the treatment; with most of the emerging plants belonging to a native *Potamogeton* species. At a small scale, this management technique is therefore a relatively short-term solution to macrophyte control with little to no long-term environmental impacts. Nonetheless, even at a small scale, the application did show changes in family assemblages of zooplankton that continued a year after the treatment. The results show that while some of these changes in

zooplankton assemblages could be associated with seasonal variance, that the application of burlap benthic barriers could be causing considerable changes in zooplankton community structure despite the proximity of both control and impact sites.

On the other hand, mechanically harvesting small patches of *M. spicatum* displayed no significant impacts on both localized abiotic factors and family assemblages of zooplankton. In 2019, the invasive macrophyte within the treatment patches already began to regrow by the end of the sampling season, with the plants growing back to their original states a year after they were treated. These results suggest that mechanically harvesting large dense patches of *M. spicatum* is an even shorter-term solution than burlap barriers and would also unlikely have any long-term environmental impacts during small scale treatments. The results suggest that changes in zooplankton communities within the impact sites of the mechanical treatments are mainly driven by their surrounding environmental variables and not the treatment itself. The environmental variables that mostly explain the variance in zooplankton community structure include temperature, TP, DO and pH, with temperature having the most influence in all three studied lakes within the Rideau River and the Cataraqui River Watershed. My findings also suggests that a lot of the changes in zooplankton community structure at all these lakes are caused by seasonally induced changes in environmental factors and explains why temperature seems to account for most of the variation in zooplankton family assemblages. Thus, I was not able to conclude if using the mechanical harvesting treatment in lakes containing higher trophic states caused any greater impacts or slower recovery rates.

There is no doubt that invasive species, such as *M. spicatum*, can have dramatic effects on freshwater ecosystems (Eiswerth et al., 2000; Leguizamon, 2017), yet, these large macrophyte beds can become a critical component to the structure and function of these same ecosystems

(Carpenter & Lodge, 1986; Phillips et al., 2016; Savino & Stein, 1982). In fact, recent studies have outlined that large macrophyte beds, including invasive species, provide more ecosystem services than anticipated (Engel, 1995; Leguizamon, 2017; O'Hare et al., 2018; Schallenberg et al., 2013). *M. spicatum* can provide ecosystem services such as useful habitat to fish and invertebrates (Engel, 1995; Leguizamon, 2017), complex habitat structure that offers superior plant refuge for zooplankton (Bergström, Svensson, & Westberg, 2000), and provide inhibitory effects on algal growth by the release of allelochemicals, or by competing for available nutrients or light (Nakai et al., 2000; Zhu et al., 2010).

However, *M. spicatum* can outcompete native species of macrophytes and generate dense monocultural stands throughout entire lakes (Boylen et al., 1999; Madsen et al., 1991). This could provide less impactful ecosystem services by reducing native plant biodiversity and reducing fish species diversity and richness when compared to diverse moderately dense native plant beds (Boylen et al., 1999; Eiswaerth et al., 2000; Engel, 1995; Keast, 1984; Smith et al., 1990). These large dense monocultural stands can also cause economical impacts such as reduced property values, reduced probability of future property development (Goodenberger & Klaiber, 2016; Horsch & Lewis, 2009), and increase cost of electricity generation and the provision of municipal water supplies (Eiswaerth et al., 2000). They can even decrease the quantity and quality of recreational activities (Eiswaerth et al., 2000). It is for these reasons and their ability to quickly proliferate and spread to other aquatic systems (Aiken et al., 1979) that *M. spicatum* has gain so much attention throughout North America, and why many management regimes have been established since their introduction. These regimes have sought to control their spread to and within lakes, and in some cases sought complete eradication from given systems.

Though the complete eradication efforts for *M. spicatum* in the past have often failed (Smith et al., 1990) causing state departments to re-evaluate their management objectives (Radomski & Perleberg, 2019). These departments have also concluded that eradication is not a cost-effective goal, and that *M. spicatum* can act similarly to indigenous species in some lakes. In fact, the importance of macrophyte beds to overall aquatic ecosystems has already lead some lake managers to use integrated approaches targeting only smaller specific areas of a lake, minimizing ecological impacts and economic efforts (Bickel & Closs, 2009; Clayton & Tanner, 1988; Greer et al., 2012; Radomski & Perleberg, 2019; Van Nes et al., 2002). These integrated approaches should especially be considered for lake ecosystems who possess a relatively higher trophic state, as these systems are thought to be generally less resilient and therefore more vulnerable to environmental disturbances (Dakos et al., 2015; Peterson et al., 1998). Either way, it should be essential to consider the value both native and invasive macrophytes could provide to lakes while making management decisions and prioritize environmental sustainability over conflicting social and commercial perspectives. After all, past management action in Canada during large increases of macrophyte biomass in lakes is more often triggered when human use of these lakes or their resources have been impacted, as oppose to being triggered by ecosystem consequences. (Chambers et al., 1999).

The desired state of a lake ecosystem is one that generally contains clear water and that is dominated by macrophytes in shallow areas of the lake, as this state provides superior natural ecosystem function and services (Hilt et al., 2017). Consequently, responsibly managing nuisance stands of *M. spicatum* to improve ecosystem sustainability will ultimately depend on their dominating presence within the lake, and its current trophic state. While *M. spicatum* may provide some ecosystem services in certain cases, their potential benefits should ideally be

considered and weighed against their potential negative impacts during macrophyte management in turbid eutrophic systems that are unable to sustain native plant beds in the first place (Engel, 1995). But generally, large monoculture stands of *M. spicatum* are greatly problematic to many oligo-mesotrophic systems throughout Canadian lakes and rivers and require rigorous management action to control and prevent future environmental and economic impacts. Therefore, improving our knowledge on the use of different cost-effective and eco-friendly macrophyte management practices on *M. spicatum* in the Rideau waterways and eastern Ontario waters is crucial to help improve the efficacy of national management efforts.

This thesis presents good examples of what to expect from smaller scale treatments for such macrophyte management practices. For instance, the scale of the studied treatments are accurate representations of lake property owners administering management techniques to control macrophytes impeding with their recreational activities (e.g., swimming, boat docking etc.). Overall, results show that these targeted small-scale macrophyte interventions could be considered viable methods for controlling *M. spicatum* in Canadian waters. Similar to past studies, their treatments have shown to temporarily reduce problematic plant emergence and biomass (Hofstra & Clayton, 2012; Painter, 1988), but have also shown to cause minimal environmental impacts. In the case of applying burlap benthic barriers, the treatment may even favour the regrowth of native *Potamogeton* over the invasive *M. spicatum*. This further confirms that these macrophyte treatments are likely great tools that could help ease ecological, economical, and social impacts that are often associated with large dominating stands of *M. spicatum* throughout Canadian waters.

However, my results demonstrated that after mechanically harvesting patches of *M. spicatum*, treated areas started showing regrowth only months after the treatment, with the stands

almost completely regrown by the following year. The application of benthic burlap barriers did appear to suppress *M. spicatum* growth from a significantly longer duration, yet still had macrophytes growing through the burlap in relative abundance 15 months after the application. This indicates that both treatments are not permanent solutions for controlling dense stands of *M. spicatum* and that repeated treatments would likely be required every year to remain effective. Another thing to consider is that both of the physical management techniques used in this thesis are not species specific and might also suppress non-targeted species of macrophytes (Hussner et al., 2017; Madsen, 2000). However, the majority of *M. spicatum* stands in eastern Ontario waters are monocultural or have very little integration of native macrophytes, indicating that the lack of targeted species specificity during benthic barriers or mechanical harvesting treatments would rarely be an issue.

It is in my opinion that to improve lake management decisions, further research should be dedicated towards the impacts of macrophyte control to freshwater ecosystems at different spatial and temporal scales. While the study's treatment plot sizes are typical of what would be carried out by lake users and lake managers in Ontario, they are possibly inadequate to properly study the impacts on the lake's biotic and abiotic factors. It is difficult to confidently say that the adjacent macrophyte plots (control sites) had no influence on the parameters above the treated plots (impact sites), and vice-versa, since both types of sites were in close proximity to one another. Ideally this thesis would have been conducted on a larger scale for the chosen lakes of this study. Otherwise, lakes containing a smaller waterbody should have been chosen for these plot sizes. Future research should also consider studying the environmental impacts of other macrophyte management practices, especially for newer practices such as the application of weevil macrophyte grazers or water jets. It is my hope that the findings from this thesis will help

initiate and guide new future studies on the environmental impacts of different macrophyte management techniques that will provide a narrower focus and higher sampling resolution. I believe that the efforts in preventing changes in dynamics between macrophytes, phytoplankton, zooplankton and fish in Canadian lakes will become progressively more important as climate change and other human induced pressures increasingly threaten our freshwater systems.

Chapter 7.0 - References

- Abernethy, V. J., Sabbatini, M. R., & Murphy, K. J. (1996). Response of *Elodea canadensis* Michx. and *Myriophyllum spicatum* L. to shade, cutting and competition in experimental culture. *Hydrobiologia*, 340(1–3), 219–224. <https://doi.org/10.1007/BF00012758>
- Addy, K., & Green, L. (1997). Natural Resources Facts, Fact Sheet No. 96-3, “Dissolved Oxygen and Temperature.” In *Cooperative Extension*. Kingston, Rhode Island.
- Aiken, S. G., Newroth, R., Wiles, I., & Control, H. (1979). The Biology of Canadian Weeds. 34. *Myriophyllum spicatum* L. *Can. J. Plant Sci.*, 59, 201–215.
- Angeler, D. G., Allen, C. R., Birgé, H. E., Drakare, S., McKie, B. G., & Johnson, R. K. (2014). Assessing and managing freshwater ecosystems vulnerable to environmental change. *Ambio*, 43, 113–125. <https://doi.org/10.1007/s13280-014-0566-z>
- Angler’s Atlas. (2021). Wolfe Lake. Retrieved January 15, 2021, from <https://www.anglersatlas.com/place/102557/wolfe-lake/fish/burbot>
- Arnott, S. E., Azan, S. S. E., & Ross, A. J. (2017). Calcium decline reduces population growth rates of zooplankton in field mesocosms. *Canadian Journal of Zoology*, 95, 323–333.
- Axelsson, L. (1988). Changes in pH as a measure of photosynthesis by marine macroalgae. *Marine Biology*, 97(2), 287–294. <https://doi.org/10.1007/BF00391314>
- Bailey, R. C., Norris, R. H., & Reynoldson, T. B. (2001). Taxonomic resolution of benthic macroinvertebrate communities in bioassessments. *Journal of the North American Benthological Society*, 20(2), 280–286. <https://doi.org/10.2307/1468322>
- Bailey, S. A. (2015). An overview of thirty years of research on ballast water as a vector for aquatic invasive species to freshwater and marine environments. *Aquatic Ecosystem Health and Management*, 18(3), 261–268. <https://doi.org/10.1080/14634988.2015.1027129>
- Baldwin, D. S. (2013). Organic phosphorus in the aquatic environment. *Environmental Chemistry*, 10(6), 439–454. <https://doi.org/10.1071/EN13151>

- Ban, S., Toda, T., Koyama, M., Ishikawa, K., Kohzu, A., & Imai, A. (2019). Modern lake ecosystem management by sustainable harvesting and effective utilization of aquatic macrophytes. *Limnology*, 20(1), 93–100. <https://doi.org/10.1007/s10201-018-0557-z>
- Barmuta, L. A., Cooper, S. D., Hamilton, S. K., Kratz, K. W., & Melack, J. M. (1990). Responses of zooplankton and zoobenthos to experimental acidification in a high-elevation lake (Sierra Nevada, California, U.S.A.). *Freshwater Biology*, 23(3), 571–586. <https://doi.org/10.1111/j.1365-2427.1990.tb00296.x>
- Baron, J. S., Proff, N. L., Angermeier, P. L., Dahm, C. N., Gleick, P. H., Hairson, N. G., ... Steinman, A. D. (2002). Meeting Ecological and Social Needs for Freshwater. *Ecological Applications*, 12(5), 1247–1260. <https://doi.org/10.1890/04-0922>
- Barrett, P. R. F., Greaves, M. P., & Newman, J. R. (1999). *Aquatic Weed Control Operation Best Practice Guidelines*. Almondsbury, Bristol.
- Bergström, S. E., Svensson, J.-E., & Westberg, E. (2000). Habitat distribution of zooplankton in relation to macrophytes in an eutrophic lake. *SIL Proceedings, 1922-2010*, 27(5), 2861–2864. <https://doi.org/10.1080/03680770.1998.11898191>
- Bickel, T. O., & Closs, G. P. (2009). Impact of Removal of the Invasive Macrophyte *Lagarosiphon major* (Hydrocharitaceae) on Invertebrates and Fish. *River Research and Applications*, 25, 734–744.
- Black, A. R., & Dodson, S. I. (2003). Ethanol: a better preservation technique for Daphnia. *Limnology and Oceanography: Methods*, 1, 45–50. <https://doi.org/10.4319/lom.2011.1.45>
- Blindow, I., Hargeby, A., & Hilt, S. (2014). Facilitation of clear-water conditions in shallow lakes by macrophytes: differences between charophyte and angiosperm dominance. *Hydrobiologia*, 737(1), 99–110. <https://doi.org/10.1007/s10750-013-1687-2>
- Bockwoldt, K. A., Nodine, E. R., Mihuc, T. B., Shambaugh, A. D., & Stockwell, J. D. (2017). Reduced Phytoplankton and Zooplankton Diversity Associated with Increased Cyanobacteria in Lake Champlain, USA. *Journal of Contemporary Water Research & Education*, 160(1), 100–118. <https://doi.org/10.1111/j.1936-704x.2017.03243.x>
- Bolduc, P., Bertolo, A., & Pinel-Alloul, B. (2016). Does submerged aquatic vegetation shape zooplankton community structure and functional diversity? A test with a shallow fluvial lake system. *Hydrobiologia*, 778(1), 151–165. <https://doi.org/10.1007/s10750-016-2663-4>
- Booms, T. L. (1999). Vertebrates removed by mechanical weed harvesting in Lake Keesus, Wisconsin. *Journal of Aquatic Plant Management*, 37, 34–36.
- Borcard, D., Legendre, P., & Drapeau, P. (1992). Partialling out the Spatial Component of Ecological Variation. *Ecology*, 73(3), 1045–1055.
- Born, S. M., Wirth, T. L., Brick, E. M., & Peterson, J. O. (1973). Restoring the Recreational Potential of Small Impoundments. In *Upper Great Lakes Regional Commission*. Madison, Wisconsin.
- Boros, G., Søndergaard, M., Takács, P., Vári, Á., & Tátrai, I. (2011). Influence of submerged macrophytes, temperature, and nutrient loading on the development of redox potential

- around the sediment-water interface in lakes. *Hydrobiologia*, 665(1), 117–127.
<https://doi.org/10.1007/s10750-011-0609-4>
- Boston, H. L., & Perkins, M. A. (1982). Water column impacts of macrophyte decomposition beneath fiberglass screens. *Aquatic Botany*, 14(C), 15–27. [https://doi.org/10.1016/0304-3770\(82\)90083-3](https://doi.org/10.1016/0304-3770(82)90083-3)
- Boylen, C. W., Eichler, L. W., & Madsen, J. D. (1999). Loss of native aquatic plant species in a community dominated by Eurasian watermilfoil. *Hydrobiologia*, 415, 207–211.
<https://doi.org/10.1023/A:1003804612998>
- Bozzuto, C., & Blanckenhorn, W. U. (2017). Taxonomic resolution and treatment effects – alone and combined – can mask significant biodiversity reductions. *International Journal of Biodiversity Science, Ecosystem Services and Management*, 13(1), 86–99.
<https://doi.org/10.1080/21513732.2016.1260638>
- Breckels, R. D., & Kilgour, B. W. (2018). Aquatic herbicide applications for the control of aquatic plants in Canada: effects to nontarget aquatic organisms. *Environmental Reviews*, 26(3), 333–338. <https://doi.org/10.1139/er-2018-0002>
- Burks, R. L., Lodge, D. M., Jeppesen, E., & Lauridsen, T. L. (2002). Diel horizontal migration of zooplankton: Costs and benefits of inhabiting the littoral. *Freshwater Biology*, 47(3), 343–365. <https://doi.org/10.1046/j.1365-2427.2002.00824.x>
- Caffrey, J. M., Millane, M., Evers, S., Moran, H., & Butler, M. (2010). A novel approach to aquatic weed control and habitat restoration using biodegradable jute matting. *Aquatic Invasions*, 5(2), 123–129. <https://doi.org/10.3391/ai.2010.5.2.01>
- Cairns, A., & Yan, N. (2009). A review of the influence of low ambient calcium concentrations on freshwater daphniids, gammarids, and crayfish. *Environ. Rev.*, 79, 67–79.
<https://doi.org/10.1139/A09-005>
- Canfield, D. E. J., Langeland, K. A., Linda, S. B., & Haller, W. T. (1985). Relations Between Water Transparency and Maximum Depth of Macrophytes Colonization in Lakes. *J. Aquat. Plant Manage*, 23, 25–28.
- Caramujo, M.-J., Crispim, M. C., & Boavida, M.-J. (1997). Assessment of the importance of fish predation versus copepod predation on life history traits of *Daphnia hyalina*. *Hydrobiologia*, 360, 243–252. <https://doi.org/10.1023/A>
- Carignan, R. (1982). An Empirical Model to Estimate the Relative Importance of Roots in Phosphorus Uptake by Aquatic Macrophytes. *Canadian Journal of Fisheries and Aquatic Sciences*, 39(2), 243–247. <https://doi.org/10.1139/f82-034>
- Carignan, R., & Kalff, J. (1980). Phosphorus Sources for Aquatic Weeds : Water or Sediments? *Science*, 207(4434), 987–989.
- Carlson, R. E., & Simpson, J. (1996). A Coordinator's Guide to Volunteer Lake Monitoring Methods. *North American Lake Management Society*, 96.
- Carpenter, Stephen R.; Lodge, D. M. (1986). Effects of submersed macrophytes on ecosystem processes. *Aquatic Botany*, 26, 341–370.

