**Aquatic macrophytes and periphyton communities as bioindicators of lake trophic status in Riding Mountain National Park, Manitoba**

by

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B.Sc. (Hon), University of British Columbia Okanagan, 2009

A THESIS SUBMITTED IN PARTIAL FULLFILLMENT OF THE REQUIREMENTS

FOR THE DEGREE OF

Master of Science

in

THE COLLEGE OF GRADUATE STUDIES

(Environmental Sciences)

THE UNIVERSITY OF BRITISH COLUMBIA

(Okanagan)

May 2015

**Abstract**

Aquatic conservation practitioners at Riding Mountain National Park of Canada (RMNP) are concerned with maintaining and restoring the ecological integrity of lakes within the park; thus, there is a need to identify lakes potentially at risk of eutrophication. The ability to identify at-risk lakes would allow lake managers to alleviate stressors, such as excess nutrients, before irreversible changes occur in the ecosystems.

Aquatic macrophytes and periphyton have potential as bioindicators of lake trophic status. Both have a widespread distribution, their growth is tightly coupled with water clarity and they obtain their nutrients directly from the water column or lake bed, making them sensitive to changes in water quality. Previous studies have identified species diversity, and macrophyte and periphyton species as reliable predictors of lake ecological status.

Aquatic macrophyte and periphyton surveys were conducted in forty-five and thirty lakes, respectively, in and immediately surrounding Riding Mountain National Park, Manitoba. Non-metric multidimensional scaling, redundancy analysis and generalized linear models identified submerged aquatic macrophyte species diversity, *Ruppia maritima*, *Potamogeton pectinatus* and *Chara* spp. as having potential use as bioindicators of ecological change in Riding Mountain National Park lakes. Periphyton taxon diversity and individual taxa were not recommended for inclusion in future monitoring in Riding Mountain National Park without further study. It is my recommendation to incorporate aquatic macrophyte species and diversity monitoring along with established aquatic monitoring programs in Riding Mountain National Park.

## Preface

This research project was conducted as part of Riding Mountain National Parks' Clear Lake Action on the Ground Initiative in conjunction with the University of British Columbia Okanagan. The written works and figures were largely produced on my own with advice from Ian Walker, Marlow Pellatt and Jeff Curtis. Figure 3.1 was available from the Riding Mountain National Park Round Table (2012) and is reproduced in this thesis with permission (See Appendix A). Carrie White produced Figure 3.2 and it is used here with permission.

Ian Walker, Marlow Pellatt and Jeff Curtis offered guidance in the development of sampling protocols, overall project design and manuscript editing. Ross Robinson, Carrie White and Maggie Stuart assisted in the field data collection. The aquatic macrophyte identifications were largely done by me with the assistance of Ian Walker. Mark Graham of the University of Alberta provided the algal identifications and enumerations.

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**Acknowledgments**

Where to start?! First I need to thank Ian Walker for guiding me throughout my undergraduate degree and all the way to the finish line of graduate studies. Who knew where a summer job counting chironomids would take me! Thank you to the rest of my graduate committee for putting up with my thesis procrastination. Marlow Pellatt and Jeff Curtis – you two have been the most understanding committee members I could ask for.

I am very grateful to Parks Canada (and my Marlow connection) for generously funding this project, as I embarked upon it five years ago. I am especially appreciative of everybody at the Wildlife Lab at Riding Mountain National Park, Manitoba. In particular, thank you Bob Reside and Roxy Grzela for all your helpful ideas. I want to give a very special thanks to Ross Robinson of RMNP for driving me around the backcountry for two years and never complaining. I'll never forget the fire you started for me in the middle of a trail in the pouring rain so I could thaw out my feet. Ross you can literally do anything!

Erin Fleming, thank you so much for being my best friend during these trying years and for always being willing to sympathize. In a small way I am so glad that Peter made me cry that day. Carrie White, my second year in the field would not have been nearly as fun or productive without you! Thank you for your friendship and always truthful, blunt advice. I especially want to thank my parents and grandparents for their prayers, phone calls and financial backing in this endeavour. To my husband, who was not around when I started this ride but was there to help me finish, I love you. Lastly, thank you Lord for not letting procrastination get the best of me!

**Dedication**

To Nixie,
Thank you for all the games of fetch, sleepless nights and snuggles.
You truly are the luckiest puppy in Manitoba!!

## Chapter 1: Introduction

Ecological integrity is defined by the National Parks Policy Act as, ". . . *a condition that is determined to be characteristic of its natural region and likely to persist, including abiotic components and the composition and abundance of native species and biological communities, rates of change and supporting processes* (Parks Canada 2012)*."* The preservation of ecological integrity (EI) is an important mandate of Parks Canada (Action on the Ground, 2005); thus so is the management and preservation of lake ecosystems. Essential to this mandate is ongoing monitoring and assessment of the current trophic condition and changes in the lake ecosystems. Early detection of potential changes would be especially useful, allowing managers a chance to mitigate significant damage. Once a lake has changed it is commonly very difficult to return a lake to its original healthy condition (Jeppesen *et. al.* 1997; Lauridsen *et. al.* 2003; Søndergaard *et. al.* 2010).

Of particular concern to lake managers are ecological thresholds, "tipping points where a small change in an ecological driver will produce a large and potentially permanent change in an ecosystem (Groffman *et al.,* 2006; Scheffer and Carpenter, 2003). This problem is particularly evident with lake eutrophication. A lake ecosystem that crosses a certain nutrient enrichment threshold may no longer be able to buffer against the effects of further increases in nutrient concentration; thus, a rapid, undesired deterioration in lake water quality occurs. Defining ecological thresholds is difficult because lakes are dynamic ecosystems with many factors that contribute to their ability to resist changes, including those resulting from nutrient loading (Groffman *et al.,* 2006).

In this study, I assess the utility of aquatic macrophytes and periphyton as potential indicators of ecosystem changes and lake trophic status in and around Riding Mountain National Park (RMNP), MB. Submerged macrophytes were used because they have a widespread distribution, play important ecological roles for wildlife, including invertebrates, and enhance lake substrate stability (Dennison *et al.* 1993; Heegaard *et al*, 2001). Additionally the leaves and roots are in close contact with their chemical environment, the surrounding water, making them sensitive to changing water quality parameters (Dennison *et al.* 1993; Heegaard *et al*, 2001). Macrophytes are the principal primary producers in shallow lakes and ponds unless nutrient loading is especially high (Søndergaard *et al*. 2010). As well, literature has shown that aquatic macrophytes have potential as water quality indicators, and as indicators of lake trophic status (Dennison *et al.* 1993; Heegaard *et al.* 2001; Lacoul and Freedman 2006; Søndergaard *et al*. 2010).

In addition to macrophyte sampling I also assess periphyton as bioindicators. Periphyton communities, like aquatic macrophytes, are believed to reflect changes in water quality conditions (McCormick and O'Dell 1996). Diatoms are particularly well known as bioindicators and have long been used to characterize current lake trophic conditions, and also to reconstruct past ecosystem changes (Dixit and Smol 1994; Flower 2005; Prygiel *et al.*, 1999; Reid *et al.,* 1995; Taylor *et al.*, 2007). Furthermore, because algae rapidly reproduce and have very short life spans, their communities change very rapidly; thus, periphyton may provide earlier indications of ecosystem change, as compared to aquatic macrophytes (McCormick *et al.,* 1996).

For my assessments I sampled aquatic macrophytes and periphyton in forty-five lakes in and around Riding Mountain National Park, Manitoba, examining their

abundance and diversity in relation to water quality variables. I predict there will be changes in macrophyte and periphyton diversity and community assemblages along the trophic (nutrient) gradient that will be useful in identifying ecological thresholds, or tipping points, between trophic classes in lakes. Indicator species of macrophytes and periphyton will also be identified to help to classify the current trophic condition of RMNP lakes.

The overall objective of this thesis is to identify ecological thresholds between lake trophic states, classify current lake trophic status and predicting future lake trophic states of Riding Mountain National Park lakes using aquatic macrophytes and periphyton. The findings in this thesis will aid in the management of the aquatic ecosystems in Riding Mountain National Park and help attain Parks Canada's goal of maintenance of ecological integrity.

# Chapter 2: Literature Review

## 2.1. Aquatic Macrophytes

There are four main groups of aquatic macrophytes: emergent, floating-leaved, submerged and free-floating (Sculthorpe 1967; Wetzel 2001). While these groups share some similar adaptations, the structure and physiology of each group are quite distinct. Similarities include thin cuticle and epidermis, relic stomata, poorly lignified xylem, roots and rhizomes subject to low oxygen conditions and, in general, reproductive structures (e.g., flowers) that rise above the water (Sculthorpe 1967).

### 2.1.1 *Emergent macrophytes*

While there is some debate as to the exact boundary between a terrestrial and emergent aquatic plant the most accepted definition defines an aquatic plant as it growing in water, in soil covered by water or in waterlogged soil for the majority of its life cycle (Sculthorpe 1967; Wetzel 2001).

Emergent aquatic plants are fairly similar to their terrestrial counterparts. The stems are vertical and rigid, yet flexible, to compensate for both wind and water disturbances (Sculthorpe 1967; Wetzel 2001). Emergent aquatic plants possess well-developed xylem and phloem, which are used to transport nutrients and water to plant tissues. The well-developed vascular system of emergent aquatic plants allows the transport of nutrients acquired from the aquatic system into the aerial portion of the plant. The root system and new growth are subject to anoxic or low oxygen conditions; they must obtain oxygen from the aerial parts of the plant, respire anaerobically (Laing 1940),

or store gases in specialized secondary air storage tissues called aerenchyma, until their foliage grows above the water surface (Sculthorpe 1967; Wetzel 2001). The emergent parts of the plant are able to transpire and therefore transport nutrients from within the aquatic system to another environment (Wetzel 2001).

### *2.1.2. Floating-leaved macrophytes*

Floating-leaved macrophytes have an unusual structure related to the unique position of their leaves at the air-water interface. Their leaves are leathery, often circular, marginated and hydrophobic, with a strong, pliable, long petiole to survive high amounts of water and wind stress (Sculthorpe 1967; Wetzel 2001).

The floating leaves are differentiated with the photosynthetic mesophyll tissue occurring on the upper portion of the leaf. The stomata are restricted to the upper portion of the leaf, with a few species showing functionless stomata on the underside (Sculthorpe 1967).

The underside of many floating-leaved plants contains hydropotens. These structures absorb ions in the water column, in a manner similar to root absorption. The absorbed ions are then transported to veins in the mesophyll for distribution (Sculthorpe 1967; Wetzel 2001). The leaf also contains masses of spongy tissue for buoyancy, collenchyma cells for strength, and veins of xylem and phloem for water and nutrient transport (Sculthorpe 1967; Wetzel 2001).

*2.1.3. Submerged macrophytes*

This group includes filamentous algae capable of producing large mats, macroalgae such as *Chara* spp., and many bryophytes and vascular plants (Sculthorpe 1967; Wetzel 2001). Common amongst all these types of organisms is their lack of structural tissue, reduced or absent vascular system, absent or reduced stomata, extremely thin cuticle and leaves, and often chloroplasts in the epidermal tissue. The thin cuticle allows for a more permeable membrane, facilitating gas, and possibly nutrient, exchange; although the preferred avenue for nutrient uptake is still via the root system (if rooting is present) (Sculthorpe 1967; Wetzel 2001).

Most species have highly modified leaves that maximize the surface area to volume ratio in order to increase the efficiency of gas and mineral absorption, when absorption of nutrients through a root system is unavailable or insufficient to meet the plant's nutrient requirements. One notable unique submerged genus is *Utricularia*, which has highly modified vegetative organs that capture and digest minute animals to supplement its nutritional needs (Sculthorpe 1967; Wetzel 2001). As well, the whorled and thin, elongated leaves offer less frictional resistance to waves and water currents; thus, the plant is less likely to be uprooted or destroyed (Sculthorpe 1967).

Photosynthesis takes place entirely underwater and is restricted by the availability of light and carbon dioxide. Consequently, submerged vascular plants commonly have thin cuticles, epidermal chloroplasts, and no stomata and do not release nutrients and gases outside of the aquatic environment.

*2.1.4. Free-floating macrophytes*

The morphology of this group is very diverse. The main plant form has aerial, surface floating, or submerged leaves, a condensed stem, little structural tissue, a highly reduced vascular system, and roots suspended in the water column. All of their nutrients (apart from carbon) must be absorbed from the water column; thus, these species are often found in waters rich in ions and other nutrients (Sculthorpe 1967; Wetzel 2001).

Leaf placement (aerial, surface-floating or submerged) dictates the cell organization. Aerial leaves resemble those of emergent plants; surface-floating leaves resemble those of floating-leaved plants, and the leaves of submerged free-floating species resemble those of the rooted submerged group. Where this group differs is in the placement of the chloroplasts; they can be located on the lower epidermis or throughout the floating leaf (Sculthorpe 1967).

*2.1.5. Reproduction*

Aquatic plants may reproduce sexually, asexually or via both mechanisms. Asexual reproduction is by far the most common and is achieved by fragmentation or lateral expansion via rhizomes, shoot bases, root suckers or tubers (Sculthorpe 1967; Wetzel 2001). Dispersal may be achieved via seeds (flowering plants), spores (bryophytes and pteridophytes), or fragments and vegetative propagules (e.g., turions).

Turions are masses composed of dormant vegetative tissue that have the ability to overwinter and then become active with increasing temperature. Dispersal of turions and fragments is through water movement, wind movement, birds, mammals or humans (Sculthorpe 1967; Wetzel 2001).

For flowering plants, pollination is essential for sexual reproduction. In some aquatic plants, pollen may transfer via water currents instead of wind or insect vectors (Sculthorpe 1967; Wetzel 2001).

## 2.2. Periphyton

There is great diversity across all algal species. They are distributed amongst nine major taxonomic groups: Cyanobacteria (blue-green bacteria), Chlorophyta (green algae), Xanthophyceae (yellow-green algae), Chrysophyceae (golden algae), Bacillariophyceae (diatoms), Cryptophyta (cryptophytes), Pyrrhophyta (dinoflagellates), Euglenophyta (euglenophytes), Rhodophyta (red algae) and Phaeophyta (brown algae) (Wetzel 2001).

Algae can be planktonic or benthic. Periphytic (attached) species grow on diverse surfaces and may be epilithic (on rocks), epizooic (on animals) or epiphytic (on plants). Algal species distributions differ both spatially (horizontally and vertically in the water column) as well as seasonally.

Although the diversity is large there are many characteristics that are common among all groups. Every group has the ability to photosynthesize, react rapidly to nutrient and sunlight inputs, and commonly reproduce both sexually and asexually (Harris 1986; Wetzel 2001). Photosynthesis is similar to that of higher plants but algae frequently contain different pigments, potentially allowing them to utilize different light spectra than aquatic macrophytes. Asexual reproduction is also quite similar. Most groups undergo vegetative division or fragmentation. Some species undergo regular sexual reproduction (Harris 1986; Wetzel 2001).

*2.2.1. Factors controlling periphyton distribution and growth*

Periphyton and other algal taxa have the ability to grow rapidly when conditions are ideal. This depends on many factors such as light, substrate, nutrients and competition (Harris 1986; Hill 1996; Rodusky *et al.*, 2001; Wetzel 2001). Light provides the energy necessary for photosynthesis and there is an increase in algal growth with an increase in irradiance up until photoinhibition occurs, either through blooms of phytoplankton in the water column or self-shading (Harris 1986; Rodusky *et al.*, 2001; Wetzel 2001). Periphytic and benthic algal species, have the ability to photosynthesize at lower light levels than planktonic species and will grow in patchy distributions where growth is concentrated in areas of light penetration, in order to meet their needs (Hill 1996). Periphyton are often nutrient limited in oligotrophic lakes and transition to being light limited in eutrophic-hypereutrophic lakes (Hannson 1992; Rodusky *et al.* 2001). Periphyton have been shown to grow optimally in lakes with high values of total phosphorus coupled with clear water. Such conditions most commonly exist in mesotrophic lakes (Hannson 1992).

Nutrients are essential for algal growth and strongly influence algal communities since other needs are often met or exceeded. When an ecosystem is disturbed and a drastic reduction or increase in nutrients occurs, algae can respond extremely rapidly as illustrated in experiments by Schindler (1974; 1977) and others (Carpenter *et al.* 1998; Correll 1998; Smith 2003; Wetzel 2001). In one of the best known nutrient experiments, conducted by Schindler (1974), a single lake was divided into two basins. Each basin was fertilized with equal amounts of carbon and nitrogen. One basin also received phosphorus. The phosphorus rapidly stimulated a large phytoplankton bloom in the latter

basin while the other basin remained unchanged. This experiment highlighted the concept of limiting nutrients in lake ecosystems, and demonstrated the importance of phosphorus as a determinant of algal biomass.

Studies specific to periphyton have shown similar results. Phosphorus, and in some cases nitrogen, is the main nutrient determinant of periphyton growth (Hannson 1992; McCormick and O'Dell 1996; Rodusky *et al.* 2001). Interestingly, one advantage that periphyton growing on or in lake sediments have over phytoplankton and periphyton growing on other substrates is their ability to utilize nutrients trapped in the sediments. This reduces the possibility of nutrients becoming re-suspended in the water column and expands the nutrient bank that certain types of periphyton can use (Blumenshine *et al.* 1997; Hannson 1990).

### 2.3. Use of Aquatic Macrophytes as Bioindicators

Using biological characteristics to describe the aquatic chemical environment is well established (Palmer *et al.* 1992; Pearsall 1920; Seddon 1972; Spence 1967). Early work defining the nutrient tolerances of aquatic macrophytes using multivariate analysis, instead of subjective means, was conducted by Seddon (1972). He concluded that species' tolerances to environmental variables occur on a continuum and only a small number of species have nutrient ranges narrow enough to be considered indicative of a particular lake environment. These species included *Potamogeton pectinatus*, which occurred most often in the eutrophic range, *Lemna trisulca*, *Lemna minor*, *Polygonum amphibium,* and *Ceratophyllum demersum* in mesotrophic lakes; and *Potamogeton perfoliatus* and *Elodea canadensis* in oligotrophic environments.

Many researchers have since compiled large country-wide datasets relating macrophytes to lakes' physical and chemical properties (Clayton and Edwards 2006; Heegaard *et al.* 2001; McElarney and Rippey 2009; Søndergaard *et al.* 2010). Many of these studies recognized, like Seddon (1972), that few aquatic plant species have sufficiently small environmental ranges to be considered good bioindicators. Subsequent research has shown that most of the trophic continuum is not well defined by individual species, and submerged aquatic plants alone are insufficient to define ecological thresholds or lake trophic status (Heegaard *et al.* 2001; McElarney and Rippey 2009; Søndergaard *et al*. 2010). Better results are obtained when aquatic macrophytes are used in combination with other ecological indicators, such as species richness, maximum colonization depth and percent cover (Penning *et al.* 2008; McElarney and Rippey 2009; Søndergaard *et al.* 2010).

When whole submerged plant communities are considered, instead of individual species, results have been more promising. Jeppesen *et al.* (2000) explored trophic structure, species richness and biodiversity in submerged macrophytes, zooplankton, phytoplankton and fish across five classes of lakes, defined by their total phosphorus concentrations. Community structure differences were assessed via species richness and Shannon-Weiner diversity. As total phosphorus concentrations increased, drastic reductions in submerged macrophyte abundance, species richness and depth of colonization were noted; p-values$< 0.05$, 0.05 and 0.001 respectively.

Another study (Penning *et al*. 2008), using three different trophic indices, found a good correspondence between these indices and aquatic plant communities as a whole. Penning *et al.* (2008) reviewed and compared a species richness index, trophic index (TI)

and lake trophic index (LTR) as methods for characterizing water quality across a total phosphorus gradient. Both TI and the LTR were insufficiently sensitive to detect sharp, well-documented differences in lake nutrient regimes. The authors Penning *et al.* (2008) and others (Heegaard *et al.* 2001; Søndergaard *et al.* 2010) agree that the best index must use a multimeric approach to predict and maintain ecological status. This would include assessments of submerged macrophyte species richness, abundance, percent cover, maximum growth depth and exotic species.

### 2.4. Use of Periphyton as Bioindicators

The use of algae as bioindicators of lake health varies considerably with algal life history. Phytoplankton have a long established relationship with nutrients in lake ecosystems, particularly phosphorus (Schindler 1977) and are generally used as one metric amongst many to assess the ecological status of a lake. Studies that have used phytoplankton as the only water quality metric conclude that the relationship can be quite variable for reliable predictions (Alahuhta *et al.* 2009; Søndergaard *et. al.* 2010). This variability arises from aquatic macrophyte and periphyton competition effects, availability of nutrients in the water column and sediments and grazing (Amengual-Morro *et al.* 2012; Gligora *et al.* 2007; Hannson 1990; Jeppesen *et. al.* 1997; McElarney and Rippey 2009; Søndergaard *et al.* 2005).

Some periphyton, specifically diatoms, are commonly used to infer past environments; specific species are known to have significant relationships with pH, oxygen levels and nutrient levels (Flower 2005; Reid *et al.* 1995). Paleolimnological studies are important for establishing lake reference conditions, against which present-

day studies of water quality can be compared (Dixit and Smol, 1994; Reid *et al.* 1995). Reid *et al.* (1995) outlines the history of diatom research and the progression from identifying present day condition using diatoms and water quality data then extrapolated the data to predict reference conditions with paleolimnological studies.

In general, diatoms have shown significant correlation to water quality variables (TN, TP, Secchi and pH) in numerous studies (Dixit and Smol 1994; Prygiel *et al.* 1999; Taylor *et al.* 2007). More recently, diatom species monitoring has been incorporated into the Water Framework Directive (WFD) as one metric for assessing water quality (European Commission 2000). The United States Environmental Protection Agency also has recommended the use of diatoms as bioindicators of aquatic habitats but currently only a few states have a diatom monitoring program (Barbour *et al.* 1999).

### 2.5. Eutrophication

Eutrophication refers to the process whereby increased nutrient loading leads to many undesirable effects. These effects include loss of aquatic plant diversity, increased aquatic plant biomass, increased algal production, decreased water clarity, increased anoxia due to decomposition of excess biomass, fish kills, food web changes, loss of drinking water quality and decreased recreational value (Carpenter *et al.* 1998; Smith 2003; Conley *et al.* 2009; Schindler 2012). The cause of eutrophication was long debated and initially proper management practices were difficult to establish. Schindler's (1977) whole lake experiments demonstrated that phosphorus was the key nutrient limiting algal production in lakes; thus, water quality improvements could be attained by managing excess phosphorus inputs. This finding was highly significant because it provided a clear

direction to manage lakes that were becoming eutrophied anthropogenically. For example, when phosphorus loading was drastically reduced many eutrophied lakes in Europe and North America saw improvements in water quality (Conley *et al.* 2009; NRC 1992). There was recognition however, that not all affected lakes saw immediate water quality improvements with decreased phosphorus loading (Carpenter *et al.* 1998; Carvalho 2011; Schindler 2012). When lakes have received large doses of excess nutrients over many years, nutrients, including phosphorus, begin to accumulate in the sediments. This leads to an internal recycling of phosphorus that can sustain the eutrophic state for many years despite decreased phosphorus inputs (Carpenter *et al.* 1998; Carvalho *et al.* 2011; Moss *et al.* 1996; Schindler 2012).

The process and reversal of eutrophication are difficult to predict (Jeppesen *et al.* 1997; Schindler 1977; Schindler 2012) because lakes vary enormously with respect to depth, area, water residence time, catchment area, climate (including wind, evaporation and precipitation) and underlying bedrock. These factors all influence how a lake will respond to excess nutrients (Carpenter *et al.* 1999). Many long-term, ecosystem scale experiments have proven that reduction in phosphorus inputs to lakes is the only reliable method to reduce eutrophication in lakes (Schindler 2012). Recovery of eutrophied lakes may be further assisted via decreased nitrogen loading, hypolimnetic oxygenation, sediment treatment and biomanipulation (Carpenter *et al.* 1999; Cooke *et al.* 1993). These claims are somewhat under dispute as none have been evaluated over the same time and scale as the phosphorus experiments (Schindler 2012). Additionally the authors who suggest alternate methods concede that their measures are often expensive and none

of the measures can completely reverse eutrophy unless the phosphorus source is also eliminated (Carpenter *et al.* 1999; Conley *et al.* 2009).

The main causes of eutrophication are associated with agricultural activities, urban development and industrial wastes (Carpenter *et al.* 1998; Conley *et al.* 2009; Schindler 2012; Smith 2003). All of these activities have economic value and can be difficult to regulate. Non-point source pollution can travel overland or underground. As well the pollution may track seasonal changes related to rainfall or summer construction (Carpenter *et al.* 1999; Smith 2003). Management of sources related to eutrophication of aquatic systems can be difficult; it may be difficult to assign responsibility for clean-up costs, and managers must weigh the economic value of the water body against the economic value of the polluting activity (Carpenter *et al.* 1999).

In the case of a park, like Riding Mountain National Park, tourist activities such as boating, fishing and camping must be closely monitored for their effects on the lake water quality. The seasonal increase in population in RMNP during the summer months can put immense pressure on aquatic systems. For example, Clear Lake is the most used lake in RMNP (pers. comm. Bob Reside). It also has the largest urban development surrounding it, Wasagaming. The increase in population of the town and surrounding area, from mainly park staff to up to 300,000 visitors puts a lot of pressure on the parks ecosystem (Vandershuit and Penwarden 2006). In recent years, strategies to reduce potential pollution to the aquatic resources have included a massive upgrade to the wastewater treatment facility in 2006 (Vandershuit and Penwarden 2006), restricted development around Clear Lake (as well as other lakes and areas in the park), banned conventional two stroke outboard motors and the restriction of motorized boats to only a

few lakes within the park. These restrictions are examples of when managers have chosen to preserve the ecosystem at the economic expense of the polluting activity.

### 2.6. Effects of Eutrophication on Aquatic Macrophytes and Periphyton

Despite slower growth, established aquatic macrophyte populations are able to rapidly sequester nutrients, reducing phytoplankton growth (Lacoul and Freedman 2006; Lauridsen *et al.* 2003). As nutrients increase in an aquatic system it is initially aquatic macrophytes, not phytoplankton, that respond with increased growth, diversity and colonization area (Balls *et al.* 1989; Lachavanne *et al.* 1992; Lacoul and Freedman 2006; Lauridsen *et al.* 2003;). Free-floating, non-rooted submerged aquatic plants are able to sequester large amounts of nutrients, specifically nitrogen and phosphorus, well beyond their immediate growth requirements; thus, they serve as effective nutrient sinks for aquatic ecosystems (Balls *et al.* 1989; Lacoul and Freedman 2006). Submerged rooted macrophytes may also act as nutrient sinks, sequestering some of the nitrogen and phosphorus that accumulates in lake sediments during times of nutrient loading (Lacoul and Freedman 2006).

Aquatic macrophytes are so efficient at incorporating excess nutrients into their tissues that lakes experiencing eutrophication can maintain a clear water state provided that a well-established macrophyte community persists (Lauridsen *et al.* 2003). For example, Balls *et al.* (1989) experimentally fertilized two shallow lakes, one lake with an established submerged aquatic plant community and another with all aquatic plants removed. The lake without a plant community experienced significant algal blooms.