- Carpenter, S. R., Cole, J. J., Pace, M. L., Batt, R., Brock, W. A., Cline, T., ... Weidel, B. (2011). Early warnings of regime shifts: A whole-ecosystem experiment. *Science*, *332*(6033), 1079–1082. <https://doi.org/10.1126/science.1203672>
- Carpenter, Stephen R. (1980). Enrichment of Lake Wingra, Wisconsin, by Submersed Macrophyte Decay. *Ecology*, *61*(5), 1145–1155. <https://doi.org/10.2307/1936834>
- Carpenter, Stephen R., & Gasith, A. (1978). Mechanical cutting of submersed macrophytes: Immediate effects on littoral water chemistry and metabolism. *Water Research*, *12*(1), 55–57. [https://doi.org/10.1016/0043-1354\(78\)90196-3](https://doi.org/10.1016/0043-1354(78)90196-3)
- Carpenter, Stephen R., & Greenlee, J. K. (1981). Lake deoxygenation after herbicide use: A simulation model analysis. *Aquatic Botany*, *11*, 173–186. [https://doi.org/10.1016/0304-3770\(81\)90058-9](https://doi.org/10.1016/0304-3770(81)90058-9)
- Carpenter, Stephen R., & Adams, M. S. (1978). Macrophyte control by harvesting and herbicides Implications for phosphorus cycling in Lake Wingra, Wisconsin. *Journal of Aquatic Plant Management*, *16*, 20–23.
- Catarauqui Region Conservation Authority. (2017). *Lake Fact Sheet (2017) Dog Lake*. Glenburnie, Ontario.
- Cattaneo, A., & Kalff, J. (1980). The relative contribution of aquatic macrophytes and their epiphytes to the production of macrophyte beds. *Limnology and Oceanography*, *25*(2), 280–289. <https://doi.org/10.4319/lo.1980.25.2.0280>
- Chambers, P. A., Lacoul, P., Murphy, K. J., & M., T. S. (2008). Global diversity of aquatic macrophytes in freshwater. *Hydrobiologia*, *595*, 9–26.
- Chambers, Patricia A., DeWreede, R. E., Irlandi, E. A., & Vandermeulen, H. (1999). Management issues in aquatic macrophyte ecology: A Canadian perspective. *Canadian Journal of Botany*, *77*(4), 471–487. <https://doi.org/10.1139/b99-092>
- Chambers, Patricia A., & Kalff, J. (1985). Depth distribution and biomass of submersed aquatic macrophyte communities in Relation to Secchi Depth¹. *Canadian Journal of Fisheries and Aquatic Sciences*, *42*(4), 701–709. <https://doi.org/10.1139/f85-090>
- Chambers, Patricia A. (1987). Light and Nutrients in the Control of Aquatic Plant Community Structure. II. In Situ Observations. *Journal of Ecology*, *75*(3), 621–628.
- Chandra, S., & Gerhardt, A. (2008). Invasive species in aquatic ecosystems: Issue of global concern. *Aquatic Invasions*, *3*(1), 1–2. <https://doi.org/10.3391/ai.2008.3.1.1>
- Chaparro-Herrera, D. J., Fernández, R., Nandini, S., & Sarma, S. S. S. (2010). Food concentration and temperature effects on the demography of *Latonopsis* cf. *australis* Sars (Cladocera: Sididae). *Hydrobiologia*, *643*(1), 55–62. <https://doi.org/10.1007/s10750-010-0136-8>
- Choi, J. Y., Jeong, K. S., Kim, S. K., La, G. H., Chang, K. H., & Joo, G. J. (2014). Role of macrophytes as microhabitats for zooplankton community in lentic freshwater ecosystems of South Korea. *Ecological Informatics*, *24*, 177–185. <https://doi.org/10.1016/j.ecoinf.2014.09.002>

- Chow-Fraser, P., & Maly, E. J. (1991). Factors Governing Clutch Size in Two Species of Diaptomus (Copepoda: Calanoida). *Can. J. Fish. Aquat. Sci.*, 48, 364–370.
- Chrislock, M. F., Doster, E., Zitomer, R. A., & Wilson, A. E. (2013). Eutrophication: Causes, Consequences, and Controls in Aquatic Ecosystems. *Nature Education Knowledge*, 4(4). <https://doi.org/10.1002/j.1551-8833.1969.tb03755.x>
- Citizens Statewide Lake Assessment Program. (2018). *CSLAP Report Otter Lake*. Retrieved from https://www.dec.ny.gov/docs/water_pdf/cslrpt18otterl.pdf
- Clayton, J. S., & Tanner, C. C. (1988). Selective control of submerged aquatic plants to enhance recreational uses of water bodies. *SIL Proceedings, 1922-2010*, 23(3), 1518–1521. <https://doi.org/10.1080/03680770.1987.11898055>
- Collen, B., Whitton, F., Dyer, E. E., Baillie, J. E. M., Cumberlidge, N., Darwall, W. R. T., ... Böhm, M. (2014). Global patterns of freshwater species diversity, threat and endemism. *Global Ecology and Biogeography*, 23(1), 40–51. <https://doi.org/10.1111/geb.12096>
- Cooke, G. Dennis. (1980). Covering Bottom Sediments as a Lake Restoration Technique. *Water Resources Bulletin*, 16(5), 921–926.
- Cooke, G. Dennis, & Gorman, M. E. (1980). Effectiveness of Dupont Typer Sheeting in Controlling Macrophyte Regrowth After Overwinter Drawdown. *American Water Resources Association*, 16(2), 353–355. <https://doi.org/10.1111/j.1752-1688.1980.tb02403.x>
- Cooke, G D, Martin, A. B., & Carlson, R. E. (1990). The effect of harvesting on macrophyte regrowth and water quality in ladue reservoir ohio USA. *J Iowa Acad Sci*, 97(4), 127–132.
- Cooper, A. L., Debues, M. J., & Sager, E. P. S. (2017). *Eurasian Watermilfoil Management on Big Cedar Lake , Ontario : Progress Report , 2017*. Peterborough, Ontario.
- Cross, T., & Mcinerny, M. (2006). *Relationships Between Aquatic Plant Cover and Fish Populations Based on Minnesota Lake Survey Data*. Hutchinson, Minnesota.
- Cuda, J. P., Charudattan, R., Grodowitz, M. J., Newman, R. M., Shearer, J. F., Tamayo, M. L., & Villegas, B. (2008). Recent Advances in Biological Control of Submersed Aquatic Weeds. *Journal of Aquatic Plant Management*, 46(1), 15–32.
- Dakos, V., Carpenter, S. R., van Nes, E. H., & Scheffer, M. (2015). Resilience indicators: Prospects and limitations for early warnings of regime shifts. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 370(1659), 1–10. <https://doi.org/10.1098/rstb.2013.0263>
- Dar, N. A., Pandit, A. K., & Ganai, B. A. (2015). Factors affecting the distribution patterns of aquatic macrophytes. *Limnological Review*, 14(2), 75–81. <https://doi.org/10.2478/limre-2014-0008>
- Data Stream. (2021). Dissolved Oxygen (DO). Retrieved July 15, 2021, from A Monitor’s Guide to Water Quality website: <https://datastream.org/en/guide/dissolved-oxygen>
- Davies, J. (1985). Evidence for a diurnal horizontal migration in *Daphnia hyalina lacustris* Sars.

Hydrobiologia, 120(2), 103–105. <https://doi.org/10.1007/BF00032130>

- Davis, M. A., & Thompson, K. (2000). Eight Ways to be a Colonizer ; Two Ways to be an Invader: A Proposed Nomenclature Scheme for Invasion Ecology. *Bulletin of the Ecological Society of America*, 81(3), 226–230.
- Declerck, S. A. J., Bakker, E. S., van Lith, B., Kersbergen, A., & van Donk, E. (2011). Effects of nutrient additions and macrophyte composition on invertebrate community assembly and diversity in experimental ponds. *Basic and Applied Ecology*, 12(5), 466–475. <https://doi.org/10.1016/j.baae.2011.05.001>
- Derry, A. M., & Arnott, S. E. (2007). Adaptive reversals in acid tolerance in copepods from lakes recovering from historical stress. *Ecological Applications*, 17(4), 1116–1126. <https://doi.org/10.1890/06-1382>
- Derry, A. M., Arnott, S. E., & Boag, P. T. (2010). Evolutionary shifts in copepod acid tolerance in an acid-recovering lake indicated by resurrected resting eggs. *Evolutionary Ecology*, 24(1), 133–145. <https://doi.org/10.1007/s10682-009-9295-3>
- Dextrase, A. J., & Mandrak, N. E. (2006). Impacts of alien invasive species on freshwater fauna at risk in Canada. *Biological Invasions*, 8(1), 13–24. <https://doi.org/10.1007/s10530-005-0232-2>
- Dodds, W. (2002). *Freshwater Ecology- Concepts and environmental Applications*. London: Academic Press Elsevier Inc.
- Dodds, W. K., Perkin, J. S., & Gerken, J. E. (2013). Human impact on freshwater ecosystem services: A global perspective. *Environmental Science and Technology*, 47(16), 9061–9068. <https://doi.org/10.1021/es4021052>
- Dokulil, M. T., & Teubner, K. (2011). Eutrophication and Climate Change: Present Situation and Future Scenarios. In *Eutrophication: causes, consequences and control* (p. 16). Vienna, Austria.
- Dorenbosch, M., & Bergsma, J. H. (2014). Bestrijding van waterwaaier in Hardinxveld met hydro-venturi. In *Bureau Waardenburg bv*. Culemborg, Netherlands.
- Duarte, C. M., & Kalff, J. (1990). Patterns in the submerged macrophyte biomass of lakes and the importance of the scale of analysis in the interpretation. *Canadian Journal of Fisheries and Aquatic Sciences*, 47(2), 357–363. <https://doi.org/10.1139/f90-037>
- Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z. I., Knowler, D. J., Lévêque, C., ... Sullivan, C. A. (2006). Freshwater biodiversity: Importance, threats, status and conservation challenges. *Biological Reviews of the Cambridge Philosophical Society*, 81(2), 163–182. <https://doi.org/10.1017/S1464793105006950>
- Dufresne, J. (2017). *Discerning Differences Between Zooplankton Communities Based on Lake Trophic Status*. University of New Hampshire.
- Dzialowski, A. R., Wang, S. H., Lim, N. C., Beury, J. H., & Huggins, D. G. (2008). Effects of sediment resuspension on nutrient concentrations and algal biomass in reservoirs of the Central Plains. *Lake and Reservoir Management*, 24(4), 313–320.

<https://doi.org/10.1080/07438140809354841>

- Eakin, H. L., & Barko, J. W. (1995). Evaluation of the Effect of Benthic Barrier Placement on Sediment Physical and Chemical Conditions. In *Aquatic Plant Control Research Program*.
- Ecottagefilms. (2021). Waterway: Dog Lake - North. Retrieved January 16, 2021, from <https://ecottagefilms.com/lake/dog-lake-north/>
- Eiswerth, M. E., Donaldson, S. G., & Johnson, W. S. (2000). Potential Environmental Impacts and Economic Damages of Eurasian Watermilfoil (*Myriophyllum spicatum*) in Western Nevada and Northeastern California 1. *Weed Technology*, *14*(3), 511–518. [https://doi.org/10.1614/0890-037x\(2000\)014\[0511:peiaed\]2.0.co;2](https://doi.org/10.1614/0890-037x(2000)014[0511:peiaed]2.0.co;2)
- Engel, S. (1983). Evaluating Stationary Blankets and Removable Screens for Macrophyte Control in Lakes 1. *J. Aquat. Plant Manage.*, Vol. 21, pp. 73–77.
- Engel, S. (1995). Eurasian Watermilfoil as a Fishery Management Tool. *Fisheries*, *20*(3), 20–27. [https://doi.org/10.1577/1548-8446\(1995\)020<0020:ewaafm>2.0.co;2](https://doi.org/10.1577/1548-8446(1995)020<0020:ewaafm>2.0.co;2)
- EPA (Environmental Protection Agency). (1976). Quality Criteria For Water. In *US Environmental Protection Agency (USEPA)*. Washington, D.C.
- EPA (Environmental Protection Agency). (2021). Indicators: Macrophytes. Retrieved September 6, 2021, from National Aquatic Resource Surveys website: <https://www.epa.gov/national-aquatic-resource-surveys/indicators-macrophytes>
- Evangelista, H. B. A., Thomaz, S. M., & Umetsu, C. A. (2014). An analysis of publications on invasive Macrophytes in Aquatic Ecosystems. *Aquatic Invasions*, *9*(4), 521–528. <https://doi.org/10.3391/ai.2014.9.4.10>
- Fairchild, G. . W. (2016). Movement and Microdistribution of *Sida Crystallina* and Other Littoral Microcrustacea. *Ecology*, *62*(5), 1341–1352.
- Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., & Holling, C. S. (2004). Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology, Evolution, and Systematics*, *35*, 557–581. <https://doi.org/10.1146/annurev.ecolsys.35.021103.105711>
- Frew, C. (2016). *Exploring the Potencial for Control of Eurasian Watermilfoil by the Milfoil Weevil in Christina Lake, British Columbia*. University of Lethbridge, Lethbridge, Alberta.
- Gabaldón, C., Devetter, M., Hejzlar, J., Šimek, K., Znachor, P., Nedoma, J., & Sed'a, J. (2019). Seasonal strengths of the abiotic and biotic drivers of a zooplankton community. *Freshwater Biology*, *64*(7), 1326–1341. <https://doi.org/10.1111/fwb.13308>
- Gallardo, B., Clavero, M., Sánchez, M. I., & Vilà, M. (2016). Global ecological impacts of invasive species in aquatic ecosystems. *Global Change Biology*, *22*(1), 151–163. <https://doi.org/10.1111/gcb.13004>
- Genkai-Kato, M., & Carpenter, S. R. (2005). Eutrophication due to phosphorus recycling in relation to lake morphometry, temperature, and macrophytes. *Ecology*, *86*(1), 210–219. <https://doi.org/10.1890/03-0545>

- Geraldes, A. M., & Boavida, M. J. (2004). Do littoral macrophytes influence crustacean zooplankton distribution? *Limnetica*, 23(1–2), 57–64.
- Gérard, J., & Triest, L. (2018). Competition between invasive *Lemna minuta* and native *L. minor* in indoor and field experiments. *Hydrobiologia*, 812(1), 57–65. <https://doi.org/10.1007/s10750-016-2754-2>
- Gettys, L. A., Haller, W. T., & Bellaud, M. (2009). Biology and Control of Aquatic Plants: A Best Management Practices Handbook. In *Biology and Control of Aquatic Plants*. Marietta, Georgia: Aquatic Ecosystem Restoration Foundation.
- Giardini, J., Yan, N. D., & Heyland, A. (2015). Consequences of calcium decline on the embryogenesis and life history of *Daphnia magna*. *The Company of Biologists*, 218, 2005–2014. <https://doi.org/10.1242/jeb.123513>
- Goodenberger, J. S., & Klaiber, H. A. (2016). Evading invasives: How Eurasian watermilfoil affects the development of lake properties. *Ecological Economics*, 127, 173–184. <https://doi.org/10.1016/j.ecolecon.2016.02.015>
- Government of Canada. (2021a). Managing aquatic vegetation on the Rideau Canal. Retrieved February 2, 2021, from Rideau Canal National Historic Site website: <https://www.pc.gc.ca/en/lhn-nhs/on/rideau/info/vegetation>
- Government of Canada. (2021b). Site management. Retrieved June 23, 2021, from Rideau Canal National Historic Site website: <https://www.pc.gc.ca/en/lhn-nhs/on/rideau/info>
- Government of Canada. (2021c). Water Management InfoNet. Retrieved June 27, 2021, from Rideau Canal National Historic Site website: <https://www.pc.gc.ca/en/lhn-nhs/on/rideau/info/infonet>
- Greer, M. J. C., Closs, G. P., Crow, S. K., & Hicks, A. S. (2012). Complete versus partial macrophyte removal: The impacts of two drain management strategies on freshwater fish in lowland New Zealand streams. *Ecology of Freshwater Fish*, 21(4), 510–520. <https://doi.org/10.1111/j.1600-0633.2012.00569.x>
- Guillaume Blanchet, F., Legendre, P., Colin Bergeron, J. A., & He, F. (2014). Consensus RDA across dissimilarity coefficients for canonical ordination of community composition data. *Ecological Monographs*, 84(3), 491–511. <https://doi.org/10.1890/13-0648.1>
- Gunnison, D., & Barko, J. W. (1992). Factors influencing gas evolution beneath a benthic barrier. *Journal of Aquatic Plant Management*, Vol. 30, pp. 23–28.
- Hall, C., & Burns, C. W. (2001). Effects of salinity and temperature on survival and reproduction of *Boeckella hamata* (Copepoda: Calanoida) from a periodically brackish lake. *Journal of Plankton Research*, 23(1), 97–103.
- Haller, W. T., Shireman, J. V., & Durant, D. F. (1980). Fish Harvest Resulting from Mechanical Control of Hydrilla. *Transactions of the American Fisheries Society*, 109(5), 517–520. [https://doi.org/10.1577/1548-8659\(1980\)109<517](https://doi.org/10.1577/1548-8659(1980)109<517)
- Hambright, K. D., Hairston, N. G., Schaffner, W. R., & Howarth, R. W. (2007). Grazer control of nitrogen fixation: Synergisms in the feeding ecology of two freshwater crustaceans.

Fundamental and Applied Limnology, 170(2), 89–101. <https://doi.org/10.1127/1863-9135/2007/0170-0089>

- Hanazato, T. (1997). Development of Low-Oxygen Layer in Lakes and its Effect on Zooplankton Communities. *Korean J. Limnol.*, 30, 506–511.
- Haney, J. F. (2013). An Image-based Key to the Zooplankton of North America. Retrieved July 20, 2019, from University of New Hampshire Center for Freshwater Biology website: <http://cfb.unh.edu/cfbkey/html/>
- Haney, James F. (1987). Field studies on zooplankton-cyanobacteria interactions. *New Zealand Journal of Marine and Freshwater Research*, 21(3), 467–475. <https://doi.org/10.1080/00288330.1987.9516242>
- Havel, J. E., Knight, S. E., & Maxson, K. A. (2017). A field test on the effectiveness of milfoil weevil for controlling Eurasian watermilfoil in Wisconsin lakes. *Hydrobiologia*, 800, 81–97. <https://doi.org/10.1007/s10750-017-3142-2>
- Heimpel, G. E., & Mills, N. J. (2017). Ecology and Applications. In *Biological Control: Ecology and Applications*. Cambridge, UK: Cambridge University Press.
- Helsel, D. R., Gerber, D. T., & Engel, S. (1996). Comparing Spring Treatments of 2,4-D with Bottom Fabrics to Control a New Infestation of Eurasian Watermilfoil. *J. Aquat. Plant Manage.*, 34, 68–71.
- Hilt, S., Brothers, S., Jeppesen, E., Veraart, A. J., & Kosten, S. (2017). Translating Regime Shifts in Shallow Lakes into Changes in Ecosystem Functions and Services. *BioScience*, 67(10), 928–936. <https://doi.org/10.1093/biosci/bix106>
- Hofstra, D. E., & Clayton, J. S. (2012). Assessment of benthic barrier products for submerged aquatic weed control. *Journal of Aquatic Plant Management*, 50, 101–105.
- Holling, C. S. (1973). Resilience and Stability of Ecological Systems. *Annu.Rev.Ecol.Syst.*, 4, 1–23.
- Hollingsworth, A., Lee, M., Tait, C., Underwood, B., & Wortsman, L. (2012). *Water Quality in the Frontenac Arch Biosphere Network*. Queen's Univeristy.
- Horppila, J., & Nurminen, L. (2003). Effects of submerged macrophytes on sediment resuspension and internal phosphorus loading in Lake Hiidenvesi (southern Finland). *Water Research*, 37(18), 4468–4474. [https://doi.org/10.1016/S0043-1354\(03\)00405-6](https://doi.org/10.1016/S0043-1354(03)00405-6)
- Horppila, J., & Nurminen, L. (2005). Effects of different macrophyte growth forms on sediment and P resuspension in a shallow lake. *Hydrobiologia*, 545(1), 167–175. <https://doi.org/10.1007/s10750-005-2677-9>
- Horsch, E. J., & Lewis, D. J. (2009). The effects of aquatic invasive species on property values: Evidence from a quasi-experiment. *Land Economics*, 85(3), 391–409. <https://doi.org/10.3368/le.85.3.391>
- Hupfer, M., & Lewandowski, J. (2008). Oxygen controls the phosphorus release from lake sediments - A long-lasting paradigm in limnology. *International Review of Hydrobiology*,

93(4–5), 415–432. <https://doi.org/10.1002/iroh.200711054>

- Hussner, A., Stiers, I., Verhofstad, M. J. J. M., Bakker, E. S., Grutters, B. M. C., Haury, J., ... Hofstra, D. (2017). Management and control methods of invasive alien freshwater aquatic plants: A review. *Aquatic Botany*, 136, 112–137. <https://doi.org/10.1016/j.aquabot.2016.08.002>
- Hussner, A., Van De Weyer, K., Gross, E. M., & Hilt, S. (2010). Comments on increasing number and abundance of non-indigenous aquatic macrophyte species in Germany. *Weed Research*, 50(6), 519–526. <https://doi.org/10.1111/j.1365-3180.2010.00812.x>
- Ivanova, M. B., & Kazantseva, T. I. (2006). Effect of water pH and total dissolved solids on the species diversity of pelagic zooplankton in lakes: A statistical analysis. *Russian Journal of Ecology*, 37(4), 264–270. <https://doi.org/10.1134/S1067413606040084>
- Jakhar, P. (2013). Role of Phytoplankton and Zooplankton as Health Indicators of Aquatic Ecosystem : A Review. *International Journal of Innovative Research & Studies*, 2(12), 490–500.
- Järvenpää, M., & Lindström, K. (2004). Water turbidity by algal blooms causes mating system breakdown in a shallow-water fish, the sand goby *Pomatoschistus minutus*. *The Royal Society*, 271, 2361–2365. <https://doi.org/10.1098/rspb.2004.2870>
- Jensen, H. S., & Andersen, F. O. (1992). Importance of temperature, nitrate, and pH for phosphate release from aerobic sediments of four shallow, eutrophic lakes. *Limnology and Oceanography*, 37(3), 577–589. <https://doi.org/10.4319/lo.1992.37.3.0577>
- Jeppesen, E., Jensen, J. P., Søndergaard, M., Lauridsen, T., & Landkildehus, F. (2000). Trophic structure, species richness and biodiversity in Danish lakes: Changes along a phosphorus gradient. *Freshwater Biology*, 45(2), 201–218. <https://doi.org/10.1046/j.1365-2427.2000.00675.x>
- Jeppesen, Erik, Søndergaard, M., Søndergaard, M., & Christoffersen, K. (1992). *The Structuring Role of Submerged Macrophytes in Lakes* (131st ed.). Springer, New York.
- Jeppesen, Erik, Søndergaard, M., Søndergaard, M., & Christoffersen, K. (1998). The Structuring Role of Submerged Macrophytes in Lakes. In *Ecological Studies* (Vol. 131).
- Jiang, X., Xie, J., Xu, Y., Zhong, W., Zhu, X., & Zhu, C. (2017). Increasing dominance of small zooplankton with toxic cyanobacteria. *Freshwater Biology*, 62(2), 429–443. <https://doi.org/10.1111/fwb.12877>
- Jones, G. B., & Cooke, G. D. (1984). Control of Nuisance Aquatic Plants With Burlap Screen. *Ohio J. Sci.*, 84(5), 248–251.
- Kagami, M., Nishihiro, J., & Yoshida, T. (2019). Ecological and limnological bases for management of overgrown macrophytes: introduction to a special feature. *Limnology*, 20(1), 1–2. <https://doi.org/10.1007/s10201-018-0565-z>
- Kamenik, C., Szeroczyńska, K., & Schmidt, R. (2007). Relationships among recent Alpine Cladocera remains and their environment: Implications for climate-change studies. *Hydrobiologia*, 594(1), 33–46. <https://doi.org/10.1007/s10750-007-9083-4>

- Karim, R., Sekine, M., & Ukita, M. (2002). Simulation of eutrophication and associated occurrence of hypoxic and anoxic condition in a coastal bay in Japan. *Marine Pollution Bulletin*, 45, 280–285.
- Karpowicz, M., Ejsmont-Karabin, J., Kozłowska, J., Feniova, I., & Dzialowski, A. R. (2020). Zooplankton community responses to oxygen stress. *Water (Switzerland)*, 12(3), 1–20. <https://doi.org/10.3390/w12030706>
- Kassambara, A. (2018). Multicollinearity Essentials and VIF in R. Retrieved March 5, 2021, from Statistical tools for high-throughput data analysis website: <http://www.sthda.com/english/articles/39-regression-model-diagnostics/160-multicollinearity-essentials-and-vif-in-r/>
- Kaushal, S. S., Likens, G. E., Pace, M. L., Utz, R. M., Haq, S., Gorman, J., & Grese, M. (2018). Freshwater salinization syndrome on a continental scale. *Proceedings of the National Academy of Sciences of the United States of America*, 115(4), E574–E583. <https://doi.org/10.1073/pnas.1711234115>
- Keast, A. (1984). The introduced aquatic macrophyte, *Myriophyllum spicatum*, as habitat for fish and their invertebrate prey. *Canadian Journal of Zoology*, 62(7), 1289–1303. <https://doi.org/10.1139/z84-186>
- Keister, J. E., & Bonnet, D. (2012). Zooplankton population connections, community dynamics, and climate variability. *Marine Climate and Climate Change*, 69(3), 347–350. <https://doi.org/10.1093/icesjms/fss034>
- Kowalczewski, A., & Ozimek, T. (1993). Further long-term changes in the submerged macrophyte vegetation of the eutrophic Lake Mikolajskie (North Poland). *Aquatic Botany*, 46(3–4), 341–345. [https://doi.org/10.1016/0304-3770\(93\)90013-M](https://doi.org/10.1016/0304-3770(93)90013-M)
- Krupska, J., Pelechaty, M., Pukacz, A., & Ossowski, P. (2012). Effects of grass carp introduction on macrophyte communities in a shallow lake. *Oceanological and Hydrobiological Studies*, 41(1), 35–40. <https://doi.org/10.2478/s13545-012-0004-4>
- Kuczyńska-Kippen, N., & Joniak, T. (2016). Zooplankton diversity and macrophyte biometry in shallow water bodies of various trophic state. *Hydrobiologia*, 774(1), 39–51. <https://doi.org/10.1007/s10750-015-2595-4>
- Kuczyńska-Kippen, N., Nagengast, B., Celewicz-Gołdyn, S., & Klimko, M. (2009). Zooplankton community structure within various macrophyte stands of a small water body in relation to seasonal changes in water level. *Oceanological and Hydrobiological Studies*, 38(3), 125–133. <https://doi.org/10.2478/v10009-009-0035-3>
- Kuiper, J. J., Verhofstad, M. J. J. M., Louwers, E. L. M., Bakker, E. S., Brederveld, R. J., van Gerven, L. P. A., ... Mooij, W. M. (2017). Mowing Submerged Macrophytes in Shallow Lakes with Alternative Stable States: Battling the Good Guys? *Environmental Management*, 59(4), 619–634. <https://doi.org/10.1007/s00267-016-0811-2>
- Laird, K. (2017). *Cataraqui region Lake Assessment Report 2017*. 56.
- Lampert, W. (1987). Laboratory studies on zooplankton-cyanobacteria interactions. *New Zealand Journal of Marine and Freshwater Research*, 21(3), 483–490.

<https://doi.org/10.1080/00288330.1987.9516244>

- Landsberg, J. H. (2010). *The Effects of Harmful Algal Blooms on Aquatic Organisms*. 10(2), 113–390. <https://doi.org/10.1080/20026491051695>
- Larry B . Crowder; William E . Cooper. (2011). Habitat Structural Complexity and the Interaction Between Bluegills and Their Prey. *Ecological Society of America*, 63(6), 1802–1813.
- Larsen, L. M. (1995). *Direct and Indirect Responses of Diaphanosoma braehvurum (Cladocera:Sididae) and Other Zooplankton to Clay Loading and Algal Food Quality in a Turbid Southeastern Reservoir*. North Carolina State University.
- Legendre, P., & Gallagher, E. D. (2001). Ecologically meaningful transformations for ordination of species data. *Oecologia*, 129(2), 271–280. <https://doi.org/10.1007/s004420100716>
- Legendre, P., & Legendre, L. (1998). Numerical Ecology. In *Elsevier* (2nd ed., Vol. 20). [https://doi.org/10.1016/S0167-8892\(12\)70001-5](https://doi.org/10.1016/S0167-8892(12)70001-5)
- Leguizamon, C. (2017). *Associations Between Invasive Eurasian Watermilfoil and Littoral Fish and Invertebrate Communities in the Keweenaw Waterway of Lake Superior*. Michigan Technological University.
- Leidonald, R., Muhtadi, A., Lesmana, I., Harahap, Z. A., & Rahmadya, A. (2019). Profiles of temperature, salinity, dissolved oxygen, and pH in Tidal Lakes. *IOP Conference Series: Earth and Environmental Science*, 260(012075). <https://doi.org/10.1088/1755-1315/260/1/012075>
- Likens, G. E., Wright, R. F., Galloway, J. N., & Butler, T. J. (1979). Acid Rain. *Scientific American*, 241(4), 43–51.
- Lima, A. F., Lansac-Tôha, F. A., Machado Velho, L. F., Bini, L. M., & Takeda, A. M. (2003). Composition and abundance of Cladocera (Crustacea) assemblages associated with Eichhornia azurea (Swartz) Kunth stands in the Upper Paraná River floodplain. *Acta Scientiarum - Biological Sciences*, 25(1), 41–48. <https://doi.org/10.4025/actascibiolsci.v25i1.2118>
- Locke, A., & Sprules, W. G. (2000). Effects of acidic ph and phytoplankton on survival and condition of bosmina longirostris and daphnia pulex. *Hydrobiologia*, 437, 187–196. <https://doi.org/10.1023/A:1026563109217>
- Lutz, R. V., Marcus, N. H., & Chanton, J. P. (1992). Effects of low oxygen concentrations on the hatching and viability of eggs of marine calanoid copepods. *Marine Biology*, 114(2), 241–247. <https://doi.org/10.1007/BF00349525>
- Madsen, J. D., Hartleb, C. F., & Boylen, C. W. (1991). Photosynthetic characteristics of Myriophyllum spicatum and six submersed aquatic macrophyte species native to Lake George, New York. *Freshwater Biology*, 26(2), 233–240. <https://doi.org/10.1111/j.1365-2427.1991.tb01732.x>
- Madsen, John D. (2000). *Advantages and Disadvantages of Aquatic Plant Management Techniques*. Vicksburg, MS: U.S Army Engineer Research and Development Center.

- Madsen, John Douglas, Sutherland, K. W., Bloomfield, J. A., Eichler, L. W., & Boylen, C. W. (1991). The decline of native vegetation under dense Eurasian watermilfoil canopies. *J. Aquat. Plant Manage.*, 29, 94–99.
- Malcolm/Ardoch Lake Stewardship Committee. (2016). *The Lake PPlan Malcolm and Ardoch Lakes Background Document*.
- Malcolm and Ardoch Lakes Landowner's Association. (2018). *Eurasian water milfoil management plan (2018-2020) 1*. Retrieved from file:///C:/Users/Eric/Documents/_School/Masters/Thesis/All Papers/Literature Review/regulations/25ee1f911c6f5f44976b3c863ec48ef4.pdf
- Malley, D. F., & Chang, P. S. S. (1986). Increase in the Abundance of Cladocera a pH 5.1 in Experimentally-Acidified Lake 223, Experimental Lakes Area, Ontario. *Water, Air, and Soil Pollution*, 30, 629–638.
- Mamani, A., Koncurat, M. L., & Boveri, M. (2019). Combined effects of fish and macroinvertebrate predation on zooplankton in a littoral mesocosm experiment. *Hydrobiologia*, 829(1), 19–29. <https://doi.org/10.1007/s10750-018-3712-y>
- Mammoser, E. (2013). Effects of grass carp use in midwest reservoirs. Iowa State University.
- Matsuda, J. T., Lansac-Tôha, F. A., Martens, K., Velho, L. F. M., Mormul, R. P., & Higuti, J. (2015). Association of body size and behavior of freshwater ostracods (Crustacea, Ostracoda) with aquatic macrophytes. *Aquatic Ecology*, 49(3), 321–331. <https://doi.org/10.1007/s10452-015-9527-2>
- Matsuzaki, S. I. S., Usio, N., Takamura, N., & Washitani, I. (2009). Contrasting impacts of invasive engineers on freshwater ecosystems: An experiment and meta-analysis. *Oecologia*, 158(4), 673–686. <https://doi.org/10.1007/s00442-008-1180-1>
- Mayer, J. R. (1978). Aquatic Weed Management By Benthic Semi-Barriers. *Journal of Aquatic Plant Management*, Vol. 16, pp. 31–33.
- McAbendroth, L., Ramsay, P. M., Foggo, A., Rundle, S. D., & Bilton, D. T. (2005). Does macrophyte fractal complexity drive invertebrate diversity, biomass and body size distributions? *Oikos*, 111(2), 279–290. <https://doi.org/10.1111/j.0030-1299.2005.13804.x>
- McAllen, R., & Brennan, E. (2009). The effect of environmental variation on the reproductive development time and output of the high-shore rockpool copepod *Tigriopus brevicornis*. *Journal of Experimental Marine Biology and Ecology*, 368(1), 75–80. <https://doi.org/10.1016/j.jembe.2008.10.013>
- Michalek, K., Ventura, A., & Sanders, T. (2016). *Mytilus* hybridisation and impact on aquaculture: A minireview. *Marine Genomics*, 27, 3–7. <https://doi.org/10.1016/j.margen.2016.04.008>
- Miller, M. A., Kudela, R. M., Mekebri, A., Crane, D., Oates, S. C., Timothy, M., ... Jessup, D. A. (2010). Evidence for a Novel Marine Harmful Algal Bloom : Cyanotoxin (Microcystin) Transfer from Land to Sea Otters. *PLoS ONE*, 5(9). <https://doi.org/10.1371/journal.pone.0012576>

- Miranda, L. E., Driscoll, M. P., & Allen, M. S. (2000). Transient physicochemical microhabitats facilitate fish survival in inhospitable aquatic plant stands. *Freshwater Biology*, 44(4), 617–628. <https://doi.org/10.1046/j.1365-2427.2000.00606.x>
- Moody, M. L., & Les, D. H. (2002). Evidence of hybridity in invasive watermilfoil (Myriophyllum) populations. *Proceedings of the National Academy of Sciences of the United States of America*, 99(23), 14867–14871. <https://doi.org/10.1073/pnas.172391499>
- Moore, M. V., Folt, C. L., & Stemberger, R. S. (1996). Consequences of elevated temperatures for zooplankton assemblages in temperate lakes. *Arch. Hydrobiol.*, 135(3), 289–319. <https://doi.org/10.1127/archiv-hydrobiol/135/1996/289>
- Morris, K., Bailey, P. C., Boon, P. I., & Hughes, L. (2003). Alternative stable states in the aquatic vegetation of shallow urban lakes. II. Catastrophic loss of aquatic plants consequent to nutrient enrichment. *Marine and Freshwater Research*, 54(3), 201–215. <https://doi.org/10.1071/MF02003>
- Murphy, K. J. (1988). Aquatic weed problems and their management: a review. II. Physical control measures. *Crop Protection*, 7(5), 283–302. [https://doi.org/10.1016/0261-2194\(88\)90075-0](https://doi.org/10.1016/0261-2194(88)90075-0)
- Nakai, S., Inoue, Y., Hosomi, M., & Murakami, A. (2000). Myriophyllum spicatum-released allelopathic polyphenols inhibiting growth of blue-green algae Microcystis aeruginosa. *Water Research*, 34(11), 3026–3032. [https://doi.org/10.1016/S0043-1354\(00\)00039-7](https://doi.org/10.1016/S0043-1354(00)00039-7)
- Nentwig W. (2007). General conclusions or what has to be done? In *Biological Invasions* (Vol. 193). <https://doi.org/10.1017/CBO9781107415324.004>
- Nes, E. H. Van, Berg, M. S. Van Den, Clayton, J. S., & Coops, H. (1999). A simple model for evaluating the costs and benefits of aquatic macrophytes. *Hydrobiologia*, 415, 335–339.
- Netherland, M. D., & Willey, L. (2017). Mesocosm evaluation of three herbicides on Eurasian watermilfoil (Myriophyllum spicatum) and hybrid watermilfoil (Myriophyllum spicatum × myriophyllum sibiricum): Developing a predictive assay. *Journal of Aquatic Plant Management*, 55, 39–41.
- Newman, R. M. (2004). Biological control of Eurasian watermilfoil by aquatic insects: Basic insights from an applied problem. *Archiv Fur Hydrobiologie*, 159(2), 145–184. <https://doi.org/10.1127/0003-9136/2004/0159-0145>
- Nichols, D. S., & Keeney, D. R. (1973). Nitrogen and phosphorus release from decaying water milfoil. *Hydrobiologia*, 42(4), 509–525. <https://doi.org/10.1007/BF00047023>
- Nichols, S. A. (1974). *Mechanical and habitat manipulation for aquatic plant management a review of techniques*. Retrieved from <https://dnr.wi.gov/files/PDF/pubs/ss/SS0077.pdf>
- Nichols, S. A., & Buchan, L. A. J. (1997). Use of native macrophytes as indicators of suitable Eurasian watermilfoil habitat in Wisconsin Lakes. *Journal of Aquatic Plant Management*, 35(1), 21–24.
- Nichols, S. A., & Shaw, B. H. (1986). Ecological life histories of the three aquatic nuisance plants, Myriophyllum spicatum, Potamogeton crispus and Elodea canadensis.

- Hydrobiologia*, 131(1), 3–21. <https://doi.org/10.1007/BF00008319>
- O'Hare, M. T., Aguiar, F. C., Asaeda, T., Bakker, E. S., Chambers, P. A., Clayton, J. S., ... Wood, K. A. (2018). Plants in aquatic ecosystems: current trends and future directions. *Hydrobiologia*, 812(1). <https://doi.org/10.1007/s10750-017-3190-7>
- Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., ... Wagner, H. (2019). vegan: Community Ecology Package. Retrieved September 2, 2020, from The Comprehensive R Archive Network website: <https://cran.r-project.org/package=vegan>
- Olsgard, F., & Somerfield, P. J. (2000). Surrogates in marine benthic investigations - Which taxonomic unit to target? *Journal of Aquatic Ecosystem Stress and Recovery*, 7(1), 25–42. <https://doi.org/10.1023/A:1009967313147>
- Olson, M. H., Carpenter, S. R., Cunningham, P., Gafny, S., Herwig, B. R., Nibbelink, N. P., ... Wilson, K. A. (1998). Managing Macrophytes to Improve Fish Growth: A Multi-lake Experiment. *Fisheries*, 23(2), 6–12. [https://doi.org/10.1577/1548-8446\(1998\)023<0006:mmtifg>2.0.co;2](https://doi.org/10.1577/1548-8446(1998)023<0006:mmtifg>2.0.co;2)
- Ontario. (2014). Pesticide licences and permits. Retrieved February 3, 2021, from Pesticides website: <https://www.ontario.ca/page/pesticide-licences-and-permits>
- Ontario. (2017). Remove native aquatic plants. Retrieved February 2, 2021, from Crown Land website: <https://www.ontario.ca/page/remove-native-aquatic-plants>
- Ontario. (2021). Invasive species in Ontario.
- Othman, R., Shaharuddin, R. I., & Mukrim, Z. (2015). Assessment OF Aquatic Ecosystem Status Using Macrophyte Species as Key Tools. *Jurnal Teknologi*, 77(30), 119–123.
- Otter Lake Landowners Association. (2020). General Information. Retrieved November 5, 2020, from <https://otterlake.org/about/general-information/>
- Otter Lake Landowners Association. (2021). *State of the Lake Report: Otter Lake (2nd Edition)*. https://doi.org/10.1007/978-94-007-0753-5_104023
- Padisak, J., & Colin S. Reynolds. (2003). Shallow lakes: the absolute, the relative, the functional and the pragmatic. *Hydrobiologia*, 506–509, 1–11. <https://doi.org/10.1023/B>
- Painter, D. (1988). Long-term effects of mechanical harvesting on Eurasian watermilfoil. *J. Environ. Manage*, Vol. 15, pp. 263–271.
- Painter, D. S., & McCabe, K. J. (1988). Investigation into the disappearance of Eurasian watermilfoil from the Kawartha Lakes. *Journal of Aquatic Plant Management*, Vol. 26, pp. 3–12.
- Pardue, W. J., & David H. Webb. (1985). A Comparison of Aquatic Macroinvertebrates Occurring in Association with Eurasian Watermilfoil (*Myriophyllum spicatum* L.) with Those Found in the Open Littoral Zone. *Journal of Freshwater Ecology*, 3(1), 69–79.
- Paturej, E., Gutkowska, A., Koszałka, J., & Bowszys, M. (2017). Effect of physicochemical parameters on zooplankton in the brackish, coastal Vistula Lagoon. *Oceanologia*, 59(1), 49–56. <https://doi.org/10.1016/j.oceano.2016.08.001>