The lake with an established plant community did not have a recordable increase in internal phosphorus loading, nor was there any increase in phytoplankton growth.

Furthermore, Le Bagousse-Pinguet *et al.* (2011) showed experimentally that plant communities are extremely important as regulators of phytoplankton growth. In laboratory experiments a target stand of submerged aquatic plants that had a large neighbouring plant community experienced less phytoplankton growth than an isolated macrophyte stand. When neighbouring plant communities were present, nitrogen became limiting, preventing phytoplankton growth, although total phosphorus remained high. This experiment highlighted the mitigating effects of large macrophyte populations on the eutrophication process. Small, isolated macrophyte communities do not have the same competitive advantage, and are unable to effectively mitigate eutrophication. Although the authors admit that laboratory experiments are imperfect analogues for the complexities of natural ecosystems, their results are in line with many observations of shallow lake ecosystems (Balls *et al.* 1989; Lauridsen *et al.* 2003).

Although submerged macrophytes do show increased growth and can maintain clear water states in eutrophic conditions, at some threshold the nutrient concentrations exceed those that aquatic macrophytes can sequester; thus phytoplankton dominance results (Balls *et al.* 1989; Lachavanne *et al.* 1992; Lacoul and Freedman 2006). Sayer *et al.* (2010) examined the eventual shift from aquatic macrophyte dominance to phytoplankton dominance with persistent eutrophication. He found that lakes with the greatest coverage of submerged macrophytes and highest submerged macrophyte species diversity were most resistant to eutrophication. He hypothesized that plants are able to remove most nutrients from the water, preventing their use by phytoplankton. Sayer *et*

*al.* (2010) surmised that the diverse life cycles of submerged aquatic plants ensured constant plant cover and therefore constant nutrient utilization, limiting phytoplankton growth. Lakes lacking aquatic macrophyte diversity and lakes lacking aquatic macrophytes were not able to mitigate the effects of eutrophication and eventually large algal blooms occurred during summer (Sayer *et al.* 2010).

Nevertheless, in almost any lake ecosystem, excessive nutrients will inevitably precipitate a shift from aquatic macrophyte to phytoplankton dominance (Balls *et al.* 1989; Jeppesen *et al.* 1997; 2000; Lachavanne *et al.* 1992; Lacoul and Freedman 2006). Even in aquatic ecosystems well buffered against eutrophication via large, diverse macrophyte populations the small, incremental increases in phytoplankton will, over time, shade out submerged macrophyte communities (Sayer *et al.* 2010).

Eutrophication affects periphyton in much the same way as submerged aquatic macrophytes. Periphyton are often nutrient limited in oligotrophic lakes and transition to being light limited in eutrophic-hypereutrophic lakes (Hannson 1992; Rodusky *et al.* 2001). Periphyton have been shown to grow optimally in lakes with high values of total phosphorus coupled with high light attenuation (Hannson 1992). Periphyton has the potential for mitigating nutrient releases from lake sediment, helping to prevent phosphorus loading and the release of excess phosphorus from lake beds (Genkai-Kato *et al.* 2012; Hannson 1990). Like aquatic macrophytes, at some threshold, phytoplankton concentration will create light limiting conditions that favour phytoplankton dominance (Genkai-Kato *et al.* 2012; Hannson 1990).

## 2.7. Previous Limnological Research in Riding Mountain National Park

Before the 1930 establishment of Riding Mountain as a national park, the area was (and still is) a hot spot for anglers. All research prior to 1930 had been focused on identifying native fish, and maintaining and establishing stocks of introduced fish. Once the area was established as a national park, fish stocking intensified. Research continued, focusing almost exclusively on fish stocking success until the 1970s (Kooyman and Hutchinson 1979). In 1979, 1980 and 1981 Kooyman and Hutchinson produced six data reports outlining chemistry data as well as extensive aquatic macrophyte, zooplankton, phytoplankton and fish inventories.

A shift in research priorities occurred during the late 1970s, with less emphasis on fish stocking and stakeholders. Parks Canada instead focused on preserving ecological integrity. An engineering company, James F. MacLaren Ltd. started monitoring the Wasagaming townsite sewage lagoons in 1976. The company determined that sewage discharge from the Wasagaming townsite into South Lake via Ominnik Marsh was having deleterious effects on the water quality of South Lake, and therefore Clear Lake (MacLaren 1977).

The sewage treatment for the town of Wasagaming, MB, prior to the 2009 upgrade consisted of three lagoons and a constructed drainage channel that periodically released effluent from the lagoon system into the Ominnik Marsh and Octopus Creek area. This marsh eventually drains into South Lake with the South Lake basin overflowing seasonally into the Clear Lake basin (Belke and McGinn, n.d.). While the effluent that is released into Ominnik Marsh must meet provincial fecal coliform and BOD guidelines there was considerable concern that Ominnik marsh, Octopus creek,

South Lake and Clear Lake were at risk for contamination from this wastewater system (Belke and McGinn n.d.). Consultants, McLaren and Pratt (as cited in Belke and McGinn n.d.), and Park managers, Rousseau (as cited in Belke and McGinn n.d.), predicted there was a faulty pipe in the wastewater system. This prediction was confirmed by Belke and McGinn (n.d.) when their monitoring showed high orthophosphate and ammonium contamination along a force main that transported the wastewater from the Wasagaming townsite to sewage lagoon cell 1. They recommended the leaky forcemain be replaced and the effluent discharge be relocated to make better use of Ominnik Marsh (Belke and McGinn n.d.).

In 1993 stakeholders with Parks Canada developed long-term monitoring goals and aquatic resource management practices for Riding Mountain National Park (Terrestrial and Aquatic Environmental Managers Manitoba 1993). The monitoring goals included: determining a water budget for Clear Lake, assessing the range of chemicals in the water and their origins, coliform levels and their sources, monitoring of Clear Lake boat traffic, and a population census for the Clear Lake basin (Hawryliuk 2000).

Awareness of increasing nutrient inputs into South Lake, Ominnik Marsh and Octopus Creek led to a major upgrade to the Wasagaming sewage treatment facility, completed in 2009 (Belke and McGinn n.d.; Vandershuit and Penwarden 2006) . Since completion of the upgrade, including new force main pipes and upgrades to the sewage lagoons, ongoing water quality monitoring has been maintained for the sewage lagoons, and at Ominnik Marsh, Octopus Creek, South Lake and Clear Lake. Preliminary reports indicated that nitrogen and phosphorus concentrations were decreasing in the lagoons,

creeks, and South Lake due to the upgrade at the Wasagaming wastewater treatment lagoons (pers. comm. RMNP staff 2010).

Phytoplankton productivity in Clear Lake was examined by Hawryliuk (2000). She outlined the controls on primary productivity and its effects on Clear Lake. Hilderman *et al.* (2005) compiled a comprehensive report on Clear Lake's status as well as data on the terrestrial ecosystem and other monitoring goals, as laid out by the Clear Lake Water Basin project. Hilderman *et al.* (2005) found that Clear Lake phosphorus levels were quite high, despite the lake's clear water state.

Parks Canada and partners identified the high phosphorus levels in Clear Lake to be a management priority and invested in a comprehensive study into the ecological integrity of Clear Lake, its watershed and selected water bodies in the Little Saskatchewan River watershed. Gray (2009) reconstructed the hypolimnetic oxygen levels in Clear Lake using midge-based paleolimnology. The hypolimnetic oxygen levels in Clear Lake showed no discernible shifts since pre-settlement. This indicated that the phosphorus that was entering Clear Lake was not being converted into algal biomass, reducing the potential decomposition load on the lake and therefore the oxygen demand in the hypolimnion.

Subsequently, Whitehouse (2010) developed a phosphorus budget for Clear Lake. He determined that the excess phosphorus was rendered biologically unavailable via co-precipitation with calcite; thus, the lake's productivity remained low despite relatively high P loading. Most recently White (2012) completed an extensive survey of RMNP diatoms, including paleolimnological assessments of water quality changes in four high-use lakes: Clear, Katherine, Moon and South Lakes. Variations in diatom community

assemblages were best explained by changes in total phosphorus values in the four lakes. In addition, she found that marked changes in assemblages, and therefore total phosphorus, have occurred in Clear Lake since settlement in the area.

The research reported in this thesis also constitutes a part of the Parks Canada Action on the Ground (AOG) initiative. My studies explore lake management options, specifically the potential for aquatic macrophytes and periphyton to be used to assess and track water quality changes with the goal of preserving the ecological integrity of the lakes in RMNP.

## 2.8. Collection Methods

### *2.8.1. Environmental Variables*

The most important chemical and physical variables affecting growth and distribution of aquatic macrophytes and periphyton are water clarity, nutrient concentration (nitrogen and phosphorus), specific conductance, alkalinity and pH (Wetzel 2001). While many other parameters are measured in water quality studies these six measurements are amongst the most important and are almost always included (Hawryliuk 1998; Heegaard *et al.* 2001; McElarney and Rippey 2009; Søndergaard *et al.* 2010). In this study I chose to include five of the aforementioned variables: pH, total phosphorus, total nitrogen, specific conductance and secchi depth. As well, chlorophyll a was added as a proxy for algal biomass in the water column and dissolved oxygen was added as another measure of lake water quality.

*2.8.2. Aquatic Macrophytes*

Commonly aquatic plants, growing in climates similar to Canada, are sampled in late summer and early fall to capture the plants at their peak development (Capers *et al.* 2010; Heegaard *et al.* 2001; McElarney and Rippey 2009; Schallenberg and Waite 2004; Søndergaard *et al.* 2010). The two most common methods for assessing aquatic plants are via boat sampling using a two-headed rake or Ekman grab, or in-water sampling using snorkeling or SCUBA. Both techniques usually employ sampling at multiple points along transects that extend across all depths inhabited by macrophytes. The transect orientation to shore, the number of transects, and the number of sampling points along those transects vary considerably among studies, according to need and capacity. Transect placement depends on where plants grow in the lake (Capers *et al.* 2010; Heegaard *et al.* 2001; Paerl *et al.* 2007; Schallenberg and Waite 2004; Søndergaard *et al.* 2010).

Although in-water sampling techniques more accurately quantify species richness and abundance of aquatic plants, time, funding and safety constraints may dictate other methods (Capers 2000); thus, boat sampling is still accepted (Capers *et al.* 2010; Heegaard *et al.* 2001; Paerl *et al.* 2007; Schallenberg and Waite 2004; Søndergaard *et al.* 2010). Furthermore, in water techniques are very difficult to conduct at turbid sites (Capers 2000).

In many water quality studies only submerged aquatic plant species are compared to the environmental variables (European Commission 2000; Jeppesen *et al.* 2000; Kosten *et al.* 2009; Penning *et al.* 2008; Søndergaard *et al.* 2010) because aquatic plants that can grow above or on top of the water column or have a well developed rooting

system are subjected to different sunlight exposure, nutrient levels and climate than fully submerged macrophyte species. By only using submerged plants the results of the relationships between the plants and water quality variables will more accurately reflect the actual relationship between lake water and aquatic macrophytes. As submerged aquatic plants have adapted to absorb nutrients in the water column through their leaves and can only use the sunlight that penetrates the water column (Sculthorpe 1967; Wetzel 2001).

*2.8.3. Periphyton*

Periphyton have the ability to colonize a wide range of surfaces, including rocks, sand, silt and aquatic plants (Reid *et al.* 1995). Periphyton samples collected from different substrates can introduce a lot of variability into the resulting data set, making statistical inferences more difficult (Reid *et al.* 1995); thus, many researchers choose to use artificial substrates to 'grow' periphyton.

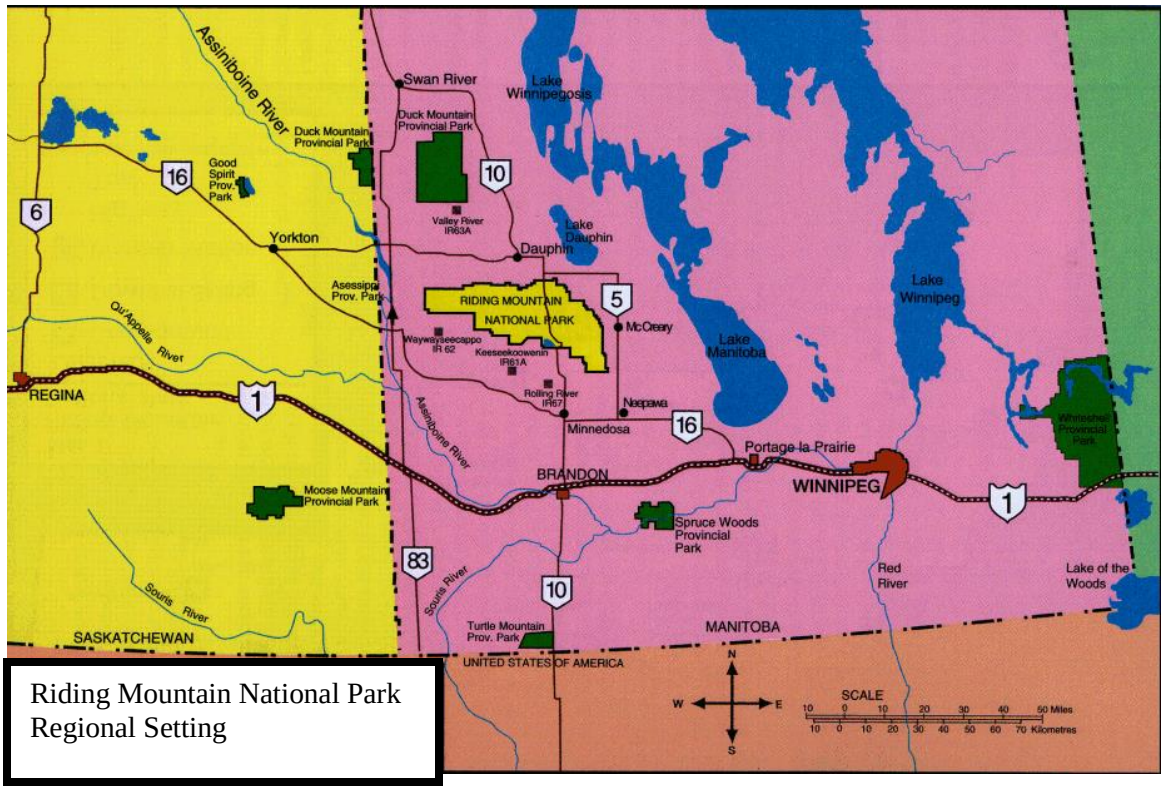
Artificial substrates have advantages. They can be placed by the researcher in specific locations, and growth time can remain the same in different sampling environments (Reid *et. al.* 1995). The artificial substrata should be left in the environment for a minimum of one month to allow for adequate community development (Aloi 1990). Some researchers have cautioned against drawing conclusions about the natural periphyton community when research is conducted using an artificial substrate (Cattaneo & Amireault 1992). Other studies, however, have shown that artificial substrates are useful when comparing communities to environmental variables, and for between lake comparisons. Since periphyton may access nutrients leached from the

substrate, the use of artificial substrates reduces the variability in nutrient availability to periphyton (Aloi 1990).

## Chapter 3: Study Area

### 3.1. Riding Mountain National Park

Riding Mountain National Park (RMNP) is located in southwestern Manitoba, Canada and covers an area of 3000 km$^2$ (Parks Canada 2012). The park is located 97 km north of Brandon, Manitoba and 247 km west of Winnipeg, Manitoba (Figure 3.1). It is easily accessible via Highway 10 from both the Trans-Canada Highway and the Yellowhead Highway. Highway 10 cuts through the middle of the Park making it accessible by car throughout the entire year.




Riding Mountain National Park
Regional Setting

*Figure 3.1 Regional setting of Riding Mountain National Park, Manitoba (Riding Mountain National Park Round Table 1996). Reproduced with permission. See Fig. A1.1.*

*3.1.1. Physical Characteristics*

RMNP is part of the Manitoba escarpment and rises 475 m above the lowlands to the east. The Manitoba escarpment and the topography of the park are a product of diverse ancient processes, including uplift, differential erosion, several Quaternary glacial advances and retreats, the formation and demise of glacial Lake Aggassiz and other glacial meltwater events (Bazillion and Braun 1992; Kooyman and Hutchinson 1979; Riding Mountain National Park Round Table 1996). Bedrock of the park consists of Cretaceous aged shales (90-100 Ma years ago) overlain by unconsolidated surficial deposits derived from the last four glacial cycles (Kooyman and Hutchinson, 1979). Glacial drift and large erratic boulders cover much of the park. Prominent glacial topographic features include ancient shorelines from glacial Lake Aggassiz, hummocky knobs, meltwater channels, rolling moraine ridges, outwash plains and many kettle lakes (Bazillion and Braun 1992; Kooyman and Hutchinson 1979; Hawryliuk 2000).

The dominant terrestrial vegetation is mixed forest including aspen, spruce, oak and mixed hardwood. Prairie grasslands occur mainly in the central and western regions of the park (Bazillion and Braun 1992; Kooyman and Hutchinson 1979; Hawryliuk 2000). A wide variety of native animals can be found including wolves, black bears, elk, moose, deer, bison, lynx, and cougar, as well as many small mammals and birds (Bazillion and Braun 1992; Kooyman and Hutchinson 1979).

### *3.1.2. Climate*

Riding Mountain National Park has a continental climate characterized by long, cold winters and short, hot summers. The park averages just 72 growing days per year (Parks Canada 2012). There are fewer ice-free days, more precipitation and lower wind speeds on the escarpment than the surrounding Manitoba prairie due to the rolling park topography, forest cover and higher elevation (Parks Canada 2012).

### *3.1.3. Aquatic Resources*

The majority of lakes in and around RMNP were formed as a product of Quaternary glaciation; they are referred to as kettle lakes. Kettle lakes are produced by large chunks of ice that became embedded in the surrounding rock and debris, and later melted (Kooyman and Hutchinson 1979). This produced the hundreds of small, shallow lakes and ponds that dot the entire park landscape (Kooyman and Hutchinson 1979). Blocked meltwater channels probably contributed to the formation of the larger RMNP lakes including Long Lake and Clear Lake (Kooyman and Hutchinson 1979). Many streams crisscross the landscape of Riding Mountain National Park, emptying into lakes or connecting with wetlands and other various forms of standing water (Kooyman and Hutchinson 1979).

Riding Mountain National Park has two main watersheds: Dauphin Lake watershed draining the northwestern region into Dauphin Lake, and the Little Saskatchewan River watershed draining the southeastern portion of the park into the Assiniboine River (Kooyman and Hutchinson 1979; Water Stewardship Division 2012). The lakes studied for this thesis all lie within the Little Saskatchewan River watershed

(Figure 3.2). This watershed contains the most recreationally important lakes of RMNP: Clear Lake, Moon Lake, Lake Audy and Katherine Lake.

### *3.1.4. History*

The first human inhabitants began to occupy the area around 10000 years ago. Archaeologists indicate that the Cree people had arrived in the area by about 1760 years ago (Bazillion and Braun 1992). Migration of other Native groups, Ojibway and Saultaeux, to the area occurred in the early 1800s (Bazillion and Braun 1992). In 1881 the Canadian Pacific Railway reached Brandon, Manitoba, facilitating an influx of European and American immigrants and settlers from Eastern Canada.

Riding Mountain was a popular hunting and fishing area, and timber harvesting occurred until the early 1900s (Bazillion and Braun 1992; Kooyman and Hutchinson 1979). Excessive timber harvesting, fishing and hunting prompted the Government of Canada, in 1906, to institute harvest regulations and designate 216 $mi^2$ (553 $km^2$) of forest as a game reserve (Bazillion and Braun 1992). Riding Mountain became an official National Park of Canada on 25 January 1930.

### *3.1.5. National Park Policy and Acts*

All Canadian National Parks must follow the principles in the Canadian National Parks Act (Parks Canada 2015). In addition, RMNP management must also adhere to the Riding Mountain National Park Management Plan (Parks Canada 2007).

In 1979 the National Parks Policy Act amended its guiding principle to focus on the ecological integrity of Canadian National Parks (Parks Canada 2012). Ecological

integrity is defined by the National Parks Policy Act as, ". . . *a condition that is determined to be characteristic of its natural region and likely to persist, including abiotic components and the composition and abundance of native species and biological communities, rates of change and supporting processes* (Parks Canada 2015)*."*

Ecological integrity is the cornerstone for the National Parks "Action on the Ground" projects that occur across the country. In 2000, the Panel on the Ecological Integrity of Canadian National Parks released a report urging Parks Canada, the Government of Canada and all Canadians to preserve our threatened National Parks. The Government of Canada responded in 2001 by committing to the investment in National Park ecological integrity through "Action on the Ground" projects (Parks Canada 2012). This study is a component of the "Keeping the Clear in Clear Lake" Action on the Ground project.

### 3.2. Study Sites

Originally fifty-one lakes were chosen to represent the diversity of lakes within Riding Mountain National Park and the surrounding area (Figure 3.2). Of those fifty-one lakes, eighteen resided in RMNP and thirty-three lay outside the Park boundary. Since Clear Lake, Moon Lake, Katherine Lake and Lake Audy are some of the most heavily developed and utilized Park lakes, the study lakes were selected to be within the same watershed. Since part of the mandate of Parks Canada is to preserve ecological integrity, it is important to look at lakes in the same watershed as the RMNP lakes of highest interest to the park because changes in one part of a watershed can have an effect on all other parts of the watershed.

A Geographic Information System (GIS) was used to semi-randomly select fifty-one lakes within the Little Saskatchewan Watershed. Due to time, access and legal constraints only lakes within 500 m of an accessible road or trail, and north of Highway 45 were included in the random sample (Figure 3.2). Because of YSI calibration error, lack of access or disappearance in subsequent years, only forty-five lakes were used in the final analysis; their exact locations and sample dates are given in Table 3.1.

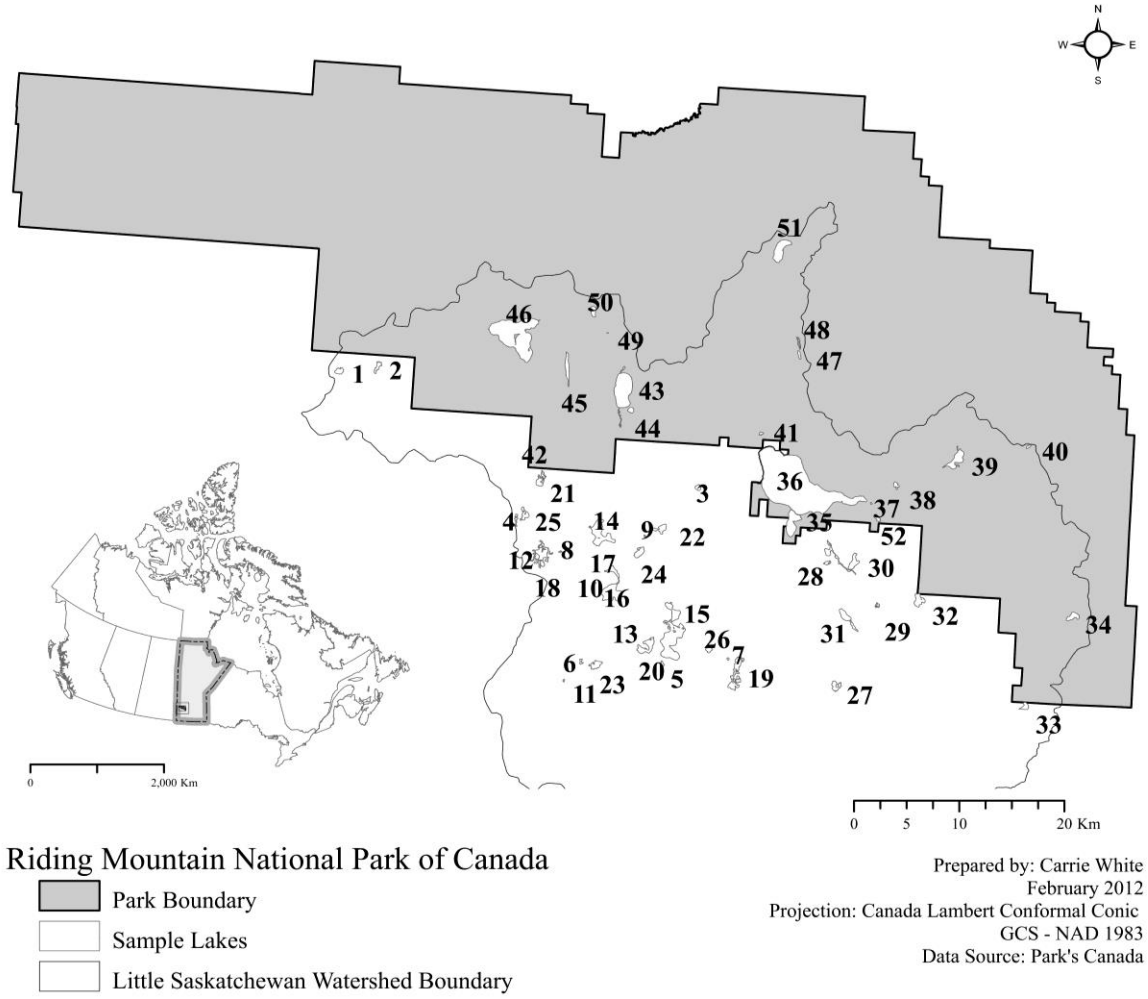



*Figure 3.2 The location of the forty-five lakes chosen to be sampled within and outside Riding Mountain National Park. Lakes 12, 17, 32, 37, 41, 42 and 47 were not used in the final analysis. Lakes were within the Little Saskatchewan River Watershed and did not extend farther south than Highway 45. Map prepared by Carrie White (2012) and used with permission.*

*Table 3.1 Latitude and longitude of all sample lakes, identified by name and/or number, that were sampled over the summers of 2010 and 2011 in and around Riding Mountain National Park, Manitoba.*

| Lake | Lake name | Latitude | Longitude | Date Sampled | Lake | Lake name | Latitude | Longitude | Date Sampled |
|---|---|---|---|---|---|---|---|---|---|
| 1 | n/a | 50°45'18 | 100°37'52 | 8/22/2011 | | | | | |
| 2 | Herchak | 50°45'43 | 100°34'48 | 9/6/2010 | 34 | Muskrat | 50°34'59 | 99°37'27 | 8/25/2010 |
| 3 | n/a | 50°40'31 | 100°08'20 | 8/9/2011 | 35 | South | 50°38'37 | 100°00'44 | 8/29/2011 |
| 4 | n/a | 50°35'52 | 100°19'57 | 8/30/2011 | 36 | Clear | 50°40'20 | 100°00'06 | 8/26/2011 |
| 5 | n/a | 50°31'14 | 100°17'29 | 8/13/2010 | 37 | Pudge | 50°40'13 | 99°54'15 | 8/16/2010 |
| 6 | n/a | 50°31'02 | 100°16'47 | 8/2/2011 | 38 | Kinosao | 50°41'14 | 99°52'20 | 8/25/2010 |
| 7 | n/a | 50°31'43 | 100°05'01 | 8/13/2010 | 39 | Whirlpool | 50°42'49 | 99°47'30 | 9/1/2010 |
| 8 | n/a | 50°35'43 | 100°19'21 | 8/31/2010 | 40 | East Deep | 50°43'37 | 99°41'51 | 8/16/2010 |
| 9 | n/a | 50°37'37 | 100°12'16 | 8/19/2010 | 43 | Audy | 50°45'09 | 100°14'58 | 8/23/2011 |
| 10 | n/a | 50°35'15 | 100°16'07 | 8/17/2011 | 44 | Audy runoff | 50°44'18 | 100°14'09 | 8/23/2011 |
| 11 | n/a | 50°30'04 | 100°18'07 | 8/2/2011 | 45 | Long | 50°46'07 | 100°19'28 | 8/9/2011 |
| 12 | n/a | 50°36'32 | 100°21'44 | 8/18/2010 | 46 | Whitewater | 50°47'23 | 100°23'11 | 8/17/2010 |
| 13 | n/a | 50°33'01 | 100°13'21 | 8/19/2010 | 48 | 4th Bead | 50°47'39 | 100°00'46 | 8/18/2011 |
| 14 | n/a | 50°34'19 | 100°11'25 | 8/19/2011 | 49 | n/a | 50°48'13 | 100°16'31 | 8/17/2010 |
| 15 | Sandy | 50°32'58 | 100°09'43 | unknown | 50 | Cripple | 50°49'14 | 100°17'44 | 8/9/2011 |
| 16 | Thomas | 50°34'47 | 100°14'50 | 8/23/2011 | 51 | Moon | 50°52'57 | 100°02'59 | 8/29/2011 |
| 17 | Stuart | 50°37'40 | 100°16'05 | unknown | 52 | Katherine | 50°39'37 | 99°53'41 | 8/18/2011 |
| 18 | Wargatie | 50°36'30 | 100°20'36 | 8/25/2011 | | | | | |
| 19 | Jackfish | 50°31'01 | 100°04'23 | 8/2/2011 | | | | | |
| 20 | Beaufort | 50°32'17 | 100°11'20 | 8/26/2011 | | | | | |
| 21 | Czoryj | 50°40'15 | 100°21'07 | 8/19/2011 | | | | | |
| 22 | Battle | 50°38'16 | 100°10'56 | 8/26/2010 | | | | | |
| 23 | Crawford | 50°30'59 | 100°15'37 | 8/2/2011 | | | | | |
| 24 | Corstorphine | 50°36'55 | 100°12'43 | 8/26/2010 | | | | | |
| 25 | n/a | 50°39'20 | 100°22'10 | 8/19/2011 | | | | | |
| 26 | n/a | 50°32'08 | 100°06'36 | 8/5/2011 | | | | | |
| 27 | Leda | 50°30'03 | 99°55'25 | unknown | | | | | |
| 28 | n/a | 50°36'59 | 99°57'33 | 8/21/2010 | | | | | |
| 29 | n/a | 50°31'49 | 99°54'28 | 8/4/2011 | | | | | |
| 30 | Octopus | 50°37'04 | 99°55'21 | 8/9/2010 | | | | | |
| 31 | Ditch | 50°34'16 | 99°55'49 | 8/4/2011 | | | | | |
| 33 | Kerr | 50°35'54 | 99°49'59 | 8/11/2011 | | | | | |

# Chapter 4: Methods

## 4.1. Environmental Variables

All water samples and measurements were taken approximately 10-20 cm below the surface as close to the estimated middle of each lake as possible. The sampling of the environmental variables occurred over the month of August 2010 and 2011 and took place on the same day as the macrophyte survey for that lake. Water samples for total nitrogen were taken in 500 mL bottles and analyzed using Ion Chromatography in accordance to SM 4110 B in Standard Methods for the Examination of Water and Wastewater (Rice *et al.* 2012). A separate 120 mL bottle with 1 mL of 1:1 sulphuric acid and water was used to sample for total phosphorus. Total phosphorus was analyzed using inductively coupled plasma-atomic emission spectrometry as outlined in Method 200.7 (Martin *et al.* 1994). A 1 L sample for chlorophyll a was collected and analyzed using UV-VIS Spectrophotometer in accordance with SM 10200 H in Standard Methods for the Examination of Water and Wastewater (Rice *et al.* 2012). All samples were analyzed by Maxxam laboratories in Winnipeg, Manitoba. Water clarity, specific conductance, dissolved oxygen (DO) and pH were measured directly in each lake with a YSI 556MPS multi-probe system meter. Lake depth was recorded using a weighted measuring rope. The depth measurements were taken as close to the middle of each lake as possible, capturing the most likely deepest part of each lake. Secchi disk transparency was also recorded. Secchi disk depth procedure followed that outlined by the Environmental Protection Agency (U. S. EPA 2012)

### 4.2. Macrophyte Abundance and Distribution

Macrophyte sampling took place from August to early September 2010 and 2011, assessing the plants near the end of their growing season. Of the fifty-one lakes selected for sampling thirty-three were sampled in 2010 and twenty-seven were sampled in 2011, with eleven lakes being sampled both summers. In each lake two 30 metre long transects were laid out with rope, perpendicular to shore.

Transect sites were chosen based on visual determination of lake areas that contained macrophytes. Every 5 m submerged macrophytes were collected over a 1 m length directly beneath the transect rope. Only three rake drags were allowed per sampling point to ensure similar collection effort at each sampling point in each lake. All collections were done from a small boat. Depth was recorded at each sampling point.

Each macrophyte species was identified in the field, and cover estimates were made using a six-point scale (Table 3.2). If plant samples could not be identified in the field they were collected and pressed within two days. These samples were then identified as soon as possible using the field guide written by Borman *et al.* (1997) or via personal communication with Ian Walker.

*Table 3.2 Six-point scale used to estimate aquatic macrophyte cover.*

| Abundance | Percent Cover |
|---|---|
| 1 | <10% |
| 2 | 10-30% |
| 3 | 30-50% |
| 4 | 50-70% |
| 5 | 70-90% |
| 6 | >90% |

## 4.3. Periphyton Collection and Analysis

The periphyton methodology followed closely that used in an earlier study of Clear Lake algae (Hawryliuk 1998). At the beginning of August 2010 acrylic rods were placed in each of the fifty-one lakes and left for approximately one month to ensure periphyton growth. Each rod was 3 ft (91.4 cm) in length and ½ inch (1.27 cm) in diameter. Rods were pushed into the sediment as far as was needed for stability, therefore a slight difference in rod exposed surface area occurred among lakes. In early September, thirty of the rods were recovered and the algae were scraped from each rod into a plastic bag. The algae were preserved at the end of every day in Lugol's solution. They were stored in a cold room until March 2011 before being sent to the University of Alberta for enumeration. Three sub-samples of each periphyton sample were taken for enumeration and taxonomic analysis. Periphyton cell count and taxonomic composition were quantified at 250-100x using Ütermohl Chambers (Vinebrooke and Graham 1997). Only cells with viable protoplasts were counted (pers. comm. Mark Graham). All subsequent statistical analysis were based on periphyton raw counts.

## 4.4. Statistics

### *4.4.1 Trophic Status Index Calculation*

The TSI was originally proposed by Carlson (1977) and modified by the US EPA (2000) with the Florida Department of Environmental Protection. The modified TSI includes equations that account for the limiting nutrient in the environment, total nitrogen, total phosphorus and chlorophyll a concentrations. The limiting nutrient was calculated using a ratio.

If TN/TP was >30 than the lake was considered phosphorus limited. The NUTRTSI was calculated as follows:

$$TP2TSI = 10 \times [2.36 \times LN(TP \times 1{,}000) - 2.38] \quad [1]$$

If TN/TP was < 10 than the lake was considered nitrogen limited. The NUTRTSI was calculated as follows:

$$TN2TSI = 10 \times [5.96 + 2.15 \times LN(TN + 0.0001)] \quad [2]$$

If TN/TP was > 10 but < 30 than the lake was considered co-limited. The NUTRTSI was calculated as follows:

$$TNTSI = 56 + [19.8 \times LN(TN)] \quad [3]$$

$$TPTSI = [18.6 \times LN(TP \times 1000)] - 18.4 \quad [4]$$

$$COTSI = (TPTSI + TNTSI)/2 \quad [5]$$

The TSI for lakes was also calculated using chlorophyll a concentrations with the following equation:

$$CHLATSI = 16.8 + 14.4 \times LN (CHLA) \quad [6]$$

Finally, the NUTRTSI and the chlorophyll a TSI were averaged to give the final TSI used in subsequent statistics.

$$TSI = (CHLATSI + NUTRTSI) / 2 \quad [7]$$

All equations can be found in "TSD for U.S. EPA's Proposed Numeric Nutrient Criteria for FL Inland Surface Fresh Waters" (US EPA 2000). The TSI interpretation scheme for northern temperate lakes is found in Carlson and Simpson (1996) (Table 4.1). For the purposes of this study oligotrophic lakes will be classified as those with TSI 0-40, mesotrophic TSI 40-60, eutrophic TSI 60-80 and hypereutrophic TSI >80. However, it

should be noted that the TSI boundaries are not discreet and are used as a general guide for interpretation. Macrophyte and periphyton species that are associated with specific lake trophic states have potential as trophic indicators. Monitoring changes in lake water quality is an important management step in maintaining the ecological integrity of lakes.

*Table 4.1 Trophic status index interpretation for northern temperate lakes (Carlson & Simpson 1996).*

| | |
|---|---|
| **TSI < 30** | Classical Oligotrophy: Clear water, oxygen throughout the year in the hypolimnion, salmonid fisheries in deep lakes. |
| **TSI 30–40** | Deeper lakes still exhibit classical oligotrophy, but some shallower lakes will become anoxic in the hypolimnion during the summer. |
| **TSI 40–50** | Water moderately clear but increasing probability of anoxia in hypolimnion during summer. |
| **TSI 50–60** | Lower boundary of classical eutrophy: Potential for decreased transparency, anoxic hypolimnia during the summer and macrophyte growth, warm-water fisheries only. |
| **TSI 60–70** | Dominance of blue-green algae, algal scums probable, extensive macrophyte problems. |
| **TSI 70–80** | Heavy algal blooms possible throughout the summer, dense macrophyte beds but extent limited by light penetration. Often would be classified as hypereutrophic. |
| **TSI > 80** | Algal scums, summer fish kills, few macrophytes, dominance of rough fish. |

### *4.4.2 Ordination*

Ordination methods reduce complex data into a few abstract summary variables. Ordination techniques do not allow for specific, quantitative interpretations of abundance-environmental relationships (Warton 2008). However, broad ecological patterns can be discovered when many types of response and predictor variables are analyzed together, which is only possible through the use of data reduction techniques (Warton 2008).

There are two types of ordination, direct and indirect gradient analysis (Legendre and Gallagher 2001).

*4.4.3. Indirect Gradient Analysis*

Indirect ordination attempts to find patterns in species community data by evaluating the similarity (or dissimilarity) of species composition and/or abundance among sites. This similarity matrix is then plotted with sites that are closer together in ordination space being more similar in species composition and/or abundance than sites that are distant in the ordination space (Kindt and Coe 2005). There are many different types of indirect gradient analysis but the one that will be utilized here is non-metric multidimensional scaling (NMDS). NMDS repeats calculations over a set number of axes (2-4) until a desired stress level is achieved (Holland 2008). The lower the stress level, the more closely the data in ordination space resembles the real data. Any stress level < 0.2 is considered biologically meaningful (Kindt and Coe 2005). Other types of indirect gradient analysis, like principal component analysis and correspondence analysis make assumptions about the data like linear or unimodal species distributions. In contrast, NMDS makes no assumptions of the data; therefore, this statistical technique can be applied to a wide variety of data (Holland 2008). NMDS was used to detect patterns in the macrophyte and periphyton data. Sites were plotted in ordination space to represent real life ecological distance. Lakes grouped together in ordination space were classified along a continuum from oligotrophic to hypereutrophic using a modified Trophic Status Index (TSI).

*4.4.4. Direct Gradient Ordination*

Direct ordination techniques build upon the patterns outlined by indirect gradient analysis and go one step further; they attempt to identify the cause of the observed pattern. These methods relate species presence/absence or abundance at a particular site to the environmental variables. The ordination maximally separates the distribution of species along the environmental gradients. Therefore, just like indirect gradient analysis the closer two sites are the more similar they will be with respect to species composition and environmental values. The eigenvalues produced by the ordination are the measure of this separation (Kindt and Coe 2005; ter Braak 1987a).

The benefit of using direct gradient analysis is that it reduces a large, multidimensional data set into a two dimensional plot representing the relationships between multiple, interacting environmental variables and the occurrence of species at particular sites (Kindt & Coe 2005; ter Braak 1987a; Warton 2008).

Redundancy analysis (RDA) will be used to assess the aquatic plants data and canonical correspondence analysis (CCA) will be used to assess the periphyton data. RDA assumes a linear relationship (short environmental gradients) between response and explanatory variables. CCA, on the other hand, assumes a unimodal relationship between the response and explanatory variables. The gradients of the first ordination axis for the aquatic macrophyte and periphyton data will be assessed to classify the data as linear or unimodal. The calculation is given in ter Braak (1987 in Jongman *et al.* 1995) and was calculated for data within this study with assistance from Ian Walker (person. comm.)

The wider the ranges of environmental values included in the study the longer the

environmental gradients. If sites included in a study cover a wide environmental range many species will be replaced by others along this gradient, producing null abundances at many sites (Legendre and Gallagher 2001). This problem can be solved by applying the Hellinger transformation to species data, which allows ecologists to use community composition data containing many zeroes (Legendre and Gallagher 2001).

RDA, along with CCA, can experience problems when excessive noise occurs in the data set (Hirst and Jackson 2007; McCune 1997). Noisy environmental data, arising during data collection or when irrelevant environmental variables are included, can distort the ordination output (McCune 1997). Species abundance often introduces noise into an ordination because the data can be difficult to obtain accurately and precisely. By using presence/absence data or eliminating redundant environmental variables, noise is greatly reduced (Hirst and Jackson 2007).

RDA and CCA were used to assess whether aquatic macrophytes or periphyton taxa were potentially useful as water quality predictors or to determine if the data could define ecological thresholds between lake trophic states. The lengths of the ordination axis for the periphyton and aquatic macrophyte data were calculated (ter Braak 1987 in Jongman *et al.* 1995) in order to determine the most appropriate direct gradient ordination technique. RDA was used with the aquatic macrophyte data, as the length of the first ordination axis, as calculated by ter Braak (1987 in Jongman *et al.* 1995) and Ian Walker (person. comm.), was less than 1.5 standard deviations. When the length of the ordination axes is less than 2.0 standard deviations it can be assumed that the response curves are linear and redundancy analysis is appropriate (ter Braak 1987 in Jongman *et al.* 1995). The periphyton species ordination axis was > 2.0 standard deviations, therefore, the

response curves were unimodal and canonical correspondence analysis was deemed appropriate (ter Braak 1987 in Jongman *et al.* 1995)

Pertinent environmental chemistry variables were identified via a literature review prior to sampling. Inclusion of irrelevant environmental data can generate data noise. To further reduce noise and bias, the aquatic macrophyte statistical analyses were based on species presence/absence data. Noise can cause distortions in ordination space, affecting the correct interpretation of relationships among variables (McCune 1997).

The RDA was restricted to submerged species only and six environmental variables. The CCA contained only periphyton species and seven environmental variables. The species data for both aquatic macrophytes and periphyton contained many null values (zero abundance) thus the Hellinger transformation was applied (see Legendre and Gallagher 2001). All environmental variables used in the final analysis, except pH (which is measured on a log scale), were log transformed, and all variables were centered on their means. Statistical analysis was carried out in R (Version 2.10.1 2009) or Microsoft Excel (Version 2003).

### *4.4.5. Gaussian Logistic Regression*

Gaussian logistic regression (GLR) describes the probability of occurrence (or non-occurrence) of an event as a function of multiple independent variables (ter Braak and Looman 1986). This method is most appropriate to use when the dependent variable is a binary variable (Kosten *et al.* 2009; ter Braak and Looman 1986). Logistic regression does not require independent variables to be normally distributed or have equal variance (Kosten *et al.* 2009). However, this method does require a high number of

data points, at least 50, to achieve stable results and assumes that independent variables are not significantly correlated to one another (Harvey *et al.* 1989; Kosten *et al.* 2009). This is ideal as the environmental variables in this study do not have an even distribution range across all trophic levels (oligotrophic ranges are underrepresented), which may affect the normality of the data. Correlated variables will be taken out of the equation as their results will be redundant (Narumalani *et al.* 1997; ter Braak and Looman 1986).

Gaussian logistic regression was used to explore the occurrence of individual macrophyte species in lakes as predicted by environmental variables. Pearson's correlation coefficient was used to determine the correlation between environmental variables. Variance inflation factor (VIF) values were used to decide which correlated variables would be best removed from the subsequent logistic regression. The results from the regression, along with the RDA, were used to determine which plant species are useful indicators of a particular trophic environment (Narumalani *et al.* 1997; ter Braak and Looman 1986).

### *4.4.4. Generalized Linear Model*

Generalized linear models (GLM) are an extension of linear multiple regression (StatSoft 2013). GLM is a widely accepted and broadly applicable statistical technique introduced by Nelder and Wedderburn (1972). It is a predictive model that, in many biological studies, is used by scientists to project the increase or decrease of a species, dependent on changes to certain measured variables (Fox 2015; Pearce and Ferrier 2001; StatSoft 2013). GLM is appropriate when there are one or more dependent and multiple independent variables and take into account the interactions between

multiple dependent variables (StatSoft 2013). Data can be analyzed without being transformed by introducing a random component that better describes the dependent variable: poisson, quasipoisson, binomial, Gaussian, among others (Fox 2015; Quinn and Keough 2001; StatSoft 2013). Dependent variables that are discrete, count data and have a non-normal distribution are best described using a poisson distribution (Quinn and Keough 2001). The poisson distribution assumes the mean is equal to the variance (Quinn and Keough 2001). If the data being analyzed by a poisson distribution contains many zeroes, which occurs in species presence/absence data, then the data will most likely be over dispersed (residual deviance > degrees of freedom). It is more appropriate then to use a quasipoisson distribution, which does not assume that the variance and mean are equal but extracts this relationship from the data itself (Quinn and Keough 2001).

GLM was used to evaluate the relationship between periphyton relative abundance data and environmental variables. In the periphyton data there are many instances where taxa are present in some lakes and not others. This creates overdispersion, where the variance does not equal the mean. This must be corrected. Therefore, the quasipoisson model was used as it does not assume that the variance equals the mean but rather calculates the ratio from the actual data (Quinn and Keough 2001).

*4.4.7. Species Diversity*

Species diversity of aquatic macrophytes is known to drastically decrease with increasing nutrient status of lakes (Jeppesen *et al.* 2000; Rørslett 1991). There is some discussion about whether the diversity of macrophytes decreases linearly with increasing phosphorus or whether the relationship is unimodal, with the optimum environment for aquatic species diversity being in the meso-eutrophic range, owing to the sufficient nutrient levels in meso-eutrophic lakes while maintaining high water clarity (Jeppesen *et al.* 2000). The uncertainty is attributed to the low representation of oligotrophic lakes in the studies that report a linear response curve (Jeppesen *et al.* 2000).

Periphyton species diversity, in particular how periphyton species abundance relates to nutrient status, has not been widely considered.

One of the most popular diversity calculations is the Shannon-Wiener Index. It calculates the uncertainty of predicting a species in a random sample at each sampling site (Spellerberg and Fedor 2003). If the diversity of an ecological community is low (i.e., dominated by one species) then the uncertainty of a prediction is low; the randomly sampled species will most likely be the dominant species. The higher the diversity the more uncertainty there is when predicting the species in a random sample (Spellerberg and Fedor 2003). In ecological studies, high diversity is correlated with lakes that have adequate nutrient levels while still maintaining good water clarity (Jeppesen *et al.* 2000). Low species diversity may indicate nutrients are less available for periphyton and macrophyte growth or that light penetration is restricted.

Species diversity is one of many parameters researchers use to establish management baselines or determine water quality (Egertson *et al.* 2004; Penning *et al.*

2008; Søndergaard *et al.* 2010). Diversity was calculated using the Shannon-Weiner Index for each of the forty-five lakes sampled for aquatic macrophytes and the thirty lakes sampled for periphyton. Taxon abundance and species relative abundance were used in the diversity calculation for aquatic plants and periphyton respectively. Linear regression was used to identify significant relationships between diversity and water chemistry variables.

## Chapter 5: Results

### 5.1. Environmental Variables

Lake environmental variables include total phosphorus (TP), total nitrogen (TN), specific conductance (Sp.C), dissolved oxygen (DO), pH, Secchi depth and chlorophyll a (chla). The lakes sampled include a wide range of values for all environmental variables (Table 5.1). The median values across all lakes were as follows: 1.17 mg/L TN, 0.035 mg/L TP, 484 µS/cm Sp.C, 95.6% DO, a pH of 8.31, and 11.6 µg/L chla. A broad spectrum of water chemistry values ensures that all lake types, oligotrophic, mesotrophic and eutrophic, are represented in the data. While many lake trophic types are represented in the data, it is important to note that 13 of the lakes sampled were classified as hypereutrophic, 13 were classified as eutrophic, 15 were mesotrophic and 4 were oligotrophic according to the TSI developed by Carlson and Simpson (1996) and the US EPA (2000). The majority of lakes with high nutrient levels, high chlorophyll a values and low transparency were found outside of the national park boundary. The rural landscape surrounding the park is almost exclusively dedicated to agricultural use. The input of agricultural fertilizers and the naturally shallow lakes of the prairie lend to the high productivity levels of most lakes in this study.

Prior to statistical analyses, all environmental data were transformed using log(x), except for pH which is already expressed on a logarithmic scale. All environmental data were normalized (i.e., scaled around their means). Based on the normal quantile plot (Figure 5.1) the transformed dataset fulfills the ordination assumption of a multivariate normal distribution.

Pearson's correlation coefficient revealed that several environmental variables were correlated with each other (Table 5.2). While including highly correlated values does not affect the ordination, it does make the interpretation more difficult (Palmer 1992). Specific conductance is highly correlated with both total phosphorus and total nitrogen (r-values of 0.51 and 0.71 respectively). A positive correlation also exists between total phosphorus and total nitrogen with an r-value of 0.69. Total phosphorus is also negatively correlated with Secchi depth (r = 0.5), identifying the tendency for lower water transparency in lakes having higher phosphorus levels. Chlorophyll a was positively correlated with total phosphorus (r = 0.72), confirming the known relationship between higher total phosphorus levels and increased primary production (Wetzel 2001). Chlorophyll a was also negatively correlated with secchi depth (r = -0.59). In general, high chlorophyll a values cause lower water transparency, which translates to decreased secchi depths (Wetzel 2001).

The variance inflation factor (VIF) was calculated to determine which variables should be dropped from the subsequent Gaussian Logistic Regression (Table 5.3). The initial VIF calculation (VIF all, Table 5.3) included all environmental variables. VIF values > 4 indicate variables with high correlation that may lead to unreliable regression results (Pardoe *et al. 2015).* When total nitrogen and chlorophyll a (VIF > 4) are removed from the equation the VIF calculations for all remaining environmental variables decrease; all other environmental variables may be included in the logistic regression.