- Pejchar, L., & Mooney, H. A. (2009). Invasive species, ecosystem services and human well-being. *Trends in Ecology and Evolution*, 24(9), 497–504. <https://doi.org/10.1016/j.tree.2009.03.016>
- Pennak, R. W. (1966). Structure of Zooplankton Populations in the Littoral Macrophyte Zone of Some Colorado Lakes. *American Microscopical Society*, 85, 329–349.
- Peterson, G., Allen, C. R., & Holling, C. S. (1998). Ecological resilience, biodiversity, and scale. *Ecosystems*, 1(1), 6–18. <https://doi.org/10.1007/s100219900002>
- Phillips, G., Willby, N., & Moss, B. (2016). Submerged macrophyte decline in shallow lakes: What have we learnt in the last forty years? *Aquatic Botany*, 135, 37–45. <https://doi.org/10.1016/j.aquabot.2016.04.004>
- Pimentel, D. (2005). Aquatic nuisance species in the New York State Canal and Hudson River systems and the Great Lakes Basin: An economic and environmental assessment. *Environmental Management*, 35(5), 692–701. <https://doi.org/10.1007/s00267-004-0214-7>
- R. I. Colautti, & H. J. MacIsaac. (2004). A neutral terminology to define ‘invasive’ species. *Diversity and Distributions*, 10, 135–141. Retrieved from www.blackwellpublishing.com/ddi
- R Core Team. (2020). R: A language and environment for statistical computing. *R Foundation for Statistical Computing*. Vienna, Austria.
- Radomski, P., & Perleberg, D. (2019). Avoiding the invasive trap: Policies for aquatic non-indigenous plant management. *Environmental Values*, 28(2), 211–232. <https://doi.org/10.3197/096327119X15515267418539>
- Rahel, F. J., & Olden, J. D. (2008). Assessing the effects of climate change on aquatic invasive species. *Conservation Biology*, 22(3), 521–533. <https://doi.org/10.1111/j.1523-1739.2008.00950.x>
- Rao, C. (1995). A review of canonical coordinates and an alternative to correspondence analysis using Hellinger distance. *Questiò: Quaderns d'Estadística, Sistemes, Informàtica i Investigació Operativa*, 19(1), 23–63.
- Reid, A. J., Carlson, A. K., Creed, I. F., Eliason, E. J., Gell, P. A., Johnson, P. T. J., ... Cooke, S. J. (2019). Emerging threats and persistent conservation challenges for freshwater biodiversity. *Biological Reviews*, 94(3), 849–873. <https://doi.org/10.1111/brv.12480>
- Rideau Valley Conservation Authority. (2014). *Rideau Lakes Subwatershed Report 2014 - Wolfe Lake Catchment*. Ottawa, Ontario.
- Rideau Valley Conservation Authority. (2015). *Otter Creek Catchment - Middle Rideau Watershed Report*. Retrieved from <https://watersheds.rvca.ca/subwatersheds/middle-rideau/catchment-reports-middle-rideau/91-middle-rideau/catchments/otter-creek>
- Ries, J. B. (2012). Shell-shocked: How different creatures deal with an acidifying ocean. *Earth Magazine*. Retrieved from <https://www.earthmagazine.org/article/shell-shocked-how-different-creatures-deal-acidifying-ocean/>

- Robertson-Bryan Inc. (2004). pH Requirements of Freshwater Aquatic Life. In *Robertson-Bryan, Inc.* Elk Grove.
- Roman, M. R., Gauzens, A. L., Rhinehart, W. K., & White, J. R. (1993). Effects of low oxygen waters on Chesapeake Bay zooplankton. *Limnology and Oceanography*, 38(8), 1603–1614. <https://doi.org/10.4319/lo.1993.38.8.1603>
- Rooney, N., Kalff, J., & Habel, C. (2003). The role of submerged macrophyte beds in phosphorus and sediment accumulation in Lake Memphremagog, Quebec, Canada. *Limnology and Oceanography*, 48(5), 1927–1937. <https://doi.org/10.4319/lo.2003.48.5.1927>
- Rosaen, A. L., Grover, E. A., & Spencer, C. W. (2012). The Costs of Aquatic Invasive Species to Great Lakes States. *Anderson Economic Group LLC*, (517), 1–51.
- Rydell, N. J. (2018). Effects of 2, 4-D Herbicide Treatments Used to Control Eurasian Watermilfoil on Fish and Zooplankton in Northern Wisconsin Lakes (University of Wisconsin). <https://doi.org/10.1017/CBO9781107415324.004>
- Sagrario, G., De Los Angeles, M., Balseiro, E., Ituarte, R., & Spivak, E. (2009). Macrophytes as refuge or risky area for zooplankton: A balance set by littoral predacious macroinvertebrates. *Freshwater Biology*, 54(5), 1042–1053. <https://doi.org/10.1111/j.1365-2427.2008.02152.x>
- Sand-Jensen, K., & Søndergeerd, M. (1981). Phytoplankton and Epiphyte Development and Their Shading Effect on Submerged Macrophytes in Lakes of Different Nutrient Status. *Internationale Revue Der Gesamten Hydrobiologie Und Hydrographie*, 66(4), 529–552.
- Savino, J. F., & Stein, R. A. (1982). Transactions of the Predator-Prey Interaction between Largemouth Bass. *Transactions of the American Fisheries Society*, 111, 255–266. [https://doi.org/10.1577/1548-8659\(1982\)111<255](https://doi.org/10.1577/1548-8659(1982)111<255)
- Schallenberg, M., De Winton, M. D., Verburg, P., Kelly, D. J., Hamill, K. D., & Hamilton, D. P. (2013). Ecosystem Services of Lakes. *Ecosystem Services in New Zealand - Conditions and Trends*, 203–225.
- Scheffer, M. (1990). Multiplicity of stable states in freshwater systems BT - Biomanipulation Tool for Water Management. *Hydrobiologia*, 200–201, 475–486.
- Scheffer, M. (2001). Alternative Attractors of Shallow Lakes. *TheScientificWorldJournal*, 1, 254–263. <https://doi.org/10.1100/tsw.2001.62>
- Scheffer, M., Carpenter, S., Foley, J. A., Folke, C., & Walker, B. (2001). Catastrophic shifts in ecosystems. *Nature*, 413(6856), 591–596. <https://doi.org/10.1038/35098000>
- Scheffer, M., & Carpenter, S. R. (2003). Catastrophic regime shifts in ecosystems: Linking theory to observation. *Trends in Ecology and Evolution*, 18(12), 648–656. <https://doi.org/10.1016/j.tree.2003.09.002>
- Schindler, D. W. (1974). Eutrophication and recovery in experimental lakes: Implications for lake management. *Science*, 184(4139), 897–899. <https://doi.org/10.1126/science.184.4139.897>

- Schindler, D. W. (1977). Evolution of Phosphorus Limitation in Lakes. *Science*, 195(4275), 260–262.
- Schmidt, J. C. (1984). An overview of chemicals for aquatic plant control. *Lake and Reservoir Management*, 1(1), 453–455. <https://doi.org/10.1080/07438148409354555>
- Schriver, P., Bøgestrand, J., Jeppesen, E., & Søndergaard, M. (1995). Impact of submerged macrophytes on fish-zooplankton phytoplankton interactions: large-scale enclosure experiments in a shallow eutrophic lake. *Freshwater Biology*, 33(2), 255–270. <https://doi.org/10.1111/j.1365-2427.1995.tb01166.x>
- Schultz, R., & Dibble, E. (2012). Effects of invasive macrophytes on freshwater fish and macroinvertebrate communities: The role of invasive plant traits. *Hydrobiologia*, 684(1), 1–14. <https://doi.org/10.1007/s10750-011-0978-8>
- Sesin, V., Dalton, R. L., Boutin, C., Robinson, S. A., Bartlett, A. J., & Pick, F. R. (2018). Macrophytes are highly sensitive to the herbicide diquat dibromide in test systems of varying complexity. *Ecotoxicology and Environmental Safety*, 165(August), 325–333. <https://doi.org/10.1016/j.ecoenv.2018.08.033>
- Shitao, P., Chunguang, L., & Richard, S. (2012). Stopping the Tide of Aquatic Macrophytes : Be Smarter in choosing Proper Control Methods. *Disaster Advances*, 5(4), 157–161.
- Simkovic, V. (2020). Eurasian Water-Milfoil (*Myriophyllum spicatum*): Best Management Practices in Ontario. In *Ontario Invasive Plant Council*. Peterborough, Ontario.
- Smith, C. S., & Barko, J. W. (1990). Ecology of Eurasian watermilfoil. *Journal of Aquatic Plant Management*, Vol. 28, pp. 55–64.
- Smith, E. P. (2006). BACI Design. In *Encyclopedia of Environmetrics* (pp. 1–12). <https://doi.org/doi:10.1002/9780470057339.vab001>.
- Søndergaard, M., Kristensen, P., & Jeppesen, E. (1992). Phosphorus release from resuspended sediment in the shallow and wind-exposed Lake Arresø, Denmark. *Hydrobiologia*, 228(1), 91–99. <https://doi.org/10.1007/BF00006480>
- Stansfield, J. H., Perrow, M. R., Tench, L. D., Jowitt, A. J. D., & Taylor, A. A. L. (1997). Submerged macrophytes as refuges for grazing Cladocera against fish predation: Observations on seasonal changes in relation to macrophyte cover and predation pressure. *Hydrobiologia*, 342–343, 229–240. https://doi.org/10.1007/978-94-011-5648-6_25
- Suen, C., & Gillett-Kaufman, J. L. (2019). Seed Shrimp, Mussel Shrimp (Freshwater Ostracods) scientific name: (Crustacea: Ostracoda: Podocopa). *Edis*, EENY-734, 3. <https://doi.org/10.32473/edis-in1260-2019>
- Tadesse, I., Green, F. B., & Puhakka, J. A. (2004). Seasonal and diurnal variations of temperature, pH and dissolved oxygen in advanced integrated wastewater pond system treating tannery effluent. *Water Research*, 38(3), 645–654. <https://doi.org/10.1016/j.watres.2003.10.006>
- Takamura, N., Kadono, Y., Fukushima, M., Nakagawa, M., & Kim, B. H. O. (2003). Effects of aquatic macrophytes on water quality and phytoplankton communities in shallow lakes.

- Ecological Research*, 18(4), 381–395. <https://doi.org/10.1046/j.1440-1703.2003.00563.x>
- Tessier, A., & Horwitz, R. I. (1990). Influence of Water Chemistry on Size Structure of plankton Assemblages. *Can. J. Fish. Aquat. Sci.*, 47, 1937–1943.
- Thompson, T. G., & Bonnar, R. U. (1931). The buffer capacity of sea water. *Industrial and Engineering Chemistry - Analytical Edition*, 3(4), 393–395. <https://doi.org/10.1021/ac50076a029>
- Timms, R. M., & Moss, B. (1984). Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in a shallow wetland ecosystem. *Limnology and Oceanography*, 29(3), 472–486. <https://doi.org/10.4319/lo.1984.29.3.0472>
- Township of Rideau Lakes. (2021). *State of the Lake Report: Otter Lake*.
- Turner, A. M., Cholak, E. J., & Groner, M. (2010). Expanding American Lotus and Dissolved Oxygen Concentrations of a Shallow Lake. *The American Midland Naturalist*, 164(1), 1–8.
- Ussery, T. A., Eakin, H. L., Payne, B. S., Miller, A. C., & Barko, J. W. (1997). Effects of benthic barriers on aquatic habitat conditions and macroinvertebrate communities. *Journal of Aquatic Plant Management*, 35(2), 69–73.
- Valley, R. D., & Drake, M. T. (2007). What does resilience of a clear-water state in lakes mean for the spatial heterogeneity of submersed macrophyte biovolume? *Aquatic Botany*, 87(4), 307–319. <https://doi.org/10.1016/j.aquabot.2007.07.003>
- van de Leemput, I. A., Dakos, V., Scheffer, M., & van Nes, E. H. (2018). Slow Recovery from Local Disturbances as an Indicator for Loss of Ecosystem Resilience. *Ecosystems*, 21(1), 141–152. <https://doi.org/10.1007/s10021-017-0154-8>
- Van Nes, E. H., & Scheffer, M. (2007). Slow recovery from perturbations as a generic indicator of a nearby catastrophic shift. *American Naturalist*, 169(6), 738–747. <https://doi.org/10.1086/516845>
- Van Nes, E. H., Scheffer, M., Van den Berg, M. S., & Coops, H. (2002). Aquatic macrophytes: Restore, eradicate or is there a compromise? *Aquatic Botany*, 72(3–4), 387–403. [https://doi.org/10.1016/S0304-3770\(01\)00212-1](https://doi.org/10.1016/S0304-3770(01)00212-1)
- Vanni, M. J. (2016). *Effects of Nutrients and Zooplankton Size on the Structure of a Phytoplankton Community*. 68(3), 624–635.
- Vaughan, I. P., & Ormerod, S. J. (2005). Increasing the value of principal components analysis for simplifying ecological data: a case study with rivers and river birds. *Journal of Applied Ecology*, 42, 487–497.
- Verhofstad, M. J. J. M., & Bakker, E. S. (2019). Classifying nuisance submerged vegetation depending on ecosystem services. *Limnology*, 20(1), 55–68. <https://doi.org/10.1007/s10201-017-0525-z>
- Vilas, M. P., Marti, C. L., Oldham, C. E., & Hipsey, M. R. (2017). Macrophyte-induced thermal stratification in a shallow urban lake promotes conditions suitable for nitrogen-fixing

- cyanobacteria. *Hydrobiologia*, 806(1), 411–426. <https://doi.org/10.1007/s10750-017-3376-z>
- Vilas, Maria P., Marti, C. L., Adams, M. P., Oldham, C. E., & Hipsey, M. R. (2017). Invasive macrophytes control the spatial and temporal patterns of temperature and dissolved oxygen in a shallow lake: A proposed feedback mechanism of macrophyte loss. *Frontiers in Plant Science*, 8(December), 1–14. <https://doi.org/10.3389/fpls.2017.02097>
- Walker, B., Holling, C. S., Carpenter, S. R., & Kinzig, A. (2004). Resilience, Adaptability and Transformability in Social–ecological Systems. *Ecology and Society*, 9(2), 5. <https://doi.org/10.1103/PhysRevLett.95.258101>
- Wall, M. (2021). Dog Lake at a Glance. Retrieved from Dog & Cranberry Lakes Association website: <https://www.dogandcranberrylakes.ca/p/Dog-Lake>
- Weinberger, D. (1980, April). The green green slime of home. *Maclean's*, 10. Retrieved from <https://archive.macleans.ca/article/1980/4/28/the-green-green-slime-of-home>
- Wetzel, R. G. (2001). Limnology. In *Academic Press*. San Diego, CA.
- Wichner, K. F., Seibel, B. A., Roman, C., Deutsch, C., Outram, D., Shaw, C. T., ... Riley, S. (2018). Ocean deoxygenation and zooplankton: Very small oxygen differences matter. *Science Advances*, 4, 1–8.
- Williamson, C. E., Dodds, W., Kratz, T. K., & Palmer, M. A. (2008). Lakes and streams as sentinels of environmental change in terrestrial and atmospheric processes. *Frontiers in Ecology and the Environment*, 6(5), 247–254. <https://doi.org/10.1890/070140>
- Williamson, M. (1999). Invasions. *Ecography*, 22(1), 5–12. <https://doi.org/10.1111/j.1600-0587.1999.tb00449.x>
- Wilson, M. A., & Carpenter, S. R. (1999). Economic valuation of freshwater ecosystem services in the United States: 1971–1997. *Ecological Applications*, 9(3), 772–783. [https://doi.org/10.1890/1051-0761\(1999\)009\[0772:EVOFES\]2.0.CO;2](https://doi.org/10.1890/1051-0761(1999)009[0772:EVOFES]2.0.CO;2)
- Wilson, S. J., & Ricciardi, A. (2009). Epiphytic macroinvertebrate communities on Eurasian watermilfoil (*Myriophyllum spicatum*) and native milfoils *Myriophyllum sibiricum* and *Myriophyllum alterniflorum* in eastern North America. *Canadian Journal of Fisheries and Aquatic Sciences*, 66(1), 18–30. <https://doi.org/10.1139/F08-187>
- Wolfenbarger, W. C. (1999). Influences of biotic and abiotic factors on seasonal succession of zooplankton in Hugo Reservoir, Oklahoma, U.S.A. *Hydrobiologia*, 400, 13–31. <https://doi.org/10.1023/A:1003738608697>
- Wood, J. D. (2017). *Biology and Management of Monoecious Hydrilla*. University of Florida.
- Wood, J. D., & Netherland, M. D. (2017). How long do shoot fragments of hydrilla (*Hydrilla verticillata*) & Eurasian watermilfoil (*Myriophyllum spicatum*) remain buoyant? *Journal of Aquatic Plant Management*, 55(July), 76–82.
- Yang, J., Zhang, X., Xie, Y., Song, C., Zhang, Y., Yu, H., & Burton, G. A. (2017). Zooplankton community profiling in a eutrophic freshwater ecosystem-lake tai basin by DNA metabarcoding. *Scientific Reports*, 7(1), 1–11. <https://doi.org/10.1038/s41598-017-01808-y>

Zhu, J., Liu, B., Wang, J., Gao, Y., & Wu, Z. (2010). Study on the mechanism of allelopathic influence on cyanobacteria and chlorophytes by submerged macrophyte (*Myriophyllum spicatum*) and its secretion. *Aquatic Toxicology*, 98(2), 196–203.
<https://doi.org/10.1016/j.aquatox.2010.02.011>

Ziegler, J. P., Solomon, C. T., Finney, B. P., & Gregory-Eaves, I. (2015). Macrophyte biomass predicts food chain length in shallow lakes. *Ecosphere*, 6(1), 1–16.
<https://doi.org/10.1890/ES14-00158.1>

Appendix A -Malcolm Lake

Table A-1. Abiotic variables and zooplankton density (3 replicates) from Malcolm Lake.