*Table 5.1 The raw environmental data. Values in bold were sampled for both the aquatic macrophytes and periphyton. Values in italics represent lakes that are located within Riding Mountain National Park, MB. Lake numbers that are excluded were not used in the final analysis.*

| Lake No. | TN | TP | Sp.C | D.O | Secchi | pH | Chla |
|---|---|---|---|---|---|---|---|
| **1** | **0.83** | **0.046** | **363** | **96.0** | **0.6** | **8.17** | **31.6** |
| **2** | **1.88** | **0.131** | **399** | **137.2** | **0.2** | **8.27** | **177.0** |
| **3** | **0.67** | **0.018** | **272** | **90.6** | **1.9** | **8.34** | **5.6** |
| **4** | **1.92** | **0.111** | **1082** | **69.9** | **0.6** | **8.16** | **30.4** |
| **5** | **1.19** | **0.031** | **1865** | **93.0** | **1.3** | **8.24** | **13.6** |
| **6** | **1.45** | **0.148** | **1053** | **93.9** | **0.6** | **8.25** | **66.2** |
| **7** | **1.30** | **0.040** | **637** | **80.5** | **0.5** | **8.38** | **12.3** |
| **8** | **1.80** | **0.058** | **1368** | **72.0** | **1.7** | **7.95** | **5.1** |
| **9** | **1.54** | **0.031** | **554** | **98.4** | **1.2** | **8.58** | **5.9** |
| **10** | **1.71** | **0.057** | **1114** | **63.2** | **2.2** | **8.71** | **5.6** |
| **11** | **1.33** | **0.222** | **1098** | **77.8** | **1.3** | **8.20** | **17.6** |
| 13 | 2.70 | 0.323 | 1565 | 29.2 | 0.5 | 7.89 | 51.3 |
| **14** | **3.42** | **0.780** | **671** | **104.9** | **0.4** | **8.88** | **272.0** |
| 15 | 1.39 | 0.057 | 1012 | 99.3 | 1.2 | 8.51 | 50.5 |
| 16 | 1.24 | 0.052 | 1273 | 102.7 | 1.6 | 8.18 | 2.6 |
| **18** | **1.70** | **0.027** | **1383** | **89.4** | **2.8** | **8.50** | **6.2** |
| 19 | 1.48 | 0.067 | 828 | 109.4 | 1.1 | 8.64 | 40.6 |
| **20** | **1.39** | **0.047** | **1349** | **85.0** | **1.1** | **8.28** | **23.2** |
| **21** | **0.80** | **0.018** | **426** | **82.0** | **1.9** | **8.00** | **7.9** |
| **22** | **1.16** | **0.273** | **495** | **99.3** | **1.0** | **8.20** | **57.3** |
| **23** | **1.67** | **0.083** | **1626** | **136.9** | **1.0** | **8.65** | **78.0** |
| 24 | 2.10 | 0.055 | 2037 | 69.1 | 1.8 | 8.43 | 20.4 |
| **25** | **1.39** | **0.038** | **492** | **84.5** | **1.4** | **9.06** | **10.1** |
| **26** | **0.96** | **0.044** | **484** | **96.0** | **1.0** | **8.42** | **23.8** |
| **27** | **1.65** | **0.343** | **1061** | **93.8** | **1.6** | **8.23** | **16.3** |
| 28 | 0.70 | 0.010 | 374 | 101.2 | 1.9 | 8.12 | 6.9 |
| **29** | **1.53** | **0.018** | **629** | **112.6** | **0.4** | **8.98** | **3.0** |
| 30 | 0.67 | 0.015 | 320 | 99.2 | 1.5 | 8.01 | 5.2 |
| 31 | 1.00 | 0.017 | 432 | 100.8 | 2.0 | 8.42 | 8.7 |
| **33** | **0.59** | **0.050** | **236** | **109.8** | **1.0** | **8.70** | **46.6** |
| ***34*** | ***0.63*** | ***0.019*** | ***197*** | ***90.5*** | ***1.3*** | ***8.18*** | ***31.6*** |
| *35* | *1.15* | *0.028* | *367.5* | *96.2* | *0.95* | *8.46* | *11.6* |
| ***36*** | ***0.67*** | ***0.008*** | ***404*** | ***91.2*** | ***4.9*** | ***8.31*** | ***4.0*** |
| *38* | *0.53* | *0.017* | *200* | *93.6* | *3.3* | *8.09* | *3.8* |
| *39* | *0.52* | *0.015* | *241* | *106.8* | *1.7* | *8.17* | *6.5* |
| ***40*** | ***1.13*** | ***0.013*** | ***128*** | ***102.3*** | ***2.7*** | ***8.18*** | ***7.5*** |
| *43* | *0.87* | *0.034* | *391* | *104.6* | *1.3* | *8.32* | *9.8* |
| ***44*** | ***1.17*** | ***0.013*** | ***952*** | ***99.6*** | ***1.0*** | ***8.07*** | ***5.3*** |
| ***45*** | ***0.67*** | ***0.013*** | ***361*** | ***89.2*** | ***2.3*** | ***8.42*** | ***9.3*** |
| ***46*** | ***0.86*** | ***0.035*** | ***317*** | ***102.5*** | ***0.9*** | ***9.40*** | ***17.6*** |
| *48* | *0.61* | *0.014* | *288* | *92.6* | *1.8* | *8.67* | *14.2* |
| *49* | *0.56* | *0.007* | *333* | *86.2* | *0.7* | *7.70* | *2.8* |
| ***50*** | ***0.90*** | ***0.017*** | ***226*** | ***95.6*** | ***1.0*** | ***8.51*** | ***4.9*** |
| *51* | *0.66* | *0.047* | *197* | *112.7* | *1.0* | *9.28* | *82.7* |
| *52* | *1.85* | *0.018* | *357* | *89.6* | *3.9* | *8.19* | *2.6* |

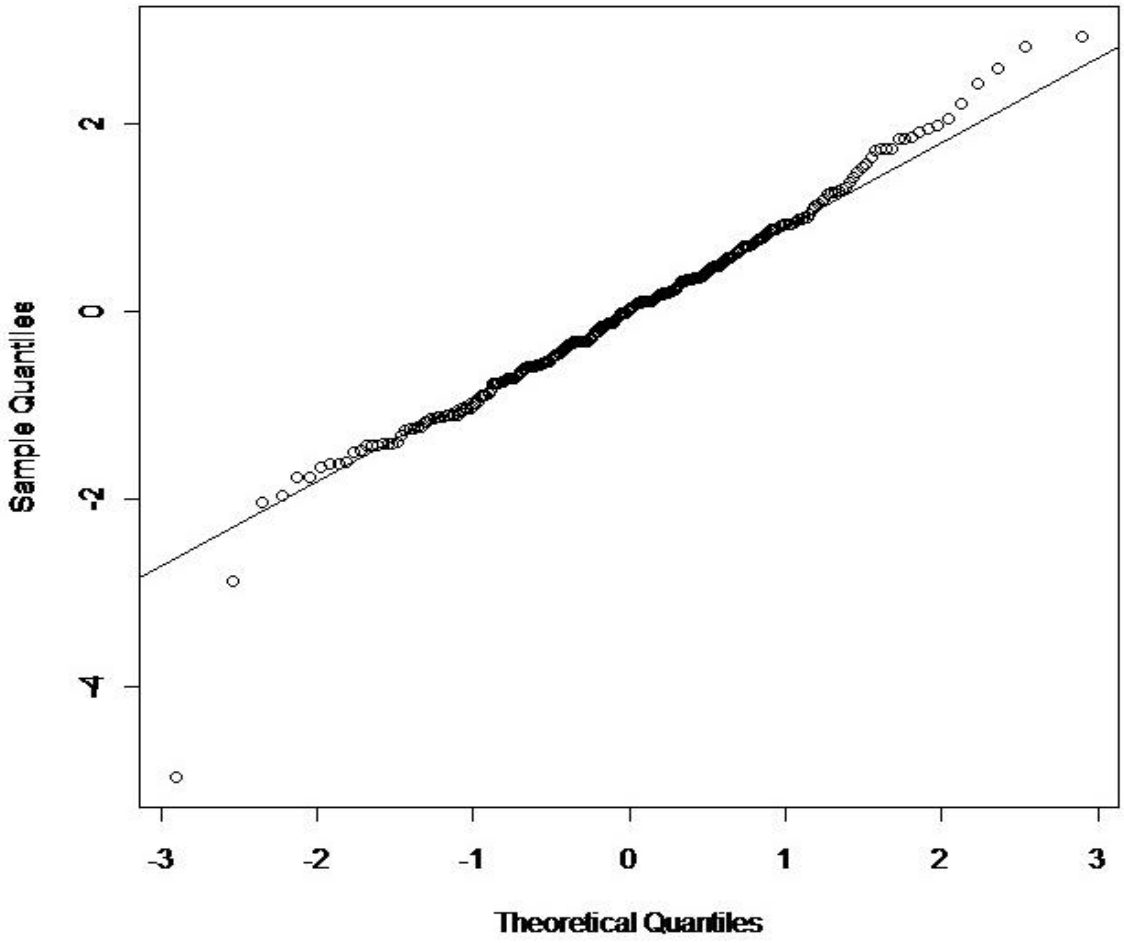



*Figure 5.1 Normal quantile plot for the environmental variables. All variables were log-transformed (with the exception of pH) and centred on their means.*

*Table 5.2 Pearson's Correlation Coefficient calculated for the environmental variables. Values in bold represent variables that are highly correlated (r > 0.5).*

| | TN | TP | Sp.C | D.O | Secchi | pH | Chla |
|---|---|---|---|---|---|---|---|
| **TN** | 1.00 | **0.69** | **0.71** | -0.31 | -0.35 | 0.09 | 0.35 |
| **TP** | **0.69** | 1.00 | **0.51** | -0.21 | **-0.50** | 0.12 | **0.72** |
| **Sp.C** | **0.71** | **0.51** | 1.00 | -0.36 | -0.16 | -0.10 | 0.16 |
| **D.O** | -0.31 | -0.21 | -0.36 | 1.00 | -0.05 | 0.33 | 0.09 |
| **Secchi** | -0.35 | **-0.50** | -0.16 | -0.05 | 1.00 | -0.14 | **-0.59** |
| **pH** | 0.09 | 0.12 | -0.10 | 0.33 | -0.14 | 1.00 | 0.27 |
| **Chla** | 0.35 | **0.72** | 0.16 | 0.09 | **-0.59** | 0.27 | 1.00 |

*Table 5.3 The variance inflation factor (VIF) for the environmental variables. Total nitrogen and chlorophyll a had VIF factors > 4 and therefore removed from the second VIF calculation. VIF without TN and Chla values are acceptable for the inclusion into the Gaussian Logistic Regression.*

| | TN | TP | Secchi | DO | Sp.C | pH | Chla |
|---|---|---|---|---|---|---|---|
| **VIF all** | **4.08** | 3.81 | 1.25 | 2.03 | 2.04 | 1.21 | **4.72** |
| **VIF without TN and Chla** | n/a | 1.16 | 1.17 | 1.30 | 1.21 | 1.15 | n/a |

## 5.2. Aquatic Macrophytes

A total of seventeen submerged plant species were found in the forty-five sample lakes. The greatest number of species, ten, was found in Lake Kinosao (38). Only one submerged species was found in Lakes 1, 6, 14, 23 (Crawford) and 36 (Clear) (Table A2.1). It is known, from previous research, that Clear Lake has more than one submerged aquatic plant species. While the majority of the lakes sampled were under 10 $km^2$ Clear Lake is large at 29.5 $km^2$. The two transects used in this study were insufficient to fully characterize the submerged aquatic macrophyte communities present in Clear Lake. The most common plant was *Myriophyllum verticillatum*, occurring in 32 sample lakes followed closely by *Potamogeton pectinatus* and *Chara* spp. which occurred in 31 and 28 lakes, respectively (Table A2.1).

*5.2.1. Non-Metric Multidimensional Scaling and Trophic Classification*

The NMDS ordination shows the similarity among the lake sites and the macrophyte species (Figure 5.2). Lakes that plot closely in the ordination space can be considered more similar than lakes plotting farther apart on the ordination space. The species that plot closer together can also be described as occurring in more similar environments than species that plot farther apart on the ordination space. The stress level = 0.12, indicates that the plot accurately represents the actual similarities of lakes in the environment. The trophic status of each lake, as determined via The TSI index is presented in Table 5.4.

Table 5.4 TSI values for all forty-five sample lakes. Lake numbers in bold represent the seventeen lakes sampled in RMNP.

| **Oligotrophic** | | **Mesotrophic** | | **Eutrophic** | | **Hypereutrophic** | |
|---|---|---|---|---|---|---|---|
| **Lake** | **TSI** | **Lake** | **TSI** | **Lake** | **TSI** | **Lake** | **TSI** |
| **49** | **27.1** | 3 | 49.1 | 5 | 69.3 | 2 | 115.9 |
| 29 | 39.3 | 16 | 43.7 | 8 | 61.4 | 4 | 87.8 |
| **36** | **34.3** | 28 | 45.4 | 10 | 62.7 | 6 | 80.5 |
| **52** | **37.1** | 30 | 45.8 | 25 | 67.1 | 13 | 101.9 |
| | | **38** | **42.4** | **35** | **65.7** | 14 | 130.5 |
| | | **39** | **49.3** | **48** | **60.6** | 15 | 91.0 |
| | | **40** | **49.8** | **43** | **60.7** | 19 | 88.7 |
| | | **44** | **44.4** | **46** | **69.9** | 22 | 94.5 |
| | | **50** | **46.3** | 7 | 70.8 | 23 | 100.5 |
| | | 1 | 54.1 | 11 | 77.6 | 24 | 119.9 |
| | | 9 | 56.3 | 20 | 78.0 | 26 | 82.2 |
| | | 18 | 55.5 | 27 | 78.7 | 33 | 84.9 |
| | | 21 | 54.4 | **34** | **76.7** | **51** | **94.1** |
| | | 31 | 55.3 | | | | |
| | | **45** | **53.2** | | | | |

Three groupings of macrophytes were evident in the NMDS ordination (Figure 5. 2). 1) Eutrophic lakes (TSI 60-80) were dominated by *Potamogeton pectinatus, Myriophyllum verticillatum, Valisneria americana, Chara* spp., *Elodea nuttalli* and *Utricularia vulgaris.* 2) Hypereutrophic lakes (TSI > 80) formed a distinct group in the upper left quadrant of the ordination and were dominated by *Ruppia maritima* and *Ceratophyllum demersum;* 3) Mesotrophic lakes (TSI 40-60) were dominated by *Potamogeton zosteriformis, Najas flexilis, Nitella, Bidens beckii, Potamogeton robbinsii, Eriocaulon aquaticum, Potamogeton richardsonii* and *Elodea canadensis*. There were no macrophyte species specific for oligotrophic lakes. The oligotrophic lakes (TSI < 40) did not have any characteristic macrophytes (as is the case with Lakes 29, 36 and 52) or, like Lake 49, had a macrophyte community resembling that of other lake trophic types.

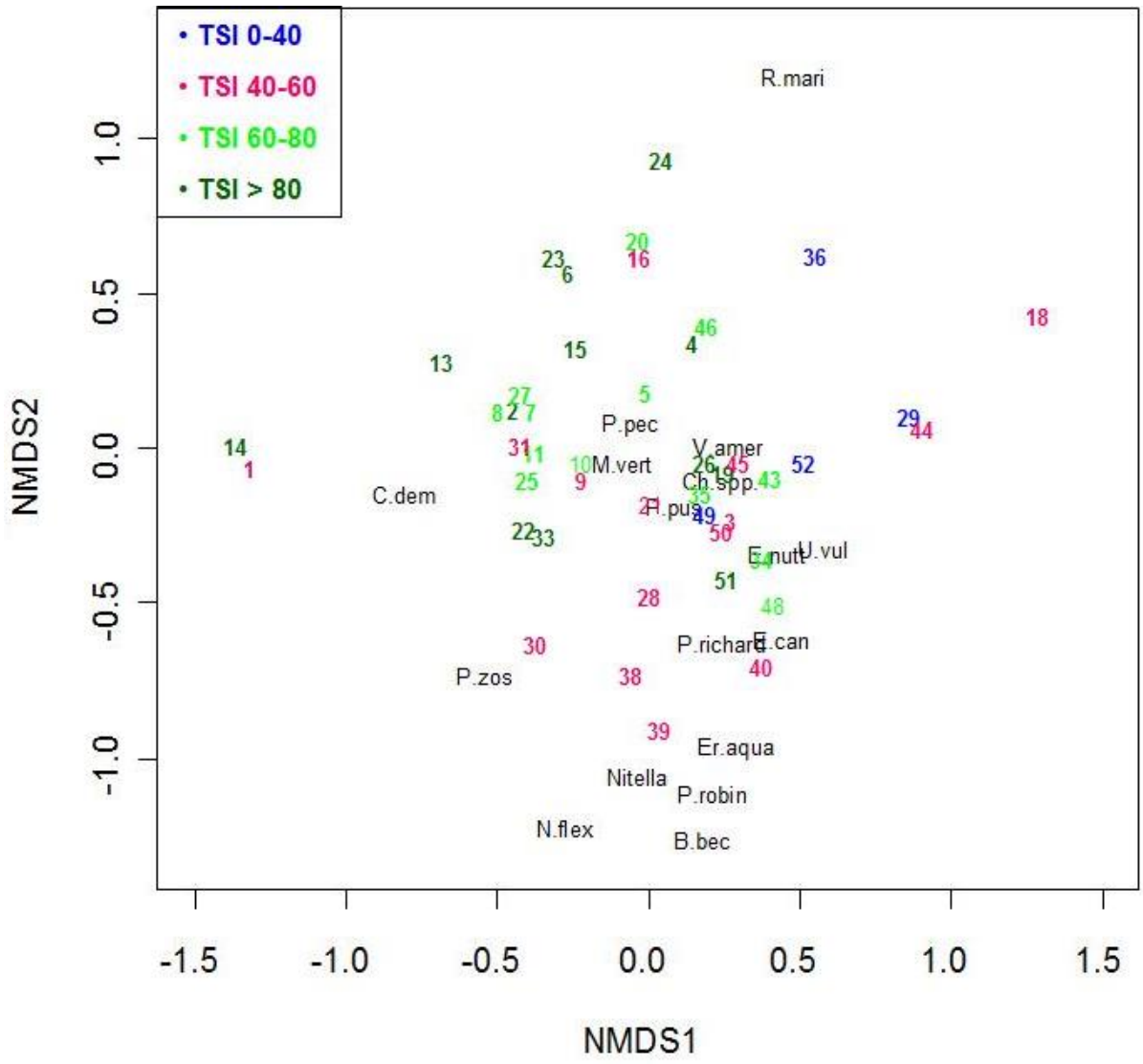



*Figure 5.2 NMDS ordination graphs for each of the forty-five sample lakes and eighteen submerged macrophyte species. Colour coding was assigned using the Trophic Status Index. Stress = 0.12. See Table 5.5 for species codes.*

*Table 5.5 Aquatic macrophyte taxon codes used in Figure 5.2 and Figure 5.3.*

| Code | Species |
|---|---|
| B.bec | *Bidens beckii* |
| Ch.spp. | *Chara* spp. |
| C.dem | *Ceratophyllum demersum* |
| Er.aqua | *Eriocaulon aquaticum* |
| E.can | *Elodea canadensis* |
| E.nutt | *Elodea nuttallii* |
| M.vert | *Myriophyllum verticillatum* |
| N.flex | *Naja flexilis* |
| P.pec | *Potamogeton pectinatus* |
| P.pus | *Potamogeton pusillus* |
| P.rich | *Potamogeton richardsonii* |
| P.robin | *Potamogeton robinsii* |
| P.zos | *Potamogeton zosteriformis* |
| R.mari | *Ruppia maritima* |
| U.vul | *Utricularia vulgaris* |
| V.amer | *Valisneria americana* |

### *5.2.2. Direct Gradient Ordination*

The RDA ordination (Figure 5.3) using the forty-five sample lakes indicates that the most important environmental gradients for explaining species variation are total phosphorus (TP), total nitrogen (TN), specific conductance (Sp.C) and secchi depth (Secchi). P-values confirm that TP, TN, Sp.C and Secchi depth are significantly related to the first two RDA axes (p-values <0.05), explaining the most variation seen in the species data (Table 5.6). The species farthest from the origin in the ordination space (*Potamogeton richardsonii, Potamogeton pectinatus, Chara* spp., *Myriophyllum verticillatum, Utricularia vulgaris, Ceratophyllum demersum* and *Ruppia maritima*) (Figure 5.3) also have the highest correlations (eigenvalues) with the first two RDA axes and the environmental variables that define those axes (TP, TN, Sp.C and Secchi). The first RDA axis explains 29.5% of the variance in the species data (Table 5.7). It is most

strongly influenced by total phosphorus in the positive direction and secchi depth in the negative direction (Table 5.6).

*Table 5.6 Environmental variables with large (close to one) eigenvalues have a greater effect on species distribution. Environmental variables that are significantly related (p-value ≤0.05) to the first two RDA axes (Figure 5.3) are shown in bold.*

| | RDA1 | RDA2 | $r^2$ | P-value |
|---|---|---|---|---|
| **TN** | **0.87** | **-0.49** | **0.28** | **< 0.05** |
| **TP** | **0.98** | **-0.18** | **0.50** | **< 0.05** |
| **Sp.C** | **0.59** | **-0.81** | **0.49** | **< 0.05** |
| D.O | -0.78 | -0.63 | 0.07 | 0.24 |
| **Secchi** | **-1.00** | **0.09** | **0.16** | **0.03** |
| pH | -0.98 | -0.20 | 0.03 | 0.59 |
| **Chla** | **0.97** | **-0.24** | **0.23** | **0.01** |

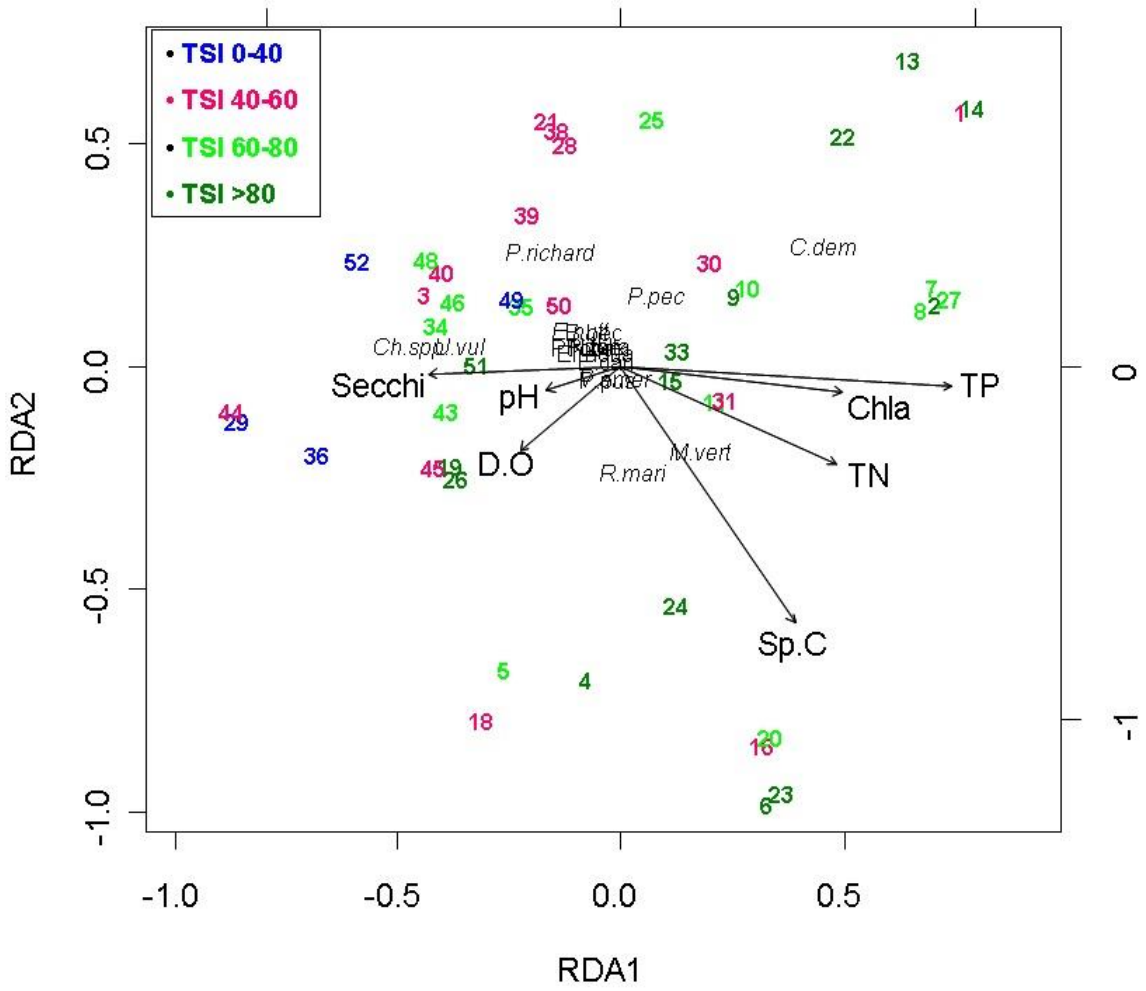



*Figure 5.3 Redundancy analysis for the aquatic macrophyte data from forty-five lakes in and around Riding Mountain National Park, Manitoba. Colour coding was assigned using the Trophic Status Index See Table 5.5 for species codes.*

The eigenvalues for *C. demersum, P. pectinatus and M. verticillatum* are positive on axis 1 indicating their association with lakes that have high total phosphorus and low secchi depths. In contrast, *Chara* spp. and *U. vulgaris* have high negative axis 1 scores (associated with lakes having high secchi depths and low total phosphorus concentrations) (Figure 5.3). Specific conductance influenced the second RDA axis most strongly (Table 5.6). *P. richardsonii* is strongly associated with low specific conductance values while *R. maritima*, a well-known halophile (Richardson, 1980), is associated with high specific conductance values (Figure 5.3). The second RDA axis explains 18.1% of the variance in the species data; thus, combined, RDA axes 1 and 2 explain 47.7% of the variance (Table 5.7).

*Table 5.7 The individual and cumulative aquatic macrophyte species variance as explained by the first three RDA axes (Figure 5.3).*

| | **RDA1** | **RDA2** | **RDA3** | **Total variance** |
|---|---|---|---|---|
| Eigenvalues for constrained axis | 0.071 | 0.032 | 0.026 | 0.534 |
| Eigenvalues for unconstrained axis | 0.087 | 0.065 | 0.062 | |
| Proportion of variance explained (%) | 29.6 | 18.2 | 16.4 | |
| Total variance explained (%) | 29.6 | 47.7 | 64.1 | |

### *5.2.3. Gaussian Logistic Regression*

Total nitrogen and chlorophyll a were determined, through a VIF > 4 (Table 5.3), to be highly correlated with the remainder of the environmental variables; affecting the reliability of the interpretation of the Gaussian Logistic Regression (GLR); therefore, total nitrogen and chlorophyll a were excluded from subsequent calculations. The seven macrophyte species whose variance was best explained by the first two RDA axes (Table 5.8) were regressed against TP, Sp.C, Secchi, DO and pH.

*Table 5.8 P-values generated from the Gaussian Logistic Regression. Values in bold indicate a significant (p-values<0.05) relationship between species and environmental variable.*

| | TP | Sp.C | Secchi | DO | pH |
|---|---|---|---|---|---|
| *Ceratophyllum demersum* | **< 0.05** | 0.07 | 0.36 | 0.22 | 0.32 |
| *Myriophyllum verticillatum* | 0.38 | 0.16 | 0.09 | **0.03** | 0.08 |
| *Potamogeton richardsonii* | 0.66 | **< 0.05** | 0.50 | 0.73 | 0.31 |
| *Chara* spp. | **0.05** | 0.10 | 0.14 | 0.32 | **0.05** |
| *Ruppia maritima* | 0.30 | **< 0.05** | 0.47 | 0.88 | 0.78 |
| *Potamogeton pectinatus* | 0.37 | 0.43 | 0.20 | **0.01** | 0.88 |
| *Utricularia vulgaris* | 0.16 | 0.20 | 0.37 | 0.59 | 0.56 |

Significant relationships (p-value < 0.05) identified in GLR are shown in bold in Table 5.8. The relationships were then plotted in Figures 5.4-5.10 to visualize the actual environmental ranges at which the species occur. *Ceratophyllum demersum* was significantly related to total phosphorus (Table 5.8). The occurrence of *C. demersum* increases as total phosphorus values increase, up until total phosphorus concentration reached levels associated with hypeutrophication (Figure 5.4). The majority of *C.demersum* individuals were found at total phosphorus levels associated with mesotrophic (0.012-0.024 mgTP/L) or eutrophic lakes (0.024-0.096 mgTP/L) (Figure 5.4).

*M. verticillatum* and *P. pectinatus* both demonstrated similar significant relationships with dissolved oxygen (Table 5.8). Fifty-eight percent of *M. verticillatum* occurrences and sixty-one percent of *P. pectinatus* occurrences were in lakes with dissolved oxygen values ranging from 75-100% (Figure 5.6 and Figure 5.7 respectively). This dissolved oxygen range, 75-100%, coincides with the dissolved oxygen range that encompasses most lakes sampled in this study (60%). Since *M. verticillatum* and *P.*

*pectinatus* were found in the majority of sample lakes (27 of 45 and 31 of 45 respectively) it stands to reason they would be recorded most often in the most often recorded dissolved oxygen range of 75-100%.

*P. richardsonii* occurred exclusively in lakes with Sp. C values <600 µS/cm (Figure 5.8), regardless of their TSI classification. Seventy-five percent of *Chara* spp. occurred in lakes with total phosphorus concentrations that most often correspond with either mesotrophic (0.012-0.027 mg/L TP) or eutrophic (0.024-0.096 mg/L TP) levels (Figure 5.9). The majority of *Chara* spp. appeared to occur in mesotrophic and eutrophic lakes with a sharp decline in species at very high or very low nutrient levels (Figure 5.9); although the lack of species at low total phosphorus levels may have been due to the underrepresentation of oligotrophic lakes in the study. *Chara* spp. and pH were also found to be significantly correlated. However, since the majority of study lakes (87%) have relatively little variation in pH (8.00-8.98) the relationship between *Chara* spp. and pH does not have much value and will not be discussed further. *R. maritima* was found exclusively in lakes with specific conductance values > 901 µS/cm (Figure 5.10), with the majority of the species' occurrences being found in lakes with specific conductance values ranging between 1201-1500 µS/cm.