| Sample Time | Sample Site | Date | Temp. (°C) | DO (%) | Cond. (µS) | pH | TN (mg/L) | TP (mg/L) | Zooplankton Density (per L) |
|-------------|-------------|------------|------------|--------|------------|------|-----------|-----------|-----------------------------|
| before | control | 2019-06-08 | 17.7 | 95.5 | 175.8 | 8.46 | 0.81 | 0.016 | 30.88 |
| before | control | 2019-06-08 | 18 | 111 | 171.9 | 8.75 | 0.9 | 0.019 | 39.00 |
| before | control | 2019-06-08 | 17.8 | 101.9 | 172.3 | 8.67 | 0.91 | 0.019 | 4.38 |
| before | bare | 2019-06-08 | 17.8 | 104.4 | 173.9 | 8.64 | 0.46 | 0.005 | N/A |
| before | bare | 2019-06-08 | 17.9 | 9.7 | 174.8 | 8.61 | 0.42 | 0.006 | N/A |
| before | bare | 2019-06-08 | 17.8 | 9.9 | 174.5 | 8.63 | 0.45 | 0.006 | N/A |
| before | impact | 2019-06-08 | 17.7 | 106.1 | 172.5 | 8.65 | 0.55 | 0.006 | 43.83 |
| before | impact | 2019-06-08 | 17.8 | 103.9 | 173.9 | 8.67 | 0.55 | 0.008 | 46.54 |
| before | impact | 2019-06-08 | 17.6 | 107.6 | 172.4 | 8.72 | 0.55 | 0.006 | 45.67 |
| after | control | 2019-07-03 | 24.5 | 57.4 | 289.1 | 7.28 | 0.7 | 0.015 | 81.08 |
| after | control | 2019-07-03 | 24.8 | 86.6 | 206.8 | 8.58 | 0.55 | 0.013 | 42.63 |
| after | control | 2019-07-03 | 24.7 | 65.2 | 209.5 | 8.49 | 0.58 | 0.014 | 40.67 |
| after | bare | 2019-07-03 | 25.1 | 99.8 | 199.7 | 8.58 | 0.5 | 0.011 | 15.79 |
| after | bare | 2019-07-03 | 25.1 | 100.8 | 199.3 | 8.62 | 0.5 | 0.01 | 15.42 |
| after | bare | 2019-07-03 | 25.2 | 90.4 | 200.3 | 8.58 | 0.51 | 0.01 | 13.75 |
| after | impact | 2019-07-03 | 25 | 103.6 | 196.8 | 8.63 | 0.49 | 0.009 | 70.00 |
| after | impact | 2019-07-03 | 24.8 | 82.1 | 205.4 | 8.23 | 0.51 | 0.009 | 37.42 |
| after | impact | 2019-07-03 | 25 | 94 | 197.3 | 8.6 | 0.54 | 0.008 | 53.79 |
| after | control | 2019-08-01 | 24.9 | 87.5 | 223.2 | 8.16 | 0.46 | 0.007 | 31.73 |

| | | | | | | | | | |
|-------|---------|------------|------|-------|-------|------|------|-------|-------|
| after | control | 2019-08-01 | 24.9 | 57 | 241 | 7.7 | 0.56 | 0.007 | 76.36 |
| after | control | 2019-08-01 | 24.9 | 76.5 | 229.6 | 7.99 | 0.44 | 0.006 | 47.41 |
| after | bare | 2019-08-01 | 24.9 | 96.4 | 195.2 | 8.71 | 0.52 | 0.008 | 15.50 |
| after | bare | 2019-08-01 | 24.9 | 105.3 | 195.1 | 8.71 | 0.49 | 0.007 | 13.32 |
| after | bare | 2019-08-01 | 24.9 | 98.1 | 195.5 | 8.7 | 0.47 | 0.008 | 16.18 |
| after | impact | 2019-08-01 | 25 | 90.5 | 197.7 | 8.56 | 0.49 | 0.008 | 23.59 |
| after | impact | 2019-08-01 | 25 | 77 | 20.5 | 8.42 | 0.52 | 0.009 | 33.18 |
| after | impact | 2019-08-01 | 25.1 | 86.2 | 199.2 | 8.56 | 0.48 | 0.007 | 44.59 |
| after | control | 2019-09-14 | 18.1 | 30 | 295.1 | 7.48 | 0.55 | 0.01 | 30.88 |
| after | control | 2019-09-14 | 18.3 | 23 | 368 | 7.17 | 0.54 | 0.009 | 39.00 |
| after | control | 2019-09-14 | 18.1 | 46.3 | 279.1 | 7.82 | 0.49 | 0.009 | 4.38 |
| after | bare | 2019-09-14 | 18.1 | 86.3 | 167.7 | 8.45 | 0.47 | 0.009 | NA |
| after | bare | 2019-09-14 | 18.1 | 85.5 | 167.8 | 8.45 | 0.48 | 0.009 | NA |
| after | bare | 2019-09-14 | 18.1 | 84.9 | 167.8 | 8.45 | 0.51 | 0.009 | NA |
| after | impact | 2019-09-14 | 18.3 | 35.1 | 282.8 | 7.5 | 0.44 | 0.008 | 43.83 |
| after | impact | 2019-09-14 | 18 | 76.2 | 174 | 8.45 | 0.47 | 0.008 | 46.54 |
| after | impact | 2019-09-14 | 18.1 | 63.2 | 180.2 | 8.36 | 0.51 | 0.01 | 45.67 |
| after | control | 2020-09-11 | 18.5 | 89.6 | 176.4 | 8.03 | 0.51 | 0.016 | 40.2 |
| after | control | 2020-09-11 | 18 | 86.4 | 173.9 | 8.28 | 0.52 | 0.015 | 40 |
| after | control | 2020-09-11 | 17.9 | 82.8 | 174.2 | 8.14 | 0.5 | 0.015 | 28.5 |
| after | bare | 2020-09-11 | 18.5 | 92.3 | 176.8 | 8.19 | 0.51 | 0.013 | 56.4 |
| after | bare | 2020-09-11 | 18.3 | 88.3 | 175 | 8.34 | 0.5 | 0.013 | 54.95 |
| after | bare | 2020-09-11 | 18 | 88.1 | 172.8 | 8.38 | 0.48 | 0.011 | 49.8 |
| after | impact | 2020-09-11 | 18.5 | 89.2 | 176 | 8.21 | 0.48 | 0.012 | 36 |
| after | impact | 2020-09-11 | 18.1 | 85.5 | 174 | 8.13 | 0.44 | 0.012 | 50.1 |
| after | impact | 2020-09-11 | 17.8 | 87.7 | 172.1 | 8.02 | 0.44 | 0.012 | 35 |
| after | control | 2020-10-28 | 8.2 | 98.3 | 137 | 6.13 | 0.41 | 0.01 | 7.25 |
| after | control | 2020-10-28 | 8.2 | 97.4 | 137.6 | 6.19 | 0.42 | 0.006 | 22.65 |
| after | control | 2020-10-28 | 8.2 | 98.7 | 137.7 | 6.24 | 0.45 | 0.006 | 21.35 |
| after | bare | 2020-10-28 | 8.2 | 103.1 | 137.9 | 6.69 | 0.4 | 0.005 | 35.65 |
| after | bare | 2020-10-28 | 8.2 | 101.3 | 137.8 | 6.64 | 0.42 | 0.011 | 43.55 |
| after | bare | 2020-10-28 | 8.2 | 100.3 | 137.8 | 6.62 | 0.41 | 0.008 | 46.75 |
| after | impact | 2020-10-28 | 8.2 | 100 | 137.8 | 6.77 | 0.41 | 0.005 | 29.4 |
| after | impact | 2020-10-28 | 8.2 | 98.7 | 137.9 | 6.73 | 0.38 | 0.005 | 42.5 |
| after | impact | 2020-10-28 | 8.2 | 100.2 | 137.9 | 6.69 | 0.4 | 0.005 | 38.55 |

Table A-2. Average zooplankton family taxa count (3 replicates) per litre of water from Malcolm Lake.

| Sample Time | Sample Site | Date | Chydoridae | Bosminidae | Daphniidae | Calanoida | Cyclopoida |
|-------------|-------------|------------|------------|------------|------------|-----------|------------|
| before | control | 2019-06-08 | 13.88 | 1.75 | 0.04 | 0.13 | 8.17 |
| before | control | 2019-06-08 | 28.33 | 0.50 | 0.42 | 0.08 | 6.42 |

| | | | | | | | |
|--------|---------|------------|-------|-------|------|------|-------|
| before | control | 2019-06-08 | 0.13 | 1.25 | 0.17 | 0.63 | 0.42 |
| before | impact | 2019-06-08 | 17.92 | 11.25 | 0.00 | 0.25 | 4.29 |
| before | impact | 2019-06-08 | 27.25 | 1.08 | 0.13 | 0.00 | 9.71 |
| before | impact | 2019-06-08 | 20.63 | 6.38 | 0.08 | 0.38 | 8.21 |
| after | control | 2019-07-03 | 52.46 | 0.50 | 0.54 | 0.08 | 15.58 |
| after | control | 2019-07-03 | 25.13 | 0.08 | 0.00 | 0.04 | 13.54 |
| after | control | 2019-07-03 | 23.33 | 0.17 | 0.17 | 0.00 | 11.92 |
| after | bare | 2019-07-03 | 0.46 | 1.96 | 0.00 | 0.00 | 1.79 |
| after | bare | 2019-07-03 | 0.17 | 1.38 | 0.04 | 0.00 | 1.58 |
| after | bare | 2019-07-03 | 0.79 | 1.96 | 0.00 | 0.00 | 1.63 |
| after | impact | 2019-07-03 | 14.54 | 34.04 | 0.00 | 0.08 | 9.42 |
| after | impact | 2019-07-03 | 8.29 | 21.25 | 0.00 | 0.17 | 2.08 |
| after | impact | 2019-07-03 | 13.71 | 24.92 | 0.00 | 0.17 | 6.21 |
| after | control | 2019-08-01 | 15.82 | 0.68 | 0.00 | 0.00 | 9.59 |
| after | control | 2019-08-01 | 38.50 | 0.82 | 0.00 | 0.00 | 23.00 |
| after | control | 2019-08-01 | 18.14 | 0.73 | 0.09 | 0.00 | 18.27 |
| after | bare | 2019-08-01 | 1.00 | 0.05 | 0.00 | 0.05 | 8.32 |
| after | bare | 2019-08-01 | 0.18 | 0.09 | 0.00 | 0.23 | 6.41 |
| after | bare | 2019-08-01 | 0.05 | 0.09 | 0.00 | 0.05 | 8.55 |
| after | impact | 2019-08-01 | 0.09 | 0.32 | 0.00 | 0.00 | 15.05 |
| after | impact | 2019-08-01 | 0.00 | 0.64 | 0.00 | 0.27 | 17.36 |
| after | impact | 2019-08-01 | 0.00 | 0.41 | 0.00 | 0.32 | 21.64 |
| after | control | 2019-09-14 | 3.91 | 2.00 | 0.09 | 2.41 | 12.27 |
| after | control | 2019-09-14 | 2.73 | 2.73 | 0.00 | 2.45 | 15.55 |
| after | control | 2019-09-14 | 2.59 | 6.27 | 0.09 | 0.00 | 20.45 |
| after | bare | 2019-09-14 | 0.64 | 2.36 | 0.05 | 1.41 | 19.36 |
| after | bare | 2019-09-14 | 1.05 | 5.82 | 0.00 | 1.27 | 8.91 |
| after | bare | 2019-09-14 | 0.82 | 5.68 | 0.00 | 1.41 | 13.41 |
| after | impact | 2019-09-14 | 0.55 | 1.68 | 0.05 | 2.73 | 20.18 |
| after | impact | 2019-09-14 | 0.18 | 1.41 | 0.05 | 1.55 | 8.36 |
| after | impact | 2019-09-14 | 0.32 | 4.32 | 0.14 | 5.32 | 26.09 |
| after | control | 2020-09-11 | 1.40 | 15.10 | 1.35 | 1.35 | 8.30 |
| after | control | 2020-09-11 | 2.90 | 13.15 | 2.95 | 0.55 | 6.45 |
| after | control | 2020-09-11 | 1.30 | 11.15 | 0.95 | 0.90 | 6.95 |
| after | bare | 2020-09-11 | 0.05 | 17.45 | 0.25 | 3.60 | 18.90 |
| after | bare | 2020-09-11 | 0.00 | 19.45 | 0.25 | 3.55 | 20.50 |
| after | bare | 2020-09-11 | 0.00 | 20.90 | 0.60 | 3.05 | 15.45 |
| after | impact | 2020-09-11 | 0.00 | 8.20 | 0.15 | 2.15 | 15.90 |
| after | impact | 2020-09-11 | 0.00 | 10.15 | 0.20 | 1.40 | 24.55 |
| after | impact | 2020-09-11 | 0.10 | 7.35 | 0.05 | 0.90 | 15.45 |
| after | control | 2020-09-20 | 1.75 | 3.05 | 0.25 | 4.65 | 10.60 |
| after | control | 2020-09-20 | 1.40 | 1.95 | 0.45 | 2.85 | 7.10 |
| after | control | 2020-09-20 | 1.00 | 1.70 | 0.25 | 2.40 | 10.25 |

| after | bare | 2020-09-20 | 0.65 | 13.35 | 0.10 | 4.00 | 16.00 |
|-------------|-------------|------------|--------------|----------------|---------|--------------|------------|
| after | bare | 2020-09-20 | 0.10 | 16.50 | 0.05 | 2.35 | 10.00 |
| after | bare | 2020-09-20 | 0.10 | 20.10 | 0.05 | 2.10 | 12.40 |
| after | impact | 2020-09-20 | 0.60 | 8.45 | 0.15 | 8.35 | 16.15 |
| after | impact | 2020-09-20 | 0.70 | 15.15 | 0.00 | 5.30 | 18.40 |
| after | impact | 2020-09-20 | 0.25 | 9.90 | 0.20 | 4.45 | 12.60 |
| after | control | 2020-10-28 | 0.30 | 1.35 | 0.00 | 1.50 | 3.55 |
| after | control | 2020-10-28 | 0.45 | 5.80 | 0.00 | 3.35 | 11.05 |
| after | control | 2020-10-28 | 0.30 | 6.80 | 0.00 | 3.40 | 7.60 |
| after | bare | 2020-10-28 | 0.30 | 14.05 | 0.00 | 7.15 | 10.10 |
| after | bare | 2020-10-28 | 0.05 | 16.20 | 0.20 | 8.15 | 14.45 |
| after | bare | 2020-10-28 | 0.10 | 16.05 | 0.05 | 10.85 | 13.80 |
| after | impact | 2020-10-28 | 0.20 | 8.15 | 0.00 | 4.90 | 8.85 |
| after | impact | 2020-10-28 | 0.05 | 11.25 | 0.10 | 7.20 | 13.00 |
| after | impact | 2020-10-28 | 0.20 | 10.65 | 0.00 | 7.40 | 11.90 |
| Sample Time | Sample Site | Date | Holopediidae | Macrothricidae | Sididae | Polyphemidae | Gammaridae |
| before | control | 2019-06-08 | 0.00 | 0.00 | 0.42 | 0.00 | 0.00 |
| before | control | 2019-06-08 | 0.00 | 1.17 | 0.00 | 0.04 | 0.13 |
| before | control | 2019-06-08 | 0.00 | 0.00 | 0.04 | 0.00 | 0.00 |
| before | impact | 2019-06-08 | 0.00 | 1.08 | 0.50 | 0.00 | 0.00 |
| before | impact | 2019-06-08 | 0.00 | 0.46 | 0.50 | 0.00 | 0.00 |
| before | impact | 2019-06-08 | 0.00 | 0.63 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-07-03 | 0.04 | 1.33 | 4.88 | 0.13 | 0.00 |
| after | control | 2019-07-03 | 0.00 | 2.00 | 1.08 | 0.00 | 0.00 |
| after | control | 2019-07-03 | 0.00 | 1.54 | 1.83 | 0.04 | 0.00 |
| after | bare | 2019-07-03 | 0.17 | 0.38 | 0.33 | 0.00 | 0.00 |
| after | bare | 2019-07-03 | 0.00 | 0.17 | 0.21 | 0.00 | 0.00 |
| after | bare | 2019-07-03 | 0.00 | 0.04 | 0.17 | 0.00 | 0.00 |
| after | impact | 2019-07-03 | 0.00 | 0.67 | 1.83 | 0.04 | 0.00 |
| after | impact | 2019-07-03 | 0.04 | 0.08 | 0.29 | 0.08 | 0.00 |
| after | impact | 2019-07-03 | 0.00 | 0.50 | 0.67 | 0.08 | 0.00 |
| after | control | 2019-08-01 | 0.00 | 0.05 | 0.27 | 0.00 | 0.00 |
| after | control | 2019-08-01 | 0.00 | 0.14 | 3.91 | 0.00 | 0.00 |
| after | control | 2019-08-01 | 0.00 | 0.32 | 2.50 | 0.00 | 0.00 |
| after | bare | 2019-08-01 | 0.00 | 0.00 | 0.09 | 0.14 | 0.00 |
| after | bare | 2019-08-01 | 0.00 | 0.00 | 0.05 | 0.00 | 0.00 |
| after | bare | 2019-08-01 | 0.00 | 0.09 | 0.09 | 0.00 | 0.00 |
| after | impact | 2019-08-01 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-08-01 | 0.00 | 0.05 | 0.41 | 0.00 | 0.00 |
| after | impact | 2019-08-01 | 0.00 | 0.00 | 0.50 | 0.00 | 0.00 |
| after | control | 2019-09-14 | 0.00 | 6.36 | 2.27 | 0.00 | 0.00 |
| after | control | 2019-09-14 | 0.00 | 3.82 | 2.00 | 0.00 | 0.00 |

| after | control | 2019-09-14 | 0.00 | 4.77 | 1.41 | 0.00 | 0.18 |
|-------------|-------------|------------|-----------|--------------|------------|-------------|----------|
| after | bare | 2019-09-14 | 0.00 | 0.45 | 0.14 | 0.00 | 0.09 |
| after | bare | 2019-09-14 | 0.00 | 0.18 | 0.45 | 0.00 | 0.00 |
| after | bare | 2019-09-14 | 0.00 | 0.50 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-09-14 | 0.00 | 0.05 | 0.59 | 0.05 | 0.00 |
| after | impact | 2019-09-14 | 0.00 | 0.09 | 0.41 | 0.00 | 0.00 |
| after | impact | 2019-09-14 | 0.00 | 0.23 | 0.45 | 0.00 | 0.00 |
| after | control | 2020-09-11 | 0.00 | 0.70 | 3.25 | 0.00 | 0.00 |
| after | control | 2020-09-11 | 0.00 | 0.55 | 2.90 | 0.00 | 0.00 |
| after | control | 2020-09-11 | 0.00 | 0.70 | 2.10 | 0.00 | 0.00 |
| after | bare | 2020-09-11 | 0.00 | 0.00 | 1.10 | 0.05 | 0.00 |
| after | bare | 2020-09-11 | 0.00 | 0.00 | 1.05 | 0.00 | 0.00 |
| after | bare | 2020-09-11 | 0.00 | 0.00 | 1.65 | 0.00 | 0.00 |
| after | impact | 2020-09-11 | 0.00 | 0.00 | 0.80 | 0.00 | 0.00 |
| after | impact | 2020-09-11 | 0.00 | 0.00 | 1.35 | 0.00 | 0.00 |
| after | impact | 2020-09-11 | 0.00 | 0.00 | 0.85 | 0.00 | 0.00 |
| after | control | 2020-09-20 | 0.00 | 0.10 | 1.30 | 0.00 | 0.00 |
| after | control | 2020-09-20 | 0.00 | 0.10 | 2.05 | 0.00 | 0.00 |
| after | control | 2020-09-20 | 0.00 | 0.00 | 1.30 | 0.00 | 0.00 |
| after | bare | 2020-09-20 | 0.00 | 0.05 | 0.80 | 0.00 | 0.00 |
| after | bare | 2020-09-20 | 0.00 | 0.00 | 0.85 | 0.05 | 0.00 |
| after | bare | 2020-09-20 | 0.00 | 0.00 | 1.20 | 0.00 | 0.00 |
| after | impact | 2020-09-20 | 0.00 | 0.00 | 0.75 | 0.00 | 0.00 |
| after | impact | 2020-09-20 | 0.00 | 0.00 | 0.85 | 0.00 | 0.00 |
| after | impact | 2020-09-20 | 0.00 | 0.00 | 0.95 | 0.00 | 0.00 |
| after | control | 2020-10-28 | 0.00 | 0.05 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-28 | 0.00 | 0.00 | 0.10 | 0.00 | 0.00 |
| after | control | 2020-10-28 | 0.00 | 0.00 | 0.15 | 0.00 | 0.00 |
| after | bare | 2020-10-28 | 0.00 | 0.00 | 0.05 | 0.00 | 0.00 |
| after | bare | 2020-10-28 | 0.00 | 0.00 | 0.25 | 0.00 | 0.00 |
| after | bare | 2020-10-28 | 0.00 | 0.00 | 0.20 | 0.00 | 0.00 |
| after | impact | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-10-28 | 0.00 | 0.05 | 0.10 | 0.00 | 0.00 |
| after | impact | 2020-10-28 | 0.00 | 0.00 | 0.05 | 0.00 | 0.00 |
| Sample Time | Sample Site | Date | Ostracoda | Chironomidae | Gastropoda | Hydrachnida | nauplius |
| before | control | 2019-06-08 | 0.00 | 0.17 | 0.00 | 0.08 | 3.58 |
| before | control | 2019-06-08 | 0.00 | 0.50 | 0.13 | 0.08 | 0.00 |
| before | control | 2019-06-08 | 0.00 | 0.04 | 0.00 | 0.00 | 1.50 |
| before | impact | 2019-06-08 | 0.04 | 0.00 | 0.00 | 0.25 | 3.71 |
| before | impact | 2019-06-08 | 0.38 | 0.13 | 0.04 | 0.08 | 3.54 |
| before | impact | 2019-06-08 | 0.00 | 0.00 | 0.00 | 0.04 | 5.63 |
| after | control | 2019-07-03 | 0.75 | 0.54 | 0.00 | 0.46 | 3.21 |