*Utricularia vulgaris* did not show any significant patterns of occurrence with respect to any environmental variable (Table 5.8). *Utricularia vulgaris* is a carnivorous plant and is able to attain nutrients by capturing small aquatic invertebrates. This ability was not taken into account in the analysis and may have allowed *Utricularia vulgaris* to thrive in the wide range of environments in which it was recorded.

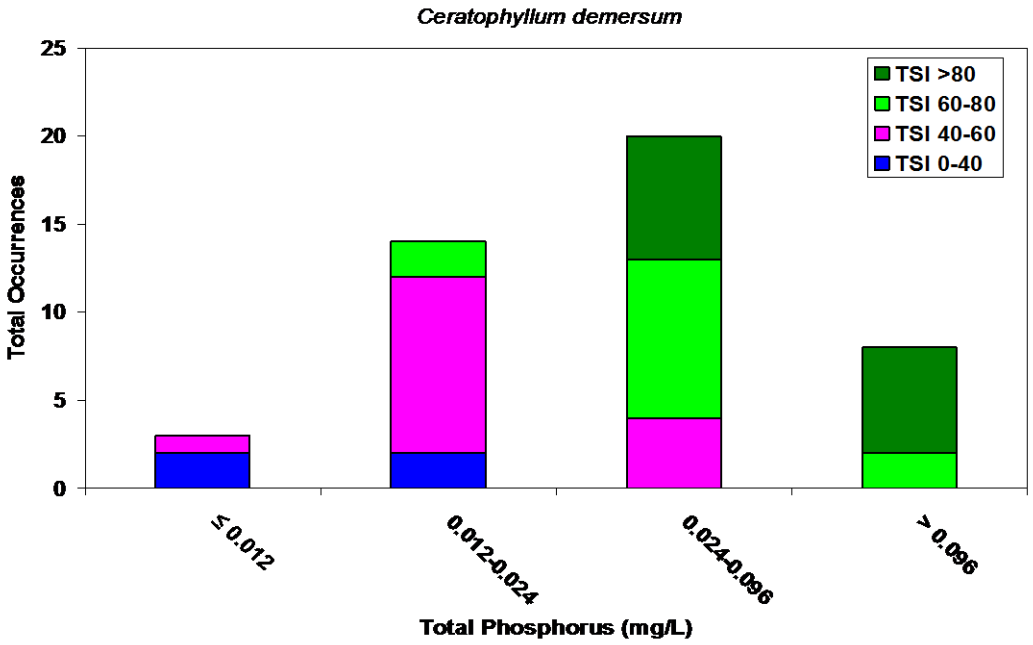



*Figure 5.4 Ceratophyllum demersum occurrences in sample lakes and their associated total phosphorus concentrations. Occurrences of C. demersum are separated by TSI categories.*

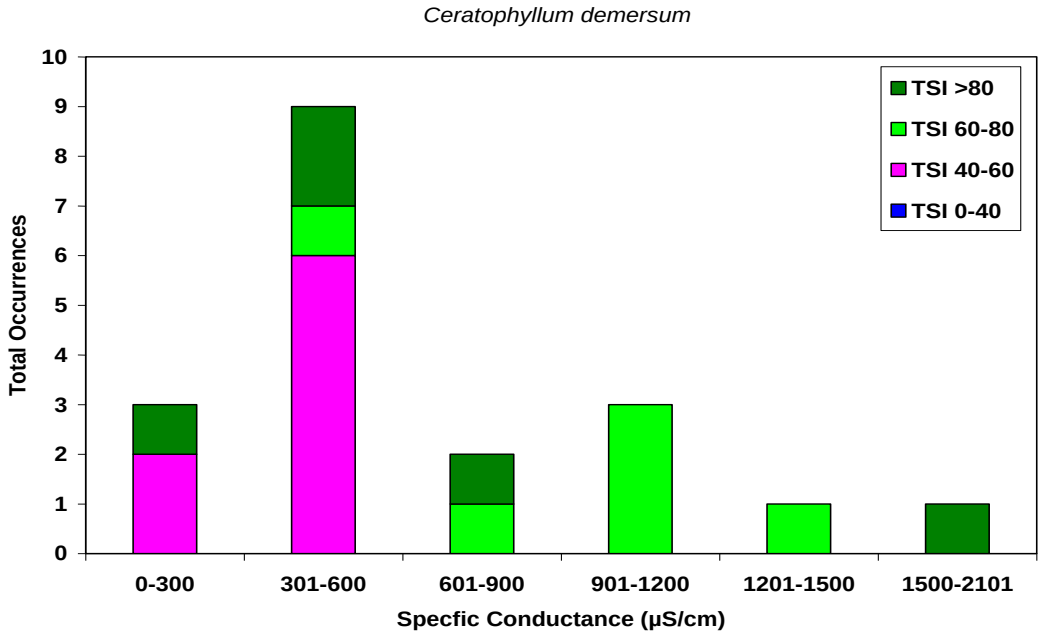



Figure 5.5 *Ceratophyllum demersum occurrences in sample lakes and their associated specific conductance levels. Occurrences of C. demersum are separated by TSI categories.*

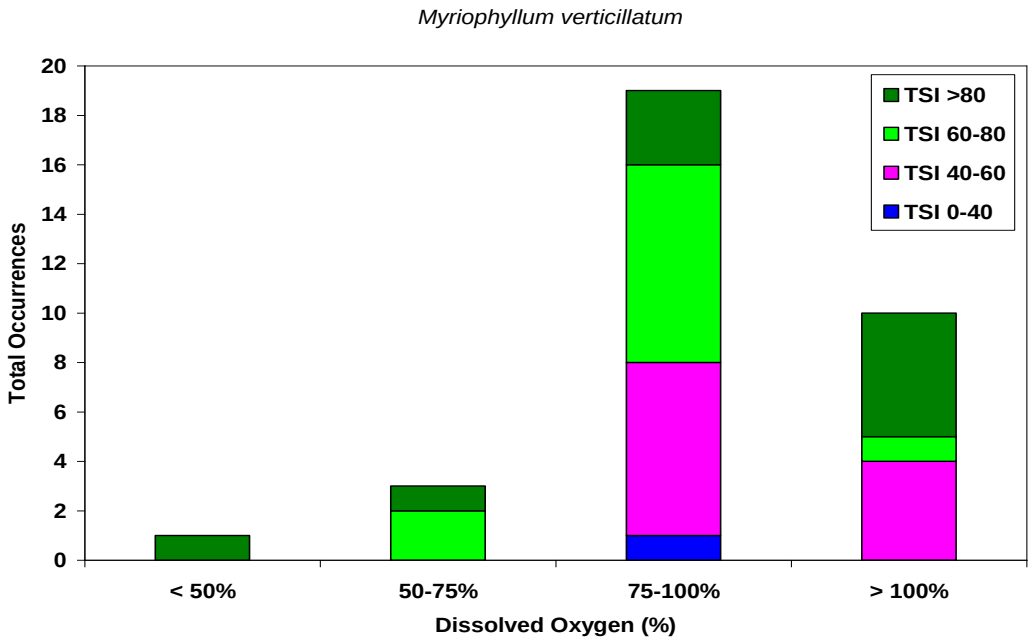



Figure 5.6 *Myriophyllum verticillatum occurrences in sample lakes and their associated dissolved oxygen levels. Occurrences of M. verticillatum are separated by TSI categories.*

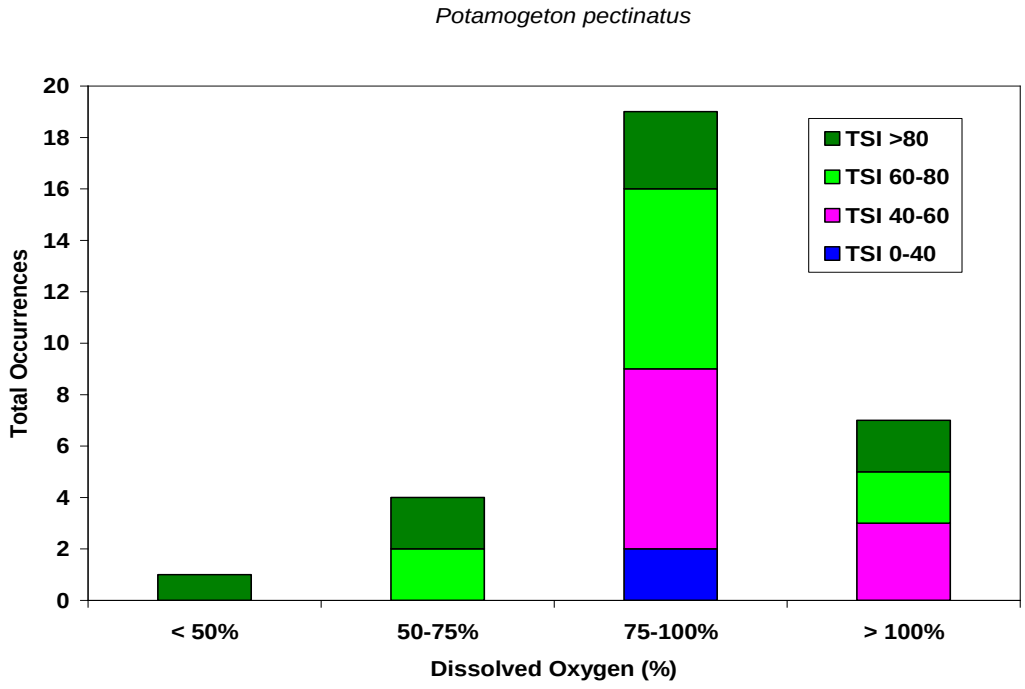



*Figure 5.7 Potamogeton pectinatus occurrences in sample lakes and their associated dissolved oxygen levels. Occurrences of P. pectinatus are separated by TSI categories.*

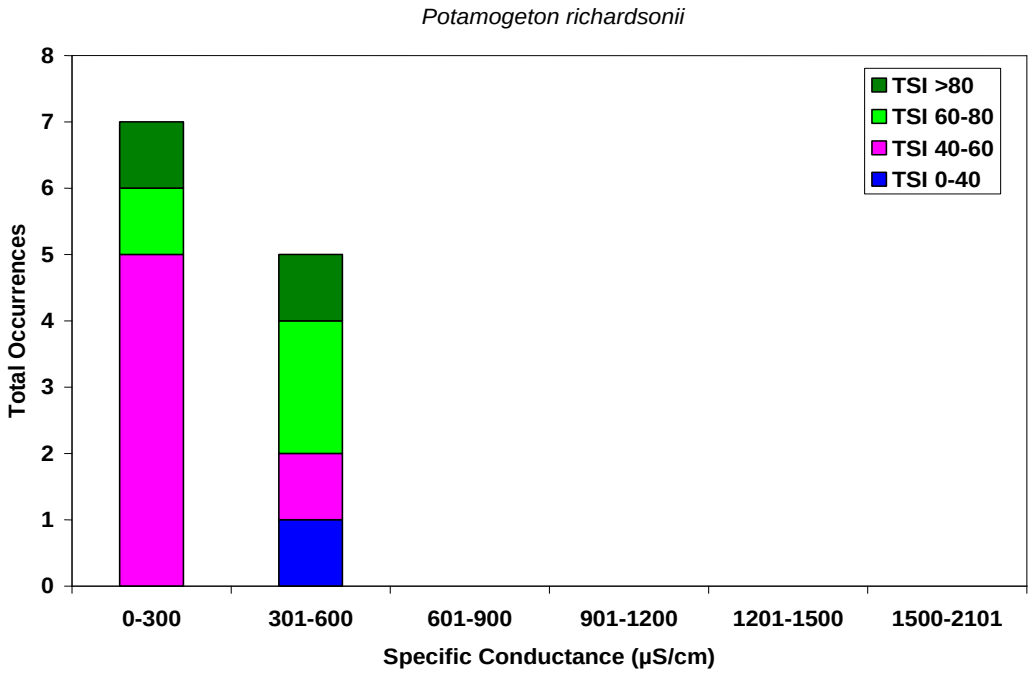



Figure 5.8 *Potamogeton richardsonii occurrences in sample lakes and their associated specific conductance levels. Occurrences of P. richardsonii are separated by TSI categories.*

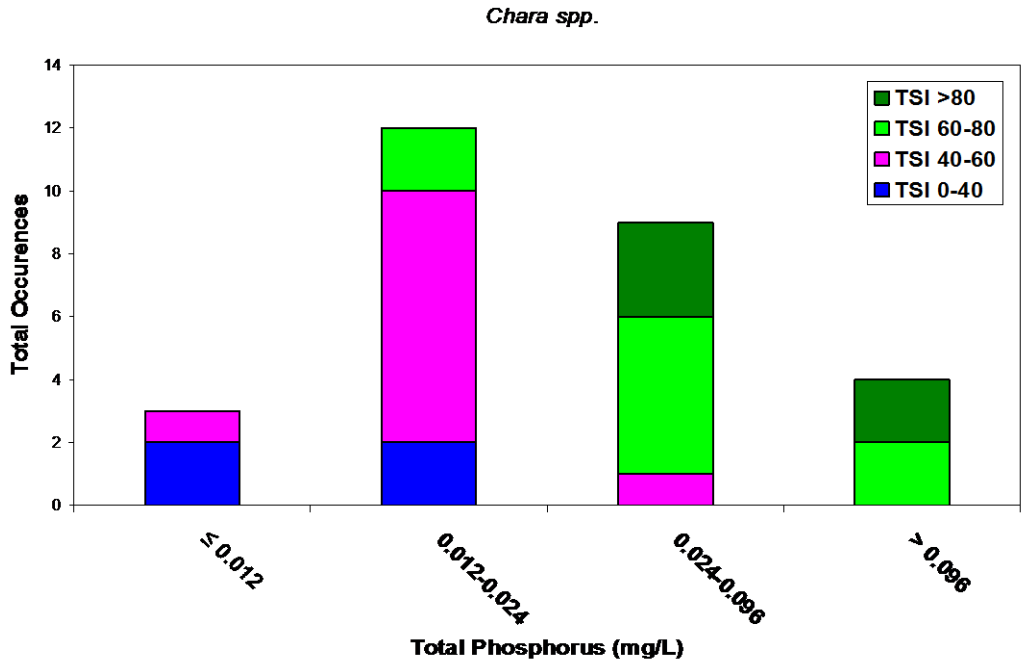



*Figure 5.9 Chara spp. occurrences in sample lakes and their associated specific conductance levels. Occurrences of Chara spp. are separated by TSI categories.*

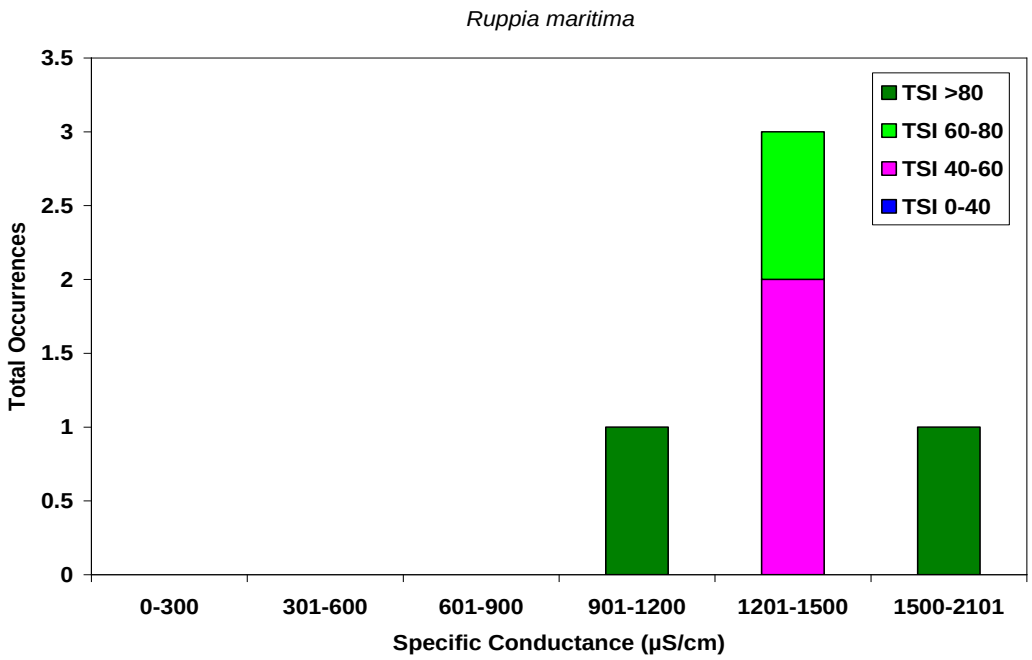



*Figure 5.10 Ruppia maritima occurrences in sample lakes and their associated specific conductance levels. Occurrences of R. maritima are separated by TSI categories.*

### *5.2.4. Species Diversity*

Lake Kinosao (38) had the greatest species diversity with ten submerged macrophyte species. Clear Lake (36), Lake 1, 6, 13 and 23 had the lowest diversity with only one submerged species recorded. Species diversity was significantly correlated with specific conductance with a p-value <0.05 (Table 5.9). The highest species diversity values were found in lakes with specific conductance values < 500 µS/cm. This pattern reflects the intolerance of most aquatic macrophyte species to high salinity, with a few notable exceptions (*P. pectinatus* and *R. maritima*) (Hammer & Heseltine 1988; Heegaard *et. al.* 2001). No correlation was detected between the TSI of a lake and species diversity values (Figure 5.11).

*Table 5.9 Linear regression was used to highlight any potentially significant relationships between aquatic macrophyte species diversity and environmental variables. The Shannon-Wiener index was used to calculate species diversity.*

| | Std. Error | P-value |
|---|---|---|
| **TP** | 1.11 | 0.67 |
| **TN** | 0.26 | 0.99 |
| **DO** | 0.01 | 0.57 |
| **Sp.C** | < 0.01 | **0.01** |
| **Secchi** | 0.09 | 0.86 |
| **pH** | 0.24 | 0.36 |
| **Chla** | < 0.01 | 0.42 |

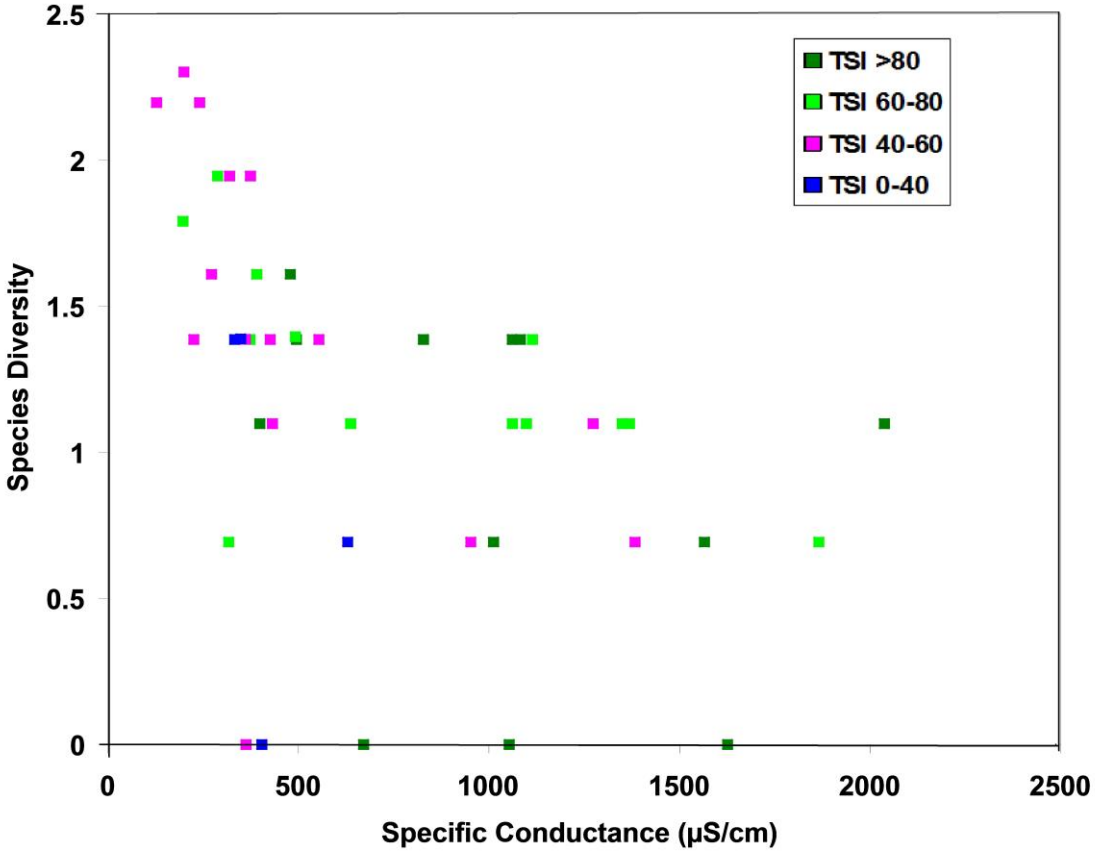



*Figure 5.11 Species diversity values for aquatic macrophytes in the forty-five sample lakes as they relate to specific conductance (µS/cm) values. P-value = <0.05. A species diversity value of 0 indicates that only one species of submerged aquatic macrophyte was found in that lake.*

## 5.3. Periphyton

Six major algal groups (Cyanophyceae, Chlorophyceae, Desmidiaceae, Bacillariophyceae, Pyrrophyta and Euglenophyceae) were identified comprising a total of 175 species in the thirty lakes sampled (Table A2.2). Only Bacillariophyceae were found in all lakes; Pyrrophyta were only recorded in three lakes, most likely due to their preference for planktonic rather than periphytic habitats (Hannson 1992) (Table A2.2). The dominant periphyton taxa in all lakes were always one of Cyanophyceae, Chlorophyceae or Bacillariophyceae (Table 5.10).

### *5.3.1. Non-Metric Multidimensional Scaling*

NMDS shows the similarity among lake sites and related periphyton. The final stress was 0.14, indicating that the ordination provides an interpretable representation of the relationships among sites and periphyton taxa. In general, eutrophic-hypereutrophic lakes dominated the lower and right portions of the ordination (Figure 5.12). Mesotrophic-eutrophic lakes dominated the centre and left areas of the ordination, while the two oligotrophic lakes were found near the upper portion of the ordination (Figure 5.12). The lack of distinct grouping of lake trophic states was not a surprise as trophic states are not discrete categories but rather occur on a continuum from oligotrophy to hypereutrophy. The periphyton taxa that are labeled in Figure 5.12 represent the periphyton which contributed $\geq 0.10\%$ to the shape of the ordination. The labeled taxa represent three of the seven major taxa recorded, Bacillariophyceae, Chlorophyceae and Cyanophyceae (Table A1.2). Most periphyton species occurred in the mesotrophic-eutrophic lakes, while there were relatively few species found in the eutrophic-

hypereutrophic lakes (Figure 5.12). Species that plotted closer together in the ordination space will tend to occur in more similar environments than species plotted farther apart. As well, lakes that occur nearer in ordination space will have more similar species composition than lakes that are farther away in ordination space.

*Table 5.10 Periphyton taxon codes used in Figures 5.12 and 5.13.*

| Code | Taxon | Type |
|---|---|---|
| Rho | *Rhopalodia gibba* (Ehrenberg) Otto Müller | Bacillariophyceae |
| P.lim | *Planktolyngbya limnetica* | Cyanophyceae |
| Pse.1 | *Pseudoanabaena spp.* 1 | Cyanophyceae |
| Coc | *Cocconeis. sp.* | Bacillariophyceae |
| Pho | *Phormidium sp.* | Cyanophyceae |
| Chr | *Chroococcus dispersus* (Keissel) Lemmermann | Cyanophyceae |
| M.lg | *Mougeotia sp. large* | Chlorophyceae |
| Sti | *Stigeclonium sp.* | Chlorophyceae |
| M.min | *Merismopedia minima* (Ehrenberg) Kützing | Cyanophyceae |
| Aul | *Aulacoseira sp.* | Bacillariophyceae |
| M.ten | *Merismopedia tenuissima* Lemmermann | Cyanophyceae |
| Pse | *Pseudoanabaena spp.* | Cyanophyceae |
| S.uln | *Synedra ulna (Nitzsch)* Ehrenberg | Bacillariophyceae |
| S.acu | *Synedra acus (Kützing)* | Bacillariophyceae |
| Gom | *Gomphosphaeria aponina* Kützing | Cyanophyceae |
| Syn | *Synechococcus sp.* | Cyanophyceae |
| Spi | *Spirogyra sp.* | Zynematophyceae |
| Lyn | *Lyngbya birgei (G. M. Smith)* | Cyanophyceae |
| Osc | *Oscillatoria sp.* | Cyanophyceae |
| M.sm | *Mougeotia sp. small* | Chlorophyceae |
| Ana | *Anabaena sp.* | Cyanophyceae |

### *5.3.2. Canonical Correspondence Analysis*

The environmental gradient that explained the most taxon variance was specific conductance (p-value <0.01), followed by total nitrogen (p-value < 0.1) and chlorophyll a (p-value < 0.15) (Table 5.11). The first CCA axis was primarily influenced by total nitrogen and specific conductance, as determined by their eigenvalues of 0.99 and 0.84, respectively (Table 5.11). The second CCA axis was influenced by chlorophyll a with an

eigenvalue of -0.89 (Table 5.11). Axis 1 explained 12.7% of the variance in the taxon data, axis 2 explained 6.3%. In total, the CCA explained 36.6% of the variation seen in the periphyton taxon data (Table 5.12). The twenty-one periphyton taxa labeled in Figure 5.13 contributed >0.1% to the overall shape of the ordination.