| | | | | | | | |
|-------|---------|------------|------|------|------|------|-------|
| after | control | 2019-07-03 | 0.13 | 0.00 | 0.00 | 0.38 | 0.00 |
| after | control | 2019-07-03 | 0.08 | 0.17 | 0.00 | 0.58 | 0.21 |
| after | bare | 2019-07-03 | 0.00 | 0.00 | 0.00 | 0.04 | 3.42 |
| after | bare | 2019-07-03 | 0.00 | 0.00 | 0.00 | 0.00 | 3.38 |
| after | bare | 2019-07-03 | 0.00 | 0.00 | 0.00 | 0.00 | 2.25 |
| after | impact | 2019-07-03 | 0.00 | 0.00 | 0.00 | 0.00 | 6.29 |
| after | impact | 2019-07-03 | 0.00 | 0.08 | 0.00 | 0.17 | 2.13 |
| after | impact | 2019-07-03 | 0.21 | 0.21 | 0.00 | 0.29 | 3.17 |
| after | control | 2019-08-01 | 0.14 | 0.27 | 0.00 | 0.09 | 3.18 |
| after | control | 2019-08-01 | 1.55 | 0.45 | 0.00 | 0.55 | 4.59 |
| after | control | 2019-08-01 | 1.55 | 0.27 | 0.00 | 0.41 | 1.14 |
| after | bare | 2019-08-01 | 0.00 | 0.00 | 0.00 | 0.00 | 3.64 |
| after | bare | 2019-08-01 | 0.00 | 0.00 | 0.00 | 0.00 | 3.45 |
| after | bare | 2019-08-01 | 0.00 | 0.00 | 0.00 | 0.00 | 4.14 |
| after | impact | 2019-08-01 | 0.00 | 0.00 | 0.00 | 0.00 | 7.32 |
| after | impact | 2019-08-01 | 0.00 | 0.00 | 0.00 | 0.00 | 12.77 |
| after | impact | 2019-08-01 | 0.00 | 0.00 | 0.00 | 0.05 | 20.73 |
| after | control | 2019-09-14 | 2.27 | 0.73 | 0.00 | 0.05 | 4.14 |
| after | control | 2019-09-14 | 0.73 | 0.68 | 0.00 | 0.50 | 5.41 |
| after | control | 2019-09-14 | 1.14 | 0.68 | 0.00 | 0.05 | 3.32 |
| after | bare | 2019-09-14 | 0.09 | 0.00 | 0.00 | 0.00 | 5.14 |
| after | bare | 2019-09-14 | 0.00 | 0.09 | 0.00 | 0.00 | 1.00 |
| after | bare | 2019-09-14 | 0.05 | 0.05 | 0.00 | 0.00 | 6.77 |
| after | impact | 2019-09-14 | 0.00 | 0.05 | 0.00 | 0.00 | 3.86 |
| after | impact | 2019-09-14 | 0.00 | 0.05 | 0.00 | 0.00 | 2.14 |
| after | impact | 2019-09-14 | 0.00 | 0.00 | 0.00 | 0.00 | 2.77 |
| after | control | 2020-09-11 | 0.00 | 0.00 | 0.00 | 0.00 | 2.20 |
| after | control | 2020-09-11 | 0.20 | 0.25 | 0.00 | 0.00 | 0.80 |
| after | control | 2020-09-11 | 0.00 | 0.00 | 0.00 | 0.00 | 1.30 |
| after | bare | 2020-09-11 | 0.00 | 0.00 | 0.00 | 0.00 | 9.65 |
| after | bare | 2020-09-11 | 0.05 | 0.00 | 0.00 | 0.00 | 6.10 |
| after | bare | 2020-09-11 | 0.00 | 0.00 | 0.00 | 0.00 | 5.00 |
| after | impact | 2020-09-11 | 0.00 | 0.00 | 0.00 | 0.00 | 4.05 |
| after | impact | 2020-09-11 | 0.00 | 0.00 | 0.00 | 0.00 | 6.90 |
| after | impact | 2020-09-11 | 0.00 | 0.00 | 0.00 | 0.00 | 6.85 |
| after | control | 2020-09-20 | 0.25 | 0.00 | 0.00 | 0.00 | 3.30 |
| after | control | 2020-09-20 | 0.10 | 0.00 | 0.00 | 0.00 | 3.00 |
| after | control | 2020-09-20 | 0.15 | 0.00 | 0.00 | 0.00 | 3.60 |
| after | bare | 2020-09-20 | 0.00 | 0.00 | 0.00 | 0.00 | 9.60 |
| after | bare | 2020-09-20 | 0.00 | 0.00 | 0.00 | 0.00 | 11.00 |
| after | bare | 2020-09-20 | 0.00 | 0.00 | 0.00 | 0.00 | 15.60 |
| after | impact | 2020-09-20 | 0.00 | 0.00 | 0.00 | 0.00 | 5.75 |
| after | impact | 2020-09-20 | 0.05 | 0.00 | 0.00 | 0.00 | 5.65 |

| after | impact | 2020-09-20 | 0.00 | 0.00 | 0.00 | 0.00 | 5.65 |
|-------------|-------------|------------|--------------|---------------|--------------|--------------|-------|
| after | control | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 0.40 |
| after | control | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 1.15 |
| after | bare | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 0.75 |
| after | bare | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 0.50 |
| after | bare | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 0.65 |
| after | impact | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 1.10 |
| after | impact | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 1.75 |
| after | impact | 2020-10-28 | 0.00 | 0.00 | 0.00 | 0.00 | 1.35 |
| Sample Time | Sample Site | Date | Brachionidae | Asplanchnidae | Synchaetidae | Conochilidae | other |
| before | control | 2019-06-08 | 2.67 | 0.00 | 0.00 | 0.00 | 0.00 |
| before | control | 2019-06-08 | 0.00 | 0.00 | 0.00 | 0.00 | 1.21 |
| before | control | 2019-06-08 | 0.21 | 0.00 | 0.00 | 0.00 | 0.00 |
| before | impact | 2019-06-08 | 4.54 | 0.00 | 0.00 | 0.00 | 0.00 |
| before | impact | 2019-06-08 | 2.38 | 0.00 | 0.00 | 0.00 | 0.88 |
| before | impact | 2019-06-08 | 3.54 | 0.17 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-07-03 | 0.58 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-07-03 | 0.08 | 0.08 | 0.08 | 0.00 | 0.00 |
| after | control | 2019-07-03 | 0.46 | 0.17 | 0.00 | 0.00 | 0.00 |
| after | bare | 2019-07-03 | 1.29 | 4.67 | 0.00 | 1.29 | 0.00 |
| after | bare | 2019-07-03 | 1.54 | 6.83 | 0.04 | 0.04 | 0.04 |
| after | bare | 2019-07-03 | 0.71 | 4.83 | 0.00 | 1.38 | 0.00 |
| after | impact | 2019-07-03 | 0.79 | 1.79 | 0.00 | 0.50 | 0.00 |
| after | impact | 2019-07-03 | 0.46 | 2.13 | 0.00 | 0.17 | 0.00 |
| after | impact | 2019-07-03 | 0.75 | 1.96 | 0.00 | 0.96 | 0.00 |
| after | control | 2019-08-01 | 0.41 | 1.23 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-08-01 | 0.59 | 2.27 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-08-01 | 0.77 | 3.23 | 0.00 | 0.00 | 0.00 |
| after | bare | 2019-08-01 | 0.18 | 2.05 | 0.00 | 0.00 | 0.00 |
| after | bare | 2019-08-01 | 0.36 | 2.45 | 0.00 | 0.09 | 0.00 |
| after | bare | 2019-08-01 | 0.14 | 3.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-08-01 | 0.45 | 0.36 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-08-01 | 0.45 | 1.23 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-08-01 | 0.59 | 0.36 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-09-14 | 14.14 | 0.32 | 0.00 | 0.09 | 0.27 |
| after | control | 2019-09-14 | 14.82 | 0.18 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-09-14 | 8.50 | 0.05 | 0.00 | 0.00 | 0.00 |
| after | bare | 2019-09-14 | 1.05 | 0.18 | 0.00 | 0.00 | 0.00 |
| after | bare | 2019-09-14 | 2.27 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | bare | 2019-09-14 | 0.86 | 0.09 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-09-14 | 11.05 | 0.09 | 0.00 | 0.00 | 0.00 |

| | | | | | | | |
|-------|---------|------------|-------|------|------|------|------|
| after | impact | 2019-09-14 | 2.36 | 0.23 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-09-14 | 11.77 | 0.09 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-11 | 0.10 | 6.45 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-11 | 0.05 | 8.75 | 0.00 | 0.00 | 0.50 |
| after | control | 2020-09-11 | 0.15 | 3.00 | 0.00 | 0.00 | 0.00 |
| after | bare | 2020-09-11 | 1.90 | 3.40 | 0.00 | 0.00 | 0.05 |
| after | bare | 2020-09-11 | 1.70 | 2.30 | 0.00 | 0.00 | 0.00 |
| after | bare | 2020-09-11 | 0.20 | 2.90 | 0.00 | 0.00 | 0.05 |
| after | impact | 2020-09-11 | 0.10 | 4.60 | 0.00 | 0.00 | 0.05 |
| after | impact | 2020-09-11 | 0.15 | 5.40 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-11 | 0.35 | 3.10 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-20 | 0.00 | 3.05 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-20 | 0.05 | 2.25 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-20 | 0.05 | 5.10 | 0.00 | 0.00 | 0.00 |
| after | bare | 2020-09-20 | 0.35 | 1.90 | 0.00 | 0.00 | 0.00 |
| after | bare | 2020-09-20 | 0.55 | 1.05 | 0.00 | 0.00 | 0.00 |
| after | bare | 2020-09-20 | 0.50 | 0.85 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-20 | 0.35 | 1.85 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-20 | 0.00 | 1.45 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-20 | 0.25 | 1.45 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-28 | 0.05 | 0.35 | 0.00 | 0.00 | 0.10 |
| after | control | 2020-10-28 | 0.55 | 0.90 | 0.00 | 0.00 | 0.05 |
| after | control | 2020-10-28 | 1.40 | 0.40 | 0.00 | 0.00 | 0.00 |
| after | bare | 2020-10-28 | 1.40 | 0.30 | 0.00 | 1.40 | 0.15 |
| after | bare | 2020-10-28 | 3.00 | 0.25 | 0.00 | 0.50 | 0.00 |
| after | bare | 2020-10-28 | 3.20 | 0.50 | 0.00 | 1.35 | 0.00 |
| after | impact | 2020-10-28 | 5.60 | 0.40 | 0.00 | 0.20 | 0.00 |
| after | impact | 2020-10-28 | 8.60 | 0.25 | 0.00 | 0.15 | 0.00 |
| after | impact | 2020-10-28 | 5.80 | 0.45 | 0.00 | 0.75 | 0.00 |

Appendix B – Wolfe Lake

Table B-1. Abiotic variables and zooplankton density (3 replicates) from Wolfe Lake.

| Sample Time | Sample Site | Date | Temp. (°C) | DO (%) | Cond. (µS) | pH | TN (mg/L) | TP (mg/L) | Zooplankton Density (per L) |
|-------------|-------------|------------|------------|--------|------------|------|-----------|-----------|-----------------------------|
| before | control | 2019-07-10 | 26.2 | 87.3 | 217.7 | 8.49 | 0.44 | 0.008 | 36.05 |
| before | control | 2019-07-10 | 26.2 | 90.1 | 216.9 | 8.52 | 0.45 | 0.007 | 37.77 |
| before | control | 2019-07-10 | 26.2 | 90.1 | 217.4 | 8.54 | 0.44 | 0.007 | 41.36 |
| before | impact | 2019-07-10 | 26.2 | 99.1 | 214.9 | 8.69 | 0.39 | 0.008 | 63.73 |
| before | impact | 2019-07-10 | 26.2 | 96.4 | 215.4 | 8.69 | 0.4 | 0.008 | 41.00 |
| before | impact | 2019-07-10 | 26.2 | 92.6 | 214.7 | 8.7 | 0.45 | 0.007 | 46.41 |

| | | | | | | | | | |
|--------|---------|------------|------|-------|-------|------|------|-------|-------|
| before | control | 2019-08-07 | 25.4 | 184.1 | 209.9 | 8.94 | 0.34 | 0.007 | 45.64 |
| before | control | 2019-08-07 | 25.4 | 85.2 | 212.1 | 8.94 | 0.36 | 0.008 | 53.36 |
| before | control | 2019-08-07 | 25.4 | 104.5 | 208.4 | 9.02 | 0.4 | 0.008 | 58.73 |
| before | impact | 2019-08-07 | 25.4 | 85.3 | 212.2 | 8.84 | 0.35 | 0.008 | 96.68 |
| before | impact | 2019-08-07 | 25.4 | 90.3 | 208.4 | 9 | 0.36 | 0.007 | 15.73 |
| before | impact | 2019-08-07 | 25.4 | 88.3 | 213 | 8.85 | 0.38 | 0.007 | 10.09 |
| after | control | 2019-09-13 | 20 | 102.6 | 187.3 | 8.92 | 0.33 | 0.01 | 7.36 |
| after | control | 2019-09-13 | 20 | 99 | 187.3 | 8.89 | 0.33 | 0.009 | 3.95 |
| after | control | 2019-09-13 | 19.9 | 101.2 | 186.7 | 8.84 | 0.36 | 0.01 | 3.14 |
| after | impact | 2019-09-13 | 20.3 | 99.2 | 188.3 | 8.87 | 0.31 | 0.01 | 8.86 |
| after | impact | 2019-09-13 | 20.4 | 98.1 | 188.7 | 8.89 | 0.35 | 0.01 | 3.68 |
| after | impact | 2019-09-13 | 20.4 | 95.9 | 188.6 | 8.9 | 0.36 | 0.011 | 4.68 |
| after | control | 2019-10-04 | 16.5 | 83.2 | 173.8 | 8.42 | 0.38 | 0.011 | 4.36 |
| after | control | 2019-10-04 | 16.5 | 78.4 | 174.2 | 8.4 | 0.37 | 0.01 | 4.32 |
| after | control | 2019-10-04 | 16.5 | 75.5 | 173.8 | 8.41 | 0.37 | 0.009 | 12.32 |
| after | impact | 2019-10-04 | 16.6 | 70.2 | 175.2 | 8.37 | 0.43 | 0.01 | 10.68 |
| after | impact | 2019-10-04 | 16.7 | 73.2 | 175.1 | 8.4 | 0.37 | 0.01 | 6.50 |
| after | impact | 2019-10-04 | 16.7 | 77.3 | 175.1 | 8.4 | 0.37 | 0.01 | 11.45 |
| after | control | 2020-09-15 | 18.8 | 97 | 181.3 | 8.83 | 0.33 | 0.013 | 25.75 |
| after | control | 2020-09-15 | 18.8 | 95.1 | 182.5 | 8.81 | 0.34 | 0.013 | 22.20 |
| after | control | 2020-09-15 | 18.9 | 96.2 | 182.2 | 8.81 | 0.33 | 0.013 | 5.55 |
| after | impact | 2020-09-15 | 18.8 | 97.9 | 181.5 | 8.79 | 0.33 | 0.013 | 10.20 |
| after | impact | 2020-09-15 | 18.9 | 100.9 | 181.5 | 8.83 | 0.34 | 0.14 | 13.55 |
| after | impact | 2020-09-15 | 19 | 101.6 | 181.2 | 8.85 | 0.34 | 0.014 | 17.80 |
| after | control | 2020-10-05 | 16.7 | 87.5 | 173.6 | 8.4 | 0.31 | 0.012 | 27.70 |
| after | control | 2020-10-05 | 16.8 | 85.5 | 173.9 | 8.34 | 0.36 | 0.015 | 27.90 |
| after | control | 2020-10-05 | 16.8 | 80.3 | 174.2 | 8.33 | 0.29 | 0.013 | 37.05 |
| after | impact | 2020-10-05 | 16.8 | 88.1 | 174.5 | 8.41 | 0.4 | 0.016 | 19.20 |
| after | impact | 2020-10-05 | 16.8 | 85.1 | 174.6 | 8.46 | 0.34 | 0.016 | 17.30 |
| after | impact | 2020-10-05 | 16.8 | 82.6 | 174.7 | 8.49 | 0.35 | 0.016 | 18.40 |

Table B-2. Average zooplankton family taxa count (3 replicates) per litre of water from Wolfe Lake.

| Sample Time | Sample Site | Date | Chydoridae | Bosminidae | Macrothricidae | Sididae | Daphniidae |
|-------------|-------------|------------|------------|------------|----------------|---------|------------|
| before | control | 2019-07-10 | 7.32 | 0.05 | 0.05 | 0.09 | 0.00 |
| before | control | 2019-07-10 | 6.77 | 0.09 | 0.00 | 0.14 | 0.00 |
| before | control | 2019-07-10 | 3.18 | 0.05 | 0.00 | 0.09 | 0.00 |
| before | impact | 2019-07-10 | 19.05 | 0.27 | 0.05 | 0.09 | 0.00 |
| before | impact | 2019-07-10 | 12.41 | 0.23 | 0.00 | 0.00 | 0.00 |
| before | impact | 2019-07-10 | 11.36 | 0.09 | 0.00 | 0.23 | 0.00 |
| before | control | 2019-08-07 | 0.23 | 0.18 | 0.05 | 0.09 | 0.00 |

| | | | | | | | |
|--------|---------|------------|------|-------|------|------|------|
| before | control | 2019-08-07 | 0.32 | 0.05 | 0.00 | 0.09 | 0.00 |
| before | control | 2019-08-07 | 0.09 | 0.18 | 0.09 | 0.27 | 0.00 |
| before | impact | 2019-08-07 | 1.91 | 11.05 | 0.50 | 9.64 | 0.09 |
| before | impact | 2019-08-07 | 0.82 | 2.82 | 0.05 | 1.09 | 0.00 |
| before | impact | 2019-08-07 | 0.18 | 0.18 | 0.09 | 0.32 | 0.00 |
| after | control | 2019-09-13 | 0.82 | 1.64 | 0.00 | 0.14 | 0.00 |
| after | control | 2019-09-13 | 0.77 | 1.64 | 0.00 | 0.05 | 0.00 |
| after | control | 2019-09-13 | 0.41 | 1.32 | 0.00 | 0.05 | 0.00 |
| after | impact | 2019-09-13 | 4.27 | 1.91 | 0.00 | 0.09 | 0.00 |
| after | impact | 2019-09-13 | 1.23 | 1.27 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-09-13 | 1.77 | 2.00 | 0.00 | 0.09 | 0.00 |
| after | control | 2019-10-04 | 0.05 | 1.18 | 0.00 | 0.05 | 0.05 |
| after | control | 2019-10-04 | 0.14 | 1.73 | 0.00 | 0.00 | 0.09 |
| after | control | 2019-10-04 | 0.09 | 2.50 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-10-04 | 0.32 | 4.59 | 0.05 | 0.05 | 0.00 |
| after | impact | 2019-10-04 | 0.32 | 2.73 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-10-04 | 0.05 | 3.55 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-15 | 0.59 | 1.77 | 0.00 | 0.09 | 0.00 |
| after | control | 2020-09-15 | 0.23 | 0.82 | 0.00 | 0.09 | 0.00 |
| after | control | 2020-09-15 | 0.05 | 0.23 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-15 | 1.32 | 0.50 | 0.05 | 0.09 | 0.00 |
| after | impact | 2020-09-15 | 0.23 | 0.41 | 0.05 | 0.09 | 0.00 |
| after | impact | 2020-09-15 | 0.55 | 1.18 | 0.05 | 0.05 | 0.00 |
| after | control | 2020-10-05 | 0.05 | 1.82 | 0.00 | 0.05 | 0.00 |
| after | control | 2020-10-05 | 0.27 | 0.73 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-05 | 0.18 | 1.27 | 0.00 | 0.05 | 0.00 |
| after | impact | 2020-10-05 | 0.45 | 0.95 | 0.00 | 0.05 | 0.00 |
| after | impact | 2020-10-05 | 0.73 | 0.64 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-10-05 | 0.36 | 0.55 | 0.00 | 0.05 | 0.00 |

| Sample Time | Sample Site | Date | Holopediidae | Polyphemidae | Cyclopoida | Calanoida | nauplius |
|-------------|-------------|------------|--------------|--------------|------------|-----------|----------|
| before | control | 2019-07-10 | 0.00 | 0.00 | 15.50 | 0.27 | 10.00 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 13.64 | 0.14 | 12.00 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 14.86 | 0.45 | 18.64 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 16.77 | 0.09 | 21.18 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 11.36 | 0.09 | 14.41 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 13.14 | 0.64 | 13.77 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 23.18 | 0.18 | 19.41 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 24.82 | 0.41 | 25.91 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 31.41 | 0.32 | 24.23 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 41.00 | 0.09 | 20.50 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 2.05 | 0.05 | 2.27 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 2.68 | 0.00 | 2.14 |

| after | control | 2019-09-13 | 0.00 | 0.00 | 1.55 | 0.00 | 2.86 |
|-------------|-------------|------------|-----------|--------------|-------------|--------------|---------------|
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.41 | 0.00 | 0.59 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.32 | 0.05 | 0.77 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.68 | 0.00 | 1.36 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.55 | 0.05 | 0.18 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.18 | 0.00 | 0.09 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.91 | 1.14 | 0.91 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.59 | 1.23 | 0.27 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 2.23 | 5.91 | 1.59 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 2.82 | 0.91 | 1.41 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 1.64 | 0.50 | 0.95 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 2.68 | 1.95 | 3.00 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 7.45 | 3.86 | 9.55 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 7.09 | 3.23 | 8.68 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 1.64 | 1.14 | 2.00 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 3.50 | 0.68 | 2.86 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 3.77 | 1.36 | 6.41 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 5.09 | 1.82 | 7.23 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 6.36 | 5.05 | 11.82 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 6.82 | 4.45 | 13.09 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 9.45 | 5.55 | 16.59 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 5.18 | 1.77 | 9.00 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 5.14 | 1.86 | 7.36 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 4.23 | 2.55 | 8.95 |
| Sample Time | Sample Site | Date | Ostracoda | Chironomidae | Hydrachnida | Brachionidae | Asplanchnidae |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.09 | 2.32 | 0.27 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.14 | 2.36 | 2.36 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.00 | 1.59 | 2.41 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 0.05 | 3.82 | 1.91 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 0.05 | 0.64 | 1.59 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 0.05 | 2.41 | 4.32 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.00 | 2.32 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.00 | 1.77 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.05 | 0.00 | 2.00 | 0.05 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 0.09 | 11.73 | 0.00 |
| before | impact | 2019-08-07 | 0.05 | 0.00 | 0.00 | 6.45 | 0.09 |
| before | impact | 2019-08-07 | 0.05 | 0.00 | 0.00 | 4.32 | 0.14 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.09 | 0.27 | 0.00 |
| after | control | 2019-09-13 | 0.27 | 0.00 | 0.00 | 0.23 | 0.00 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| after | impact | 2019-09-13 | 0.14 | 0.00 | 0.09 | 0.32 | 0.00 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.00 | 0.32 | 0.05 |

| | | | | | | | |
|-------|---------|------------|------|------|------|------|------|
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.00 | 0.55 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 | 0.09 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 | 0.18 | 0.09 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-10-04 | 0.05 | 0.00 | 0.00 | 0.50 | 0.00 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 | 0.32 | 0.05 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.09 | 0.00 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.05 | 0.00 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-15 | 0.05 | 0.00 | 0.00 | 0.18 | 0.00 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.05 | 0.00 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.59 | 0.00 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.05 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |

| Sample Time | Sample Site | Date | Synchaetidae | Conochilidae | other |
|-------------|-------------|------------|--------------|--------------|-------|
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.09 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.14 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.09 |
| before | impact | 2019-07-10 | 0.00 | 0.05 | 0.41 |
| before | impact | 2019-07-10 | 0.00 | 0.09 | 0.14 |
| before | impact | 2019-07-10 | 0.00 | 0.41 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.05 | 0.00 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 0.09 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 0.00 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.05 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 |

| | | | | | |
|-------|---------|------------|------|------|------|
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.05 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-10-05 | 0.00 | 0.05 | 0.00 |

Appendix C – Otter Lake

Table C-1. Abiotic variables and zooplankton density (3 replicates) from Otter Lake.

| Sample Time | Sample Site | Date | Temp. (°C) | DO (%) | Cond. (µS) | pH | TN (mg/L) | TP (mg/L) | Zooplankton Density (per L) |
|-------------|-------------|------------|------------|--------|------------|------|-----------|-----------|-----------------------------|
| before | control | 2019-07-10 | 26.40 | 120.00 | 285.10 | 8.66 | 0.43 | 0.01 | 28.59 |
| before | control | 2019-07-10 | 26.40 | 114.50 | 286.00 | 8.67 | 0.39 | 0.01 | 14.59 |
| before | control | 2019-07-10 | 26.60 | 118.00 | 287.90 | 8.65 | 0.38 | 0.01 | 35.18 |
| before | impact | 2019-07-10 | 26.70 | 153.30 | 278.10 | 8.98 | 0.53 | 0.01 | 16.00 |
| before | impact | 2019-07-10 | 26.60 | 130.00 | 280.60 | 8.87 | 0.48 | 0.01 | 15.64 |
| before | impact | 2019-07-10 | 26.50 | 134.90 | 283.00 | 8.79 | 0.56 | 0.01 | 9.55 |
| before | control | 2019-08-08 | 24.80 | 112.80 | 269.70 | 8.95 | 0.40 | 0.01 | 35.59 |
| before | control | 2019-08-08 | 24.80 | 104.50 | 270.20 | 8.92 | 0.46 | 0.01 | 41.09 |
| before | control | 2019-08-08 | 24.80 | 107.60 | 270.50 | 8.93 | 0.47 | 0.01 | 25.09 |
| before | impact | 2019-08-08 | 24.80 | 86.50 | 269.50 | 8.91 | 0.47 | 0.01 | 61.73 |
| before | impact | 2019-08-08 | 24.90 | 93.40 | 268.80 | 8.90 | 0.42 | 0.01 | 47.86 |
| before | impact | 2019-08-08 | 24.90 | 103.90 | 268.70 | 8.95 | 0.44 | 0.01 | 37.32 |
| after | control | 2019-09-12 | 19.80 | 87.50 | 241.80 | 8.63 | 0.36 | 0.01 | 16.05 |
| after | control | 2019-09-12 | 19.70 | 84.30 | 241.70 | 8.65 | 0.37 | 0.01 | 22.09 |
| after | control | 2019-09-12 | 19.80 | 85.70 | 242.30 | 8.64 | 0.37 | 0.01 | 27.23 |
| after | impact | 2019-09-12 | 20.10 | 76.00 | 243.20 | 8.30 | 0.36 | 0.01 | 23.45 |
| after | impact | 2019-09-12 | 20.00 | 78.50 | 286.90 | 8.48 | 0.35 | 0.01 | 19.36 |
| after | impact | 2019-09-12 | 20.00 | 68.20 | 278.80 | 8.15 | 0.36 | 0.01 | 22.50 |
| after | control | 2019-10-03 | 16.90 | 85.70 | 227.80 | 8.47 | 0.39 | 0.01 | 43.55 |
| after | control | 2019-10-03 | 16.90 | 83.60 | 227.90 | 8.47 | 0.39 | 0.01 | 42.68 |

| | | | | | | | | | |
|-------|---------|------------|-------|--------|--------|------|------|------|-------|
| after | control | 2019-10-03 | 16.90 | 79.20 | 228.10 | 8.45 | 0.42 | 0.01 | 41.64 |
| after | impact | 2019-10-03 | 16.70 | 82.40 | 226.20 | 8.35 | 0.44 | 0.01 | 40.86 |
| after | impact | 2019-10-03 | 16.70 | 84.70 | 226.10 | 8.43 | 0.40 | 0.01 | 55.09 |
| after | impact | 2019-10-03 | 16.70 | 86.70 | 226.70 | 8.45 | 0.39 | 0.01 | 33.27 |
| after | control | 2020-09-15 | 18.50 | 100.30 | 232.10 | 8.66 | 0.43 | 0.01 | 20.60 |
| after | control | 2020-09-15 | 18.50 | 95.50 | 232.30 | 8.69 | 0.41 | 0.01 | 25.05 |
| after | control | 2020-09-15 | 18.20 | 95.40 | 230.60 | 8.72 | 0.50 | 0.01 | 30.85 |
| after | impact | 2020-09-15 | 18.40 | 84.50 | 232.90 | 8.53 | 0.45 | 0.02 | 69.15 |
| after | impact | 2020-09-15 | 18.30 | 86.70 | 232.60 | 8.57 | 0.45 | 0.01 | 65.40 |
| after | impact | 2020-09-15 | 18.30 | 94.50 | 229.60 | 8.64 | 0.43 | 0.01 | 50.45 |
| after | control | 2020-10-05 | 16.40 | 94.00 | 223.80 | 8.51 | 0.39 | 0.01 | 32.15 |
| after | control | 2020-10-05 | 16.40 | 92.10 | 223.80 | 8.55 | 0.39 | 0.01 | 39.85 |
| after | control | 2020-10-05 | 16.40 | 88.80 | 224.50 | 8.58 | 0.41 | 0.01 | 37.85 |
| after | impact | 2020-10-05 | 16.30 | 95.10 | 222.90 | 8.53 | 0.44 | 0.01 | 54.60 |
| after | impact | 2020-10-05 | 16.30 | 93.10 | 223.10 | 8.67 | 0.44 | 0.01 | 84.90 |
| after | impact | 2020-10-05 | 16.20 | 92.30 | 222.20 | 8.68 | 0.42 | 0.01 | 77.95 |

Table C-2. Average zooplankton family taxa count (3 replicates) per litre of water from Otter Lake.

| Sample Time | Sample Site | Date | Chydoridae | Bosminidae | Macrothricidae | Sididae | Daphniidae |
|-------------|-------------|------------|------------|------------|----------------|---------|------------|
| before | control | 2019-07-10 | 0.36 | 0.14 | 0.00 | 0.00 | 0.00 |
| before | control | 2019-07-10 | 0.27 | 0.09 | 0.00 | 0.00 | 0.00 |
| before | control | 2019-07-10 | 0.14 | 0.36 | 0.00 | 0.00 | 0.00 |
| before | impact | 2019-07-10 | 0.32 | 0.14 | 0.00 | 0.00 | 0.00 |
| before | impact | 2019-07-10 | 0.18 | 0.27 | 0.00 | 0.05 | 0.05 |
| before | impact | 2019-07-10 | 0.09 | 0.09 | 0.00 | 0.00 | 0.00 |
| before | control | 2019-08-07 | 0.23 | 0.09 | 0.00 | 0.00 | 0.00 |
| before | control | 2019-08-07 | 0.41 | 0.27 | 0.05 | 0.09 | 0.00 |
| before | control | 2019-08-07 | 0.45 | 0.14 | 0.05 | 0.14 | 0.18 |
| before | impact | 2019-08-07 | 0.00 | 0.14 | 0.00 | 0.18 | 0.00 |
| before | impact | 2019-08-07 | 0.05 | 0.05 | 0.00 | 0.00 | 0.00 |
| before | impact | 2019-08-07 | 0.14 | 0.14 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-09-13 | 0.77 | 0.68 | 0.05 | 0.00 | 0.00 |
| after | control | 2019-09-13 | 0.05 | 0.14 | 0.00 | 0.32 | 0.05 |
| after | control | 2019-09-13 | 0.14 | 0.09 | 0.00 | 0.32 | 0.05 |
| after | impact | 2019-09-13 | 0.64 | 0.09 | 0.00 | 0.05 | 0.14 |
| after | impact | 2019-09-13 | 0.27 | 0.05 | 0.05 | 0.00 | 0.00 |
| after | impact | 2019-09-13 | 1.68 | 4.55 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-10-04 | 1.18 | 2.91 | 0.00 | 0.64 | 0.77 |
| after | control | 2019-10-04 | 0.77 | 4.50 | 0.05 | 0.86 | 1.18 |
| after | control | 2019-10-04 | 0.05 | 1.14 | 0.00 | 0.68 | 4.09 |
| after | impact | 2019-10-04 | 0.14 | 0.23 | 0.00 | 0.23 | 0.05 |

| | | | | | | | |
|-------|---------|------------|------|-------|------|------|------|
| after | impact | 2019-10-04 | 0.18 | 0.18 | 0.00 | 0.05 | 0.00 |
| after | impact | 2019-10-04 | 0.27 | 0.32 | 0.00 | 0.27 | 0.05 |
| after | control | 2020-09-15 | 3.82 | 2.09 | 0.36 | 0.18 | 0.05 |
| after | control | 2020-09-15 | 0.68 | 1.36 | 0.00 | 0.05 | 0.00 |
| after | control | 2020-09-15 | 0.23 | 1.59 | 0.05 | 0.05 | 0.00 |
| after | impact | 2020-09-15 | 0.91 | 11.86 | 0.05 | 0.14 | 0.09 |
| after | impact | 2020-09-15 | 0.23 | 11.86 | 0.05 | 0.23 | 0.00 |
| after | impact | 2020-09-15 | 0.50 | 6.64 | 0.05 | 0.23 | 0.09 |
| after | control | 2020-10-05 | 0.14 | 0.45 | 0.00 | 0.32 | 0.09 |
| after | control | 2020-10-05 | 0.05 | 0.68 | 0.05 | 0.18 | 0.18 |
| after | control | 2020-10-05 | 0.82 | 0.68 | 0.00 | 0.23 | 0.23 |
| after | impact | 2020-10-05 | 0.45 | 7.32 | 0.05 | 0.05 | 0.00 |
| after | impact | 2020-10-05 | 0.05 | 15.23 | 0.09 | 0.09 | 0.05 |
| after | impact | 2020-10-05 | 0.09 | 13.45 | 0.23 | 0.05 | 0.09 |

| Sample Time | Sample Site | Date | Holopediidae | Polyphemidae | Cyclopoida | Calanoida | nauplius |
|-------------|-------------|------------|--------------|--------------|------------|-----------|----------|
| before | control | 2019-07-10 | 0.00 | 0.00 | 20.27 | 0.00 | 6.95 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 8.86 | 0.00 | 5.14 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 25.73 | 0.00 | 8.59 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 3.91 | 0.00 | 11.18 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 2.41 | 0.05 | 12.41 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 1.14 | 0.09 | 7.91 |
| before | control | 2019-08-07 | 0.00 | 0.91 | 18.59 | 2.32 | 13.32 |
| before | control | 2019-08-07 | 0.00 | 2.09 | 15.09 | 2.09 | 20.95 |
| before | control | 2019-08-07 | 0.00 | 1.45 | 7.27 | 4.36 | 10.95 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 40.23 | 0.50 | 19.50 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 33.91 | 0.14 | 13.18 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 24.64 | 0.09 | 11.86 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 7.36 | 0.32 | 6.86 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 13.50 | 0.23 | 7.73 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 14.45 | 0.23 | 11.91 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 18.91 | 0.14 | 3.45 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 14.50 | 0.05 | 4.36 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 11.82 | 0.18 | 4.00 |
| after | control | 2019-10-04 | 0.14 | 0.00 | 21.82 | 10.68 | 5.27 |
| after | control | 2019-10-04 | 0.05 | 0.00 | 20.23 | 9.64 | 5.09 |
| after | control | 2019-10-04 | 0.27 | 0.00 | 18.18 | 11.23 | 6.00 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 28.55 | 7.59 | 3.95 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 39.95 | 6.45 | 8.18 |
| after | impact | 2019-10-04 | 0.05 | 0.00 | 20.91 | 9.50 | 1.91 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 10.00 | 0.23 | 1.86 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 14.73 | 0.36 | 5.59 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 17.41 | 0.18 | 8.55 |

| after | impact | 2020-09-15 | 0.00 | 0.09 | 24.55 | 6.64 | 18.55 |
|-------------|-------------|------------|-----------|--------------|-------------|--------------|---------------|
| after | impact | 2020-09-15 | 0.00 | 0.00 | 25.23 | 3.73 | 18.14 |
| after | impact | 2020-09-15 | 0.00 | 0.05 | 19.05 | 2.64 | 16.64 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 16.32 | 3.64 | 8.27 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 20.68 | 4.27 | 10.14 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 20.23 | 3.55 | 8.14 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 19.14 | 6.59 | 16.00 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 26.82 | 15.18 | 19.68 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 23.36 | 13.09 | 20.50 |
| Sample Time | Sample Site | Date | Ostracoda | Chironomidae | Hydrachnida | Brachionidae | Asplanchnidae |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.05 | 0.77 | 0.00 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.05 | 0.18 | 0.00 |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.05 | 0.27 | 0.00 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 0.05 | 0.32 | 0.00 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 0.00 | 0.18 | 0.00 |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 0.00 | 0.23 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.00 | 0.14 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.00 | 0.05 | 0.00 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 0.00 | 1.05 | 0.05 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 0.05 | 0.45 | 0.00 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 0.05 | 0.36 | 0.00 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-09-13 | 0.05 | 0.00 | 0.05 | 0.00 | 0.00 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.05 | 0.00 | 0.00 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.00 | 0.05 | 0.00 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.00 | 0.09 | 0.00 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.05 | 0.00 | 0.09 |
| after | control | 2019-10-04 | 0.09 | 0.00 | 0.00 | 0.05 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.27 | 0.05 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 | 0.14 | 0.00 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 | 0.09 | 0.00 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-15 | 0.05 | 0.00 | 0.05 | 0.05 | 0.00 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |

| after | control | 2020-10-05 | 0.00 | 0.00 | 0.05 | 0.00 | 0.00 |
|-------------|-------------|------------|--------------|--------------|-------|------|------|
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.05 | 0.00 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Sample Time | Sample Site | Date | Synchaetidae | Conochilidae | other | | |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.05 | | |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.00 | | |
| before | control | 2019-07-10 | 0.00 | 0.00 | 0.05 | | |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 0.09 | | |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 0.05 | | |
| before | impact | 2019-07-10 | 0.00 | 0.00 | 0.00 | | |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.00 | | |
| before | control | 2019-08-07 | 0.05 | 0.00 | 0.00 | | |
| before | control | 2019-08-07 | 0.00 | 0.05 | 0.00 | | |
| before | impact | 2019-08-07 | 0.00 | 0.09 | 0.00 | | |
| before | impact | 2019-08-07 | 0.00 | 0.05 | 0.00 | | |
| before | impact | 2019-08-07 | 0.00 | 0.05 | 0.00 | | |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.14 | | |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2020-09-15 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2020-09-15 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2020-10-05 | 0.00 | 0.00 | 0.00 | | |
| after | control | 2020-10-05 | 0.00 | 0.05 | 0.45 | | |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 | | |
| after | impact | 2020-10-05 | 0.00 | 0.00 | 0.00 | | |

Appendix D – Dog Lake

Table D-1. Abiotic variables and zooplankton density (3 replicates) from Dog Lake.

| Sample Time | Sample Site | Date | Temp. (°C) | DO (%) | Cond. (µS) | pH | TN (mg/L) | TP (mg/L) | Zooplankton Density (per L) |
|-------------|-------------|------------|------------|--------|------------|------|-----------|-----------|-----------------------------|
| before | control | 2019-07-09 | 25.60 | 125.90 | 157.90 | 9.32 | 0.50 | 0.01 | 24.86 |
| before | control | 2019-07-09 | 25.60 | 130.70 | 157.90 | 9.29 | 0.52 | 0.01 | 22.09 |
| before | control | 2019-07-09 | 25.70 | 136.00 | 157.60 | 9.34 | 0.59 | 0.02 | 28.77 |
| before | impact | 2019-07-09 | 25.60 | 123.30 | 155.90 | 9.27 | 0.57 | 0.02 | 20.68 |
| before | impact | 2019-07-09 | 25.50 | 122.10 | 157.30 | 9.30 | 0.53 | 0.01 | 22.36 |
| before | impact | 2019-07-09 | 25.60 | 125.10 | 157.40 | 9.33 | 0.54 | 0.02 | 23.36 |
| before | control | 2019-08-07 | 25.10 | 77.30 | 161.10 | 8.98 | 0.57 | 0.02 | 182.86 |
| before | control | 2019-08-07 | 25.10 | 81.70 | 160.50 | 9.05 | 0.58 | 0.02 | 191.95 |
| before | control | 2019-08-07 | 25.10 | 71.40 | 160.80 | 9.05 | 0.67 | 0.02 | 76.32 |
| before | impact | 2019-08-07 | 25.10 | 75.60 | 162.30 | 8.79 | 0.56 | 0.02 | 201.00 |
| before | impact | 2019-08-07 | 25.10 | 73.60 | 162.60 | 8.73 | 0.66 | 0.02 | 129.41 |
| before | impact | 2019-08-07 | 25.10 | 74.20 | 162.00 | 8.80 | 0.63 | 0.02 | 161.82 |
| after | control | 2019-09-13 | 19.40 | 87.90 | 147.10 | 8.90 | 0.58 | 0.02 | 139.32 |
| after | control | 2019-09-13 | 19.40 | 87.70 | 147.10 | 8.91 | 0.58 | 0.02 | 63.27 |
| after | control | 2019-09-13 | 19.40 | 91.10 | 147.00 | 8.93 | 0.61 | 0.02 | 159.86 |
| after | impact | 2019-09-13 | 19.40 | 92.90 | 146.90 | 8.97 | 0.62 | 0.02 | 165.73 |
| after | impact | 2019-09-13 | 19.40 | 94.30 | 146.90 | 8.99 | 0.61 | 0.02 | 59.09 |
| after | impact | 2019-09-13 | 19.40 | 96.50 | 146.80 | 8.98 | 0.63 | 0.02 | 130.86 |
| after | control | 2019-10-04 | 15.90 | 75.50 | 135.20 | 8.05 | 0.68 | 0.02 | 201.09 |
| after | control | 2019-10-04 | 15.90 | 75.10 | 135.00 | 8.05 | 0.68 | 0.02 | 303.50 |
| after | control | 2019-10-04 | 15.80 | 77.70 | 135.00 | 8.02 | 0.69 | 0.02 | 230.09 |
| after | impact | 2019-10-04 | 15.60 | 71.10 | 134.50 | 8.10 | 0.69 | 0.03 | 373.91 |
| after | impact | 2019-10-04 | 15.70 | 73.10 | 134.60 | 8.07 | 0.72 | 0.03 | 393.68 |
| after | impact | 2019-10-04 | 15.60 | 73.10 | 134.40 | 8.01 | 0.77 | 0.03 | 394.14 |
| after | control | 2020-09-16 | 18.80 | 89.60 | 142.50 | 8.85 | 0.68 | 0.02 | 31.70 |
| after | control | 2020-09-16 | 18.70 | 89.60 | 142.00 | 8.79 | 0.64 | 0.02 | 23.15 |
| after | control | 2020-09-16 | 18.40 | 80.60 | 141.30 | 8.78 | 0.65 | 0.02 | 27.00 |
| after | impact | 2020-09-16 | 18.60 | 88.30 | 141.70 | 9.08 | 0.62 | 0.02 | 18.65 |
| after | impact | 2020-09-16 | 18.30 | 89.00 | 140.60 | 9.00 | 0.65 | 0.02 | 33.00 |
| after | impact | 2020-09-16 | 18.20 | 83.00 | 142.30 | 8.91 | 0.63 | 0.02 | 27.50 |
| after | control | 2020-09-23 | 17.30 | 92.30 | 138.80 | 8.80 | 0.74 | 0.02 | 17.55 |
| after | control | 2020-09-23 | 17.20 | 92.90 | 136.80 | 8.75 | 1.03 | 0.02 | 17.05 |
| after | control | 2020-09-23 | 16.80 | 81.50 | 136.80 | 8.73 | 0.73 | 0.02 | 19.60 |
| after | impact | 2020-09-23 | 17.80 | 105.70 | 139.70 | 8.70 | 0.65 | 0.03 | 52.50 |
| after | impact | 2020-09-23 | 17.70 | 94.40 | 139.00 | 8.67 | 0.60 | 0.03 | 33.05 |
| after | impact | 2020-09-23 | 15.70 | 92.10 | 136.20 | 8.68 | 0.68 | 0.02 | 71.95 |

Table D-2. Average zooplankton family taxa count (3 replicates) per litre of water from Dog Lake.

| Sample Time | Sample Site | Date | Chydoridae | Bosminidae | Macrothricidae | Sididae | Daphniidae |
|-------------|-------------|------------|--------------|--------------|----------------|-----------|------------|
| before | control | 2019-07-09 | 0.45 | 0.18 | 0.00 | 2.59 | 0.00 |
| before | control | 2019-07-09 | 0.18 | 0.45 | 0.09 | 2.64 | 0.00 |
| before | control | 2019-07-09 | 0.50 | 0.23 | 0.00 | 2.27 | 0.00 |
| before | impact | 2019-07-09 | 0.41 | 0.68 | 0.00 | 4.68 | 0.00 |
| before | impact | 2019-07-09 | 0.36 | 0.23 | 0.00 | 5.00 | 0.00 |
| before | impact | 2019-07-09 | 0.55 | 0.32 | 0.00 | 4.45 | 0.00 |
| before | control | 2019-08-07 | 10.64 | 15.05 | 0.23 | 66.41 | 0.27 |
| before | control | 2019-08-07 | 11.05 | 4.55 | 0.05 | 84.77 | 0.55 |
| before | control | 2019-08-07 | 7.50 | 5.45 | 0.05 | 34.86 | 0.00 |
| before | impact | 2019-08-07 | 8.32 | 10.23 | 0.09 | 29.23 | 0.00 |
| before | impact | 2019-08-07 | 4.36 | 3.00 | 0.09 | 25.50 | 0.00 |
| before | impact | 2019-08-07 | 5.86 | 4.73 | 0.23 | 35.36 | 0.00 |
| after | control | 2019-09-13 | 2.91 | 18.27 | 0.05 | 22.32 | 0.00 |
| after | control | 2019-09-13 | 6.41 | 12.64 | 0.00 | 14.59 | 0.00 |
| after | control | 2019-09-13 | 10.50 | 39.32 | 0.05 | 16.77 | 0.00 |
| after | impact | 2019-09-13 | 8.82 | 22.36 | 0.05 | 11.32 | 0.00 |
| after | impact | 2019-09-13 | 7.50 | 15.45 | 0.00 | 6.91 | 0.05 |
| after | impact | 2019-09-13 | 6.95 | 20.86 | 0.05 | 9.27 | 0.14 |
| after | control | 2019-10-04 | 11.82 | 55.27 | 0.00 | 9.82 | 0.00 |
| after | control | 2019-10-04 | 14.68 | 108.00 | 0.00 | 15.82 | 0.00 |
| after | control | 2019-10-04 | 8.95 | 83.68 | 0.09 | 12.64 | 0.00 |
| after | impact | 2019-10-04 | 22.91 | 157.64 | 0.55 | 12.27 | 0.00 |
| after | impact | 2019-10-04 | 42.00 | 197.73 | 0.00 | 16.77 | 0.00 |
| after | impact | 2019-10-04 | 35.18 | 171.27 | 0.59 | 21.55 | 0.00 |
| after | control | 2020-09-16 | 2.95 | 3.41 | 0.05 | 5.14 | 0.45 |
| after | control | 2020-09-16 | 1.86 | 5.00 | 0.00 | 4.64 | 0.09 |
| after | control | 2020-09-16 | 2.27 | 3.59 | 0.05 | 4.82 | 0.23 |
| after | impact | 2020-09-16 | 2.23 | 2.82 | 0.00 | 5.14 | 0.14 |
| after | impact | 2020-09-16 | 2.73 | 4.82 | 0.05 | 9.50 | 0.36 |
| after | impact | 2020-09-16 | 2.14 | 4.50 | 0.00 | 6.55 | 0.36 |
| after | control | 2020-09-23 | 1.64 | 2.23 | 0.05 | 4.36 | 0.32 |
| after | control | 2020-09-23 | 1.41 | 2.55 | 0.05 | 3.91 | 0.18 |
| after | control | 2020-09-23 | 2.55 | 1.68 | 0.05 | 2.64 | 0.18 |
| after | impact | 2020-09-23 | 1.14 | 5.86 | 0.05 | 7.50 | 0.14 |
| after | impact | 2020-09-23 | 1.59 | 6.36 | 0.00 | 7.36 | 0.18 |
| after | impact | 2020-09-23 | 1.86 | 17.09 | 0.09 | 12.59 | 0.55 |
| Sample Time | Sample Site | Date | Holopediidae | Polyphemidae | Cyclopoida | Calanoida | nauplius |
| before | control | 2019-07-09 | 0.00 | 0.00 | 11.73 | 0.00 | 8.45 |
| before | control | 2019-07-09 | 0.00 | 0.14 | 7.55 | 0.00 | 9.14 |

| | | | | | | | |
|--------|---------|------------|------|------|--------|------|-------|
| before | control | 2019-07-09 | 0.00 | 0.27 | 13.64 | 0.00 | 10.59 |
| before | impact | 2019-07-09 | 0.00 | 0.00 | 6.18 | 0.00 | 6.64 |
| before | impact | 2019-07-09 | 0.00 | 0.00 | 7.86 | 0.00 | 6.73 |
| before | impact | 2019-07-09 | 0.00 | 0.00 | 6.86 | 0.00 | 8.95 |
| before | control | 2019-08-07 | 0.09 | 0.00 | 82.64 | 0.00 | 1.45 |
| before | control | 2019-08-07 | 0.00 | 0.95 | 82.41 | 0.00 | 1.18 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 25.32 | 0.00 | 0.50 |
| before | impact | 2019-08-07 | 0.05 | 0.00 | 143.82 | 0.00 | 5.64 |
| before | impact | 2019-08-07 | 0.05 | 0.00 | 91.82 | 0.00 | 1.00 |
| before | impact | 2019-08-07 | 0.09 | 0.00 | 112.14 | 0.00 | 0.73 |
| after | control | 2019-09-13 | 0.05 | 0.00 | 56.50 | 0.14 | 34.45 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 21.18 | 0.00 | 5.45 |
| after | control | 2019-09-13 | 0.09 | 0.00 | 53.91 | 0.05 | 25.41 |
| after | impact | 2019-09-13 | 0.09 | 0.00 | 64.55 | 0.00 | 45.77 |
| after | impact | 2019-09-13 | 0.05 | 0.00 | 14.45 | 0.00 | 5.09 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 58.59 | 0.00 | 31.41 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 106.09 | 0.00 | 13.14 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 139.91 | 0.00 | 17.86 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 102.05 | 0.00 | 18.86 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 130.36 | 0.00 | 17.18 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 75.00 | 0.00 | 0.55 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 108.82 | 0.00 | 4.36 |
| after | control | 2020-09-16 | 0.00 | 0.00 | 12.77 | 0.00 | 1.77 |
| after | control | 2020-09-16 | 0.00 | 0.00 | 7.59 | 0.00 | 0.91 |
| after | control | 2020-09-16 | 0.00 | 0.00 | 9.32 | 0.05 | 2.18 |
| after | impact | 2020-09-16 | 0.00 | 0.00 | 5.00 | 0.00 | 0.32 |
| after | impact | 2020-09-16 | 0.00 | 0.00 | 10.59 | 0.00 | 1.00 |
| after | impact | 2020-09-16 | 0.00 | 0.00 | 9.18 | 0.00 | 0.77 |
| after | control | 2020-09-23 | 0.00 | 0.00 | 4.86 | 0.00 | 1.14 |
| after | control | 2020-09-23 | 0.00 | 0.00 | 3.95 | 0.00 | 2.95 |
| after | control | 2020-09-23 | 0.00 | 0.00 | 5.82 | 0.00 | 4.32 |
| after | impact | 2020-09-23 | 0.00 | 0.00 | 13.64 | 0.00 | 18.14 |
| after | impact | 2020-09-23 | 0.00 | 0.00 | 6.18 | 0.00 | 8.14 |
| after | impact | 2020-09-23 | 0.00 | 0.00 | 16.91 | 0.00 | 16.09 |

| Sample Time | Sample Site | Date | Ostracoda | Chironomidae | Hydrachnida | Brachionidae | Asplanchnidae |
|-------------|-------------|------------|-----------|--------------|-------------|--------------|---------------|
| before | control | 2019-07-09 | 0.00 | 0.00 | 0.00 | 0.18 | 0.00 |
| before | control | 2019-07-09 | 0.00 | 0.00 | 0.09 | 0.36 | 0.05 |
| before | control | 2019-07-09 | 0.00 | 0.00 | 0.00 | 0.32 | 0.00 |
| before | impact | 2019-07-09 | 0.00 | 0.09 | 0.05 | 0.18 | 0.18 |
| before | impact | 2019-07-09 | 0.14 | 0.05 | 0.23 | 0.18 | 0.09 |
| before | impact | 2019-07-09 | 0.09 | 0.00 | 0.14 | 0.23 | 0.09 |
| before | control | 2019-08-07 | 0.05 | 0.00 | 0.00 | 3.77 | 2.27 |

| | | | | | | | |
|--------|---------|------------|------|------|------|-------|------|
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.05 | 2.91 | 3.32 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.14 | 1.64 | 0.77 |
| before | impact | 2019-08-07 | 1.00 | 0.00 | 0.09 | 1.32 | 1.23 |
| before | impact | 2019-08-07 | 0.86 | 0.00 | 0.14 | 0.55 | 0.91 |
| before | impact | 2019-08-07 | 0.82 | 0.00 | 0.05 | 0.86 | 0.64 |
| after | control | 2019-09-13 | 0.59 | 0.00 | 0.00 | 4.00 | 0.05 |
| after | control | 2019-09-13 | 0.23 | 0.00 | 0.00 | 2.77 | 0.00 |
| after | control | 2019-09-13 | 0.05 | 0.00 | 0.00 | 13.68 | 0.05 |
| after | impact | 2019-09-13 | 0.05 | 0.00 | 0.09 | 12.59 | 0.00 |
| after | impact | 2019-09-13 | 0.05 | 0.00 | 0.05 | 9.50 | 0.00 |
| after | impact | 2019-09-13 | 0.14 | 0.00 | 0.09 | 3.09 | 0.14 |
| after | control | 2019-10-04 | 0.05 | 0.00 | 0.05 | 4.86 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 | 7.14 | 0.09 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 | 3.41 | 0.23 |
| after | impact | 2019-10-04 | 0.27 | 0.00 | 0.00 | 32.45 | 0.00 |
| after | impact | 2019-10-04 | 1.91 | 0.00 | 0.00 | 59.73 | 0.00 |
| after | impact | 2019-10-04 | 1.09 | 0.00 | 0.00 | 51.27 | 0.00 |
| after | control | 2020-09-16 | 1.23 | 0.00 | 0.14 | 0.05 | 0.73 |
| after | control | 2020-09-16 | 0.14 | 0.00 | 0.09 | 0.14 | 0.45 |
| after | control | 2020-09-16 | 1.36 | 0.00 | 0.00 | 0.00 | 0.36 |
| after | impact | 2020-09-16 | 0.68 | 0.00 | 0.32 | 0.00 | 0.00 |
| after | impact | 2020-09-16 | 0.18 | 0.00 | 0.00 | 0.05 | 0.32 |
| after | impact | 2020-09-16 | 0.68 | 0.00 | 0.05 | 0.05 | 0.68 |
| after | control | 2020-09-23 | 0.77 | 0.00 | 0.00 | 0.14 | 0.41 |
| after | control | 2020-09-23 | 0.05 | 0.00 | 0.00 | 0.09 | 0.32 |
| after | control | 2020-09-23 | 0.36 | 0.00 | 0.05 | 0.05 | 0.14 |
| after | impact | 2020-09-23 | 0.36 | 0.00 | 0.00 | 0.00 | 0.41 |
| after | impact | 2020-09-23 | 0.14 | 0.00 | 0.00 | 0.09 | 0.00 |
| after | impact | 2020-09-23 | 0.05 | 0.00 | 0.05 | 0.00 | 0.14 |

| Sample Time | Sample Site | Date | Synchaetidae | Conochilidae | other |
|-------------|-------------|------------|--------------|--------------|-------|
| before | control | 2019-07-09 | 0.00 | 1.27 | 0.00 |
| before | control | 2019-07-09 | 0.00 | 1.41 | 0.00 |
| before | control | 2019-07-09 | 0.00 | 0.95 | 0.00 |
| before | impact | 2019-07-09 | 0.00 | 1.59 | 0.00 |
| before | impact | 2019-07-09 | 0.00 | 1.50 | 0.00 |
| before | impact | 2019-07-09 | 0.00 | 1.68 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.00 | 0.00 |
| before | control | 2019-08-07 | 0.00 | 0.14 | 0.05 |
| before | control | 2019-08-07 | 0.00 | 0.09 | 0.00 |
| before | impact | 2019-08-07 | 0.00 | 0.00 | 0.00 |
| before | impact | 2019-08-07 | 0.00 | 1.09 | 0.05 |
| before | impact | 2019-08-07 | 0.00 | 0.32 | 0.00 |

| | | | | | |
|-------|---------|------------|------|------|------|
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-09-13 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.05 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-09-13 | 0.00 | 0.00 | 0.14 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.00 | 0.00 |
| after | control | 2019-10-04 | 0.00 | 0.18 | 0.00 |
| after | impact | 2019-10-04 | 0.00 | 0.27 | 0.00 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 |
| after | impact | 2019-10-04 | 0.00 | 0.00 | 0.00 |
| after | control | 2020-09-16 | 0.00 | 0.00 | 0.14 |
| after | control | 2020-09-16 | 0.00 | 0.14 | 0.00 |
| after | control | 2020-09-16 | 0.00 | 0.32 | 0.00 |
| after | impact | 2020-09-16 | 0.00 | 0.00 | 0.32 |
| after | impact | 2020-09-16 | 0.00 | 0.41 | 0.00 |
| after | impact | 2020-09-16 | 0.00 | 0.05 | 0.00 |
| after | control | 2020-09-23 | 0.00 | 0.00 | 0.05 |
| after | control | 2020-09-23 | 0.00 | 0.05 | 0.00 |
| after | control | 2020-09-23 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-23 | 0.00 | 0.50 | 0.00 |
| after | impact | 2020-09-23 | 0.00 | 0.00 | 0.00 |
| after | impact | 2020-09-23 | 0.00 | 0.00 | 0.00 |