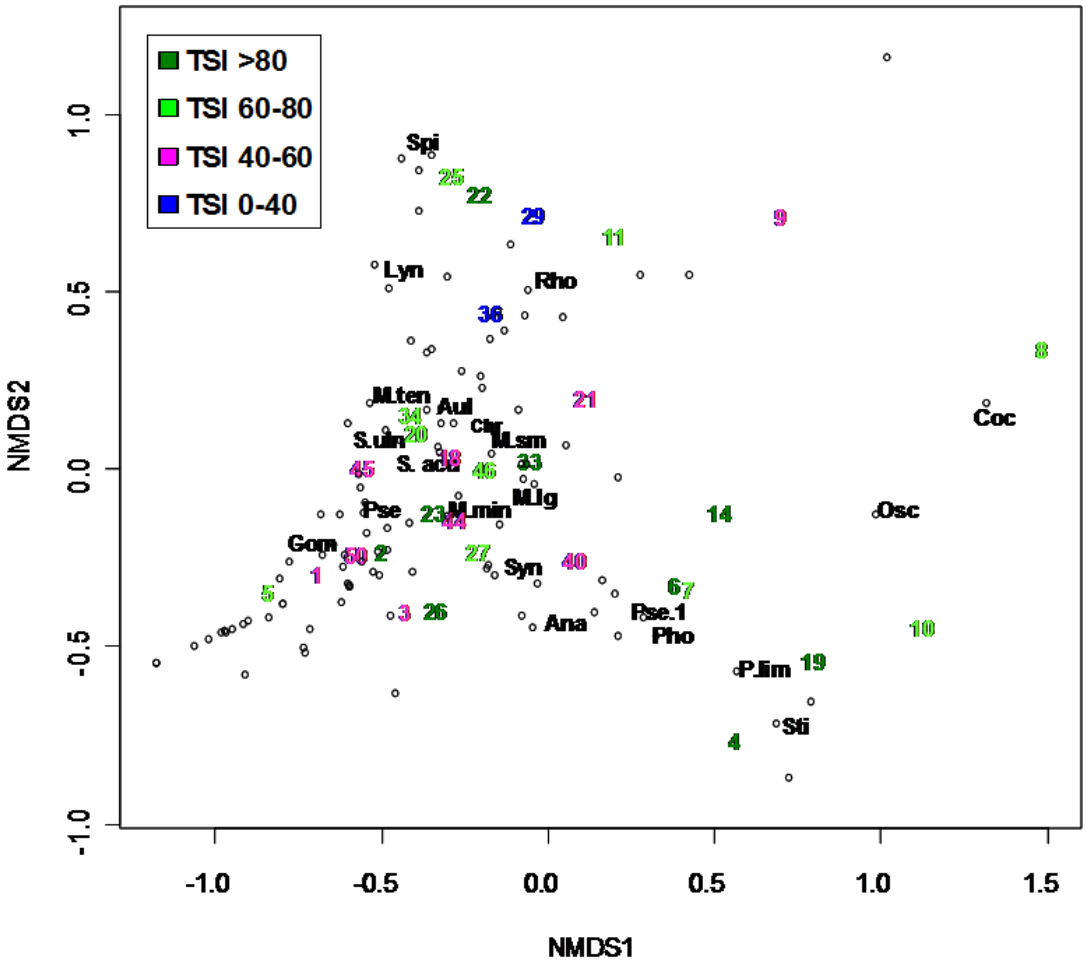



*Figure 5.12 NMDS ordination displaying the thirty lake sites and their associated periphyton taxa. Only taxa that contributed ≥ 0.10% to the shape of the ordination were labeled with species codes. All other taxa are represented as hollow circles. Stress = 0.14. See Table 5.10 for taxon codes.*

These twenty-one taxa were identified as having the highest potential as bioindicators and are given in Table 5.10. Only these twenty-one taxa will be described further.

Overall, most periphyton species (labeled and plotted points) plotted in the upper central/right portion of the ordination. There appeared to be three periphyton groups identified by the ordination. Groups 2 and 3 appeared to separate on a continuum whereas Group 1 was clearly distinct (Figure 5.13):

Group 1 occupied the left portion of the ordination. This group consisted of the taxa *Aulacoseira* sp., *Merismopedia tenuissima* Lemmermann, *Merismopedia minima* (Ehrenberg) Kützing, and *Chroococcus dispersus* (Keissel) Lemmermann (Figure 5.13). In general, this periphyton group occurred in lakes with relatively low nutrient levels (TN and TP), high water clarity (high secchi depths) and moderate phytoplankton biomass (as denoted by high chlorophyll a) (Figure 5.13). Group 2 occupied the upper right portion of the ordination and included *Rhopalodia gibba* (Ehrenberg) Otto Müller, *Planktolyngbya limnetica*, *Pseudoanabaena* spp. 1, *Cocconeis*. sp., *Phormidium* sp, *Stigeoclonium* sp., and *Mougeotia* sp. large. These taxa characterized lakes with moderate nutrient levels (TN and TP) but low phytoplankton biomass (low chla) and thus high water clarity (high secchi depths). Group 3 periphyton occupied the lower right portion of the ordination. The group included: *Pseudoanabaena* spp., *Synedra ulna* (Nitzsch) Ehrenberg, *Synedra acus* (Kützing), *Gomphosphaeria aponina* Kützing, *Synechococcus* sp., *Spirogyra* sp., *Lyngbya birgei* (G. M. Smith), *Oscillatoria* sp., and *Mougeotia* sp. small. This group of periphyton would tend to occur in lakes with low visibility (low secchi depths), high chlorophyll a levels and moderate nutrient levels (TN and TP).

*Table 5.11 Relationship of environmental variables to the first two CCA axes. Bold values indicate variables significantly related to the first two CCA axes. The values in italics are the variables with the highest eigenvalues, contributing most strongly to the structure of the CCA.*

| | **CCA1** | **CCA2** | **$r^2$** | **P-value** |
|---|---|---|---|---|
| Chla | **-0.45** | **-0.89** | **0.52** | **0.14** |
| ***TN*** | ***0.98*** | ***-0.21*** | ***0.6*** | ***0.10*** |
| TP | 0.99 | -0.13 | 0.29 | 0.45 |
| ***Sp.C*** | ***0.84*** | ***-0.55*** | ***0.78*** | ***0.01*** |
| DO | 0.81 | -0.59 | 0.05 | 0.86 |
| Secchi | -0.66 | 0.75 | 0.18 | 0.66 |
| pH | 0.89 | 0.47 | 0.19 | 0.58 |

*Table 5.12 The individual and cumulative periphyton taxon variance explained by the first three CCA axes.*

| | **CCA1** | **CCA2** | **CCA3** | **Total** |
|---|---|---|---|---|
| Eigenvalue | 0.489 | 0.244 | 0.204 | |
| Proportion explained | 0.127 | 0.063 | 0.053 | |
| Variance explained | 12.7 | 6.3 | 5.3 | 24.3 |
| Variance explained by all axes | | | | 36.6 |

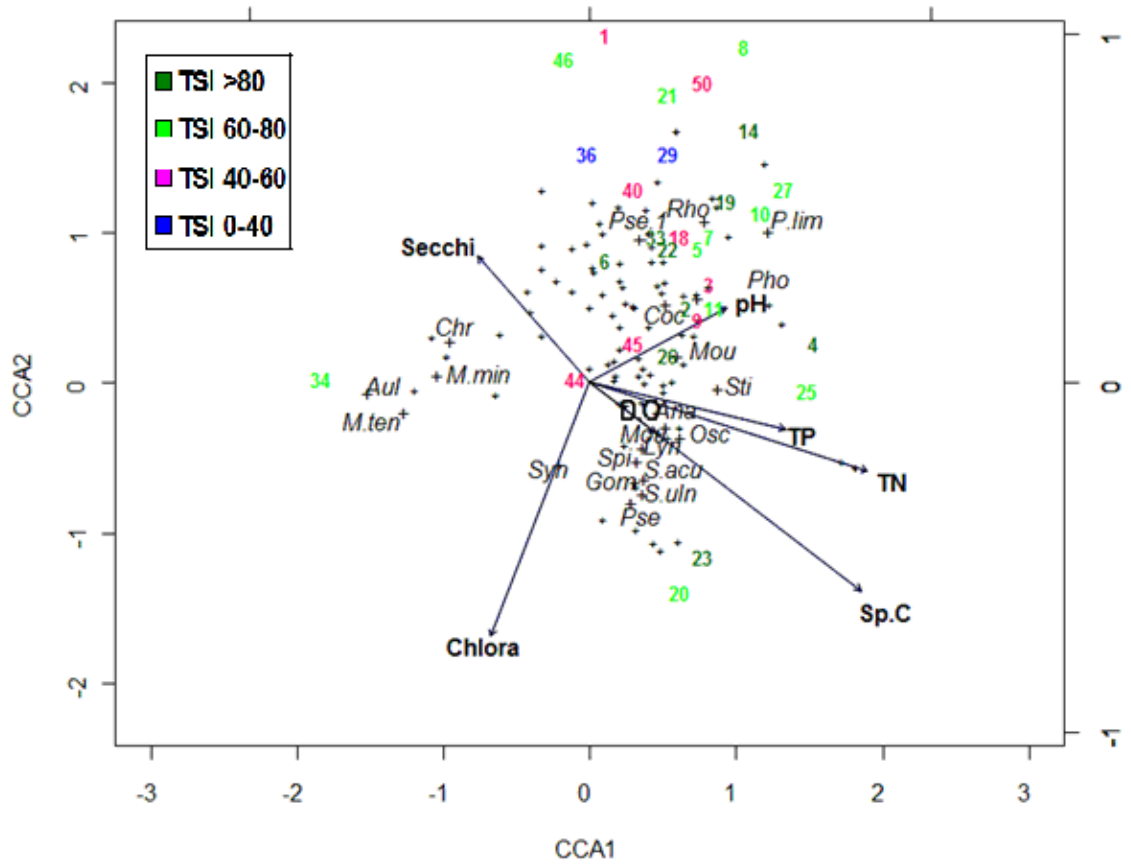



*Figure 5.13 Canonical Correspondence Analysis is for the periphyton data from thirty lakes in and around Riding Mountain National Park, Manitoba. Only taxa that contributed >0.10% to the shape of the ordination were labeled. All other taxa are represented as points. See Table 5.10 for taxon codes.*

*5.3.3. Generalized Linear Model*

The abundances of the twenty-one algal taxa outlined in Figure 5.13 were fitted against the environmental variables most strongly correlated with the 1st and 2nd CCA axes (Table 5.11) using the quasipoisson family of the Generalized Linear Model. Of the twenty-one taxa analyzed, twenty taxa had GLM algorithms that did not converge or displayed no statistical significance. These taxa outputs are not discussed. *Cocconeis* sp. was the only taxon to show a possibly significant relationship with any environmental variable. According to the GLM, dissolved oxygen concentration has a high probability (p-value = 0.02) of predicting the occurrence of *Cocconeis* sp. The population of *Cocconeis* sp. appeared to decrease in abundance as the dissolved oxygen concentration increased (Figure 5.14). Figure 5.14 also shows, with the exception of one lake, all lakes with supersaturated dissolved oxygen values were hypereutrophic lakes.

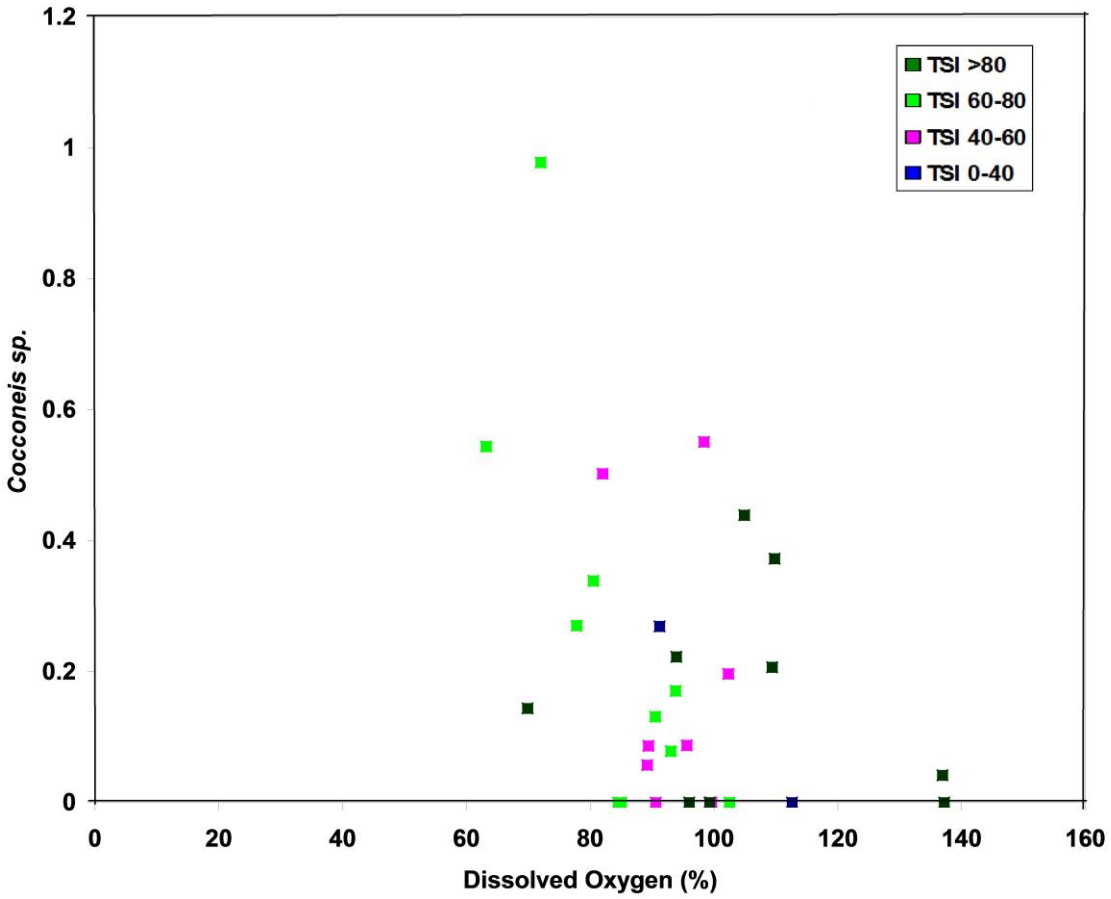



*Figure 5.14 The relative abundance of Cocconeis sp. in each of the twenty-seven lakes sampled for periphyton as it relates to dissolved oxygen concentration (%).*

*5.3.4. Taxon Diversity*

Bacillariophyceae was the only major group recorded in all thirty lakes. The most abundant taxon was *Oedogonium* (class Chlorophyceae), making up 7.09% of all periphyton cells counted from the sample lakes. The least abundant taxon included in the final analysis was *Cosmarium* (family Desmidiaceae) making up only 0.0003% of the periphyton cells. The highest diversity occurred in the mesotrophic Lake 50 (Cripple), while the lowest diversity was recorded in the eutrophic Lake 8 (Table A1.2). Periphyton taxon diversity was regressed against the environmental variables. Two environmental variables, total nitrogen and dissolved oxygen, were identified (p-value $< 0.05$) as having a significant impact on periphyton taxon diversity (Table 5.13).

Periphyton taxon diversity decreased, with the exception of the outlier lake 14, as total nitrogen increased regardless of the trophic status index classification (Figure 5.15). This relationship was highly variable. An increase in total nitrogen may be an indication, in lakes classified as eutrophic or hypereutrophic, that periphyton species are being outcompeted for nutrients by other photosynthetic organisms; leading to their decline. However, Figure 5.15 shows a decrease in periphyton taxon diversity in mesotrophic and oligotrophic lakes, lakes that would most likely not be overrun with primary producers. Increasing total nitrogen concentration and the decline of periphyton taxon diversity may point to some other correlating factor not taken into account for in this study.

Periphyton diversity increased with increasing dissolved oxygen concentration, although the relationship was highly variable (Figure 5.16). Photosynthetic organisms contribute to the dissolved oxygen concentration but dissolved oxygen also varies with the time of day samples were taken, temperature and lake mixing (Wetzel 2001). The

highest periphyton diversity occurred in lakes that were at or near saturation with respect to dissolved oxygen (Figure 5.16). Rather than dissolved oxygen concentration predicting the periphyton diversity it is more likely that the high dissolved oxygen concentrations reflect the great abundance of photosynthetic organisms, including a healthy periphyton population. This relationship was also highly variable. Not all lakes with dissolved oxygen concentrations at or near 100% saturation had high periphyton diversity (Figure 5.16).

*Table 5.13 Periphyton diversity regressed against the environmental variables to test for significance. Dissolved oxygen and total nitrogen were significant predictors of periphyton diversity. The Shannon-wiener index was used to calculate diversity.*

| | **Std. Error** | **P-value** |
|---|---|---|
| TP | 1.28 | 0.11 |
| **TN** | **0.40** | **0.01** |
| **DO** | **0.01** | **0.05** |
| Sp.C | < 0.01 | 0.24 |
| Secchi | 0.14 | 0.86 |
| pH | 0.40 | 0.73 |

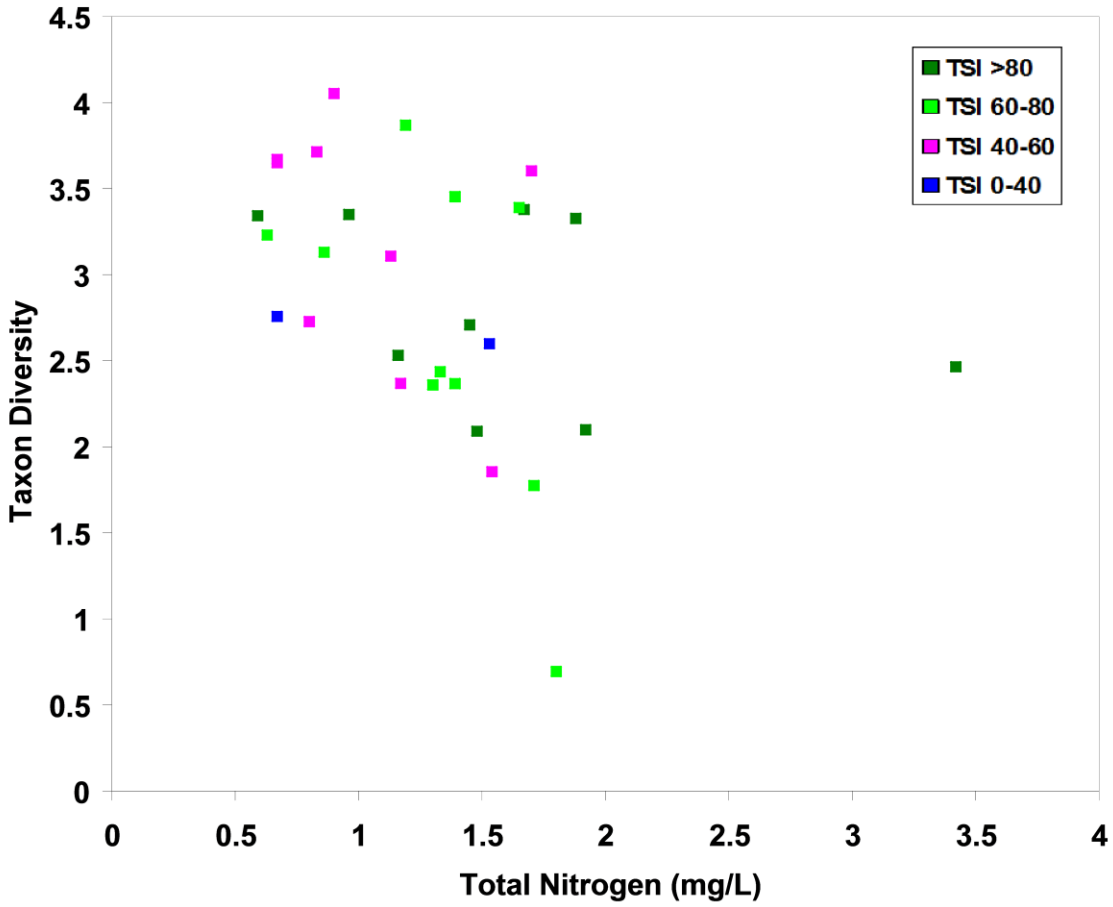



*Figure 5.1 Periphyton diversity as a function of total nitrogen in each of the thirty lakes sampled for periphyton in and around Riding Mountain National Park, MB.*

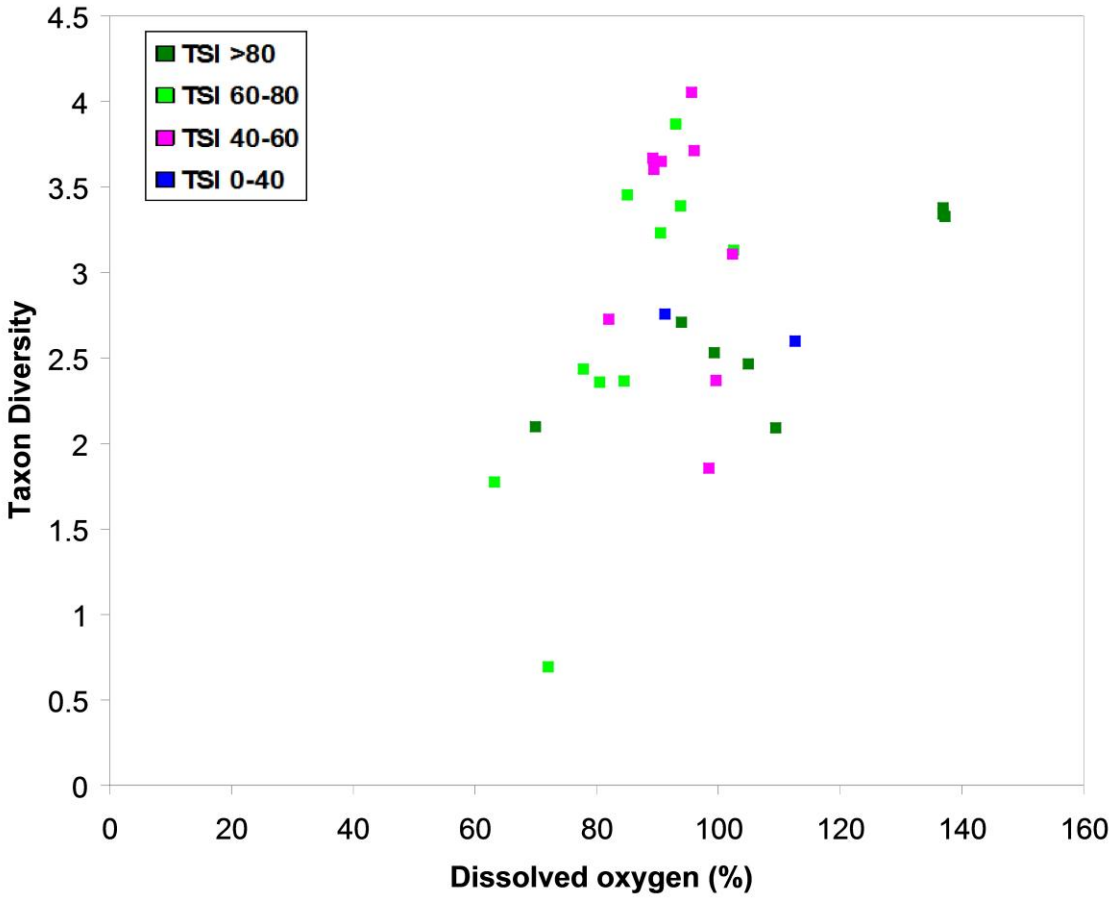



*Figure 5.16 Periphyton diversity as a function of dissolved oxygen in each of the thirty lakes sampled for periphyton in and around Riding Mountain National Park, MB.*

## Chapter 6: Discussion

The principal goal of my research has been to identify bioindicators that can be usefully and pragmatically employed as indicators of ecosystem change in Riding Mountain National Park lakes. If such indicators can be identified, it might be possible to identify lakes approaching an ecological threshold between nutrient states and mitigate against permanent, drastic changes in these ecosystems before they occur. My discussion, therefore, focuses solely on the practicalities of implementing macrophyte and periphyton indicators for this purpose. The results of the RDA and GLR identified three macrophyte species, *Chara* spp., *Ruppia maritima* and *Potamogeton richardsonii*, and one periphyton genus, *Cocconeis*, which have potential as indicators of lake trophic status and changing water quality for the shallow, prairie lakes of Manitoba. Species diversity of macrophytes and periphyton will also prove useful as indicators of changing water quality in lakes. This study presents evidence supporting the continued monitoring of aquatic macrophytes and recommends improvements to the periphyton sampling procedure.

### 6.1. Aquatic Macrophytes

The RDA indicated that Secchi depth, total phosphorus, total nitrogen and specific conductance most strongly influence the distributions of species. Of the seventeen submerged species recorded, seven (*Ceratophyllum demersum, Myriophyllum verticillatum, Potamogeton richardsonii, Potamogeton pectinatus, Chara* spp., *Ruppia maritima* and *Utricularia vulgaris)* were identified from the RDA as being possibly significantly related to one or more environmental variables. The GLR further revealed

three species, *Chara* spp., *Ruppia maritima* and *Potamogeton richardsonii,* had significant relationships with specific environmental variables. These three species have the most potential for managers as bioindicators of ecological changes in Riding Mountain National Park lakes.

The majority of *Chara* appeared to occur in mesotrophic and eutrophic lakes with a sharp decline in this species at very high or very low nutrient levels (Figure 5.8); although the lack of this species at low total phosphorus levels may have been due to the underrepresentation of oligotrophic lakes in the study. The distribution of *Chara* spp. was best described by total phosphorus. *Chara* spp. most often correspond with either mesotrophic (0.012-0.027 mg/L TP) or eutrophic (0.024-0.096 mg/L TP) levels (Figure 5.8). At extremely high or low total phosphorus levels *Chara* spp. were nearly absent (Figure 5.8). *Chara* spp. disappearance could serve as a useful indicator of total phosphorus levels increasing towards hypereutrophication. However, it is impossible to determine if the lack of *Chara* spp. at low total phosphorus levels (associated with oligotrophy) was a true relationship or due to the underrepresentation of lakes with low total phosphorus levels in this study.

*Chara* spp. presence might have been better explained if hardness was examined. *Chara* spp. are known to thrive in hard water environments (Van den Berg *et al.* 1999). This study did not measure hardness; thus, I am unable to assess its role in shaping the distribution of *Chara* spp.

*Potamogeton richardsonii* declined significantly with increasing specific conductance. This species proved to be highly intolerant of specific conductance values > 500 µS/cm. Disappearance of *Potamogeton richardsonii* from lakes with established

communities could alert to a change in specific conductance and, thus, increasing dissolved solids. Dissolved solids have many deleterious effects on freshwater biota including reduced dissolved oxygen solubility, and increased nutrients, salts and other ions (Fondriest Environmental 2014). *Potamogeton richardsonii* occurred in ten of the twenty lakes sampled in RMNP and its disappearance from lakes with previously established communities could indicate rising salinity due to changes to climate (increased evaporation) or other anthropogenic inputs.

*Ruppia maritima* occurrences were also best defined by specific conductance. *Ruppia maritima* is a well known halophile (Richardson 1980) and was absent from lakes with Sp.C < 1000 µS/cm. All lakes in this study that had Sp. C > 1000 µS/cm also had total phosphorus values > 27µg/L, which is within the eutrophic range according to the TSI index (Carlson and Simpson 1996; Wetzel 2001). While the presence of *Ruppia maritima* indicates a eutrophic condition, the species' absence does not necessarily indicate a lower trophic condition. There were several sample lakes that had total phosphorus values and TSI values higher than the eutrophic range that did not contain *Ruppia maritima*; thus, this species is a better indicator of lakes with high specific conductance than as a trophic state indicator.

Species diversity decreased significantly with increasing specific conductance. Most macrophyte species, with the exception of *Ruppia maritima,* do not tolerate high salinity. Therefore, it is expected that the majority of aquatic macrophyte species would be found in lakes with moderate to low specific conductance (Fondriest Environmental 2014; Mitchell and Prepas 2005). The majority of lakes whose specific conductance values were < 500 µS/cm had high species diversity values, with the exception of Lake 1

and 36 (Clear). The lack of species diversity in these two lakes may have been caused by other factors which were not taken into account, such as the sandy bottom of Lake 1 and the greater average depth and area of Clear Lake (36). It is worth stating that low diversity values do not necessarily equate to low abundance of macrophyte species. The lakes with low diversity may have had a high abundance of one or two dominant macrophyte species. In the end, macrophyte species diversity would be a better indicator of specific conductance rather than a defining characteristic of a particular trophic state.

No specific aquatic macrophyte species was indicative of a particular trophic state that could have been useful for managers in defining when a particular lake was in transition or had changed ecological states. Other studies have also had difficulty defining such ecological thresholds. For example, a study of Danish lakes was unable to quantify the lower nutrient threshold for aquatic plants citing an underrepresentation of oligotrophic lakes in their data set (Søndergaard *et al.* 2010). Identifying the upper limits for aquatic macrophyte species was possible, but Søndergaard *et al.* (2010) found that many macrophyte species spanned a broad ecological range; thus, making them unfit as indicators of ecologically important thresholds, such as those between clear water and turbid water states. The results from Jeppesen *et al.* (2000), Heegaard *et al.* (2001) and this study all agree that there is an upper threshold of aquatic macrophyte tolerance to hypereutrophication. At some combination of high nutrient levels and decreased light penetration in all three studies, submerged aquatic macrophytes all but disappear. However, this disappearance is gradual and the critical nutrient and light levels at which submerged aquatic macrophytes disappeared differed among studies and lakes within each study. While not all factors known to affect the growth of submerged aquatic

macrophytes were included in this study the variables that were considered, according to the ordination, explained 62% of the variation seen in the data. This is enough to consider the interpretation of the results to be reliable. Other studies are in agreement; Heegaard *et al.* (2001) examined aquatic macrophytes and their relationship to a large number of environmental factors, including: local chemistry, local nutrients, geographical features and lake physical characteristics. They found the most influential variables on the regional distribution of aquatic macrophytes were local chemical and nutrient composition of lakes. The lack of data for lakes considered oligotrophic may have influenced the conclusion that there was no significant statistical indication of lower ecological thresholds defined by aquatic macrophytes or their communities. Marked differences in water quality, such as the ability of light to penetrate the water column and increased phytoplankton biomass, often occur as oligotrophic lakes become eutrophied (Penning *et al.* 2008; Søndergaard *et al.* 2010).

### 6.2. Periphyton

Most eutrophic lakes were poorly differentiated from the mesotrophic lakes in the NMDS. This lack of clear differentiation may have arisen from the consistent depth the rods were placed in each lake; removing potential differences in light availability between mesotrophic and eutrophic lakes. Since periphyton are typically fixed to a substrate at or near the lake bottom; light needs to penetrate into the water column (Genkai-Kato *et al.* 2012; Hannson 1992; Rodusky *et al.* 2001). There may have been differences in the ability of light to penetrate the water column between mesotrophic and eutrophic lakes that were masked due to the consistent depth the periphyton sampling

rods were placed at. Also, many of the mesotrophic and eutrophic lakes possibly had such subtle differences on the TSI scale that these lakes looked similar based on taxon occurrences. Many of the hypereutrophic lakes and some of the eutrophic lakes had very few periphyton taxa. Hypereutrophic and eutrophic lakes can have low water transparency due to high periphyton biomass concentration (chlorophyll a). This may have accounted for the low number of species associated with these lakes. Taking a closer look at the environmental variables via CCA yields a more complete picture of the factors controlling periphyton taxon occurrences.

Out of the twenty-one taxa best described by the CCA only one taxon, *Cocconeis* sp., was identified by GLM as having a distribution well described by one of the environmental variables, dissolved oxygen (p-value < 0.05). It should be noted that dissolved oxygen is highly variable on a daily basis. The dissolved oxygen variability changes with changes in temperature, photosynthetic rates and time of day (USGS: Water Science School 2014; Wetzel 2001). This large variability makes the prediction of dissolved oxygen levels by *Cocconeis* sp. difficult and not entirely useful from a management stand point. However, supersaturation of dissolved oxygen most often occurs during an explosion of photosynthesis that accompanies a phytoplankton bloom (USGS: Water Science School 2014; Wetzel 2001). The secchi depth and chlorophyll a values recorded in Lakes 2 and 23 (0.1 m, 0.15 m; 176 µg/L, 77 µg/L respectively) also point towards an abundance of phytoplankton in these lakes. *Cocconeis* sp. was almost totally absent from lakes with dissolved oxygen levels >110%. The three lakes with dissolved oxygen levels >110% were Lake 2, 23 and 29. The absence (or nearly complete absence, <0.04% relative abundance) of *Cocconeis* sp. in the two

hypereutrophic lakes (2, 23) reiterates the point that in hypereutrophic lakes benthic flora (like periphyton) are most probably shaded out by blooms of phytoplankton. The reasons for *Cocconeis* sp. absence from the oligotrophic Lake 29 were unclear. Lake 29 had dissolved oxygen levels in the supersaturated range but had low submerged macrophyte abundance and diversity, a low chlorophyll a concentration of 3.0 µg/L, and the lowest overall periphyton relative abundance. It is clear that *Cocconeis* followed a unimodal distribution, being nearly absent from both highly unproductive and highly productive lakes. This unimodal distribution of periphyton with respect to environmental variables is well documented, although not for *Cocconeis* in particular (Genkai-Kato *et al.* 2012; Hannson 1992; Rodusky *et al.* 2001). While *Cocconeis* sp. absence may be useful in defining a lake tending toward hypereutrophy, the sampling effort is high and identifying periphyton taxa is time consuming; making it a less useful indicator for managers to integrate into a lake monitoring program.

Total nitrogen and dissolved oxygen were identified by the GLR as having the greatest effects on periphyton diversity. Total nitrogen is known to be a limiting factor for the growth of periphyton in lakes (Rodusky *et al.* 2012) and had a clear regulatory effect on the periphyton in this study. Periphyton diversity decreased as total nitrogen concentration increased. While one would assume that an increase in total nitrogen concentration would correlate with an increase in trophic status, it did not. In fact, no discernible pattern between the trophic status of lakes and total nitrogen concentration was seen in the data. This may have been a result of the Trophic Status Index improperly classifying the sample lakes or poor results of the periphyton taxa themselves due to possibility that the sampling design in this study did not obtain an accurate sampling of

the periphyton population. Regardless, the results of this relationship were highly variable, most likely due to the complex interactions between the nutrient levels required for growth, competition with phytoplankton for resources, and light requirements (Genkai-Kato *et al.* 2012; Hannson 1992; Roduíguez *et al.* 2012; Rodusky *et al.* 2001).

Similar results were observed between dissolved oxygen and periphyton diversity as with dissolved oxygen and *Cocconeis* sp. Periphyton taxon diversity peaked at or near 100% dissolved oxygen saturation and declined when dissolved oxygen became supersaturated, regardless of the lake's trophic status. The declining periphyton diversity at supersaturated dissolved oxygen levels were most likely due to shade-out effects from phytoplankton blooms that often coincide with supersaturated dissolved oxygen levels (USGS: Water Science School 2014; Wetzel 2001). Just like with the *Cocconeis* sp. relationship with dissolved oxygen, the variability of dissolved oxygen levels on a daily basis makes the prediction of dissolved oxygen levels by periphyton taxon diversity difficult and not useful for the managers at Riding Mountain National Park.

## 7: Conclusion

In future studies, RMNP should compare future monitoring data with previously collected submerged macrophyte and periphyton data (from this study; Kooyman and Hutchinson 1979; White 2012) in order to identify clear benchmarks for changes to the lakes' ecological integrity. This is to ensure the RMNP lakes are monitored for changes from their natural state. The lakes in RMNP spanned the continuum from oligotrophic to hypereutrophic, although most are mesotrophic. Preserving ecological integrity means maintaining the lakes' natural state, whether that is oligotrophic, eutrophic or anywhere in between.

Incorporating submerged macrophyte surveys for species diversity, and the presence of the indicator species *Chara* spp., *Ruppia maritima* and *Potamogeton richardsonii* would add to the understanding of the ecological state of RMNP lakes and alert managers to potential changes to the lake environment. Periphyton taxa and diversity monitoring are not recommended. The sampling effort and time required for periphyton identification is high and the results from this study with respect to periphyton were poor. The periphyton sampling method should be revised and the possibility of periphyton monitoring could then be reconsidered if better results were obtained. Possible revisions should include sampling the actual substrate which the periphyton are growing on or place the artificial substrate in a way that better replicates the periphyton benthic environment, such as on the lake bed in the littoral area. Scraping a defined and uniform area for periphyton is also recommended to ensure more reliable and replicable results.

The management of aquatic resources and maintenance of ecological integrity is of utmost importance to RMNP and Parks Canada. The findings of this study will be assessed to see if they can be incorporated into the RMNP ecological integrity monitoring program in conjunction with already established monitoring practices.

Incorporating these monitoring recommendations into the established aquatic monitoring program of RMNP will increase the understanding of these complex aquatic systems. Detection of changes in the ecological integrity of a lake will allow managers to potentially mitigate or reverse significant biological changes. Once the ecological integrity of a lake ecosystem has been compromised it is very difficult to return a lake to its original pre-disturbance condition. Emphasis on monitoring multiple water quality parameters to gain an overall picture of the complex aquatic system that is a lake is important to shift the focus of preserving the lakes from issues pertaining chiefly to human preference (swimming, drinking, and fishing) to the broader goal of preserving ecological integrity.

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## Appendix I: Permission Correspondence

*Figure A1.1 Permission correspondence from Parks Canada to use Figure 3.1.*

From: Heather Gray <heather_joy13@hotmail.com>
To: "information@pc.gc.ca" <information@pc.gc.ca>
Date: 01/05/2013 03:15 PM
Subject: Permission

Hi,

I am a grad student at University of British Columbia Okanagan wanting to use the regional map of SW Manitoba, located on the Parks Canada website, as a figure in my MSc thesis. I need written permission, via email, to submit along with my thesis. If you would allow me to use the map that would be great.

Here is the link

http://www.pc.gc.ca/pn-np/mb/riding/visit/visit13.aspx

Thank you!

Heather Gray

Hi Heather,

The original version of that map comes from our 1996 management plan, so you may wish to cite it from there and use that version as it is much more legible (attached). There is an updated and simplified version in our current management plan if you would rather avoid the colours.

(See attached file: Manage Plan Eng. Art.ps2.pdf)

Cheers, Sean

Sean Frey
Geomatics Coordinator | Coordonnateur géomatique
Parks Canada | Parcs Canada
Riding Mountain National Park of Canada | Parc national du Canada du Mont-Riding
135 Wasagaming Drive, Wasagaming, Manitoba R0J 2H0
Office | Bureau: 204 848-7244
Fax | Télécopieur : 204 848-2596
E-mail | Courriel : Sean.Frey@pc.gc.ca
loc: 50.658415, -99.971439
Live well | Bien vivre
http://www.parkscanada.gc.ca/riding

**Appendix 2: Additional Data**

*Table A2.1 Aquatic macrophyte species presence/absence for each sample lake. Data is a compilation from 2010 and 2011 sampling years.*

| **Lake** | *Bidens beckii* | *Chara* spp. | *Ceratophyllum demersum* | *Eriocaulon aquaticum* | *Elodea canadensis* | *Elodea nuttalli* | *Myriophyllum verticillatum* | *Najas flexilis* | *Nitella* |
|---|---|---|---|---|---|---|---|---|---|
| 1 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 3 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 4 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 5 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 6 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 7 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 8 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 9 | 0 | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 10 | 0 | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 11 | 0 | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 13 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| 15 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 16 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 18 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 19 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 20 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 21 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| 22 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 23 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 24 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| **Lake** | *Bidens beckii* | *Chara* spp. | *Ceratophyllum demersum* | *Eriocaulon aquaticum* | *Elodea canadensis* | *Elodea nuttalli* | *Myriophyllum verticillatum* | *Najas flexilis* | *Nitella* |
|---|---|---|---|---|---|---|---|---|---|
| 25 | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| 26 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 27 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 28 | 0 | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 29 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 30 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 1 | 1 |
| 31 | 0 | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 0 |
| 33 | 0 | 1 | 1 | 0 | 0 | 1 | 1 | 0 | 0 |
| 34 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 35 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 36 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 38 | 1 | 1 | 1 | 1 | 0 | 0 | 1 | 1 | 0 |
| 39 | 1 | 1 | 1 | 0 | 0 | 0 | 1 | 1 | 0 |
| 40 | 1 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 1 |
| 43 | 0 | 1 | 0 | 0 | 0 | 1 | 1 | 0 | 0 |
| 44 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 45 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 46 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 48 | 0 | 1 | 0 | 1 | 0 | 1 | 1 | 0 | 0 |
| 49 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 50 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 51 | 0 | 1 | 0 | 0 | 1 | 1 | 1 | 0 | 0 |
| 52 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 |

| **Lake** | *Potamogeton pectinatus* | *Potamogeton pusillus* | *Potamogeton richardsonii* | *Potamogeton robbinsii* | *Potamogeton zosteriformis* | *Ruppia maritima* | *Utricularia vulgaris* | *Valisneria americana* |
|---|---|---|---|---|---|---|---|---|
| 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 3 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 0 |
| 4 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 7 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 8 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 9 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 10 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 11 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 13 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 15 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| 16 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 18 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 |
| 19 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |
| 20 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| 21 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |
| 22 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| 23 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 24 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 1 |
| 25 | 1 | 0 | 0 | 0 | 1 | 0 | 0 | 0 |
| 26 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |
| 27 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 28 | 1 | 0 | 1 | 1 | 0 | 0 | 1 | 0 |
| 29 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |

| **Lake** | *Potamogeton pectinatus* | *Potamogeton pusillus* | *Potamogeton richardsonii* | *Potamogeton robbinsii* | *Potamogeton zosteriformis* | *Ruppia maritima* | *Utricularia vulgaris* | *Valisneria americana* |
|---|---|---|---|---|---|---|---|---|
| 30 | 1 | 0 | 0 | 0 | 1 | 0 | 1 | 0 |
| 31 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 33 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 34 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 1 |
| 35 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| 36 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 38 | 1 | 0 | 1 | 0 | 1 | 0 | 1 | 0 |
| 39 | 0 | 0 | 1 | 1 | 0 | 0 | 1 | 1 |
| 40 | 1 | 1 | 1 | 1 | 0 | 0 | 1 | 0 |
| 43 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |
| 44 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |
| 45 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |
| 46 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 48 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 0 |
| 49 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| 50 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 0 |
| 51 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| 52 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |

*Table A2.2 Raw periphyton counts from the 30 sample lakes. Counts are given as viable cell counts per sample.*

| | Lake | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cyanophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Aphanocapsa delicatissima W. and G.S. West* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aphanocapsa sp.* | 1000 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aphanothece sp.* | 542 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Pelogloea sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Chroococcus limneticus Lemmermann* | 0 | 11 | 0 | 65 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 32 | 0 | 0 |
| *Chroococcus turgidus (Kutzing) Nageli* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Chroococcus tenuis.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 17 | 0 | 0 |
| *Chroococus minutus (Kutzing) Nageli* | 0 | 71 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 284 | 0 | 0 |
| *Chroococcus dispersus (Keissl.) Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 329 | 0 | 0 |
| *Chroococcus sp.* | 49 | 0 | 25 | 0 | 0 | 137 | 0 | 0 | 0 | 12 | 0 | 0 | 69 | 0 | 0 |
| *Coelosphaerium Naegelianum Unger* | 0 | 0 | 61 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Dactylococcopsis linearis Geitler* | 108 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Dactylococcopsis Smithii R. and F. Chodat* | 0 | 0 | 991 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gomphosphaeria aponina Kuetzing* | 157 | 0 | 0 | 0 | 167 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 411 | 0 | 611 |
| *Gomphosphaeria lacustris Chodat* | 35 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 118 |
| *Gomphosphaeria natus Komarek and Hindak* | 10 | 0 | 114 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 96 | 0 | 38 |
| *Merismopedia tenuissima Lemmermann* | 29 | 12 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 67 |
| *Merismopedia minima (Ehrenberg) Kutzing* | 187 | 69 | 0 | 0 | 279 | 95 | 0 | 0 | 0 | 0 | 0 | 0 | 237 | 0 | 58 |
| *Merismopedia punctata (Meyen)* | 64 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 94 | 0 | 91 |
| *Merismopedia glauca (Ehrenberg) Kutzing* | 103 | 0 | 458 | 0 | 69 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 61 | 0 | 113 |

| | Lake | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cyanophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Synechococcus sp.* | 0 | 0 | 361 | 0 | 0 | 301 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Rhabdogloea lineare Schmidle and Lauterborn* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Rhabdogloea Gorskii Woloszynska* | 0 | 0 | 17 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Anabaena sp.* | 28 | 197 | 56 | 184 | 114 | 51 | 0 | 0 | 0 | 33 | 0 | 0 | 287 | 90 | 337 |
| *Limnothrix redekei (Van Goor) Meffert* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Planktolyngbya limnetica* | 0 | 0 | 63 | 0 | 0 | 109 | 0 | 0 | 0 | 71 | 0 | 0 | 0 | 0 | 0 |
| *Lyngbya Birgei G.M. Smith* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 275 |
| *Lyngbya limnetica Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Lyngbya sp.* | 127 | 0 | 92 | 94 | 0 | 144 | 69 | 0 | 0 | 501 | 0 | 0 | 0 | 312 | 56 |
| *Oscillatoria geminata (Meneghini) Gomont* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Oscillatoria limnetica Lemmermann* | 0 | 129 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Oscillatoria tenuis Agardh* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 68 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Oscillatoria sp.* | 55 | 608 | 77 | 0 | 0 | 47 | 0 | 101 | 219 | 331 | 0 | 0 | 81 | 84 | 28 |
| *Pseudoanabaena spp.* | 0 | 0 | 102 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 117 | 0 | 638 |
| *Anabaena lavendrii Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cyanophyceae sp. A [unknown-cells]* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Anabaena spiroides Klebs* | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cyanodictyon sp.* | 1500 | 1500 | 0 | 0 | 2544 | 0 | 0 | 0 | 0 | 2500 | 0 | 0 | 1500 | 0 | 0 |
| *Pseudoanabaena constricta (Szafer) Lauterborn* | 157 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Pseudoanabaena spp.* | 621 | 177 | 384 | 0 | 0 | 109 | 0 | 0 | 0 | 0 | 0 | 314 | 341 | 224 | 71 |
| *Gloeocapsa sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gleotrichia sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 568 | 0 | 0 | 0 | 0 |
| *Anabaena circinalis Rabenhorst* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | | | |
| *Anabaena flos-aquae (Lyngbye) Brebisson* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aphanizomenon flos-aquae Ralfs* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nostoc sp.* | 0 | 0 | 0 | 450 | 0 | 0 | 357 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cyanophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Planktothrix sp.* | 331 | 0 | 63 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 178 | 0 | 0 | 0 |
| *Phormidium sp.* | 79 | 0 | 177 | 78 | 0 | 17 | 74 | 0 | 0 | 0 | 0 | 279 | 0 | 0 | 0 |
| *Rhabdoderma irregulare* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 408 | 51 | 86 |
| *Rhabdoderma sp.* | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Rhabdogloea sp.* | 0 | 0 | 15 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Spirulina sp.* | 0 | 14 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 17 |

| | Lake | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cyanophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Aphanocapsa delicatissima W. and G.S. West* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aphanocapsa sp.* | 0 | 0 | 0 | 0 | 785 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 300 |
| *Aphanothece sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 450 |
| *Pelogloea sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Chroococcus limneticus Lemmermann* | 18 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 56 | 0 | 0 | 61 | 33 |
| *Chroococcus turgidus (Kutzing) Nageli* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 19 | 0 | 0 |
| *Chroococcus tenuis.* | 0 | 0 | 0 | 173 | 39 | 0 | 0 | 0 | 88 | 0 | 0 | 0 | 0 | 0 | 6 |
| *Chroococus minutus (Kutzing) Nageli* | 49 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 37 | 0 | 0 | 0 | 38 | 0 | 0 |
| *Chroococcus dispersus (Keissl.) Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 821 | 0 | 0 | 0 | 521 | 509 | 0 |
| *Chroococcus sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 43 | 0 | 56 | 0 | 114 | 0 |  | 71 | 25 |
| *Coelosphaerium Naegelianum Unger* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Dactylococcopsis linearis Geitler* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Dactylococcopsis Smithii R. and F. Chodat* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gomphosphaeria aponina Kuetzing* | 0 | 0 | 900 | 0 | 1750 | 0 | 0 | 0 | 147 | 0 | 203 | 0 | 750 | 0 | 0 |
| *Gomphosphaeria lacustris Chodat* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Cyanophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Gomphosphaeria natus Komarek and Hindak* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Merismopedia tenuissima Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 578 | 0 | 0 | 0 | 0 | 0 | 10 |
| *Merismopedia minima (Ehrenberg) Kutzing* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 91 | 1025 | 0 | 96 | 261 | 59 | 294 | 61 |
| *Merismopedia punctata (Meyen)* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 276 | 214 | 0 | 0 | 0 | 167 | 0 | 0 |
| *Merismopedia glauca (Ehrenberg) Kutzing* | 0 | 0 | 212 | 0 | 61 | 0 | 0 | 36 | 0 | 0 | 0 | 0 | 0 | 36 | 14 |
| *Synechococcus sp.* | 0 | 0 | 1200 | 0 | 527 | 0 | 0 | 0 | 661 | 0 | 0 | 0 | 117 | 0 | 0 |
| *Rhabdogloea lineare Schmidle and Lauterborn* | 0 | 0 | 0 | 0 | 12 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 12 | 11 | 0 |
| *Rhabdogloea Gorskii Woloszynska* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Anabaena sp.* | 0 | 0 | 48 | 46 | 39 | 1433 | 0 | 117 | 48 | 12 | 88 | 51 | 56 | 37 | 91 |
| *Limnothrix redekei (Van Goor) Meffert* | 0 | 0 | 95 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 375 |
| *Planktolyngbya limnetica* | 0 | 0 | 0 | 0 | 314 | 885 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 103 |
| *Lyngbya Birgei G.M. Smith* | 0 | 0 | 189 | 277 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 412 | 0 | 294 |
| *Lyngbya limnetica Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 51 | 0 | 0 | 0 | 0 | 601 | 0 | 63 |
| *Lyngbya sp.* | 0 | 0 | 0 | 0 | 306 | 574 | 0 | 0 | 0 | 0 | 145 | 0 | 139 | 68 | 118 |
| *Oscillatoria geminata (Meneghini) Gomont* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Oscillatoria limnetica Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Oscillatoria tenuis Agardh* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Oscillatoria sp.* | 0 | 0 | 374 | 0 | 56 | 0 | 0 | 23 | 0 | 0 | 0 | 0 | 0 | 0 | 196 |
| *Pseudoanabaena spp.* | 0 | 0 | 118 | 0 | 94 | 0 | 0 | 0 | 97 | 0 | 0 | 0 | 119 | 0 | 227 |
| *Anabaena lavendrii Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10 |
| *Cyanophyceae sp. A [unknown-cells]* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 19 |
| *Anabaena spiroides Klebs* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 69 |
| *Cyanodictyon sp.* | 0 | 0 | 0 | 0 | 500 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1000 |
| *Pseudoanabaena constricta* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

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| **Cyanophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *(Szafer) Lauterborn* | | | | | | | | | | | | | | | |
| *Pseudoanabaena spp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 56 | 0 | 0 | 154 | 0 | 0 | 816 | 477 |
| *Gloeocapsa sp.* | 0 | 0 | 25 | 0 | 29 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 12 |
| *Gleotrichia sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Anabaena circinalis Rabenhorst* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | | | |
| *Anabaena flos-aquae (Lyngbye) Brebisson* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 33 | 55 |
| *Aphanizomenon flos-aquae Ralfs* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nostoc sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Planktothrix sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Phormidium sp.* | 0 | 0 | 0 | 0 | 436 | 256 | 0 | 203 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Rhabdoderma irregulare* | 0 | 0 | 0 | 25 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 77 | 0 | 79 |
| *Rhabdoderma sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Rhabdogloea sp.* | 0 | 0 | 0 | 32 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 15 | 3 |
| *Spirulina sp.* | 0 | 0 | 12 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | | | | | | | | | | | | 0 | 0 | 0 |

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| **Chlorophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| Pyramidomonas tetrarhynchus Schmarda | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 18 |
| Carteria spp. | 0 | 13 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 |
| Chlamydomonas spp. | 45 | 49 | 11 | 0 | 108 | 0 | 0 | 0 | 0 | 0 | 0 | 71 | 0 | 0 | 3 |
| Chlorogonium maximum Skuja | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Coenochloris planctonica (W. and G.S. West) Hindak | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Gloeococcus Schrocteri (Chodat) Lemmermann | 91 | 0 | 0 | 0 | 111 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Sphaerocystis schroeteri Chodat | 0 | 3 | 0 | 0 | 76 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Pediastrum duplex Meyen | 2 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 11 | 0 | 0 | 0 | 0 | 0 | 2 |
| Pediastrum Boryanum (Turpin) Meneghini | 1 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Pediastrum tetras (Ehrenberg) Ralfs | 6 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

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| **Chlorophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| Pediastrum sp. | 11 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 9 |
| Oocystis submarina v. variabilis Skuja | 0 | 51 | 53 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 53 | 0 | 0 |
| Oocystis lacustris Chodat | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Oocystis Borgei Snow | 3 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Oocystis crassa Wittrock | 16 | 7 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Oocystis sp. | 29 | 22 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 38 |
| Chodatella sp. Lemmermann | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Tetraedron minimum (Brunow) Hansgrig | 6 | 0 | 68 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Tetraedron sp. | 11 | 0 | 6 | 0 | 17 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 17 | 0 | 14 |
| Scenedesmus acuminatus (Lagerheim) Chodat | 2 | 3 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 17 | 0 | 0 | 6 |
| Scenedesmus arcuatus Lemmermann | 4 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| Scenedesmus balatonicus Hortobagyi | 0 | 27 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 20 |
| Scenedesmus ecornis (Ralfs) Chodat | 17 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 |
| Scenedesmus quadricauda (Turp.) Brebisson | 0 | 39 | 25 | 0 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 51 | 102 | 0 | 17 |
| Scenedesmus spinosus Chodat | 0 | 0 | 0 | 0 | 27 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 |
| Scenedesmus denticulatus Lagerhiem | 0 | 11 | 11 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 11 | 0 | 0 | 0 |
| Scenedesmus brevispina (G.M Smith) Chodat | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Scenedesmus sp. | 28 | 3 | 5 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 9 | 0 | 0 | 6 |
| Dictyosphaerium simplex Sukja | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Monoraphidium komarkovae (Nyg.) Komarkova-Legnerova | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Monoraphidium contortum (Thuret in Brebisson) Komarkova-Legnerova | 75 | 17 | 39 | 0 | 93 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Monoraphidium sp. a | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Monoraphidium minutum (Naegeli) Komarkova-Legnerova | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Crucigenia quadrata Morren | 0 | 3 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Crucigenia rectangularis (A. Braun) Gay | 0 | 0 | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Crucigenia spp. | 9 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Coelastrum cambricum Archer | 1 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

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| **Chlorophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| Selenastrum minutum (Naegeli) Collins | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Selenastrum sp. | 12 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Kirchneriella lunaris (Kirchner) Moebius | 0 | 0 | 16 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Ankistrodesmus braunii (Naegeli) Collins | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Ankistrodesmus falcatus (Corda) Ralfs | 27 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 38 | 0 | 0 | 0 |
| Ankistrodesmus sp. | 39 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Quadrigula closterioides (Bohlin) Printz | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Elakatothrix gelatinosa Willen | 3 | 0 | 0 | 0 | 11 | 0 | 0 | 0 | 0 | 0 | 0 | 10 | 0 | 0 | 0 |
| Elakatothrix genevensis Hindak | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 |
| Planctonema Lauterborni Schmidle | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Small greens | 91 | 0 | 36 | 0 | 29 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 55 |
| Scourfieldia cordiformis Takeda | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Chlorella mucosa Kors. | 11 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 |
| Ankrya judai (G.M. Smith) Fott | 0 | 0 | 0 | 0 | 15 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Mougeotia sp. sm | 0 | 59 | 307 | 66 | 55 | 0 | 0 | 0 | 0 | 0 | 151 | 56 | 211 | 0 | 208 |
| Mougeotia sp. lg. | 0 | 308 | 112 | 28 | 19 | 0 | 0 | 0 | 0 | 0 | 74 | 199 | 496 | 0 | 116 |
| Chlorogonium sp. | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Schroederia setigera (Schroder) Lemmermann | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 17 | 0 | 0 | 0 |
| Spirogyra sp. | 0 | 0 | 0 | 0 | 105 | 0 | 0 | 0 | 0 | 0 | 31 | 0 | 0 | 0 | 473 |
| Westella sp. | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Closteriopsis longissima Lemmermann | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Zygnema sp. | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Chloricystis sp. | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 56 | 0 | 0 |
| Bulbochaetae sp. | 0 | 0 | 53 | 0 | 321 | 0 | 0 | 0 | 0 | 0 | 48 | 212 | 0 | 0 | 51 |
| Oedogonium spp. | 115 | 1182 | 168 | 0 | 293 | 0 | 103 | 0 | 418 | 0 | 197 | 364 | 622 | 0 | 403 |
| Stigeoclonium sp. | 0 | 0 | 0 | 1886 | 0 | 0 | 0 | 0 | 0 | 73 | 0 | 689 | 0 | 0 | 0 |
| Pandorina morum (Mueller) Bory | 16 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Micractinium pusillum Fresen [cells] | 24 | 0 | 13 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Coleochaetae sp. | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

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| **Chlorophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Pyramidomonas tetrarhynchus Schmarda* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Carteria spp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Chlamydomonas spp.* | 0 | 67 | 0 | 0 | 0 | 103 | 0 | 11 | 0 | 0 | 57 | 0 | 41 | 0 | 17 |
| *Chlorogonium maximum Skuja* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Coenochloris planctonica (W. and G.S. West) Hindak* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gloeococcus Schrocteri (Chodat) Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Sphaerocystis schroeteri Chodat* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 59 |
| *Pediastrum duplex Meyen* | 0 | 1 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 3 | 1 |
| *Pediastrum Boryanum (Turpin) Meneghini* | 0 | 0 | 0 | 0 | 0 | 8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 |
| *Pediastrum tetras (Ehrenberg) Ralfs* | 0 | 0 | 0 | 0 | 0 | 17 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| *Pediastrum sp.* | 0 | 0 | 0 | 0 | 0 | 20 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 11 |
| *Oocystis submarina v. variabilis Skuja* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 30 | 0 | 0 |
| *Oocystis lacustris Chodat* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 19 |
| *Oocystis Borgei Snow* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| *Oocystis crassa Wittrock* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| *Oocystis sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Chodatella sp. Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tetraedron minimum (Brunow) Hansgrig* | 0 | 59 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 |
| *Tetraedron sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 15 | 0 | 1 | 0 | 0 |
| *Scenedesmus acuminatus (Lagerheim) Chodat* | 0 | 2 | 14 | 0 | 15 | 10 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 3 |
| *Scenedesmus arcuatus Lemmermann* | 0 | 10 | 1 | 0 | 3 | 0 | 0 | 0 | 29 | 0 | 0 | 0 | 0 | 0 | 1 |
| *Scenedesmus balatonicus Hortobagyi* | 0 | 0 | 4 | 0 | 0 | 47 | 0 | 0 | 11 | 0 | 0 | 0 | 8 | 0 | 0 |
| *Scenedesmus ecornis (Ralfs) Chodat* | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Scenedesmus quadricauda (Turp.) Brebisson* | 0 | 39 | 61 | 0 | 41 | 59 | 12 | 0 | 0 | 0 | 0 | 0 | 29 | 19 | 24 |
| *Scenedesmus spinosus Chodat* | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 9 | 0 | 0 | 0 | 11 | 0 | 33 |
| *Scenedesmus denticulatus Lagerhiem* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

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| **Chlorophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Scenedesmus brevispina (G.M Smith) Chodat* | 0 | 15 | 0 | 0 | 3 | 19 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 |
| *Scenedesmus sp.* | 0 | 0 | 0 | 0 | 15 | 38 | 0 | 0 | 13 | 0 | 29 | 0 | 0 | 0 | 0 |
| *Dictyosphaerium simplex Sukja* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Monoraphidium komarkovae (Nyg.) Komarkova-Legnerova* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Monoraphidium contortum (Thuret in Brebisson) Komarkova-Legnerova* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 37 | 0 | 0 | 0 | 14 | 0 | 22 |
| *Monoraphidium sp. a* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 68 | 0 | 0 | 0 | 0 |
| *Monoraphidium minutum (Naegeli) Komarkova-Legnerova* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Crucigenia quadrata Morren* | 0 | 0 | 0 | 0 | 2 | 94 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 32 |
| *Crucigenia rectangularis (A. Braun) Gay* | 0 | 0 | 0 | 0 | 0 | 32 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Crucigenia spp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 15 | 0 | 0 |
| *Coelastrum cambricum Archer* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 |
| *Selenastrum minutum (Naegeli) Collins* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Selenastrum sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Kirchneriella lunaris (Kirchner) Moebius* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Ankistrodesmus braunii (Naegeli) Collins* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Ankistrodesmus falcatus (Corda) Ralfs* | 0 | 0 | 0 | 0 | 811 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Ankistrodesmus sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Quadrigula closterioides (Bohlin) Printz* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Elakatothrix gelatinosa Willen* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Elakatothrix genevensis Hindak* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Planctonema Lauterborni Schmidle* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Small greens* | 0 | 0 | 23 | 0 | 58 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 91 | 0 | 0 |
| *Scourfieldia cordiformis Takeda* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Chlorella mucosa Kors.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Ankrya judai (G.M. Smith) Fott* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Mougeotia sp. sm* | 0 | 0 | 298 | 0 | 33 | 84 | 0 | 51 | 0 | 0 | 12 | 163 | 118 | 0 | 96 |
| *Mougeotia sp. lg.* | 0 | 0 | 103 | 0 | 0 | 396 | 0 | 78 | 44 | 0 | 77 | 37 | 71 | 0 | 17 |
| *Chlorogonium sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Schroederia setigera (Schroder)* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

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| **Chlorophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Lemmermann* | | | | | | | | | | | | | | | |
| *Spirogyra sp.* | 0 | 0 | 0 | 667 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 423 | 0 | 53 |
| *Westella sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Closteriopsis longissima Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Zygnema sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 37 |
| *Chloricystis sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Bulbochaetae sp.* | 0 | 0 | 119 | 0 | 117 | 127 | 16 | 0 | 183 | 0 | 47 | 0 | 105 | 0 | 112 |
| *Oedogonium spp.* | 0 | 294 | 203 | 1674 | 603 | 264 | 59 | 674 | 337 | 57 | 86 | 74 | 91 | 67 | 193 |
| *Stigeoclonium sp.* | 0 | 0 | 658 | 0 | 0 | 578 | 0 | 167 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Pandorina morum (Mueller) Bory* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| *Micractinium pusillum Fresen [cells]* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 39 | 0 | 39 |
| *Coleochaetae sp.* | 0 | 0 | 0 | 0 | 0 | 122 | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | **Lake** | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Desmidaceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Closterium kuetzingii Brebisson* | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| *Closterium pronum Brebisson* | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Closterium sp.* | 2 | 0 | 23 | 0 | 11 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| *Cosmarium depressum* | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cormarium depressum v. achondrum (Boldt) W. And G.S. West* | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cosmarium humile* | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 |
| *Cosmarium trilobulatum* | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cosmarium sp.* | 5 | 3 | 19 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 3 |
| *Euastrum sp.* | 1 | 0 | 3 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Micrasterias sp. (dentriculata)* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Xanthidium sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurastrum dilatum* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurodesmus extensus (Andersson) Teiling* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Arthrodesmus octocornis (Ehrenberg and Ralfs) Arch.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Desmidaceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Staurodesmus cuspidatus (Brebisson and Ralfs) Teiling* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurodesmus paradoxum Meyen* | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurodesmus bullardii G.M. Smith* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurodesmus sp.* | 9 | 2 | 17 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 |
| *Teilingia granutatum (Roy and Biss.) Bour. and Comp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Spondylosium planum (Wolle) W. and G.S. West* | 0 | 0 | 0 | 0 | 59 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Small greens* | 0 | 69 | 91 | 0 | 21 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Scourfieldia cordiformis Takeda* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Chlorella mucosa Kors.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Ankrya judai (G.M. Smith) Fott* | 16 | 27 | 0 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Bambusina brebissonii Kutzing* | 0 | 0 | 0 | 0 | 108 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gonatozygon sp.* | 0 | 0 | 3 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Desmidaceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Closterium kuetzingii Brebisson* | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Closterium pronum Brebisson* | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Closterium sp.* | 0 | 0 | 7 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 |
| *Cosmarium depressum* | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1 | 0 |
| *Cormarium depressum v. achondrum (Boldt) W. And G.S. West* | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cosmarium humile* | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 |
| *Cosmarium trilobulatum* | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cosmarium sp.* | 0 | 0 | 3 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 9 | 0 | 0 | 7 | 11 |
| *Euastrum sp.* | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 7 |
| *Micrasterias sp. (dentriculata)* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| *Xanthidium sp.* | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurastrum dilatum* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Desmidaceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Staurodesmus extensus (Andersson) Teiling* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 |
| *Arthrodesmus octocornis (Ehrenberg and Ralfs) Arch.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurodesmus cuspidatus (Brebisson and Ralfs) Teiling* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurodesmus paradoxum Meyen* | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurodesmus bullardii G.M. Smith* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Staurodesmus sp.* | 0 | 0 | 3 | 0 | 3 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |
| *Teilingia granutatum (Roy and Biss.) Bour. and Comp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 51 |
| *Spondylosium planum (Wolle) W. and G.S. West* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 84 |
| *Small greens* | 0 | 0 | 61 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 56 | 0 |
| *Scourfieldia cordiformis Takeda* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Chlorella mucosa Kors.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Ankrya judai (G.M. Smith) Fott* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Bambusina brebissonii Kutzing* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 131 |
| *Gonatozygon sp.* | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 |

| | Lake | | | | | | | | | | | | | | |
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| **Euglenophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Euglena acus Ehrenberg* | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Euglena variabilis* | 7 | 0 | 0 | 0 | 17 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Euglena ehrenbergii* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Euglena gracilus* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Phacus caudata Hubner* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Phacus mirabilis Pochmann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Phacus sp.* | 0 | 13 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelononas armata (Ehrenberg)* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelomonas cylindrica Ehrenberg* | 0 | 0 | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelomonas planktonica Swirenko* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Euglenophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Ehrenberg* | | | | | | | | | | | | | | | |
| *Trachelomonas volvocina Ehrenberg* | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelomonas hispida (Perty) Stein* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelomonas spp.* | 0 | 0 | 0 | 0 | 29 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Astasia sp.* | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Euglenophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Euglena acus Ehrenberg* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Euglena variabilis* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 |
| *Euglena ehrenbergii* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Euglena gracilus* | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Phacus caudata Hubner* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Phacus mirabilis Pochmann* | 0 | 17 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Phacus sp.* | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelononas armata (Ehrenberg)* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelomonas cylindrica Ehrenberg* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelomonas planktonica Swirenko Ehrenberg* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelomonas volvocina Ehrenberg* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelomonas hispida (Perty) Stein* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Trachelomonas spp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Astasia sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Bacillariophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Achnanthes flexella* | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Achnanthes inflata* | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Achnanthes lanceolata* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Achnanthes minutissima* | 99 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 233 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Bacillariophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Achnanthes sp.* | 0 | 58 | 117 | 0 | 101 | 0 | 47 | 0 | 0 | 0 | 0 | 0 | 41 | 0 | 0 |
| *Aulacoseira granulata (Ehrenberg) Simonsen* | 0 | 0 | 0 | 0 | 56 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira islandica (O. Muller) Simonsen* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira italica (Ehrenberg) Simonsen* | 101 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira distans (Ehrenberg) Simonsen* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira sp.* | 0 | 0 | 0 | 0 | 91 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cyclotella stelligera Cleve and Grunow* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cyclotella pseudostelligera* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cyclotella sp.* | 0 | 0 | 0 | 0 | 111 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 72 | 0 | 0 |
| *Rhopalodia gibba (ehrenb.) O. Mull.* | 17 | 19 | 0 | 0 | 45 | 0 | 0 | 47 | 0 | 0 | 37 | 0 | 136 | 0 | 0 |
| *Rhizosolenia eriense H.L. Smith* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tabellaria fenestrata (Lyngbye) Kutzing* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tabellaria flocculosa (Roth) Kutzing* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Stauroneis anceps Ehrenberg* | 0 | 0 | 9 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 9 | 0 | 0 |
| *Synedra acus Kutzing* | 28 | 0 | 37 | 0 | 51 | 6 | 16 | 0 | 0 | 0 | 0 | 0 | 79 | 32 | 421 |
| *Synedra acus v. radians (Kutzing) Hustedt* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Synedra ulna (Nitzsch) Ehrenberg* | 41 | 0 | 11 | 0 | 13 | 47 | 0 | 0 | 0 | 0 | 0 | 0 | 20 | 71 | 264 |
| *Stephanodiscus sp.* | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 |
| *Anomoeoneis serians* | 0 | 23 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Asterionella formosa Hassall* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Asterionella ralfsii v. americana W. Smith* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Small diatoms* | 5 | 5 | 28 | 0 | 55 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Large diatoms* | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tabellaria fenestrata v. intermedia Grunow* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira italica v subarctica (O. Muller) Simonsen* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nitzschia spp.* | 2 | 0 | 16 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nitzschia clausii Hantzsch* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tabellaria quadriseptata Knudson* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Bacillariophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Fragilaria capucina v. rumpens (Kutzing) Lange-Bertalot* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Fragilaria crotenensis* | 16 | 0 | 9 | 0 | 67 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 294 | 0 | 0 |
| *Fragilaria sp.* | 0 | 0 | 0 | 0 | 341 | 16 | 9 | 0 | 0 | 0 | 0 | 0 | 83 | 0 | 0 |
| *Frustulia rhomboides* | 9 | 0 | 0 | 0 | 77 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cyclotella bodanica Eulenst.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Eunotia arcus* | 6 | 6 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 16 | 0 | 0 | 0 | 0 |
| *Eunotia bilunaris* | 0 | 17 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Eunotia exigua* | 0 | 11 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Eunotia incisa* | 1 | 2 | 0 | 0 | 27 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 |
| *Eunotia pectinalis (Kutzing) Rabenhorst* | 17 | 33 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 |
| *Actinella punctata Lewis* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Navicula spp.* | 39 | 58 | 87 | 0 | 32 | 2 | 17 | 0 | 0 | 0 | 0 | 0 | 0 | 11 | 0 |
| *Pinnularia spp.* | 3 | 15 | 41 | 0 | 55 | 19 | 2 | 0 | 0 | 0 | 0 | 0 | 20 | 0 | 0 |
| *Cocconeis sp.* | 0 | 0 | 0 | 61 | 21 | 58 | 93 | 3207 | 327 | 429 | 55 | 588 | 44 | 39 | 0 |
| *Cocconeis placentula Ehrenb.* | 0 | 0 | 0 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 101 | 0 | 0 | 0 | 0 |
| *Cymbella sp.* | 26 | 0 | 11 | 0 | 23 | 4 | 14 | 0 | 0 | 0 | 0 | 0 | 12 | 0 | 15 |
| *Cymbella affinis Kutzing* | 0 | 17 | 3 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 9 | 0 | 19 |
| *Cymbella delicatula Kutzing* | 2 | 0 | 21 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 |
| *Cymbella ehrenbergii Kutzing* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 32 | 0 | 0 |
| *Cymbella gracilis (Ehrenberg) Kutzing* | 22 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 16 | 0 | 33 |
| *Cymbella hebridica* | 5 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| *Cymbella helvetica Kutzing* | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 |
| *Cymbella lunata* | 7 | 0 | 0 | 0 | 3 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 58 | 0 | 29 |
| *Cymbella microcephala* | 12 | 37 | 0 | 0 | 41 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 137 | 0 | 37 |
| *Diatoma sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Epithemia sp.* | 3 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Entomoneis paludosa (W. Sm.) Reimer in Patr. And Reimer* | 0 | 0 | 0 | 0 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gomphonema sp.* | 11 | 16 | 17 | 26 | 11 | 0 | 0 | 0 | 4 | 0 | 11 | 0 | 143 | 0 | 58 |
| *Gomphonema acuminatum* | 0 | 21 | 0 | 0 | 39 | 6 | 0 | 0 | 9 | 0 | 3 | 0 | 18 | 0 | 2 |
| *Gomphonem angustum Agardh* | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 16 |
| *Gomphonema gracile* | 29 | 5 | 13 | 0 | 5 | 0 | 0 | 0 | 19 | 0 | 37 | 0 | 39 | 0 | 47 |

| | Lake | | | | | | | | | | | | | | |
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| **Bacillariophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Gomphonema truncatum* | 33 | 0 | 5 | 0 | 2 | 1 | 0 | 0 | 2 | 0 | 2 | 0 | 11 | 0 | 11 |
| *Gyrosigma acuminatum* | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Bacillariophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Achnanthes flexella* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 |
| *Achnanthes inflata* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 |
| *Achnanthes lanceolata* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 17 | 0 | 0 |
| *Achnanthes minutissima* | 0 | 0 | 0 | 0 | 0 | 116 | 27 | 21 | 0 | 0 | 0 | 0 | 221 | 0 | 43 |
| *Achnanthes sp.* | 62 | 0 | 39 | 87 | 91 | 37 | 0 | 47 | 179 | 223 | 29 | 0 | 45 | 61 | 17 |
| *Aulacoseira granulata (Ehrenberg) Simonsen* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 15 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira islandica (O. Muller) Simonsen* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 97 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira italica (Ehrenberg) Simonsen* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira distans (Ehrenberg) Simonsen* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 51 | 2638 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cyclotella stelligera Cleve and Grunow* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cyclotella pseudostelligera* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Cyclotella sp.* | 0 | 0 | 0 | 0 | 0 | 31 | 0 | 9 | 0 | 0 | 8 | 0 | 0 | 0 | 11 |
| *Rhopalodia gibba (ehrenb.) O. Mull.* | 0 | 0 | 2 | 0 | 0 | 139 | 59 | 0 | 0 | 9 | 0 | 0 | 30 | 42 | 418 |
| *Rhizosolenia eriense H.L. Smith* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tabellaria fenestrata (Lyngbye) Kutzing* | 0 | 0 | 0 | 0 | 0 | 19 | 0 | 0 | 15 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tabellaria flocculosa (Roth) Kutzing* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Stauroneis anceps Ehrenberg* | 0 | 0 | 5 | 0 | 17 | 3 | 0 | 3 | 3 | 0 | 0 | 0 | 0 | 2 | 23 |
| *Synedra acus Kutzing* | 11 | 11 | 19 | 55 | 139 | 77 | 3 | 33 | 17 | 11 | 38 | 71 | 103 | 39 | 201 |
| *Synedra acus v. radians (Kutzing) Hustedt* | 0 | 0 | 0 | 0 | 0 | 49 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 13 |
| *Synedra ulna (Nitzsch) Ehrenberg* | 9 | 102 | 64 | 12 | 157 | 0 | 17 | 7 | 2 | 5 | 17 | 26 | 71 | 14 | 35 |
| *Stephanodiscus sp.* | 0 | 0 | 0 | 0 | 3 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Bacillariophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Anomoeoneis serians* | 0 | 0 | 0 | 0 | 0 | 17 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 9 |
| *Asterionella formosa Hassall* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Asterionella ralfsii v. americana W. Smith* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Small diatoms* | 0 | 0 | 0 | 0 | 0 | 16 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Large diatoms* | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tabellaria fenestrata v. intermedia Grunow* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Aulacoseira italica v subarctica (O. Muller) Simonsen* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Nitzschia spp.* | 0 | 0 | 26 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 20 |
| *Nitzschia clausii Hantzsch* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tabellaria quadriseptata Knudson* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Fragilaria capucina v. rumpens (Kutzing) Lange-Bertalot* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Fragilaria crotenensis* | 49 | 0 | 33 | 0 | 0 | 331 | 0 | 0 | 19 | 7 | 0 | 0 | 53 | 36 | 33 |
| *Fragilaria sp.* | 0 | 0 | 0 | 0 | 0 | 23 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 69 |
| *Frustulia rhomboides* | 3 | 0 | 0 | 0 | 0 | 4 | 0 | 10 | 0 | 0 | 0 | 0 | 69 | 0 | 161 |
| *Cyclotella bodanica Eulenst.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Eunotia arcus* | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 1 | 0 | 0 | 16 | 0 | 3 |
| *Eunotia bilunaris* | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 1 | 0 | 0 | 2 | 0 | 0 | 0 |
| *Eunotia exigua* | 0 | 0 | 0 | 0 | 0 | 6 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 |
| *Eunotia incisa* | 0 | 0 | 0 | 0 | 0 | 14 | 0 | 0 | 2 | 8 | 0 | 41 | 1 | 0 | 0 |
| *Eunotia pectinalis (Kutzing) Rabenhorst* | 0 | 0 | 0 | 0 | 0 | 72 | 0 | 0 | 9 | 3 | 0 | 65 | 2 | 4 | 10 |
| *Actinella punctata Lewis* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Navicula spp.* | 17 | 0 | 38 | 0 | 62 | 27 | 0 | 49 | 28 | 0 | 14 | 21 | 51 | 0 | 109 |
| *Pinnularia spp.* | 6 | 0 | 11 | 0 | 0 | 32 | 0 | 12 | 2 | 6 | 6 | 11 | 2 | 17 | 18 |
| *Cocconeis sp.* | 118 | 0 | 9 | 0 | 0 | 182 | 0 | 391 | 131 | 33 | 59 | 0 | 17 | 0 | 44 |
| *Cocconeis placentula Ehrenb.* | 0 | 0 | 77 | 0 | 0 | 0 | 0 | 0 | 17 | 0 | 0 | 0 | 188 | 0 | 0 |
| *Cymbella sp.* | 0 | 59 | 27 | 11 | 39 | 10 | 17 | 21 | 5 | 42 | 45 | 0 | 17 | 17 | 71 |
| *Cymbella affinis Kutzing* | 19 | 2 | 16 | 0 | 29 | 19 | 0 | 0 | 0 | 3 | 0 | 0 | 10 | 0 | 16 |
| *Cymbella delicatula Kutzing* | 11 | 0 | 9 | 29 | 14 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 66 | 0 | 49 |
| *Cymbella ehrenbergii Kutzing* | 0 | 2 | 0 | 16 | 0 | 0 | 0 | 0 | 2 | 6 | 0 | 0 | 1 | 0 | 0 |
| *Cymbella gracilis (Ehrenberg) Kutzing* | 0 | 0 | 0 | 0 | 0 | 0 | 33 | 15 | 10 | 2 | 0 | 0 | 5 | 0 | 83 |

| | Lake | | | | | | | | | | | | | | |
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| **Bacillariophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Cymbella hebridica* | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 2 | 0 | 0 | 0 | 0 | 37 | 0 | 6 |
| *Cymbella helvetica Kutzing* | 29 | 1 | 39 | 0 | 47 | 26 | 0 | 19 | 0 | 0 | 0 | 0 | 0 | 6 | 13 |
| *Cymbella lunata* | 0 | 0 | 0 | 6 | 0 | 31 | 4 | 9 | 14 | 0 | 0 | 0 | 6 | 0 | 44 |
| *Cymbella microcephala* | 0 | 0 | 0 | 0 | 0 | 57 | 21 | 72 | 39 | 17 | 0 | 0 | 21 | 0 | 67 |
| *Diatoma sp.* | 0 | 0 | 14 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 5 | 0 | 5 |
| *Epithemia sp.* | 9 | 0 | 4 | 0 | 0 | 13 | 0 | 29 | 3 | 1 | 0 | 0 | 0 | 33 | 15 |
| *Entomoneis paludosa (W. Sm.) Reimer in Patr. And Reimer* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 |
| *Gomphonema sp.* | 0 | 107 | 39 | 21 | 37 | 2 | 41 | 66 | 21 | 5 | 63 | 0 | 19 | 419 | 91 |
| *Gomphonema acuminatum* | 31 | 23 | 13 | 5 | 2 | 37 | 0 | 3 | 9 | 1 | 0 | 0 | 9 | 6 | 41 |
| *Gomphonem angustum Agardh* | 0 | 2 | 1 | 0 | 0 | 0 | 0 | 7 | 0 | 1 | 0 | 0 | 33 | 203 | 18 |
| *Gomphonema gracile* | 8 | 37 | 2 | 37 | 24 | 5 | 9 | 109 | 19 | 0 | 0 | 0 | 41 | 29 | 103 |
| *Gomphonema truncatum* | 19 | 2 | 7 | 9 | 18 | 41 | 1 | 0 | 2 | 3 | 0 | 0 | 10 | 0 | 11 |
| *Gyrosigma acuminatum* | 2 | 0 | 0 | 1 | 0 | 11 | 0 | 13 | 4 | 1 | 0 | 0 | 1 | 10 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Pyrrophyceae** | **1** | **2** | **3** | **4** | **5** | **6** | **7** | **8** | **9** | **10** | **11** | **14** | **18** | **19** | **20** |
| *Gymnodinium helveticum Penard* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gymnodinium sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium Willei Huitfeldt-Kaas* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium limbatum (Stokes) Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium inconspicuum Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium pusillum (Penard) Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium goslaviense Woloszynska* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium aciculiferum Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium wisconsiense Eddy* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium sp.* | 14 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Ceratium hirundenella (Muller) Schrank* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| | Lake | | | | | | | | | | | | | | |
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| **Pyrrophyceae** | **21** | **22** | **23** | **25** | **26** | **27** | **29** | **33** | **34** | **36** | **40** | **44** | **45** | **46** | **50** |
| *Gymnodinium helveticum Penard* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Gymnodinium sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium Willei Huitfeldt-Kaas* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium limbatum (Stokes) Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium inconspicuum Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium pusillum (Penard) Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium goslaviense Woloszynska* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium aciculiferum Lemmermann* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium wisconsiense Eddy* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Peridinium sp.* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 |
| *Ceratium hirundenella (Muller) Schrank* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |