Characterizing Plankton Communities in Lake Ontario Coastal Wetlands Along an Urban Land-Use Gradient

by

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An oral defense of this thesis took place on April 15th 2020 in front of the following examining committee:

Examining Committee:

Chair of Examining Committee	Dr. Janice Strap
Research Supervisor	Dr. Andrea Kirkwood
Examining Committee Member	Dr. Julia Green-Johnson
Examining Committee Member	Dr. Michelle Bowman
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The above committee determined that the thesis is acceptable in form and content and that a satisfactory knowledge of the field covered by the thesis was demonstrated by the candidate during an oral examination. A signed copy of the Certificate of Approval is available from the School of Graduate and Postdoctoral Studies.

Abstract

This thesis presents the results of a study on the effect of habitat condition and water quality on plankton communities across an urban land-use gradient in the Lake Ontario coastal wetlands: Frenchman’s Bay, Lynde Marsh, McLaughlin Bay, and Bowmanville Marsh over two years (2018-2019). One of the study wetlands (McLaughlin Bay) was assessed over three years (2017-2019) for its suitability as a candidate wetland for biomanipulation restoration. I found water quality was generally not degraded along the urban gradient as expected. Nutrient rich waters and high chloride concentrations were determined to be important drivers of decreased diversity and higher algal biomass dominated by cyanobacteria. In my assessment of McLaughlin Bay, I found that due to the nutrient- and chloride-rich conditions, the plankton community was dominated by inedible algal communities, and small zooplankton taxa. These results do not support applying biomanipulation as a restoration approach in McLaughlin Bay at this time.

Keywords: Plankton communities; water quality; Land use; Lake Ontario coastal wetlands; Biomanipulation

AUTHOR’S DECLARATION

I hereby declare that this thesis consists of original work of which I have authored. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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STATEMENT OF CONTRIBUTIONS

I hereby certify that I am the sole author of this thesis and that no part of this thesis has been published or submitted for publication. I have used standard referencing practices to acknowledge ideas, research techniques, or other materials that belong to others. Furthermore, I hereby certify that I am the sole source of the creative works and/or inventive knowledge described in this thesis.

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List of Abbreviations

TN	Total Nitrogen
TP	Total Phosphorus
TDP	Total Dissolved Phosphorus
Chl a	Chlorophyll a
Temp	Temperature
Secchi	Secchi Depth
DO	Dissolved Oxygen
BV	Bowmanville Marsh
MB	McLaughlin Bay
LM	Lynde Marsh
FB	Frenchman’s Bay
CLOCA	Central Lake Ontario Conservation Authority
ANOVA	Analysis of Variance
VIF	Variance Inflation Factor
PCA	Principal Component Analysis
DCA	Detrended Correspondence Analysis
RDA	Redundancy Analysis

Chapter 1: General Introduction

Land use is known to be one of the most influential drivers of water quality in aquatic ecosystems (Ren et al., 2003; Tu, 2011). Land use within a watershed can change an ecosystem drastically, depending on the type and intensity of the land-use. In cases where land cover has been changed from natural to agricultural land, there is typically an excess of nutrients in the water due to the use of fertilizers and concentrated animal waste (Parry, 1998; Scanlon et al., 2007). Agricultural land-use can also lead to soil erosion causing sediment-laden run-off to enter waterways. This leads to increased turbidity in waters, lower water clarity, and potential impacts on photosynthetic organisms due to light limitation.

When looking at urban land cover, it is common for nutrient loadings to increase due to residential and commercial fertilizer use on lawns and gardens (He et al., 2014). A notable land-cover type associated with urban areas is impervious surface cover. Urban growth includes the building and expansion of roads, parking lots, buildings, and even manicured lawns, all of which increase the amount of impervious surface cover on the landscape. Impervious surfaces are land surfaces that do not allow sufficient infiltration of rain and snow-melt into soils and groundwater. Therefore, when it rains, water runs over the impervious surfaces in urban areas, and drains quickly into surface waters such as stormwater ponds and tributaries, and then eventually coastal wetlands. Stormwater run-off from areas high in impervious surface cover tend to have more contaminants as well, including nutrients, metals, pesticides, etc. (Arnold & Gibbons, 1996).

Another concern related to urban land cover is the increased levels of chloride from road salt use (Scott et al., 2019). When de-icing salts are applied to roads and parking lots in urban areas, they are readily dissolved in rain and snow-melt water. As run-off flows over the developed landscape, it picks up and concentrates salts from the watershed prior to entering receiving waters. Chloride in road-salt is the key toxicant of concern, as it can be toxic to aquatic organisms in sufficient concentrations. (Canadian Council of Ministers of the Environment, 2011).

1.1 Great Lakes Coastal Wetlands

Wetlands are ecosystems that are mainly characterized by being flooded with water, either temporarily or permanently. This unique habitat type leads to the presence of a variety of biota that have adapted to changing water levels (Keddy, 2010). These habitats are known to be productive ecosystems with high species diversity, and that has been an important factor in the goal of restoration (Bobbink et al., 2006).

Great Lakes coastal wetlands are very important ecosystems with over 2000 existing along the Canadian Great Lakes shoreline (Ingram et al., 2004). Wetlands are valuable ecosystems to humans, especially in areas with increasing human development in their watersheds (Mitsch & Gosselink, 2000). Wetlands provide various benefits to humans through both ecosystem services as well as acting as temporary and permanent habitats for a wide variety of organisms (Sierszen et al., 2012). These ecosystems are important in flood mitigation as they play a large role in preventing flood damage, especially in more urbanized areas, which provides significant economic benefits (Hey & Philippi, 1995). Coastal wetlands can also play a role in erosion and wave damage protection through wave attenuation, especially in macrophyte dominant wetlands (Gedan et al., 2011). These ecosystems can also play major roles in nutrient retention, which can be very important in reducing the impact of urbanization in more populated regions (Comin et al., 1997; Mitsch & Gosselink, 2000).

Great Lakes coastal wetlands have become degraded in more recent years as human influence through agriculture, pollution, and densely populated regions have been linked to poorer water quality conditions (Morrice et al., 2008). These wetlands have also been affected by the introduction of invasive species, most notably *Phragmites australis* (common reed) and *Cyprinus carpio* (common carp), which can alter biological communities and displace native species (Lougheed et al., 1998; Tulbure et al., 2007). It has been shown that many Great Lakes coastal wetlands require restoration, especially more urbanized areas, since a majority of wetlands along Lake Ontario, Lake Erie, and Lake Michigan are considered degraded (Cvetkovic & Chow-Fraser, 2011).

1.2 Water Quality and the Biotic Community

When water quality conditions degrade in any aquatic system there is likely to be concern for the impacts it will have on the organisms within that ecosystem. One of the main characteristics of healthy wetlands is that they are typically dominated by emergent and submergent macrophytes, which provide many of the ecosystem services that makes wetlands so valuable (Engelhardt & Ritchie, 2001). When an ecosystem receives an influx of nutrients, there is sometimes a shift from a clear water macrophyte dominated state, to a turbid phytoplankton dominated state (Holling, 1973). This phenomenon, known as alternative stable states, is the result of degraded water quality and can cause a shift in the entire food web through the change in species composition and altered habitat (Bayley & Prather 2003).

An increase of nutrients, namely phosphorus, is impactful because this element is a major limiting nutrient of algal growth. When phosphorus levels increase, often there is a positive linear relationship with algal biomass until the relationship asymptotes, indicating phosphorus is no longer limiting growth (Bachmann & Jones., 1974; Watson et al., 1992). It has been shown that nitrogen can also act as a limiting nutrient to phytoplankton growth, especially when added in combination with phosphorus (Elser et al., 1990). Measuring parameters such as depth, Secchi depth, and turbidity are important as they all can play a role in the availability of light which is essential in the growth of algae (Li et al., 2011). It has also been shown that the growth of algae can increase dissolved oxygen and pH levels in the water (Li et al., 2011).

While nutrients are a very important driver of plankton communities, another important factor to consider is chloride. In urbanizing areas, chloride is becoming a larger issue because there is great potential for toxicity with increased salinity. It has been shown previously that increased salinity in aquatic ecosystems can decrease the abundance of large filter feeding zooplankton, leading to less grazing pressure on the phytoplankton community, and therefore increasing algal biomass. This degradation of some species by chloride can lower species richness and cause a shift in plankton communities (Kipriyanova et al., 2007). Chloride may also play a role in promoting cyanobacterial blooms as it was found that different types of cyanobacteria are more resistant to high chloride levels than other phytoplankton. Certain cyanobacterial taxa have the ability to export ions (sodium and chloride) out of their cells to prevent toxicity (Apte et al., 1987; Hagemann, 2011; Tonk et al., 2007).

One type of restoration approach that has been developed to control algal blooms is biomanipulation. This restoration technique involves altering one or more aspects of the biological community, such as adding or removing organisms at a particular trophic level. The objective is to shift food web interactions in order to improve ecosystem conditions, such as decreasing algal abundance via increased zooplankton grazing (Shapiro, 1990). A form of biomanipulation has been used in a Lake Ontario coastal wetland, namely, Cootes Paradise Marsh, a degraded Lake Ontario coastal wetland. The goal of this project was to remove and invasive fish species in order to improve water quality conditions and reduce algal biomass. This method had some positive effects on water quality and plankton communities (Thomasen & Chow-Fraser, 2012). A biomanipulation has been shown to be a successful restoration approach under certain water quality conditions (Benndorf, 1990).

1.3 Study Area

This study takes place on the north shore of Lake Ontario in Durham Region, Ontario. The Durham Region is an area with a large population (estimated over 600,000 in 2016) that is projected to grow rapidly in the coming years (Ontario Ministry of Finance, 2019; Statistics Canada, 2017). The population growth within this region has led to changes in water quality and the biological communities along the Great Lakes (Frieswyk & Zedler, 2007; Kelso et al., 1996). The Durham region can act as a model for changing land use in the Great Lakes region and a baseline on how plankton communities change in the following years will likely be reflected in the results of this study.

There is a gradient of urban development across the study wetland watersheds, starting with the highest urban development surrounding Frenchman’s Bay in Pickering, followed by Lynde Marsh in Whitby, McLaughlin Bay in Oshawa and Courtice, and Bowmanville Marsh in Bowmanville (Figure 1.1). Frenchman's Bay is the most westerly wetland of this study, and has a relatively small watershed (26-km^2). Lynde Marsh has the second largest drainage area (141-km^2) of the study wetlands and is dominated by both urban and agricultural land-use (Central Lake Ontario Conservation Authority, 2012). McLaughlin Bay has a very small watershed (2-km^2) that is adjacent to Darlington Provincial Park and drains two large parking lots by the General Motors Headquarters (Central Lake Ontario Conservation Authority, 2013). The easternmost wetland is Bowmanville Marsh in Bowmanville, Ontario, and has the largest watershed (190-km^2) which is predominantly agricultural land.

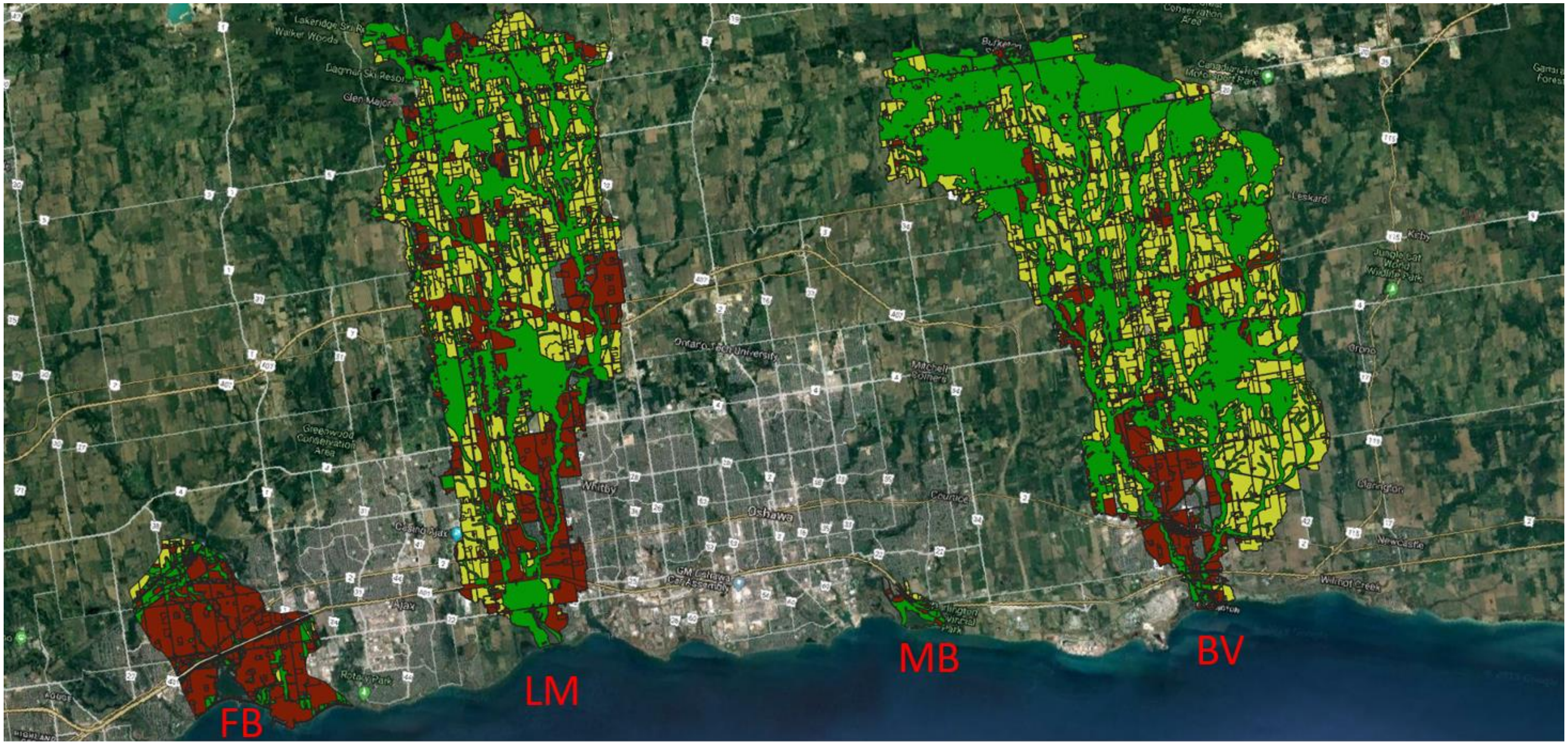

Figure 1.1. Land cover profiles of four studied wetlands from left to right: Frenchman's Bay (FB), Lynde Marsh (LM), McLaughlin Bay (MB), and Bowmanville Marsh (BV). Red color indicates developed land, yellow indicates agricultural, and bright green indicates natural.

1.4 Research Objectives

The overall goal of my thesis research was to characterize and assess the current ecological condition of four central Lake Ontario coastal wetlands based on their land-use influences, water quality profiles, and plankton community structure. This overall assessment will provide important baseline information regarding the status of plankton communities in these coastal wetlands, which can inform future management decisions related to wetland restoration. Additionally, as part of this assessment of wetland ecological condition, McLaughlin Bay was evaluated as a potential candidate for restoration using biomanipulation. Therefore, to achieve these goals, the following research objectives were implemented:

1. Characterize the seasonal (May-September) water quality and plankton communities of four coastal Lake Ontario wetlands in Durham Region over two years (Chapter 2);
2. Determine if coastal-wetland water quality degradation corresponds to the urbanization gradient across watersheds (Chapter 2);
3. Assess the relative role of abiotic and biotic factors in structuring plankton communities in the study wetlands (Chapter 2); and
4. Evaluate the suitability of McLaughlin Bay to be a candidate for restoration by biomanipulation (Chapter 3)

In Chapter 2, I assessed water quality in the study wetlands as a function of changing land-use across watersheds, primarily focusing on the shift in urban land-use. I predicted that watersheds with high levels of urban development would generally have poorer water quality profiles, particularly higher nutrients and chloride concentrations. In order to assess plankton communities, I characterized the composition of phytoplankton and zooplankton communities, and compared their composition and abundance among study wetlands and previous studies of Lake Ontario coastal wetlands. I expected plankton communities in degraded areas to have fewer species, while containing more groups often associated with urbanization (cyanobacteria, filamentous green algae, etc.).

In order to understand the effects of water quality and habitat conditions on plankton communities, I looked at how changing water quality conditions impacted the biomass and composition of plankton communities as well as how plankton communities impacted each other. In addition to water quality, habitat conditions were inferred by the presence of submergent macrophytes and physical characteristics like depth, in order to understand if these factors influenced plankton community structure and abundance. I expected in shallower, macrophyte dominated sites there would a lower abundance of algae, and greater zooplankton abundance. In this chapter, it was expected that water quality would be an important driver of plankton communities in these wetlands and the degraded state of water quality would lead to notable differences among communities.

In chapter three, McLaughlin Bay water quality and plankton communities were assessed to determine the feasibility of biomanipulation as a restoration approach. Characterization of plankton communities was done by assessing the abundance and types of algae present and comparing their general composition to previously studied systems that have attempted biomanipulation restoration projects. By understanding plankton communities and assessing relationships with water quality variables, I was able to obtain more information on the possible viability of applying a biomanipulation project in McLaughlin Bay.

In the final chapter, I summarize my key findings and conclusions regarding the current ecological state of my study wetlands. I also discuss my study limitations and make recommendations for future research in Lake Ontario coastal wetlands. Finally, I discuss the significance of my thesis results and their importance in informing coastal wetland restoration in Lake Ontario.

Chapter 2: Assessment of Water Quality and Plankton Communities in Lake Ontario Coastal Wetlands across an Urban Gradient

2.1 Introduction

The land cover in the Great Lakes region has changed greatly throughout history, and this region continues to increase in human population density (Ontario Ministry of Finance, 2019). Alterations in land cover have had major changes throughout the landscape of the Great Lakes region (Cole et al., 1998). The type of land use in a watershed tends to dictate water quality in various ways. Urbanized land is known to increase chloride levels through the use of road salts, but also in many cases municipal loading of fertilizers can contribute high levels of nutrients not involved in farming (He et al., 2014). This is a significant contribution because through population growth, there is likely to be more of these nutrients that are essential in the growth of algae. In the Great Lakes region as the land cover changes to more developed regions, there is a decrease in water quality as seen historically in this area (Chow-Fraser, 2006; Croft-White et al., 2017). This change can be harmful to aquatic organisms as well as wetland-dependent terrestrial animals as these organisms are impacted by human influence and increased road density from development in the Great Lakes region (Panci et al., 2017).

Watershed area (i.e., catchment size) can have a major impact on wetland water quality as well. In some cases, watershed area can play a larger role in water quality variables compared to the type of land cover as larger watersheds tend to have more nutrients and other contaminants feeding into the ecosystem (Decatanzaro et al., 2009). Another physical parameter to consider is lake connectivity. Lake connectivity seems to play a major role in the function of coastal wetlands as it was found that in a more degraded stream leading to the free-flowing wetland, there was still a greater fish diversity when compared to relatively clear-water diked wetlands (Kowalski et al., 2014). This research outlines that in an ecosystem where water exchange is typical, changing this can lead to a large alteration in the normal ecological function (Kowalski et al., 2014). A study by Bouvier et al. (2009) looked at how hydrological connectivity to the Great Lakes may affect the fish communities in coastal wetlands. It was found that with increasing connectivity, there was an increase in fish species richness and that this connectivity played a major role in structuring fish communities (Bouvier et al., 2009).

Natural hydrology seems to play a major role in the remediation of coastal wetlands and has been seen as an issue in these increasingly urbanized regions. Previous research showed that the altered hydrology of wetlands can act as a major contributor to ecosystem degradation. This can be key in restoring historical water quality conditions (nutrient transport) as well as biological communities (Wilcox & Whillans, 1999). These differences in watershed connectivity can be seen in the Great Lakes coastal wetlands of this present study. Previously Frenchman's Bay's outlet to Lake Ontario was expanded and fortified to allow a greater connectivity to the lake in a recent restoration project (Toronto and Region Conservation, 2009). In contrast, McLaughlin Bay is the only wetland in this study that has not had significant exchange with Lake Ontario for over a decade due to the development of a natural barrier beach (Central Lake Ontario Conservation Authority, 2013). These differences may be important in the scope of this project as hydrological connectivity may play a major role in the water quality and biological health of these coastal wetlands (Kowalski et al., 2014).

Phytoplankton and zooplankton communities can play important roles in aquatic ecosystems as major primary and secondary producers, respectively. Plankton communities have short life cycles and are sensitive to degradation which can allow them to be early indicators of stress in an environment (Schindler, 1987). By assessing the types of plankton communities present, valuable information about the current ecological health of a wetland can be understood (Lougheed & Chow-Fraser, 2002).

Important drivers of aquatic community structure in wetlands typically include abiotic (e.g., nutrients) and biotic (e.g., predation) factors. For example, when there is a large increase in phytoplankton biomass, there can be a corresponding decline in species diversity as some species outcompete others for limited resources such as nutrient and light availability (Skácelová & Lepš, 2014). Previous research has also shown that zooplankton growth, abundance, and diversity increase with increasing phytoplankton diversity. It is assumed that with more species of phytoplankton present, there are more feeding niches, allowing certain specialist species to persist with different characteristics like different cell size, structure, shape, and habits. Therefore, with an increased diversity of phytoplankton it is likely there are far reaching effects on the entire aquatic food web (Striebel et al., 2012).

As water quality conditions change there are evident shifts in phytoplankton community structure. High levels of nitrogen and phosphorus have been shown to lead to increased proportions of cyanobacteria in phytoplankton communities (Downing et al., 2001). Trochine et al. (2011) showed that filamentous green algae will likely increase in abundance with increased temperature and nutrient conditions while inhibiting competing phytoplankton growth. As a result, it would lead to a dominance of filamentous green algae, and an overall decrease in species richness (Trochine et al., 2011). When water quality conditions change in aquatic ecosystems alterations in the structure of phytoplankton communities is often affected as well.

As these phytoplankton communities change they are potentially impacting zooplankton communities through interspecies interactions. Zooplankton abundances can be impacted when phytoplankton communities see a shift to increased cyanobacterial biomass as they are typically not ideal for feeding by zooplankton (De Bernardi & Giussani, 1990). Zooplankton community composition is also altered by changes in the phytoplankton community, as smaller zooplankton are not as negatively impacted by some cyanobacterial species as larger filter-feeding zooplankton are (Fulton & Paerl, 1988). As phytoplankton communities become evident of a degraded ecosystem, zooplankton community structure is likely to be affected as well.

Previous studies on the plankton communities in Lake Ontario coastal wetlands assessed the composition of plankton communities. A study by Lougheed & Chow-Fraser (2002) used zooplankton communities based on their association with macrophytes and water quality to develop an index to assess wetland quality. Diverse zooplankton communities with large cladocerans are known to be associated with macrophytes and better water quality conditions so this can be used to assess relative ecosystem health (Lougheed & Chow-Fraser, 2002). A study in Cootes Paradise Marsh in Hamilton, Ontario assessed how plankton communities changed over a long period of time in response to human induced changes (Chow-Fraser et al., 1998). This study found that when nutrient inputs were increased in this region, nitrogen fixing cyanobacterial abundance was increased. The zooplankton community saw a loss of the large cladoceran, *Daphnia* and during this period, they were replaced by smaller types of zooplankton. It was found later that cyanobacteria decreased when nutrient inputs were controlled, though the *Daphnia* communities did not recover (Chow-Fraser et al., 1998).

A study by Lougheed & Chow-Fraser (1998) examined the zooplankton communities in Cootes Paradise Marsh to predict the changes a common carp (*Cyprinus carpio*) exclusion study would cause in the biological community. It was found that this wetland had turbid water which lead to mostly small bodied zooplankton assemblages. When the common carp exclusion was done and a majority of the carp were removed from the wetland and the ecosystem was assessed, it was found that zooplankton communities had a more balanced size distribution with a greater proportion of larger zooplankton, though there was not a notable increase in cladoceran biomass, or in the zooplankton index previously developed (Thomasen & Chow-Fraser, 2012).

There has been marked urban growth in the Greater Toronto Area, and Durham Region is no exception. In order to assess how the urban gradient across Durham Region influences coastal wetland water quality and plankton communities, I aimed to address the following research objectives:

1. Characterize the seasonal (May-September) water quality and plankton communities of four coastal Lake Ontario wetlands in Durham Region over two years
2. Determine if coastal-wetland water quality degradation corresponds to the urbanization gradient across watersheds
3. Assess the relative role of abiotic and biotic factors in structuring plankton communities in the study wetlands

By documenting current water quality conditions and plankton community structure, the information reported here can be used to inform future wetland restoration initiatives. Determining the important ecological drivers of plankton community structure, particularly negative factors that affect biomass and diversity, also provides meaningful information to inform wetland mitigation.

2.2 Methods

2.2.1 Study Area

The four coastal wetlands chosen for this study include: Frenchman's Bay, Lynde Marsh, McLaughlin Bay, and Bowmanville Marsh as previously described. Three sites in each wetland, aside from McLaughlin Bay which had four sites, were sampled monthly from May to September in 2018 and 2019. Sites spanned each wetland from near the main tributary inlet to near the confluence with the lake. Bowmanville Marsh sites span from the inlet of Bowmanville and Soper Creek, to the outlet to Lake Ontario (Figure 2.1). McLaughlin Bay has four sites, which span from the inlet near the General Motors Headquarters, to the barrier beach near Lake Ontario, with an additional site in the eastern section of the wetland (Figure 2.2) (Central Lake Ontario Conservation Authority, 2013). Lynde Marsh has three sampling sites in this wetland, which occur along a transect starting from the inflow at Lynde Creek, to the outflow into Lake Ontario (Figure 2.3). Frenchman's Bay is the most westerly wetland of this study, sampling locations span a transect starting from the inlet (fed mainly by Pine Creek, Amberlea Creek, Dunbarton Creek, and Kronso Creek) to the outflow area to Lake Ontario (Figure 2.4).

Figure 2.1. Map of Bowmanville Marsh with study sites shown in red.

Figure 2.2. Map of McLaughlin Bay with study sites shown in red.

Figure 2.3. Map of Lynde Marsh with study sites shown in red.

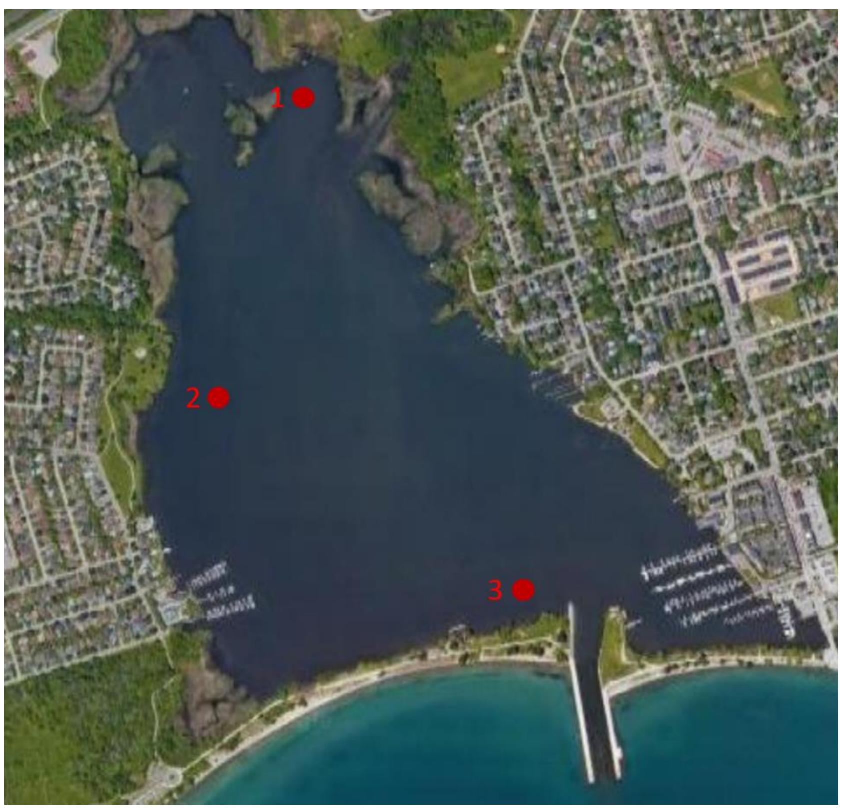

Figure 2.4. Map of Frenchman's Bay with study sites shown in red.

2.2.2 Data Collection

Wetlands were sampled on the same day or within two days of each other for each monthly sample collection. Water quality parameters measured on-site included site depth (m), Secchi depth (m), pH, temperature (°C), dissolved oxygen (mg L^{-1}), and conductivity (µS cm^{-1}). Temperature, pH, dissolved oxygen, and conductivity were measured using a 650 MDS multi parameter probe (YSI, Yellow Springs, Ohio, USA). Water samples were collected in 1 L Nalgene bottles at 0.5 m depth using a horizontal Van Dorn water sampler and kept in iced coolers to transport to the lab for analysis.

Water samples were used to measure spectrophotometric turbidity at 750 nm (Balch, 1931). I was not able to measure turbidity in August 2019 due to equipment issues; therefore, turbidity measurements were removed from all regressions for missing data. Chlorophyll a (µg L^{-1}), was collected by filtering 100 mL of each replicate from every site through glass microfiber filters (0.45 µm). Chlorophyll extraction and measurement was done using 90% acetone as previously described (Kirkwood et al., 1999). Total dissolved phosphorus (µg L^{-1}), samples were immediately filtered through 0.20 µm nylon membrane filters. Total and dissolved phosphorus was measured using the Ontario Ministry of the Environment (1983) modified Ascorbic Acid method originally developed by Murphy & Riley (1962). Measurements of chloride (mg L^{-1}), total Kjeldahl nitrogen (mg L^{-1}), ammonia + ammonium (mg L^{-1}), nitrite (mg L^{-1}), and nitrate (mg L^{-1}) were analyzed by an accredited lab SGS Canada Inc.

Macrophyte samples were collected at each site by throwing a lake rake and taking all plant material collected following the protocol of Ginn (2011). Macrophyte samples were not able to be collected in September 2018 in all wetlands and June 2019 in

Lynde Marsh as sampling in this period was done in a smaller boat where collection was not possible. Phytoplankton samples were taken at each site using a horizontal Van Dorn sampler at 0.5 m depth and preserved using Lugol's solution (Sigma Aldrich). Zooplankton were collected using a horizontal Van Dorn sampler at 0.5m depth. Water samples (3L) were filtered through a 63 µm filter and preserved in 70% ethanol. An additional sample per site was collected at 0.5 m using a zooplankton Wisconsin net and preserved in 70% ethanol solution. Macrophyte samples were identified to species level, dried and weighed to determine relative biomass.

Phytoplankton were identified to the genus level using an EVOS xl-core microscope at 400x magnification following a dichotomous key from Prescott (1962), Sheath and Wehr (2003), and Baker at al. (2012). Cells were also counted and measured for length and width of individual cells in order to determine biomass. Microscopic identification of zooplankton species was done to reach 100 individuals and identified to the species level when possible (Copepods were identified to order, cladocerans were identified to genus) (Balcer et al., 1984; Haney et al., 2013). Biomass estimates were calculated using previously established length-weight linear regressions (EPA, 2003).

2.2.3 Data Analysis

All statistical analysis was performed with the R statistical platform (version 3.6.1., R Core Team, 2019). Landscape metrics from each watershed (percent land-use and watershed area) were determined using the open source mapping software QGIS (QGIS Development Team, 2019). Land cover information was calculated using the Central Lake Ontario Conservation Authority and Toronto and Region Conservation 2017 Land cover open data sets (Central Lake Ontario Conservation Authority, 2017;

Toronto and Region Conservation, 2017). Genus richness plots were used in this case as phytoplankton were only identified to genus and genus richness has been shown to provide similar information to species richness (Balmford et al., 1996). Richness was calculated using rarefied richness, based on rarefaction curves, in order to account for number of individuals sampled as sites with more individuals would typically have a greater number of species (Sanders, 1968).

Phytoplankton were also analyzed to the common algal group level, with distinction between the Chlorophyceae group and the class Zygnematophyceae to distinguish between filamentous green algae from other green algae in order to assess their role within the food web (Chow-Fraser et al., 1998). To explore possible relationships between variables, Pearson correlation analysis was applied when bivariate normality was attainable, otherwise Spearman correlation analysis was used. In order to assess differences among groups, ANOVA was used when univariate normality was attainable, otherwise, Kruskal-Wallis rank sum test was used.

Multiple linear regression was used to predict the variation in algal, cyanobacterial and zooplankton biomass. Chlorophyll a was used as it was the best fit dependent variable to represent algal biomass. When using multiple linear regression, variance inflation factor (VIF) was used to assess for collinearity between variables to determine if results were inflated by relationships within independent variables. Any variable with a value above 4 would have been removed from the regression, but no variables exceeded the VIF cut-off (Pan & Jackson, 2008). Data were assessed for normality before testing, log transformations were made when necessary to fit parametric assumptions. A detrended correspondence analysis (DCA) was used in order to assess whether the relationship between the water quality variables and communities were linear or unimodal. This can help determine whether a redundancy analysis or canonical correspondence analysis should be used. If the longest axis gradient length was less than 3, a redundancy analysis was used (Lepš & Šmilauer, 1999).

Redundancy analysis was used to examine relationships between water quality variables and plankton community data. Correspondence analysis was used in order to assess variation in community data. For redundancy analysis and correspondence analysis with zooplankton, the entire community and the top 10 most abundant groups were assessed. The top 10 zooplankton groups represented all zooplankton taxa greater than 1% of the total overall relative abundance in biomass. A Student's t-test was used to test for differences in phytoplankton and zooplankton biomass when macrophytes were present or absent in order to test if macrophytes as a habitat feature alter plankton biomass. Biomass variables were log transformed in order to fit parametric assumptions. Only relative abundance was collected for macrophyte biomass, so a logistic regression was used for plants based on their presence or absence at each collection. Macrophytes were present at nearly half (55/114) of the sites when collection was possible.

Results

2.3.1 Characterizing Water Quality in Coastal Wetlands

The bar chart and summary statistics (Figure 2.5; Table 2.1) show that there are distinct composition profiles of land-use cover among each of the watersheds and their corresponding wetlands. Most similar are Bowmanville and Lynde Creek, which have almost even proportions of agriculture and natural land cover with small amounts of developed land use while being relatively large (>100-km^2). Frenchman's Bay has over 70% of developed land cover, in a relatively small watershed (26-km^2) while McLaughlin Bay is near 70% natural land and a very small watershed (2km^2).

In the following summary statistics, the average and standard deviation of water quality variables in each wetland for both 2018 and 2019 are shown (Table 2.2; Table 2.3).

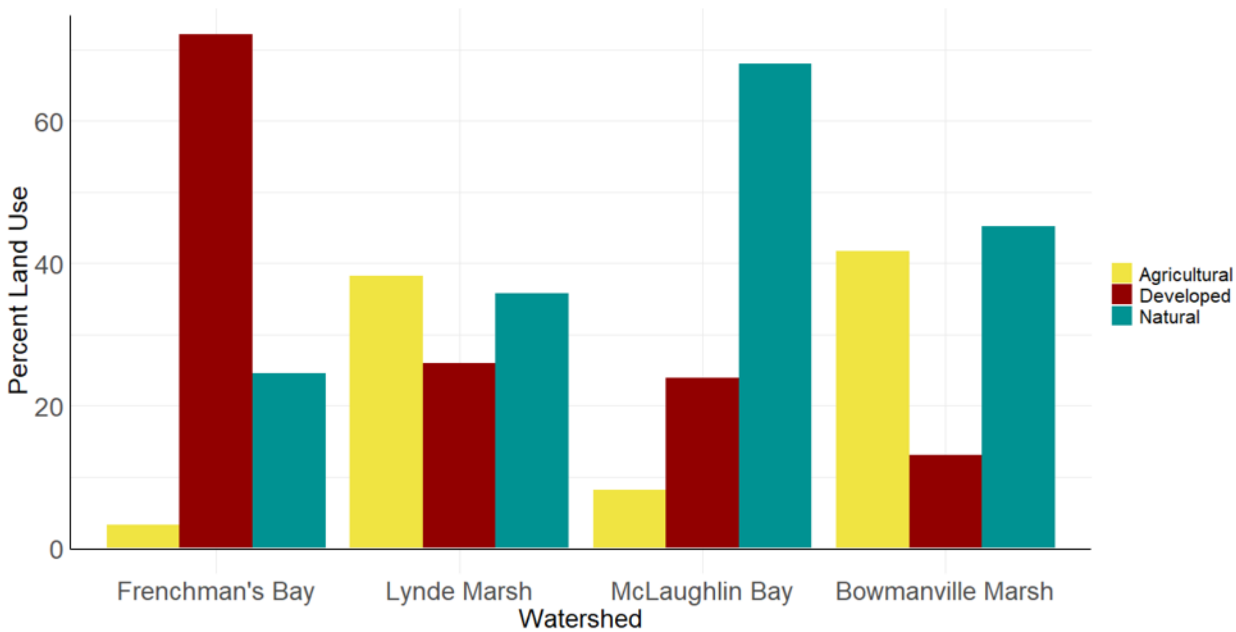

Figure 2.5. Land cover profiles of four studied wetlands. Watersheds of each wetlands is shown on the x-axis. percent land use per wetlands in shown on y-axis.

Table 2.1. Watershed land cover profiles for the four study wetlands from Durham Region, Ontario. Total area of each land cover type and percent land use.

Wetland	**Land Cover**	**Area (km^2)**	**Percent Land Use (%)**
Bowmanville Marsh	Natural	85.29	45.15
	Developed	24.73	13.09
	Agricultural	78.89	41.76
McLaughlin Bay	Natural	1.25	68.00
	Developed	0.44	23.87
	Agricultural	0.15	8.13
Lynde Marsh	Natural	50.38	35.79
	Developed	36.61	26.01
	Agricultural	53.77	38.20
Frenchman's Bay	Natural	6.36	24.59
	Developed	18.69	72.20
	Agricultural	0.83	3.22

Table 2.2. Mean and standard deviation (in parentheses) of water quality variables across the study wetlands in 2018.

Wetland	Depth (m)	Secchi Depth (m)	pH	Conductivity (µs cm^{-1})	Temperature (°C)	Chloride (mg L^{-1})	Total Phosphorus (µg L^{-1})	Total Dissolved Phosphorus (µg L^{-1})	Total Nitrogen (mg L^{-1})	Chlorophyll a (µg L^{-1})
Bowmanville Marsh	0.72 (0.39)	0.46 (0.27)	7.72 (0.93)	658 (121)	20.5 (3.6)	33 (6)	100.30 (49.49)	14.39 (15.61)	0.51 (0.27)	29.55 (64.32)
McLaughlin Bay	1.32 (0.51)	0.42 (0.19)	8.26 (0.80)	2115 (521)	21.5 (3.1)	289 (54)	101.49 (41.94)	7.11 (4.02)	0.59 (0.42)	61.89 (71.68)
Lynde Marsh	0.56 (0.29)	0.26 (0.16)	7.77 (0.88)	1236 (325)	20.8 (3.3)	129 (49)	85.14 (61.92)	8.27 (4.42)	0.65 (0.44)	25.65 (65.02)
Frenchman's Bay	1.70 (0.40)	1.00 (0.37)	8.48 (0.76)	950 (379)	22.7 (3.1)	102 (32)	45.10 (25.08)	8.04 (8.10)	0.30 (0.13)	12.82 (8.56)

Table 2.3. Mean and standard deviation (in parentheses) of water quality variables across the study wetlands in 2019.

Wetland	Depth (m)	Secchi Depth (m)	pH	Conductivity (μs cm^{-1})	Temperature (°C)	Chloride (mg L^{-1})	Total Phosphorus (μg L^{-1})	Total Dissolved Phosphorus (μg L^{-1})	Total Nitrogen (mg L^{-1})	Chlorophyll a (μg L^{-1})
Bowmanville Marsh	1.21 (0.41)	0.78 (0.27)	7.59 (0.19)	882 (208)	19.7 (3.1)	28 (4)	23.02 (10.99)	0.83 (1.70)	0.47 (0.31)	7.32 (9.58)
McLaughlin Bay	1.70 (0.46)	0.51 (0.23)	8.15 (0.27)	2202 (579)	21.4 (4.1)	178 (42)	50.66 (17.36)	1.99 (1.92)	0.34 (0.10)	17.03 (12.52)
Lynde Marsh	1.15 (0.32)	0.70 (0.30)	7.74 (0.37)	1194 (405)	19.0 (3.7)	73 (32)	26.20 (13.79)	2.27 (5.92)	0.36 (0.16)	8.13 (7.92)
Frenchman's Bay	2.10 (0.53)	1.29 (0.63)	8.22 (0.56)	1085 (471)	20.7 (4.0)	78 (49)	19.41 (8.44)	0.50 (0.93)	0.24 (0.15)	7.54 (8.16)

The trend plots showed how water quality variables and zooplankton biomass changed monthly throughout both years of the sampling period (Appendix A1-A8). It was found that chlorophyll a and turbidity peak in August in all wetlands except for Frenchman's Bay where it remains relatively stable. Total phosphorus and total dissolved phosphorus appear to be higher in 2018 than in 2019 in each wetland. It also appears that depth and Secchi depth are higher in 2019 than in 2018 in Lynde Marsh and Bowmanville Marsh.

Principal component analyses conducted on the water quality variables explained 48% of the variation observed (Figure 2.6). McLaughlin Bay sites seem to cluster and are positively associated with chloride while also being negatively associated with Secchi depth. Frenchman's Bay was grouped and was associated with Secchi depth and lower concentrations of chloride and nutrients. Bowmanville Marsh and Lynde Marsh clustered together and were associated with lower concentrations of chloride and shallower site depth throughout this study.

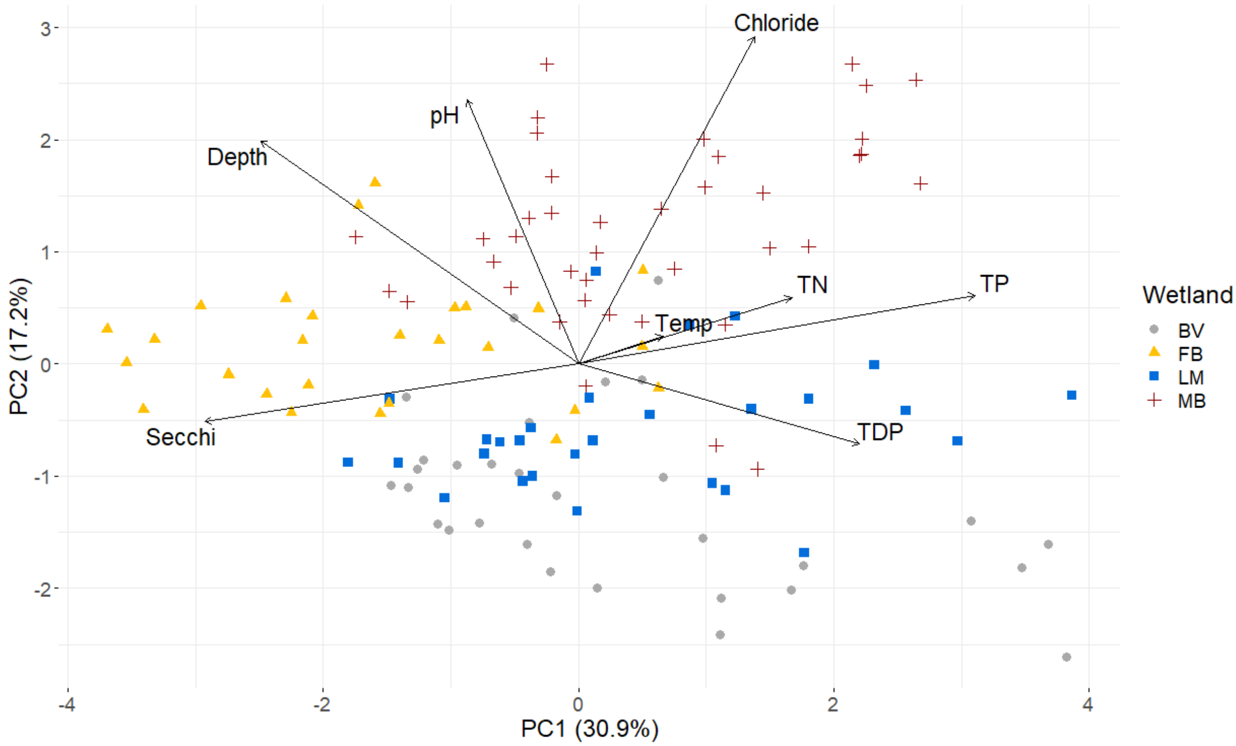

Figure 2.6. Principal component analysis showing water quality variables as vectors and wetlands as indicated by symbols and colors.

The boxplots show the differences between individual wetland water quality variables for each year of study (Figure 2.7-2.9). Using ANOVA, there was found to be a difference among wetlands in 2018 and 2019 in depth, pH, total phosphorus, as well as Secchi depth in 2019. Kruskal Wallis results showed a difference in both 2018 and 2019 for total nitrogen, total dissolved phosphorus, chloride, conductivity, chlorophyll a, and Secchi depth in 2018. The only variable that did not differ among wetlands was temperature.

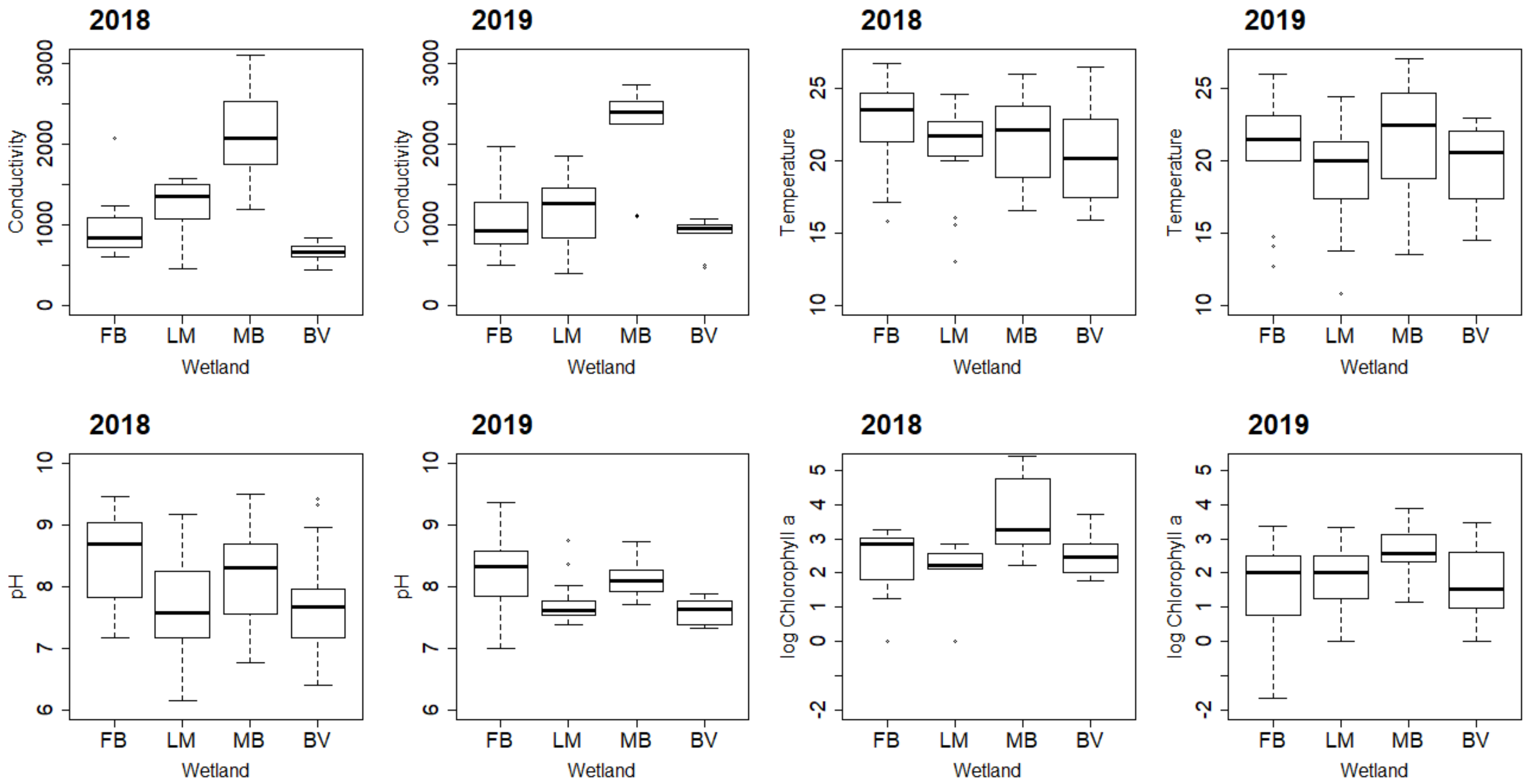

Figure 2.7. Boxplots of conductivity (µs cm^{-1}), temperature (^oC), pH, and log chlorophyll a in 2018 and 2019.

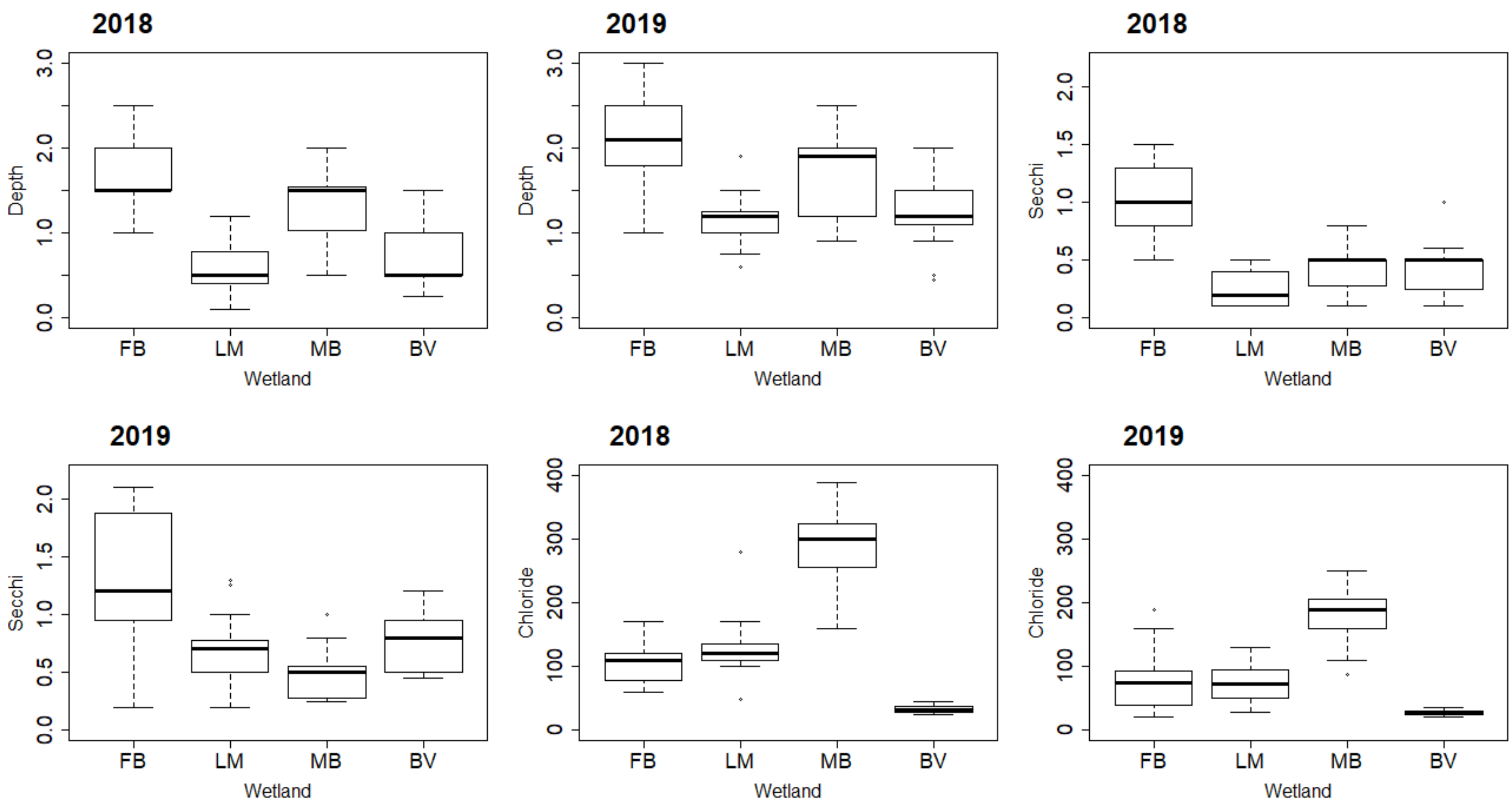

Figure 2.8. Boxplots of depth (m), Secchi depth (m), and chloride (mg L^{-1}) in 2018 and 2019.

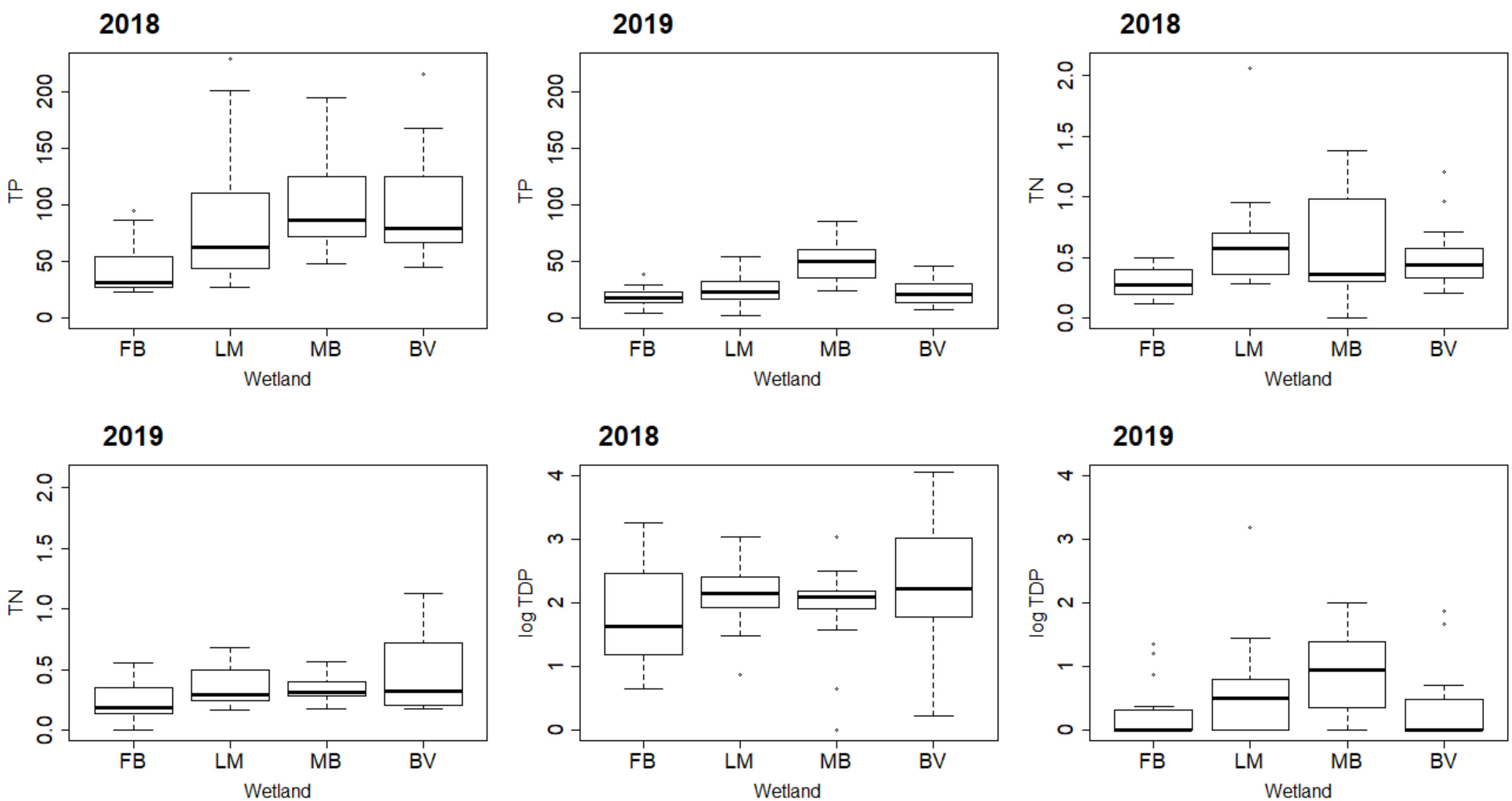

Figure 2.9. Boxplots of total phosphorus (TP) (µg L^{-1}), total nitrogen (TN) (mg L^{-1}), and log transformed total dissolved phosphorus (TDP) in 2018 and 2019.

2.3.2 Characterizing Plankton Communities in Coastal Wetlands

Phytoplankton relative abundance based on their average biomass across wetlands in each year of study is presented in Figure 2.10. There is a greater proportion of cyanobacteria (Cyanophyceae) in McLaughlin Bay compared to other wetlands throughout the study. There is a large number of filamentous algae (Zygnematophyceae) in Bowmanville Marsh in both 2018 and 2019. It is also evident that when filamentous algae are present they are often the dominant algal group due to their large biomass contribution.

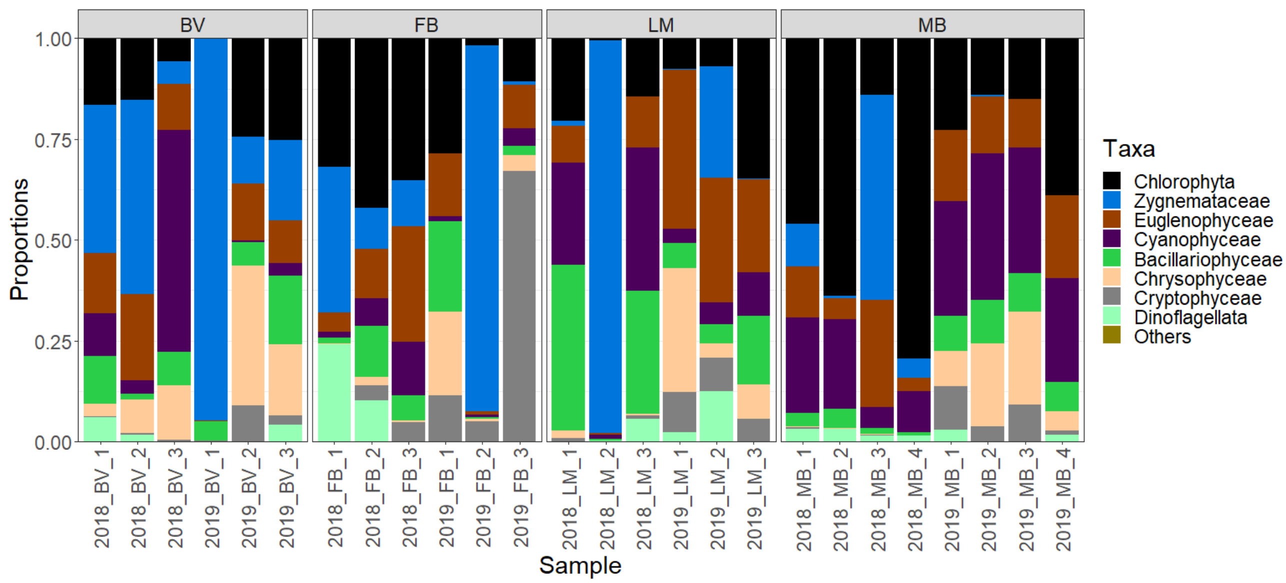

Figure 2.10. Relative abundance plot of phytoplankton groups calculated with seasonal totals. Each bar represents phytoplankton relative abundance at each site within a given wetland, each year of study.

The composition of the zooplankton community can be seen in the zooplankton relative abundance plots (Figure 2.11). There are some notable differences among wetlands in overall zooplankton community composition. The zooplankton community in

McLaughlin Bay is made up mostly of *Bosmina*, a small filter feeding zooplankton while also having some large filter feeding *Daphnia* and Calanoid Copepods. When looking at the other wetlands, they have very few *Daphnia* while having more abundant numbers of Copepod nauplii and *Ceriodaphnia*.

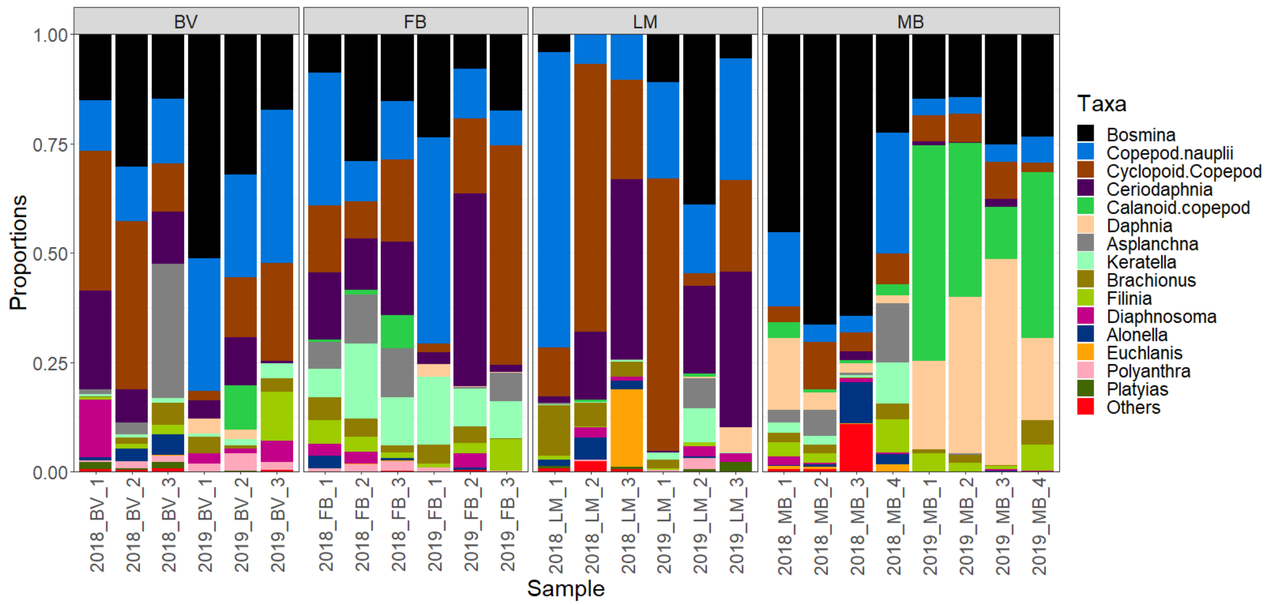

Figure 2.11. Relative abundance plot of zooplankton groups calculated with seasonal totals. Each bar represents zooplankton relative abundance at each site within a given wetland, each year of study

The phytoplankton community composition is shown in the correspondence analysis in appendix A (Figure A13). This plot shows that some of the groups (Dinoflagellata and Zygnematophyceae) have cases of dominance that influence the plot and make differences among sites and other phytoplankton groups difficult to interpret. In the zooplankton community there is also no clear separation among wetlands except for McLaughlin Bay. McLaughlin Bay is mostly different from the other wetlands as it mostly has *Daphnia* and Calanoid copepods that appear to drive its composition (Figure A14). This is even more evident when looking at the top ten most abundant zooplankton groups, the separation for McLaughlin Bay among other wetlands in zooplankton structure is clear (Figure A15).

2.3.3 Understanding the Relationship Between Water Quality and Habitat Conditions with Plankton Communities in Coastal Wetlands

In the following correlation matrix, the relationships between groups of zooplankton and phytoplankton are shown (Figure 2.12). The most notable relationships here are the positive relationship between Chlorophyceae and all groups of zooplankton, as well as the Rotifer zooplankton sub-class and Cyanophycea.

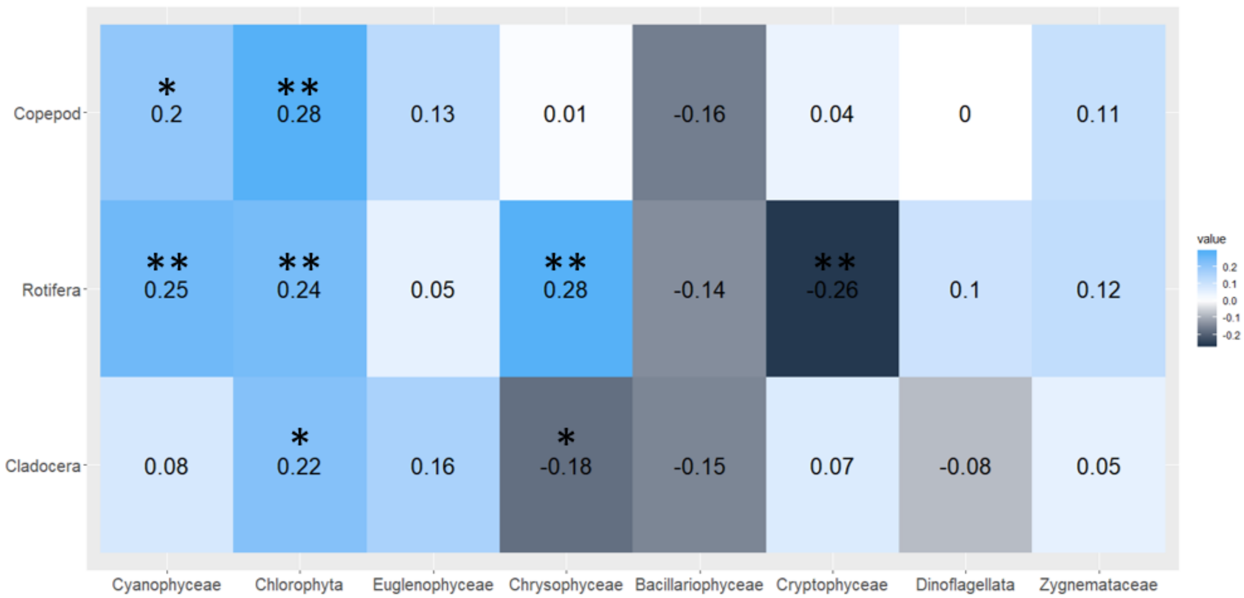

Figure 2.12. Spearman correlation matrix showing relationship between zooplankton sub-class and phytoplankton groups, based on biomass through the study period. * indicates $p<0.05$, ** indicates $p<0.01$, *** indicates $p<0.001$.

The Pearson correlation figure shows the general relationship between the water quality variables and algal biomass (Figure 2.13). In the correlation biplot there is a positive relationship between total phosphorus, total nitrogen, total dissolved phosphorus, and chloride with algal biomass. When looking at zooplankton biomass and these water quality variables, there is a positive relationship with total phosphorus and chloride with zooplankton biomass, but a negative with total nitrogen. There is also a positive relationship with algal biomass and zooplankton biomass.

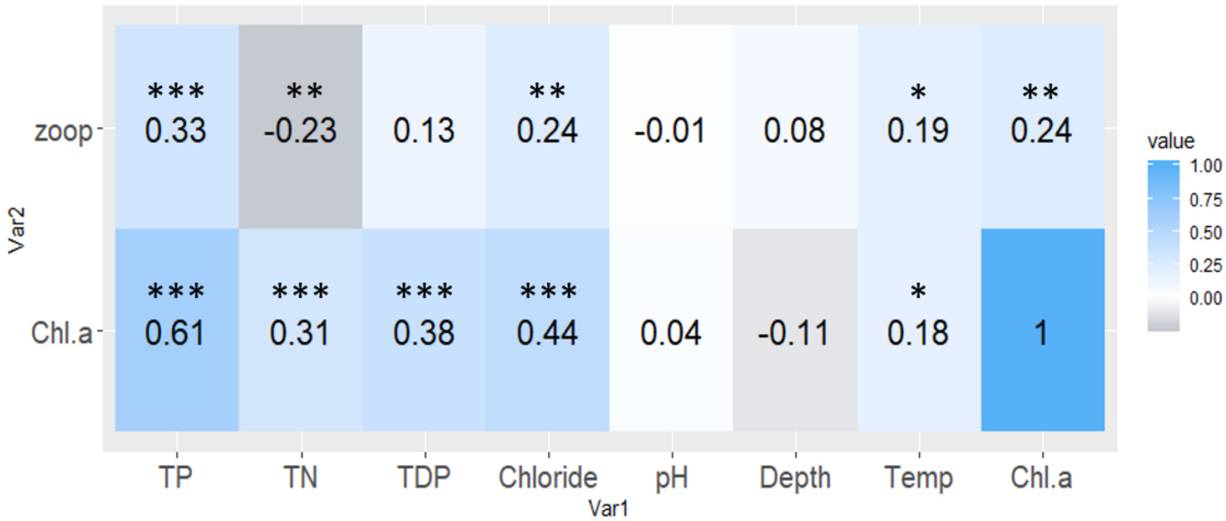

Figure 2.13. Pearson correlation matrix showing relationships between water quality with algal biomass and zooplankton biomass as represented by chlorophyll a. * indicates p<0.05, ** indicates p<0.01, *** indicates p<0.001.

When looking at a linear regression between total phosphorus and algal biomass, it is shown that total phosphorus plays a moderate positive role in predicting algal biomass (Figure 2.14).

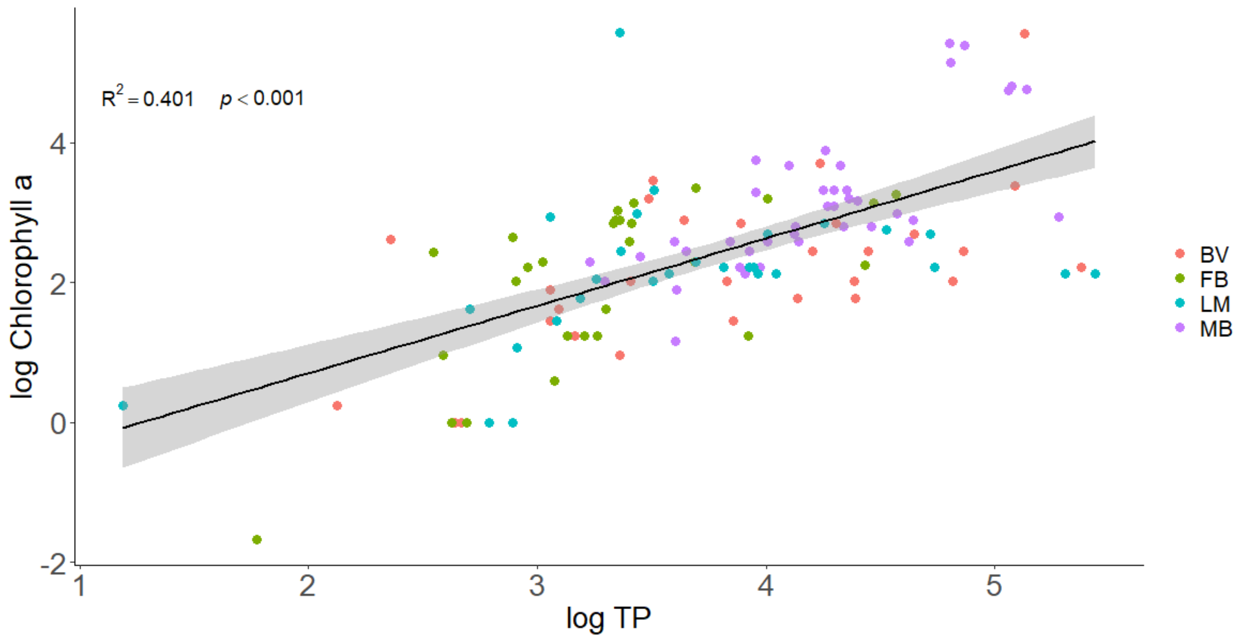

Figure 2.14. Linear regression of log transformed total phosphorus and log transformed chlorophyll a.

The linear regression with chloride shows there is also a significant positive relationship in predicting algal biomass (Figure 2.15).

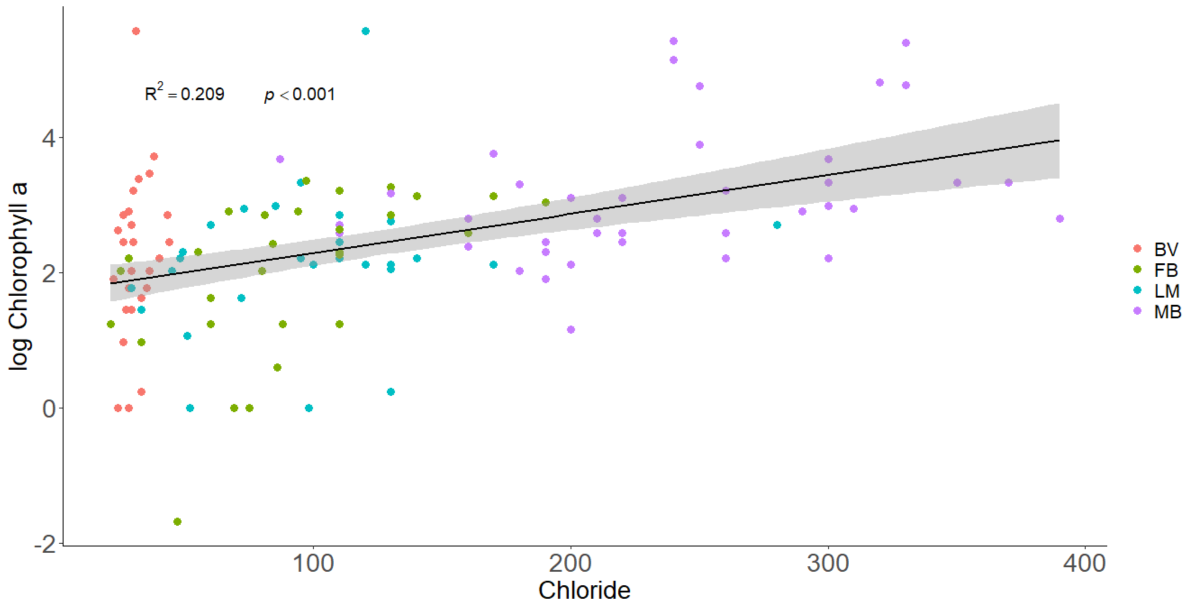

Figure 2.15. Linear regression of chloride and log transformed chlorophyll a.

Another aspect is to look at how chloride is predicting certain groups within the phytoplankton community. Chloride was found to be an important predicting variable of cyanobacterial biomass in the coastal wetlands, having a significant positive relationship (Figure 2.16).

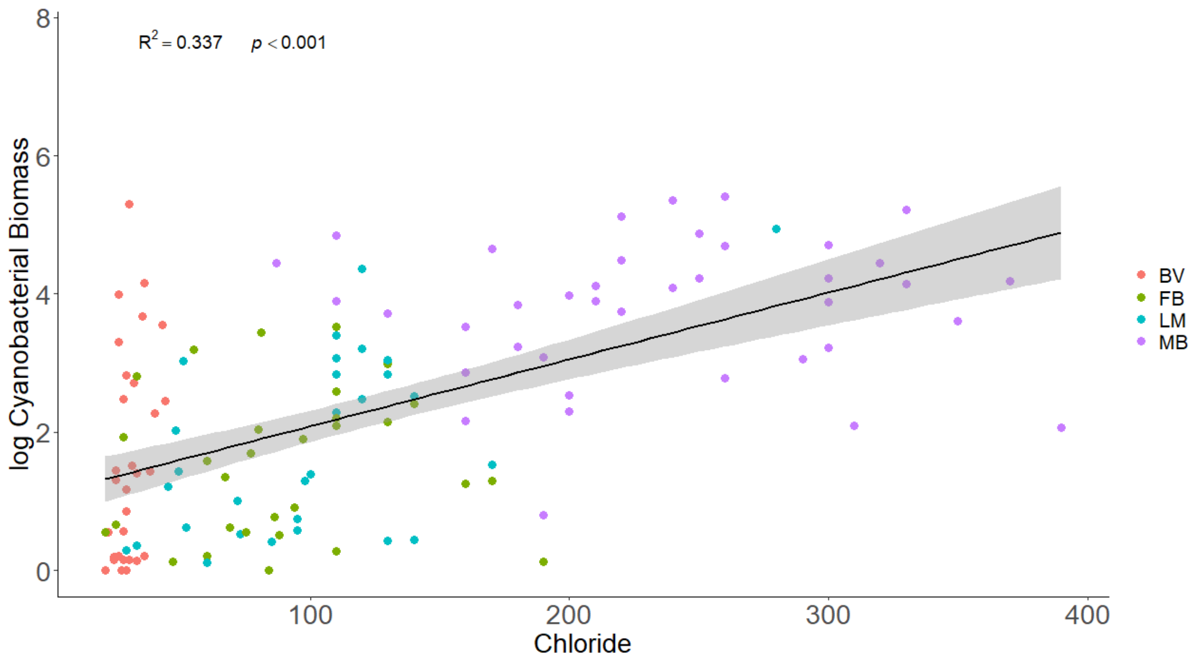

Figure 2.16. linear regression of log transformed total phosphorus and log transformed cyanobacterial biomass.

The relationship between these water quality variables and the phytoplankton community can be seen in the species richness Pearson correlation (Figure 2.17). There is a negative relationship between total phosphorus, chloride, conductivity, and total dissolved phosphorus with species richness. Conversely, there is a positive relationship with Secchi depth and depth with genus richness.

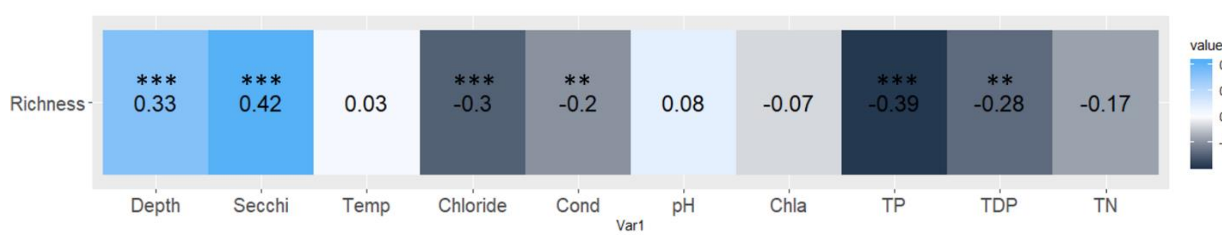

Figure 2.17. Pearson correlation between phytoplankton genus richness and water quality variables. * indicates p<0.05, ** indicates p<0.01, *** indicates p<0.001.

To further look at how the phytoplankton community is impacted by water quality variables, the redundancy analysis (RDA) shows that chloride was positively influencing Cyanophyceae, while Chrysophyceae are positively associated with depth (Figure 2.18). An RDA was used, as a detrended correspondence analysis (DCA) found the longest gradient to be less than 3, indicating a linear relationship.

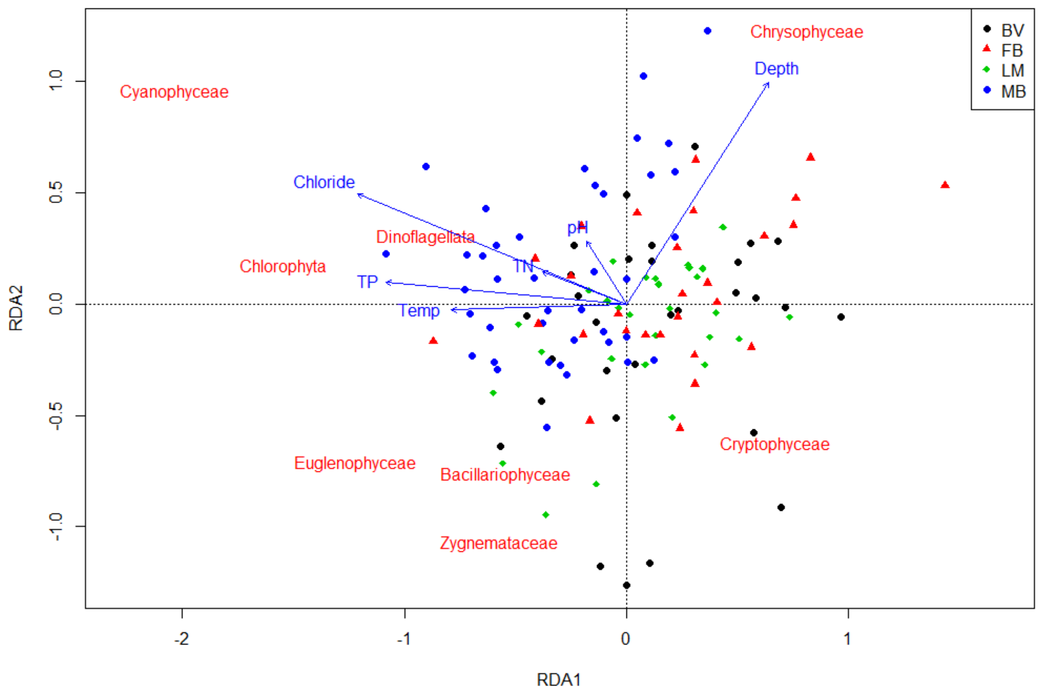

Figure 2.18. Redundancy analysis of phytoplankton community in relation to water quality variables.

It is also clear that the wetlands are somewhat separated by differences in water quality and zooplankton community structure (Figure 2.19). An RDA was used as a detrended correspondence analysis (DCA) found the longest gradient to be less than 3. Copepods and *Daphnia* are associated with high chloride and depth when looking at the top ten most abundant zooplankton groups (Figure 2.20). Lynde Marsh and McLaughlin

Bay are relatively separate from the other wetlands in terms of zooplankton in relation to water quality, while Frenchman's Bay and Bowmanville Marsh are not distinct (Figure 2.28).

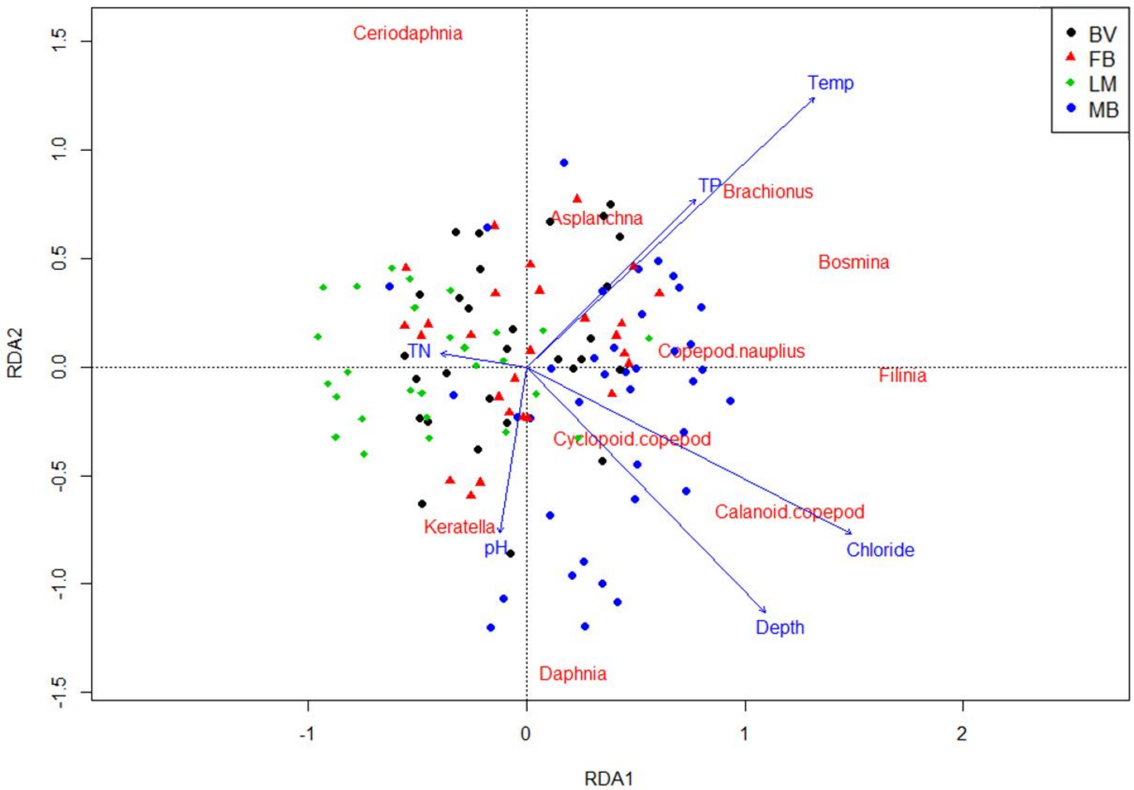

Figure 2.19. Redundancy analysis of water quality and top ten most abundant zooplankton taxa.

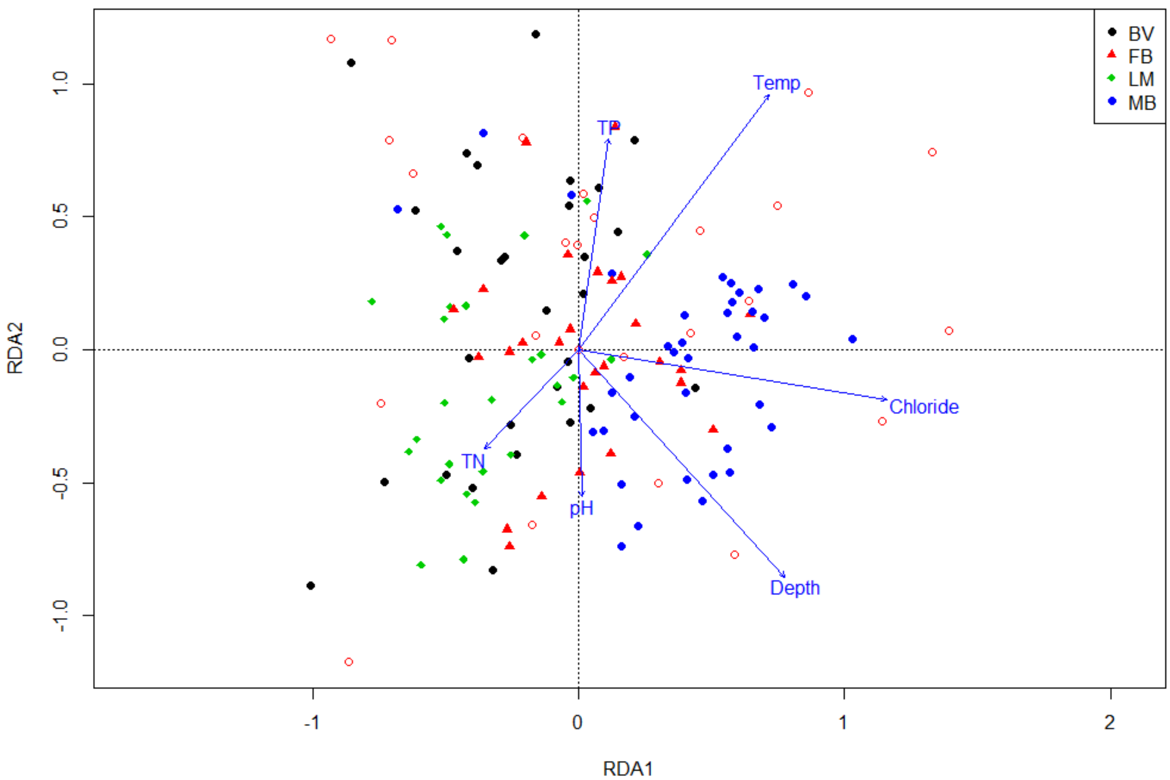

Figure 2.20. Redundancy analysis of total zooplankton community (27 taxa) in relation to water quality. Taxa names have not been included for clarity.

In the following multiple linear regression, chlorophyll a is positively predicted by total phosphorus, total nitrogen, and chloride (Table 2.4). This is similar to the simple linear regressions, but in this case, 51% of the variation in chlorophyll a values can be explained by this model.

Table 2.4. Summary statistics for multiple linear regression of water quality variables predicting chlorophyll a across wetlands.

Variable	Estimate	t-value	Sig.t	Whole model Adj. R^2	Whole Model P-value
Depth	0.176	1.341	0.182	0.506	< 0.001
Temperature	0.0344	1.544	0.125		
Chloride	0.336	2.629	<0.01		
TP	3.4192	-6.460	<0.001		
TN	2.620	4.295	<0.001		

In the following multiple linear regression, cyanobacterial biomass is positively predicted by total phosphorus, total nitrogen, and chloride (Table 2.5). With this model, 44% of the variation in cyanobacterial biomass is explained by these water quality variables.

Table 2.5. Multiple linear regression results of water quality variables predicting cyanobacterial biomass (µg mL^{-1}).

	Estimate	t-value	Sig.t	Model Adj. R^2	Whole Model p-value
Depth	0.0646	0.338	0.7363	0.44	<0.001
Temperature	0.03754	1.176	0.2419		
Chloride	0.1563	5.329	<0.001		
TP	0.6513	3.738	<0.001		
TN	1.2315	1.999	<0.001		

With the zooplankton linear regression, in this model water quality variables as well as chlorophyll a were used to predict zooplankton biomass, 11% of the variance in zooplankton biomass was explained by these variables. This model also has no variables with a significant impact, showed that this model does not explain zooplankton biomass very well. In the logistic regression with plant biomass there is a positive relationship of depth and Secchi depth with plant presence (Table 2.6). These results show that in shallow, clear waters, there is more likely to be macrophytes present. Relative abundances of macrophyte biomass where plants were collected can be seen in appendix A (Figures A16-A17).

Table 2.6. logistic regression results comparing the effect of wetland characteristics and water quality on submergent plant presence (+) or absence (-).

	Estimate	z-value	Sig.t
Depth	-0.61	-2.08	0.0379
Secchi Depth	1.04	3.22	0.0013
Temperature	0.096	0.393	0.694
Chloride	0.45	1.71	0.087
pH	0.157	0.714	0.475
Total Phosphorus	0.33	0.984	0.325
Total Nitrogen	-0.0326	-0.091	0.927
Total Dissolved Phosphorus	-0.264	-1.021	0.307
Chlorophyll a	-0.706	-1.451	0.1469

As shown in the following boxplot, there is no significant difference between zooplankton biomass based on the presence or absence of plants (P=0.164) (Figure 2.21). There was also no significant difference between algal biomass based on the presence or absence of plants (P=0.071) (Figure 2.22).

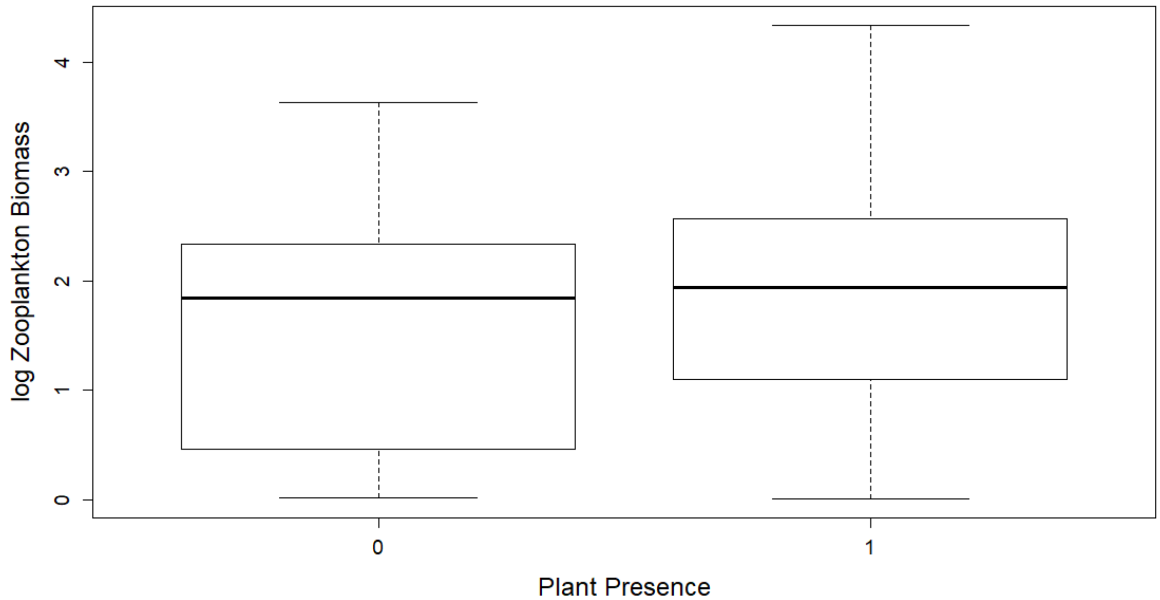

Figure 2.21. Biomass of zooplankton based on submergent plant absence (0) or presence (1).

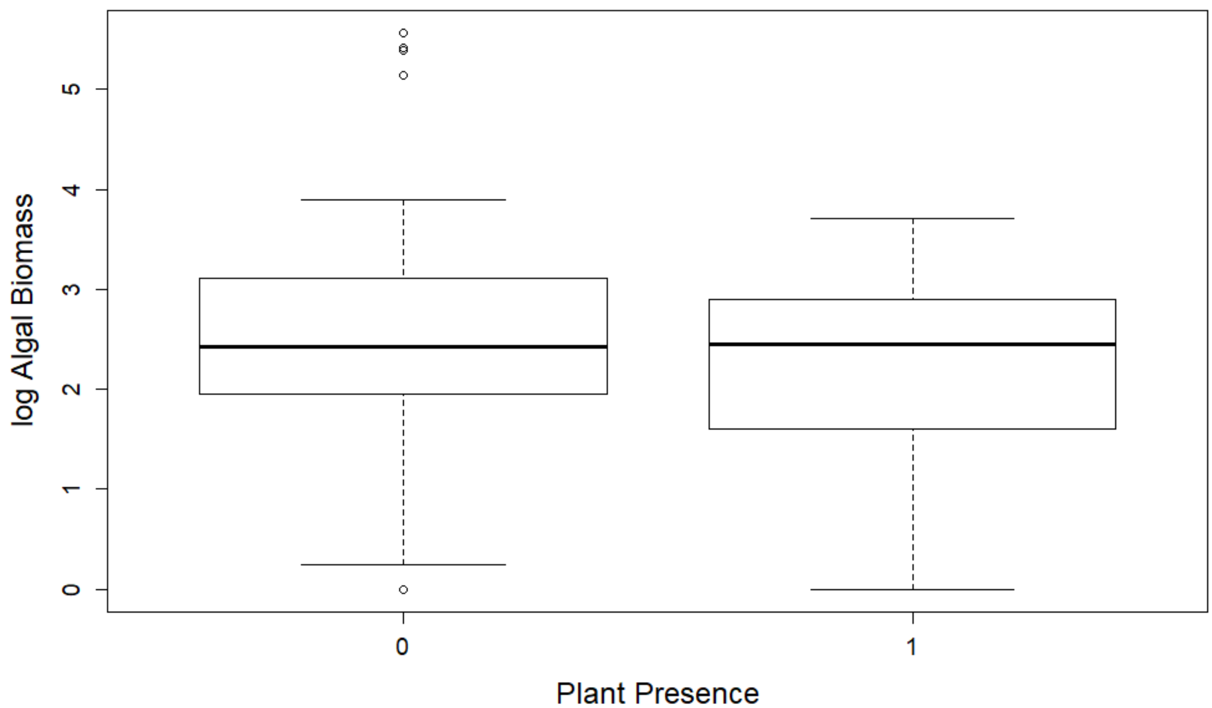

Figure 2.22. Biomass of algae (chlorophyll a) based on submergent plant absence (0) or presence (1).

Discussion

2.4.1 Characterizing Water Quality as a Function of Watershed Land-Cover

The watershed land-cover delineation clearly shows an urban gradient decreasing from west to east (Figure 2.5 and Table 2.1) where Frenchman's Bay has the highest developed land-cover (>70%) while Bowmanville Marsh has the least (<15%). When comparing wetlands in terms of water quality and their land cover, there are some unexpected trends. In more developed urban areas, there is typically increased pollution from run-off, including nutrients and chloride (Tong & Chen, 2002). In this case the more developed watershed of Frenchman's Bay has a better water quality profile, with respect to nutrient and chloride concentrations, than the less developed wetlands, especially

McLaughlin Bay. In terms of water quality, Frenchman's Bay has lower total nitrogen and phosphorus, and higher Secchi depth (i.e., water clarity) than the other wetlands.

The disconnect between urban land-cover and water quality across the wetlands is also evident when evaluating the principal component analysis (Figure 2.6). The PCA biplot does a decent job of grouping wetlands in terms of overall water quality profile. Frenchman's Bay generally has the lowest nutrients, lowest chloride levels, and highest water clarity, which deems it the least degraded wetland of this study based on those parameters. The separation among Mclaughlin Bay, Bowmanville Marsh, and Lynde Marsh appears to be most strongly influenced by chloride.

These differences in water quality condition among wetlands may in part be attributed to the hydrology of these wetlands and their connectivity to Lake Ontario. With the improved connectivity of Frenchman's Bay to the lake, chloride and nutrients are significantly diluted by lake water exchange (Toronto and Region Conservation Authority, 2009). Based on watershed land-cover alone, I would have predicted Frenchman's Bay to have the most degraded water quality profile, and having the highest chloride concentrations. In contrast, McLaughlin Bay, which is hydrologically cut-off from Lake Ontario, has the poorest water quality conditions, even though its watershed is small and dominated by natural land-cover. However, McLaughlin Bay's watershed has two large parking lots that have been heavily de-iced with road salt for over 20 years, which explains why the chloride concentrations are exceptionally high in McLaughlin Bay compared to the other wetlands (Central Lake Ontario Conservation Authority, 2013). Without an outlet, nutrients such as phosphorus would accumulate in the sediments over the last few decades as well. Lynde Marsh and Bowmanville Marsh are the most comparable wetlands based on their size and proportion of different land-covers. However, Lynde Marsh does have more developed land cover, and correspondingly higher chloride, conductivity, and lower Secchi depth. Conversely, Bowmanville Marsh also has a relatively high total nitrogen content while having the largest portion of agricultural area.

All of the wetlands in 2018 had an average total phosphorus concentration over 30µg L^{-1}, while only Mclaughlin Bay did in 2019. In aquatic ecosystems, 30µg L^{-1} of total phosphorus is classified as eutrophic as well as being at the upper limit recommended by the provincial water quality guidelines (Ministry of the Environment, 1994). Chloride was measured above the Canadian water quality guideline for chronic chloride exposure (120mg L^{-1}), but only in Lynde Marsh during 2018, and McLaughlin Bay in 2018 and 2019. This suggests that organisms living in McLaughlin Bay and Lynde Marsh may be experiencing chronic toxicity from elevated chloride concentrations (Canadian Council of Ministers of the Environment, 2011).

Watershed area is also a possible explanation for why land use percentages are not indicative of water quality, as a larger watershed typically indicates more contaminants leading into an aquatic ecosystem (Decatanzaro et al., 2009). Watershed area does not explain water quality well in this case, as McLaughlin Bay has poor water quality, though having the smallest watershed. While Frenchman's Bay has the second smallest watershed, it is also the wetland with the best overall water quality. Bowmanville marsh has the largest watershed, though appears to have fairly good water quality compared to others in this study.

2.4.2 Characterizing Plankton Communities in Coastal Wetlands

The relative abundance plots show how these phytoplankton communities differ among wetlands. There appears to be a relatively high abundance of Chlorophyceae, or green algae as well as cyanobacteria in McLaughlin Bay. Other wetlands such as Bowmanville Marsh have more filamentous green algae. Alternatively, Frenchman's Bay and Lynde Marsh appear to have more diverse phytoplankton communities at the group level. Wetlands with greater proportions of cyanobacteria are typically more degraded systems, which indicates that the phytoplankton communities in Mclaughlin Bay reflect a degraded ecosystem (Chow-Fraser et al., 1998).

In terms of zooplankton communities, the relative abundance plots showed that the small cladoceran, *Bosmina* is the most dominant zooplankton taxon across wetlands, and was especially most dominant in McLaughlin Bay in 2018. There were also relatively high *Daphnia* numbers during the 2019 season for McLaughlin Bay, while being barely present overall in the other wetlands. The large-bodied zooplankton community in McLaughlin Bay is mostly made up of *Daphnia* and calanoid copepod zooplankton, differing from the other three wetlands which consist mostly of the medium-sized *Ceriodaphnia*. It is commonly seen that in areas with more turbid waters and higher nutrients that zooplankton communities are dominated by small cladocerans such as *Bosmina*, which typically replace larger-bodied zooplankton groups such as *Daphnia* and *Ceriodaphnia* (Thomasen & Chow-Fraser, 2012).

The Spearman correlation analysis between phytoplankton groups and zooplankton sub-classes found positive relationships between Chlorophyceae and all zooplankton sub-classes. Cyanophyceae had positive relationships with Rotifers and

Copepods, but not with Cladocerans, possibly because the large filter feeding Cladocerans often cannot consume a variety of cyanobacterial species (Fulton & Paerl, 1988). It is also likely that since most of the Chlorophyceae in this study are smaller cells and colonies, they are likely easily consumed by all types of zooplankton, resulting in a significant positive relationship between these groups (Chow-Fraser & Knoechel, 1985).

2.4.3 The Role of Abiotic and Biotic Factors in Structuring Plankton Communities

The multiple linear regression shows that total phosphorus, total nitrogen, and chloride in combination explain 52% and 46% of chlorophyll a and cyanobacterial biomass respectively. This shows that these water quality variables are explaining a significant portion of the increase in cyanobacteria, and algal blooms in general. In these models, total phosphorus, total nitrogen, and chloride are highly significant in explaining the variation in total algal (as chlorophyll a) and cyanobacterial biomass. These variables are all pollutants typically associated with poor water quality caused by human activities (Tong & Chen, 2002). Nutrients are often associated with agriculture and urbanization, and have been shown to promote algal dominance in wetlands and inhibit macrophyte colonization (Thomasen & Chow-Fraser, 2012).

In the multiple linear regression analysis, total phosphorus is a highly significant positive predictor of algal biomass. This indicates that the coastal wetlands in this study are phosphorus-limited, meaning that with increasing phosphorus supply, there is a corresponding increase in algal biomass. Based on the eutrophic status of these wetlands and their phosphorus concentrations, they are certainly at risk of experiencing algal blooms (Watson et al., 2016). What is also interesting in these results is that chloride positively influenced total algal biomass and cyanobacterial biomass in these systems.

This is concerning because the use of road salts can increase the chloride concentrations in these ecosystems. When aquatic environments become too saline, it can lead to decreased large-zooplankton abundance, therefore leading to decreased grazing rates, and an increase in phytoplankton biomass (Lind et al., 2018). Another concern is that cyanobacteria are known as a more salt and chloride tolerant group of algae, which means that as chloride increases with salinity, these potentially harmful algae will also increase (Apte et al., 1987).

Phytoplankton richness is another important biological metric because it can indicate that areas with more genera are generally more stable communities (Balmford et al., 1996). There was found to be a significant positive relationship between depth and Secchi depth with species richness. It has been shown, as in this study, that in areas with higher Secchi depth there is typically greater phytoplankton richness (Karacaoğlu et al., 2006). There also were significant negative relationships between TP, TDP, and chloride with genus richness. It has previously been shown that there is a negative relationship between nutrients and richness in nutrient rich lakes (Fontúrbel & Castaño-Villa, 2011). Increasing salinity can lead to a decrease of phytoplankton richness and diversity as well (Flöder et al., 2010).

The redundancy analyses showed that chloride is a strong driver of community composition, especially positively influencing Cyanophyceae as seen in the linear regressions. When assessing differences between sites, McLaughlin Bay and Frenchman's Bay appear different from each other more than any other site, which is similar to the relationship earlier when it was shown that McLaughlin Bay and Frenchman's Bay were different in water quality. In the top ten abundant zooplankton

RDA, *Ceriodaphnia* was negatively associated with chloride and depth, showing that this genus is present mostly in shallow, low salinity environments. It has been found previously that *Ceriodaphnia* are relatively sensitive to chloride, when compared to *Daphnia*, which may help explain why *Daphnia* are present in McLaughlin Bay (a more saline environment), and *Ceriodaphnia* are not (Harmon et al., 2003). When looking at the difference in sites, McLaughlin Bay and Lynde Marsh differ in terms of community composition more than any other wetlands, potentially attributable to their various differences in water quality and phytoplankton community structure.

The logistic regression analysis showed that submergent plant presence/absence was negatively associated with depth. This is not surprising because it indicates that as light penetration diminishes with depth, plants are less likely to persist with low light levels. This is further supported as Secchi depth has a positive relationship with plants, indicating that as water clarity increases, there is a greater chance for light penetration to support submergent plant growth. In turbid, degraded wetlands, it is often difficult for plants to colonize (Lacoul & Freedman, 2006). It was also found that chlorophyll a does not have a significant effect on plant presence, indicating that plant presence may be limited by physical variables such as light availability instead of competitive relationships with algal communities.

In terms of zooplankton biomass, there is no difference between zooplankton abundance whether there are submergent plants present or absent, even though zooplankton communities have often been associated with macrophytes (Lougheed & Chow-Fraser, 2002). In order to understand how differing depths may play a role in abundances of plankton communities as a habitat feature, it was assessed by Pearson correlation analysis. There was not a significant relationship between depth and algal abundance. When looking at zooplankton abundance, there was not a relationship with depth either. This means that the depth of sites between wetlands are not significantly altering plankton communities. The evident differences between the abundance in these different communities are apparently related to water quality variables, and not physical depth of these wetlands.

When looking at the differences among these wetlands in water quality and the biological communities, there are some major differences that do not appear to be related to the types of land-use or size of the watersheds. There are some differences among these wetlands that may be partially attributed to the hydrological connectivity to Lake Ontario, as more connectivity appears to be linked to improved water quality in some wetlands, while no connectivity is linked to poor water quality in others. One of the major concerns in these wetlands is the water quality and food web interaction. When looking at the relationship between chloride and algae as well as cyanobacteria, this relationship is concerning as chloride levels rising in developing areas can potentially alter entire plankton communities. As nutrient concentrations are a major concern in many restoration efforts, it appears while they are important, that chloride concentrations may also play important roles in the persistence of cyanobacterial blooms. Factors such as hydrology and chloride levels should be more strongly considered in future restoration efforts as their role in degrading water quality and promoting algal blooms were evident in this study.

Chapter 3: Evaluating the Suitability of McLaughlin Bay as a Candidate Wetland for Biomanipulation Restoration

3.1 Introduction

Human activities on the landscape can alter aquatic ecosystems through nutrient and contaminant loading from agriculture and urbanization. As a result, water quality and biological communities in aquatic ecosystems can become degraded and impaired, respectively. As human populations grow, particularly in the Great Lakes region, these changes are having profound effects on aquatic ecosystems in the region (Cvetkovic & Chow-Fraser, 2011). This has been shown in the Great Lakes coastal wetlands as many, especially in Lake Ontario, are considered degraded (Cvetkovic & Chow-Fraser, 2011). When conditions worsen in these wetlands, nutrient input can increase and lead to algal blooms (Watson et al., 2016).

Some restoration strategies have been proposed as a means of mitigating the shift and maintenance of algal dominated systems. One type of restoration approach is known as biomanipulation (Shapiro 1990). Based on trophic cascade theory (Carpenter et al. 1985), biomanipulation involves the addition of a native piscivorous (top-predator) fish species to a lake or wetland ecosystem to induce "top-down" control on algal biomass. The anticipated increased piscivory is expected to lower planktivorous (i.e., zooplankton-consuming species) fish abundance. Fewer planktivores releases predation pressure on large-bodied zooplankton, and in turn, high large-bodied zooplankton abundance results in lower algal abundance via increased zooplankton grazing (Carpenter et al., 1985;

Lathrop et al., 2002). This strategy has been shown to be effective in certain scenarios (e.g., mesotrophic ecosystems) where conditions are appropriate (Lathrop et al., 2002).

The structure of the aquatic food web, as well as abiotic factors such as nutrient conditions and physical variables, can dictate the effectiveness of biomanipulation (Angeler et al., 2003). A biomanipulation to improve water quality was conducted in Cootes Paradise Marsh, a coastal wetland at the far western end of Lake Ontario (Thomasen & Chow-Fraser, 2012). This project involved removing invasive common carp (*Cyprinus carpio*) from entering the wetlands. Post-biomanipulation, it was determined that although water quality had improved in the marsh, this improvement was relatively ineffective in terms of algal biomass. The main issue was that bottom-up effects (i.e., nutrient supply) in the wetland was not controlled, especially in open water areas. This indicates that when nutrient inputs are too high, phytoplankton growth likely cannot be controlled through food web interactions (Thomasen & Chow-Fraser, 2012).

Agasild et al. (2007) showed that in ecosystems with mainly inedible algae (i.e., filamentous algae and cyanobacteria), there was a dominance of small zooplankton (rotifers and small cladocerans), which are grazers that have a low impact on total phytoplankton biomass. When looking only at small sized phytoplankton, it was evident that there are multiple interactions as phytoplankton composition can dictate the types of zooplankton that are present, and zooplankton can shift algal community composition and biomass through selective grazing (Agasild et al., 2007). This study showed that the characteristics of the algal community may play a major role in the success of a potential biomanipulation, as more inedible algae can lead to less effective grazer communities.

Some studies have shown that biomanipulation can be a successful approach in wetlands with little to no physical disturbance (such as frequency and duration of flooding) or have primarily internal nutrient-loading (Angeler et al., 2003). McLaughlin Bay has a very small watershed and has not been open to Lake Ontario in over a decade, so it likely does not experience significant flooding or change in water-level from the lake. The small watershed, as well as presence of common carp, which resuspend sediment into the water column, suggests that its high nutrient concentration may be associated with internal loading from bioturbation of the sediments (Central Lake Ontario Conservation Authority, 2013).

While the influx of nutrients plays an important role in structuring plankton communities, chloride may also be a concern in developed watersheds. The use of de-icing salts has been a concern in developed areas and over 90% of sodium chloride introduced into aquatic ecosystems in developing regions can be attributed to de-icing salts (Kelly et al., 2008). Previous research has also shown a negative relationship between de-icing salts and overall zooplankton health at mid (470 mg L^{-1}) and high (780 mg L^{-1}) sodium chloride concentrations. High salt environments lowered zooplankton abundance up to 70% (Jones et al., 2017). This lowered abundance of zooplankton resulted in an increase of phytoplankton abundance through shifted food web interactions (Jones et al., 2017).

Cyanobacteria have been shown to have relatively high salt tolerance compared to other freshwater algae, as cyanobacterial cells actively export sodium and chloride ions from their cells to reduce salt stress (Apte et al., 1987; Hagemann, 2011; Tonk et al., 2007). When water quality positively influences cyanobacteria, it also negatively influences the zooplankton communities, resulting in very strong bottom-up effects. This could be an issue in the context of a biomanipulation where the interactions between organisms are strongly controlled by the water quality conditions present in the system (Jones et al., 2017). Through interactions where cyanobacterial biomass is increased, zooplankton abundance often decreases, thus with more pollutants supporting cyanobacteria, zooplankton communities will also struggle as it is poor quality food for zooplankton.

Some reasons why cyanobacteria are poor food for zooplankton include: colonies or filamentous cyanobacteria are difficult to consume, many cyanobacteria are toxic, and generally have low nutritional value for zooplankton (De Bernardi & Giussani, 1990). A study by Karjalainen et al. (2005) showed there is a transfer of cyanobacterial toxins to some zooplankton species, which could harm organisms throughout the food web such as fish, either directly or indirectly through biomagnification. One important note is that small zooplankton are less affected by colonies and filamentous taxa compared to larger bodied zooplankton. When cyanobacteria bloom, this can cause a change in the make-up of the zooplankton community, specifically lowering the amount of large filter feeding cladocerans (Fulton & Paerl, 1988).

Other studies suggest the success of biomanipulation depends on controlling bottom-up effects, such as nutrient supply as it has been shown that at very high nutrient concentrations, this approach is not always successful (Benndorf, 1990). Previous research has shown that even in a case where biomanipulation by introduction of a native piscivorous fish was successful in controlling planktivore biomass, and enhancing large-bodied zooplankton populations, there was still excessive cyanobacterial biomass. This study suggests that when bottom up control of algal blooms is very strong, phytoplankton do not decrease because the community typically shifts to colonial phytoplankton, which are inedible to the majority of zooplankton in the system (Böing et al., 1998). A study by Vanni (1987) showed that even with the removal of planktivorous fish and an increase in larger cladoceran species, there was still a positive influence from nutrient enrichment on the primary producers. Removing planktivorous fish did not alter phytoplankton density, and the phytoplankton community shifted to grazing-resistant phytoplankton with gelatinous sheaths. Even though there were effects from zooplankton on the phytoplankton community, this approach promoted some of the grazing resistant algae and overall abundance was not altered (Vanni, 1987). In order for a biomanipulation to be successful, one must consider multiple aspects of the food chain, as top down and bottom up controls play major roles in phytoplankton biomass and community structure.

McLaughlin Bay has been earmarked by the Central Lake Ontario Conservation Authority (CLOCA) as a candidate wetland for restoration potentially using biomanipulation. In order to determine if this is an appropriate restoration approach for McLaughlin Bay, my main research objective was to evaluate the suitability of McLaughlin Bay as a candidate for biomanipulation restoration. This included an assessment of the seasonal plankton communities and water quality conditions over three study years (2017-2019), and assessing relationships between biological communities as well as with water quality variables. In order to be a suitable wetland for biomanipulation, McLaughlin Bay would need to have consistent year-year (1) large-bodied zooplankton in high abundance, (2) edible size-class phytoplankton available to support large-bodied zooplankton grazers, and (3) Mesotrophic (moderate nutrient) conditions.

3.2 Methods

3.2.1 Study Design

McLaughlin Bay (Oshawa, Ontario) was the focal wetland for this study (Figure 3.1). Sites were sampled monthly from May to September in 2017-2019. Initially, when the study first started in May, 2017, three sites were chosen to sample the gradient of water from the inlet to near the barrier beach. However, a fourth site was added in June 2017 in the eastern area of the wetland in order to get a better assessment of the entire wetland for the remainder of the study period (Figure 2.2).

Figure 3.1. McLaughlin Bay, Oshawa, Ontario with surrounding delineated watershed. Natural land cover seen in green, agricultural land cover seen in yellow, and developed seen in red.

3.2.2 Data Collection

Methods for water quality and biological samples are described in the methods section of Chapter 2 (page 17). In brief, water quality parameters measured on site included site depth (m), Secchi depth (m), pH, temperature (^oC), dissolved oxygen (mg L^{-1}), and conductivity (µS cm^{-1}). Water samples were collected in 1 L Nalgene bottles and kept in iced coolers to transport to the lab for analysis. Water samples were used to measure spectrophotometric turbidity, chlorophyll a (µg L^{-1}), total dissolved phosphorus (µg L^{-1}), and total phosphorus (µg L^{-1}). Measurements of chloride (mg L^{-1}), total Kjeldahl nitrogen (mg L^{-1}), ammonia + ammonium (mg L^{-1}), nitrite (mg L^{-1}), and nitrate (mg L^{-1}) were analyzed by an accredited lab (SGS Canada Inc., Lakefield, Ontario). Biological samples collected included macrophytes, phytoplankton, and zooplankton, as previously described. Macrophytes were not collected in 2017 or in September of 2018 as sampling in this period was done in a smaller boat where collection was not possible.

3.2.3 Data Analysis

Landscape metrics from McLaughlin Bay's watershed (area and percent land-use) was determined using the open source mapping software QGIS (QGIS Development Team, 2019). Land cover information was calculated using the CLOCA Land cover open data set (CLOCA 2017). Statistical analyses were performed using the R statistical platform (version 3.6.1, R Core Team, 2019). The relationships between water quality variables and their impact on community structure (e.g., genus richness) was determined because genus richness is an important determinant of community health and function (Bajer et al., 2016).

Average monthly total phosphorus concentrations were shown to compare to provincial water quality objectives concentrations of 30μg L^{-1} in freshwater environments (Ministry of the Environment and Energy, 1994). Average monthly chloride concentrations were also shown to compare to Canada's water quality guidelines that state any chloride concentration over 120mg L^{-1} is considered unsafe for chronic (30-day) exposure of aquatic life (Canadian Council of Ministers of the Environment, 2011).

Genus richness was calculated as phytoplankton were identified to genus and this richness metric can provide useful information on community composition (Balmford et al. 1996). Richness was calculated using rarefied species richness to adjust for number of individuals sampled (Sanders, 1968). Pearson correlation was used to assess relationships between water quality variables and genus richness. TP, TDP, and TN were log transformed before analysis in order to fit parametric assumptions. ANOVA was used in order to compare chlorophyll a values among years. Tukey post-hoc test was used to test for individual differences among years of chlorophyll a. Chlorophyll a was log transformed in order to fit parametric assumptions.

Sizes of edible and inedible algae were based off of Chow-Fraser & Knoechel (1985) as they determined that algae >30μm are considered inedible, including individual cells, colonies, and filaments. To see if phytoplankton fit within the appropriate size classes, a t-test was used to assess for differences in phytoplankton between edible (<30 μm) and inedible (>30 μm). Values for the t-test were log transformed before analysis in order to fit parametric assumptions. Phytoplankton communities were also visualized based on measured sizes classes, in filamentous, small and large colonial and by general cell size (<30 µm and >30 µm) in order to assess specific types of algal size classes present.

A multiple linear regression was used to predict biomass of plankton communities. Variance inflation factor (VIF) was used to test for collinearity to determine if results were influenced by relationships between independent variables. Any variable with a value above 4 would have been removed from the regression, but no variables exceeded the VIF cut-off (Pan & Jackson 2008). To predict chlorophyll a, all variables in the model were used aside from dissolved oxygen and pH, this was done because DO and pH are positively influenced by phytoplankton growth and therefore may not act as adequate predictor variables for algal biomass. For zooplankton, these variables and chlorophyll a were used and nutrient variables (TP and TN) were excluded since they do not directly impact zooplankton. In order to achieve multivariate normality, the variables TP, TDP, TN, chlorophyll a, and zooplankton biomass were log transformed. In total, the Mclaughlin Bay data set had a sample size of n = 59.

3.3 Results

The land-use profile is shown in Figure 3.1 and summarized in Table 3.1. McLaughlin Bay has a predominantly natural watershed, with some influence from developed land as well. The watershed area is small, totaling less than 2 km^2.

Table 3.1. Area and percent land use summary statistics of McLaughlin Bay watershed.

Land Use	Area (km^2)	Percent Land Use
Natural	1.25	68%
Developed	0.437	23.9%
Agricultural	0.149	8.1%

Descriptive statistics summarizing the collected water quality variables are presented in Table 3.2. Some observed variables had a lot of variation that can be seen in differences across both time and by site in some cases.

Table 3.2. Descriptive statistics for McLaughlin Bay Water Quality over study period of 2017-2019.

Variable	Mean	Standard Deviation	Min	Max
Depth (m)	1.55	0.48	0.50	2.50
Secchi Depth(m)	0.48	0.19	0.10	1.00
Conductivity (µS cm^{-1})	1841	646	1102	3108
Dissolved Oxygen (mg L^{-1})	9.83	2.27	3.29	15.23
Temperature (°C)	21.7	3.5	13.5	27.1
pH	8.11	0.53	6.77	9.51
Turbidity (abs @750nm)	0.030	0.0452	0.0037	0.34
Total Phosphorus (µg L^{-1})	93.44	51.19	24.29	253.37
Total Nitrogen (mg L^{-1})	0.45	0.28	0	1.38
Total Dissolved Phosphorus (µg L^{-1})	4.17	4.11	0	20.02
Chloride (mg L^{-1})	251	69	87	390
Chlorophyll a (µg L^{-1})	33.74	46.68	2.17	223.7

Average total phosphorus and chloride levels over the study period are shown in figure 3.2 and figure 3.3 respectively.

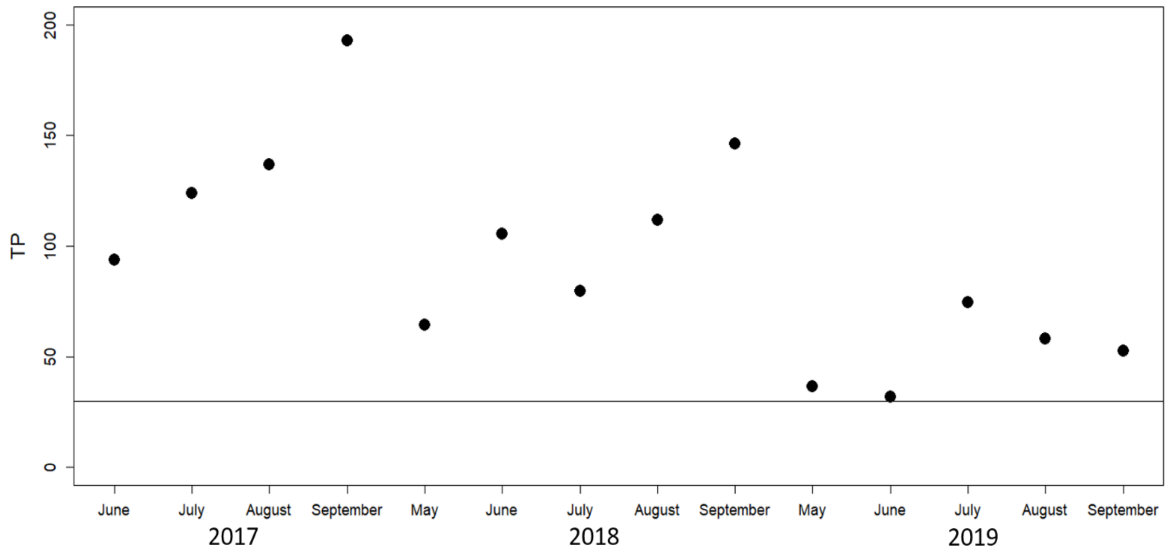

Figure 3.2. Average monthly total phosphorus values over three years of study. Phosphorus concentration marked at provincial water quality objective of 30 µg L^{-1}.

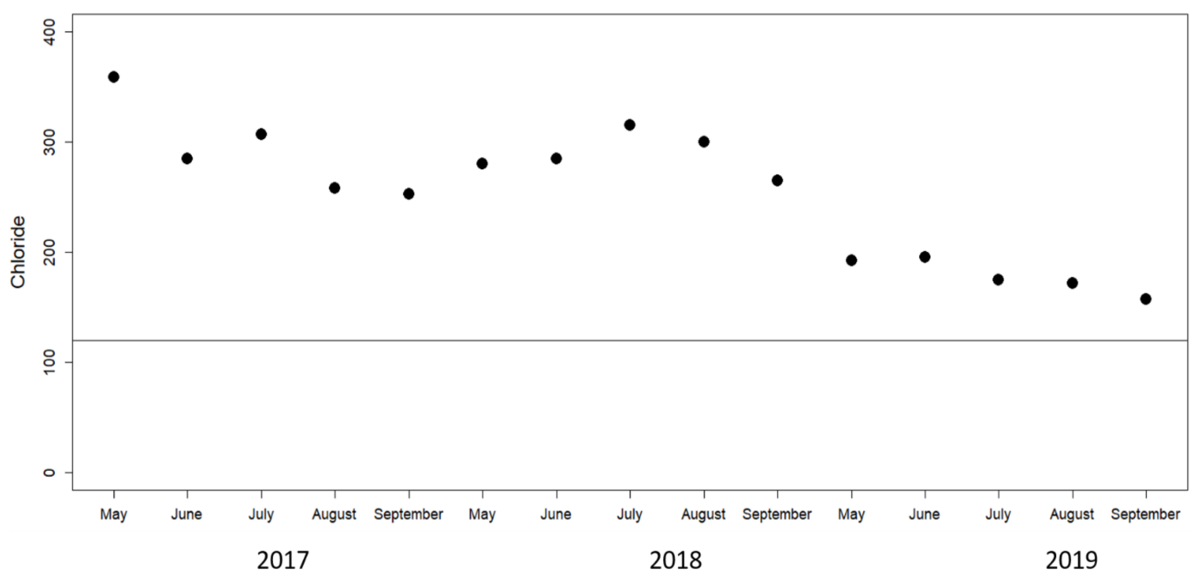

Figure 3.3. Average monthly chloride values over three years of study. Chloride concentration marked at Canada water quality guideline of 120 mg L^{-1}.

The principal component analysis (PCA) shows how the water quality variables associate with each year of study (Figure 3.4). There is a lot of variation across the three-year study with a total of 48.4% of the variance being explained by the first two axes. Sites in 2019 were negatively associated with phosphorus and chloride while in 2017, sites are positively associated with these variables. In 2018, many sites are associated with nitrogen and TDP. The PCA shows nutrients and chloride are negatively associated with sites in 2019, while dissolved oxygen is positively associated. Chlorophyll a values are at their lowest in 2019 (Figure 3.5). A one-way ANOVA indicated that chlorophyll a significantly differed among years of the study ($p=0.002$). Tukey post-hoc analysis tested differences among years in chlorophyll a, and found that 2019 was significantly lower than 2018 (Figure 3.5).

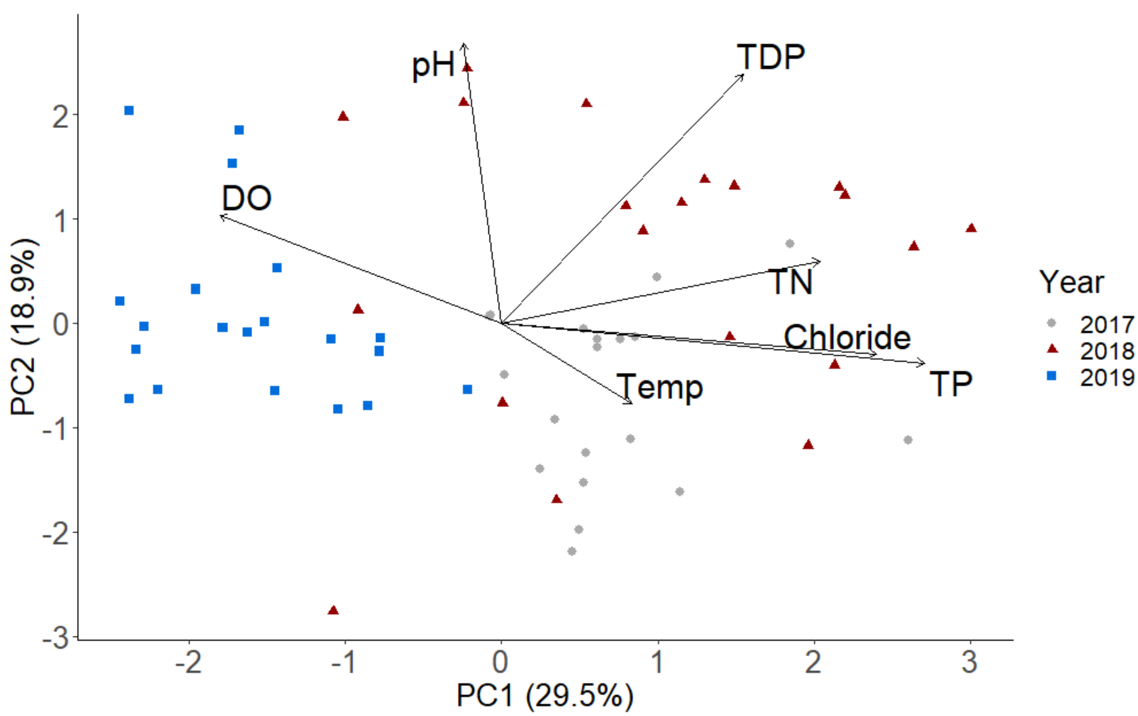

Figure 3.4. Principal component analysis performed on water quality variables from four sites spanning across monthly sampling from May-September in 2017-2019.

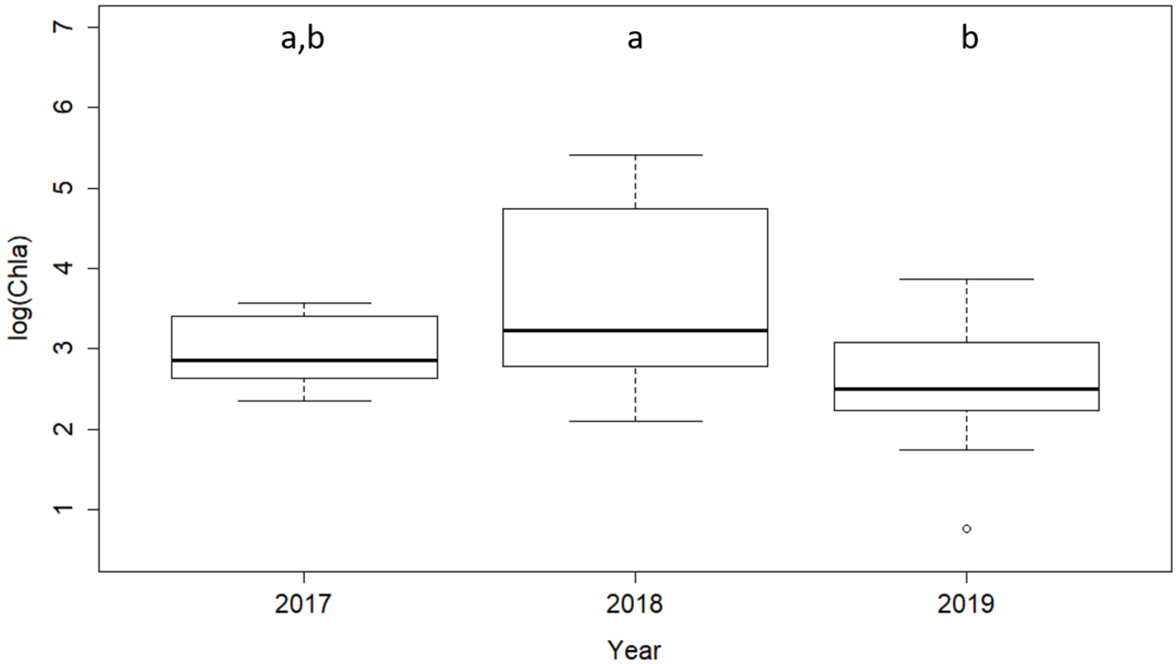

Figure 3.5. Boxplot of log transformed chlorophyll a values in 2017, 2018, and 2019 sampling years in McLaughlin Bay at all four sites. ANOVA indicated that chlorophyll a significantly differed among years (p=0.002).

Pearson correlation analysis of water quality variables and phytoplankton genus richness determined that only Secchi depth had a statistically significant positive relationship with richness (p=0.014, r=0.32) (Table 3.3).

Table 3.3. Pearson correlation with water quality variables and phytoplankton genus richness.

Variable	r	p-value
Chloride	0.12	0.362
Depth	0.25	0.0565
DO	0	0.9818
pH	0.12	0.364
Secchi	0.32	0.014
Total dissolved phosphorus	-0.03	0.8514
Temperature	-0.21	0.119
Total nitrogen	-0.05	0.726
Total phosphorus	-0.14	0.3035

The t-test between different sizes of algal communities showed that there was a statistically significant difference between edible and inedible algal biomass, where inedible algal biomass was higher ($DF=115.02$, $t= -4.574$, $p<0.001$) (Figure 3.6).

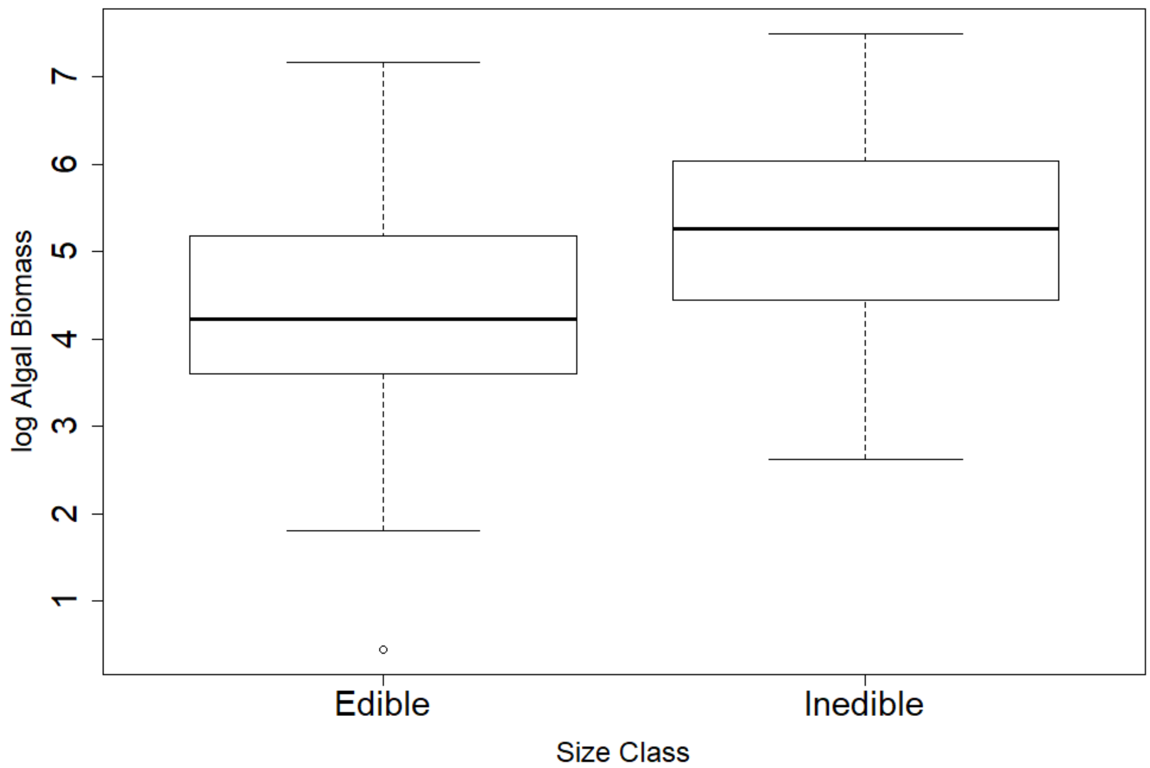

Figure 3.6. Boxplot showing log transformed biomass of edible (<30µm and small colonies) and inedible (>30µm, large colonies, and large filamentous) algae.

Most of the algae fit within the large cell, large colony or filamentous size class. There were a notable proportion of algal cells in <30µm size class, but very few in small colonial form (Figure 3.7).

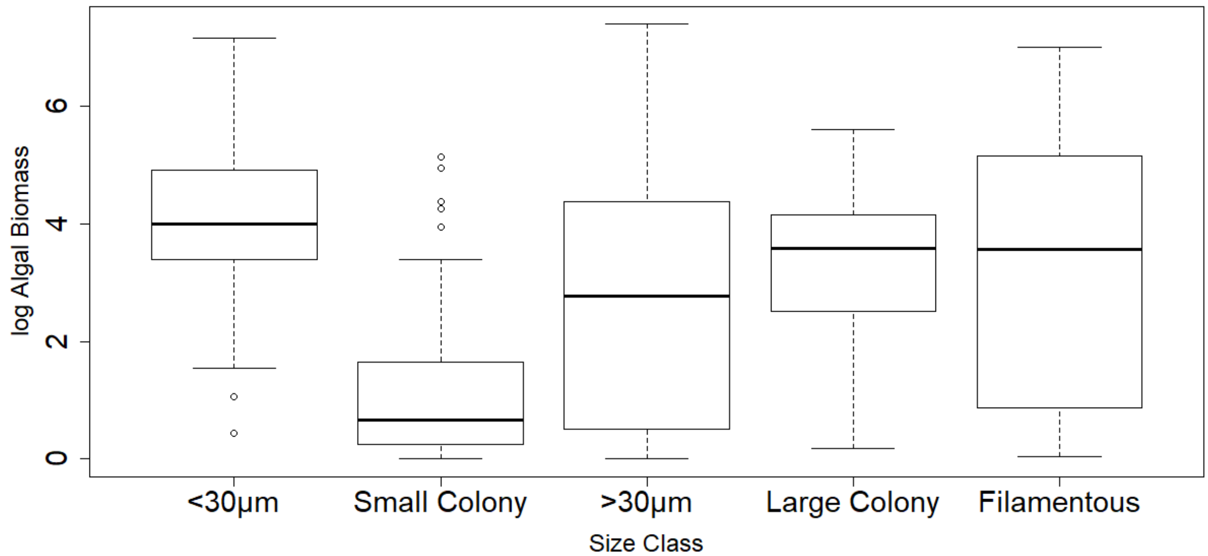

Figure 3.7. Boxplot showing size classes of phytoplankton in McLaughlin Bay. Total biomass calculated from sum phytoplankton biomass of all sites from samples taken in 2017-2019.

When running a multiple linear regression on chlorophyll a, total phosphorus and total nitrogen had a significant positive relationship (Table 3.4). Temperature had a significant positive relationship ($p<0.05$) with algal biomass as well. The multiple linear regression explained just over 60% of the variation in algal biomass as chlorophyll a ($R^2=0.605$, $p<0.001$).

Table 3.4. Multiple linear regression results of water quality variables predicting chlorophyll a.

	Estimate	t-value	Sig.t	Whole model Adj. R^2	Whole Model P-value
Depth	0.03728	0.206	0.8375	0.605	< 0.001
Temperature	0.0712	2.682	0.0102		
Chloride	-0.00072	-0.512	0.6109		
Total Phosphorus	0.6033	2.751	<0.01		
Total Dissolved Phosphorus	0.09291	0.944	0.3501		
Total Nitrogen	2.7218	4.579	<0.001		

The multiple linear regression to explain zooplankton biomass determined that pH and dissolved oxygen were significant independent variables (Table 3.5). This model explains 24% of the variation in zooplankton biomass ($p<0.01$).

Table 3.5. Multiple linear regression results of water quality variables predicting zooplankton biomass.

	Estimate	t-value	Sig.t	Whole model Adj. R^2	Whole Model P-value
Depth	0.6581	1.726	0.091	0.242	< 0.01
Temperature	-0.07280	-1.208	0.233		
Chloride	0.001242	0.430	0.669		
Dissolved Oxygen	0.32007	-3.552	<0.001		
pH	-6.0376	-2.293	0.027		
Chlorophyll a	-0.42672	-1.986	0.053		

In the phytoplankton relative abundance plots of the top 15 most abundant genera, the top two were the cyanobacterial taxa *Anabaena* and *Microcystis*, which have the greatest abundance overall across the study period (Figure 3.8).

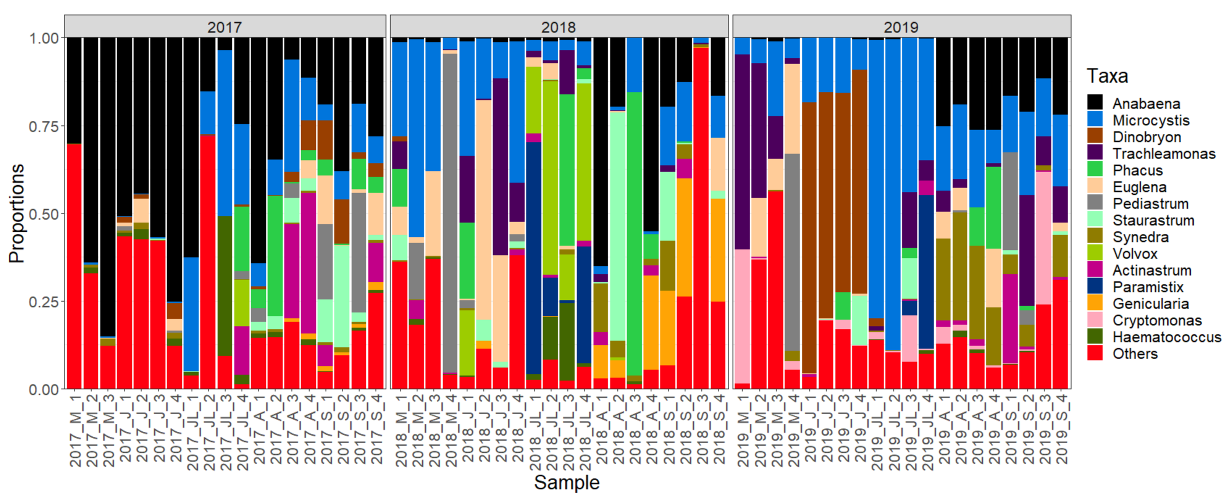

Figure 3.8. Relative abundance chart of phytoplankton genera from 2017-2019 sampling years. X-axis shows year, month, and site number. Y-axis is relative proportion of algae.

In the zooplankton relative abundance plot (Figure 3.9), there is a large proportion of small filter feeding zooplankton overall (*Bosmina*, copepod nauplii). Some of the more efficient filter feeding zooplankton, such as *Daphnia*, are present mostly in the earlier months (May and June) in each year of the study. Calanoid copepods are fairly abundant through 2017 and 2019, while being barely present in 2018.

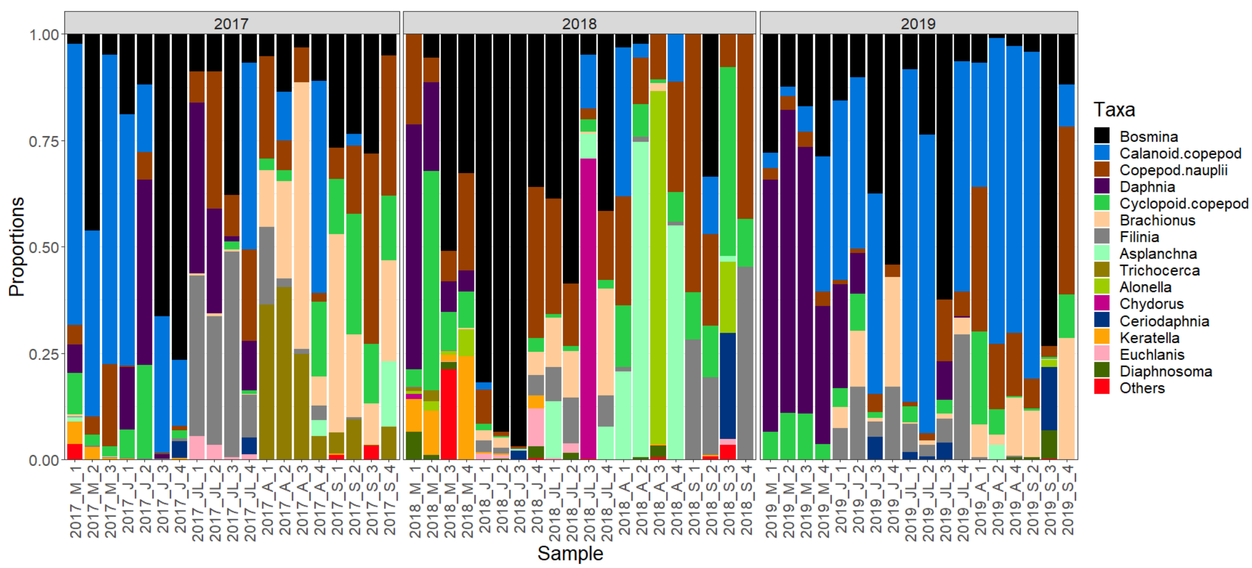

Figure 3.9. Relative abundance plot of zooplankton communities from 2017-2019 sampling years. X-axis shows year, month, and site number. Y-axis is relative proportion of zooplankton.

3.4 Discussion

One of the important aspects to consider in a biomanipulation restoration project is water quality, as this can be the major factor of whether a biomanipulation is viable as a long-term solution. If a wetland is considered degraded, then biomanipulation may not be a reasonable approach since bottom-up effects on algal growth may be too strong to control through food web interactions. In terms of phosphorus, provincial water quality objectives suggest that total phosphorus concentrations be below 30µg L^{-1} in freshwater environments (Ministry of the Environment and Energy, 1994). My results indicate that McLaughlin Bay is very degraded in terms of phosphorus concentrations, with an average of 93.4µg L^{-1}. There were only two occasions over the entire study period where phosphorus was below the provincial guidelines in McLaughlin Bay. This value is in the eutrophic range, but close to the hyper-eutrophic range of 100 µg L^{-1} set by the Canadian water quality guidelines (Canadian Council of Ministers of the Environment, 2004).

Another aspect that deems McLaughlin Bay a degraded wetland ecosystem is its chloride concentrations. Canada's water quality guidelines state that any chloride concentration over 120mg L^{-1} is considered unsafe for chronic (30-day) exposure of aquatic life. Chloride was very high in McLaughlin Bay, with an average value of 251 mg L^{-1} over the study period, with only occasion was the concentration below the 120mg L^{-1} guideline over the study (Canadian Council of Ministers of the Environment, 2011). Having such high chloride concentrations in McLaughlin Bay is assumed to be harmful to sensitive aquatic life at each trophic level. This can cause issues in biodiversity since some organisms cannot survive in high salinity. It has been shown that the use of road salts can have direct toxicity on freshwater species and zooplankton are very sensitive to increased salt concentrations which can inhibit their communities greatly (Hintz & Relyea, 2017; Jones et al., 2017). Previous studies have shown that some cyanobacteria are tolerant to rising salt concentrations, potentially allowing them to thrive in these conditions as a result of reduced competition (Apte et al., 1987; Hagemann, 2011; Jones et al., 2017).

Though the land use information indicates that McLaughlin Bay is a mainly natural, small watershed, it is apparent that the water quality is in a highly degraded state. One of the possible explanations for this could be that McLaughlin Bay has been hydrologically closed off from Lake Ontario through a natural barrier beach over the last 10 years. Chloride is also high because of road salt application being heavy in the local watershed with the presence of highway 401 and a large parking lot in the catchment. Water exchange with the lake is a typical characteristic of coastal wetlands, and the lack of hydrological connection may be playing a role in McLaughlin Bay's chronically degraded water quality, in addition to internal phosphorus loading (Central Lake Ontario Conservation Authority, 2013).

The multiple linear regression analysis showed that nutrients (TP and TN) play an important role in explaining the variation in algal biomass, as these elements are known to be limiting nutrients in algal growth. Temperature was a significant positive predictor of algal biomass too, and this is likely due to the fact that phytoplankton, and specifically cyanobacteria, typically have increased production at higher temperatures and can tolerate higher temperatures (Butterwick et al., 2004; Konopoka & Brock, 1978).

Regression analysis also showed that algal biomass (as chlorophyll a) had a negative relationship with zooplankton biomass, which may be a result of zooplankton grazing, but it is more likely related to the abundance of inedible algae driving down large-bodied zooplankton biomass overall. This is supported by the fact that the most effective algal grazers, large bodied zooplankton, were in low abundance for most of the sampling season each year. Some water quality variables affected zooplankton abundance directly as well. Interestingly, dissolved oxygen had a negative relationship with zooplankton abundance. It has been shown that zooplankton typically have positive relationships with dissolved oxygen, but that is mostly seen in low oxygen environments.

In the case of McLaughlin Bay, all dissolved oxygen concentrations were above hypoxic concentrations (<3.5 mg L^{-1}) in aquatic ecosystems (Steckbauer, et al., 2011). The results detected in this study may reflect a confounding effect of dissolved oxygen production during algal blooms, which may explain the negative relationship. Finally, the multiple regression model deemed pH as a negative explanatory variable of zooplankton abundance. O'Brien & deNoyelles (1972) found that in cases of photosynthetically increased rates of pH (e.g., during an algal bloom), zooplankton may have lower survivability. It has also been found that some zooplankton species have differing survival rates based on rising pH. Previous research has shown that filtration rates or respiratory rates may be altered as pH changes (Ivanova & Klekowski, 1972). It is likely that DO and pH are directly influenced by algal photosynthesis, potentially having additive synergistic effects on the zooplankton communities.

As mentioned, the size-class of phytoplankton has a significant bearing on whether an aquatic ecosystem may be eligible for top-down control by grazers. In this case, there were significantly more inedible than edible algae. This is an important drawback for biomanipulation as a restoration approach, because biomanipulation success relies on phytoplankton biomass being controlled by zooplankton grazing. When algal biomass becomes very high, there is typically a take-over by these inedible populations that are not easily controlled by top down effects (Benndorf et al. 1990).

As seen in the phytoplankton relative abundance plot (Figure 3.8), the prevalence of cyanobacteria infers poor ecological conditions in McLaughlin Bay. The cyanobacteria

Anabaena and *Microcystis* are the two most dominant genera present in the algal community, and both taxa are notorious bloom formers and toxin producers. This is important in the scope of biomanipulation, because as mentioned, colonial and filamentous algae are poor food quality for zooplankton, and these toxin-producing taxa are also potentially harmful to grazers and fish (Benndorf et al. 1990).

When looking at the zooplankton relative abundance plot (Figure 3.9), ideal filter feeders such as *Daphnia* are only present in relatively high abundance in May/June each year, and abundance dramatically declines during the summer months when algal abundance peaks. It is likely that these zooplankton are present, but not abundant enough to control algal blooms before they are replaced with less efficient, small grazers such as *Bosmina*, Copepod nauplii, and some rotifers (Makarewicz et al., 1998). It should be noted that *Daphnia* typically decline in abundance by mid-summer because of increased predation by planktivores, so it is possible in McLaughlin Bay, planktivory may be an important driver of the seasonal decline in large bodied zooplankton biomass each year. Though there is the presence of Calanoid copepods in early 2017 and in 2019, it appears the pressures from algal blooms and likely planktivory are working in combination to lead to this shift to smaller individuals in the zooplankton community.

Overall, the prospect of a biomanipulation has a lot of potential as a useful management tool, but likely only when used in combination with other restoration approaches that aim to control nutrient concentrations (Benndorf et al. 1990). It appears that in McLaughlin Bay specifically, there is strong influence on the phytoplankton community by water quality conditions, which may not be resolved by only top-down forces such as increased grazing rates. Though there was shown to be some ideal zooplankton grazers in the community, their decline by mid-summer means that they are not likely to control the nuisance algae that dominate for most of the summer months.

My study of McLaughlin Bay over three years confirms that it is a highly degraded Great Lakes coastal wetland. Key areas requiring mitigation include controlling road salt run-off into the wetland, as well as nutrient inputs and internal phosphorus loading. If possible, I recommend opening up hydrological connectivity to Lake Ontario again to allow the discharge of built-up chloride and nutrients from the wetland water column. Even if this was a temporary measure while road salt run-off controls were put in place, it could make all of the difference for future restoration efforts potentially involving biomanipulation.

Chapter 4: General Conclusion

As the landscape of the Durham Region and the Great Lakes region changes, it is essential to understand how shifting land cover may impact aquatic ecosystems as important as coastal wetlands. The goal of my research was to look across an urban gradient in order to characterize water quality and plankton communities in Lake Ontario coastal wetlands and to understand the drivers of those biological communities. I also looked at McLaughlin Bay as a candidate for a biomanipulation restoration. By understanding water quality conditions and plankton communities in this region, this information can act as a baseline for other Great Lakes coastal wetlands and what to expect in their response to increasing watershed disturbance, in order to help development of restoration strategies.

By looking at four Lake Ontario coastal wetlands in the Durham Region, I found that land use, water quality, and biological communities vary considerably among ecosystems. Although land use and watershed size are typically important in driving water quality in wetlands, they did not appear to be the most influential driver of water quality among my study wetlands. According to land-use composition, it appears that water quality was not as affected in some wetlands as they were in others. In McLaughlin Bay, I found that even with a small, natural watershed, water quality was very degraded. In Frenchman's Bay, water quality was relatively high, though having a mostly developed (while relatively small) watershed. There were also differences in plankton communities as McLaughlin Bay had a greater proportion of cyanobacteria relative to other wetlands, while in Bowmanville Marsh, a more agricultural watershed, there was a greater proportion of filamentous green algae. The changes in plankton community composition were not only evident in phytoplankton, as the zooplankton communities were characteristic of degraded ecosystems in the studied wetlands. Zooplankton were mostly small cladocerans or rotifers with few larger, more efficient grazing zooplankton in the community.

The link between water quality and the plankton communities was evident as water quality variables such as total nitrogen, total phosphorus, and chloride were significant factors in explaining the variation in algal abundance, as well as cyanobacterial biomass. There was also a negative relationship between phytoplankton genus richness and the water quality variables phosphorus and chloride. The relationships between chloride and the plankton communities in both abundance and richness may indicate that chloride should be a greater concern in the Great Lakes coastal wetlands than may have been realized. As these water quality variables such as chloride and nutrients increase, as is typically seen with urbanization, there will likely be stronger effects on the biological communities, leading to greater algal abundances, altered biological community structure, and more potentially harmful cyanobacteria.

When evaluating McLaughlin Bay as a candidate for restoration by biomanipulation, several key conditions related to water quality and plankton community structure had to be determined. It was clear that McLaughlin Bay was in a consistently degraded state based on its chronically high nutrient (i.e. eutrophic) and chloride concentrations, high algal biomass, and relatively low community diversity throughout the study period. Conditions such as degraded water quality and a large proportion of inedible cyanobacteria within the algal community lead to the determination that McLaughlin Bay may not be a suitable ecosystem for biomanipulation.

I found that the high algal abundance in McLaughlin Bay was linked to high nutrient conditions and warm water temperatures. Unless remediation measures are implemented to control the poor water quality conditions leading to these algal blooms, the issue is unlikely to be resolved by enhancing top down control in the food web. Factors that lead to these degraded conditions such as intensive urban land-use in the small watershed, as well as conditions leading to internal nutrient loading in the wetland must be resolved before attempting to restore the biological communities. The hydrological connectivity to Lake Ontario likely needs to be re-established in order to remediate degraded conditions.

Though land-use type and intensity can have significant impacts to aquatic ecosystems, I determined that variable land-use gradients in this region alone does not infer water quality conditions in four Lake Ontario coastal wetlands. Nutrients, particularly phosphorus, and chloride were the most important variables that explained the variation in algal abundance and plankton community structure. While excess nutrients are known to be an important factor in promoting algal blooms, my thesis research shows the potential for chloride, caused by de-icing salts, to structure plankton communities in Great Lakes coastal wetlands. Until nutrient and chloride inputs into coastal wetlands can be controlled, certain restoration strategies such as biomanipulation will likely not be effective in restoring degraded wetlands.

The Great Lakes region is likely to undergo major land-use changes as urban areas grow. Increased urbanization of coastal wetland watersheds will only increase inputs of nutrients and road salt unless mitigative action is taken to trap these pollutants before they enter coastal wetland ecosystems. Future research aimed at developing infrastructure to reduce inputs of contaminated run-off to coastal wetlands is essential. Future research should also develop an array of restoration approaches that address the different environmental impacts experienced by coastal wetlands, such as road salt pollution for urban coastal wetlands and agricultural pollution for other coastal wetlands. Overall, I hope the data that I collected and analyzed for my thesis research offers important baseline information for wetland managers to make informed decisions regarding future restoration initiatives in Lake Ontario coastal wetlands.

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Appendix

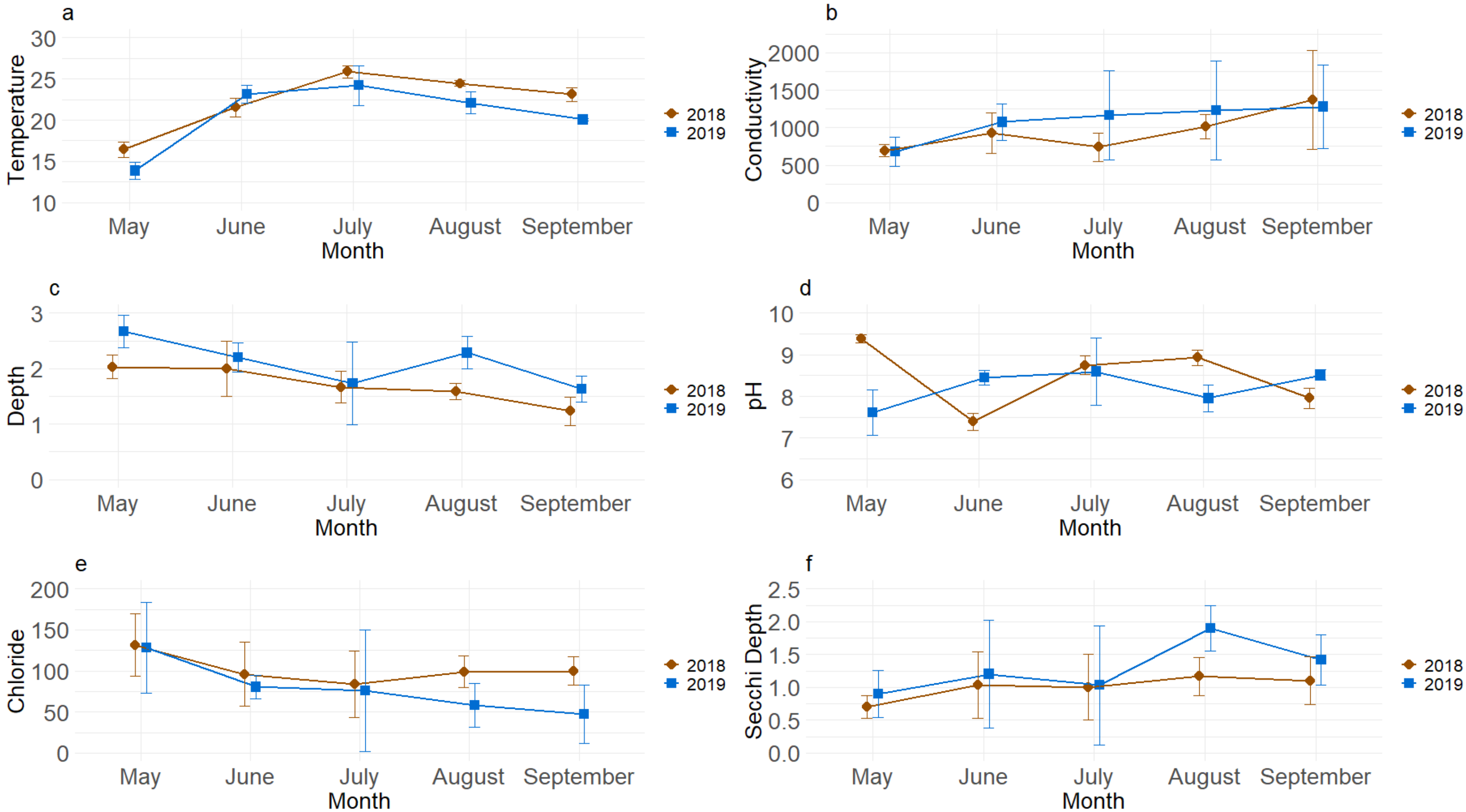

Figure A1. Trend plots showing mean and standard deviation of Frenchman's Bay in (a) temperature (°C), (b) conductivity (μs cm^{-1}), (c) depth (m), (d) pH, (e) chloride (mg L^{-1}), and (f) Secchi Depth (m).

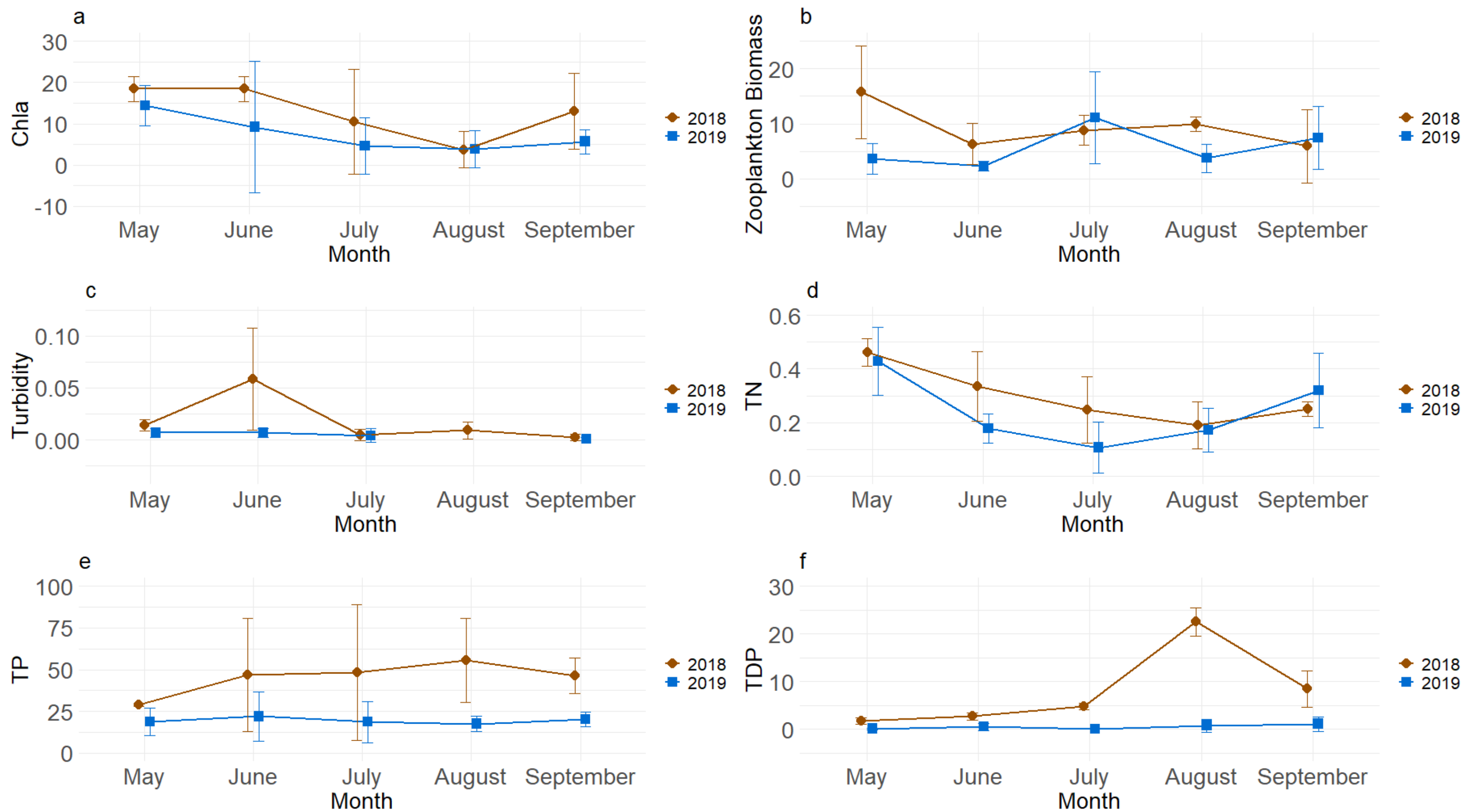

Figure A2. Trend plots showing mean and standard deviation of Frenchman's Bay in (a) chlorophyll a (µg L^{-1}), (b) zooplankton biomass (mg), (c) turbidity (abs @750nm), (d) total nitrogen (mg L^{-1}), (e) total phosphorus (µg L^{-1}), and (f) total dissolved phosphorus (µg L^{-1}).

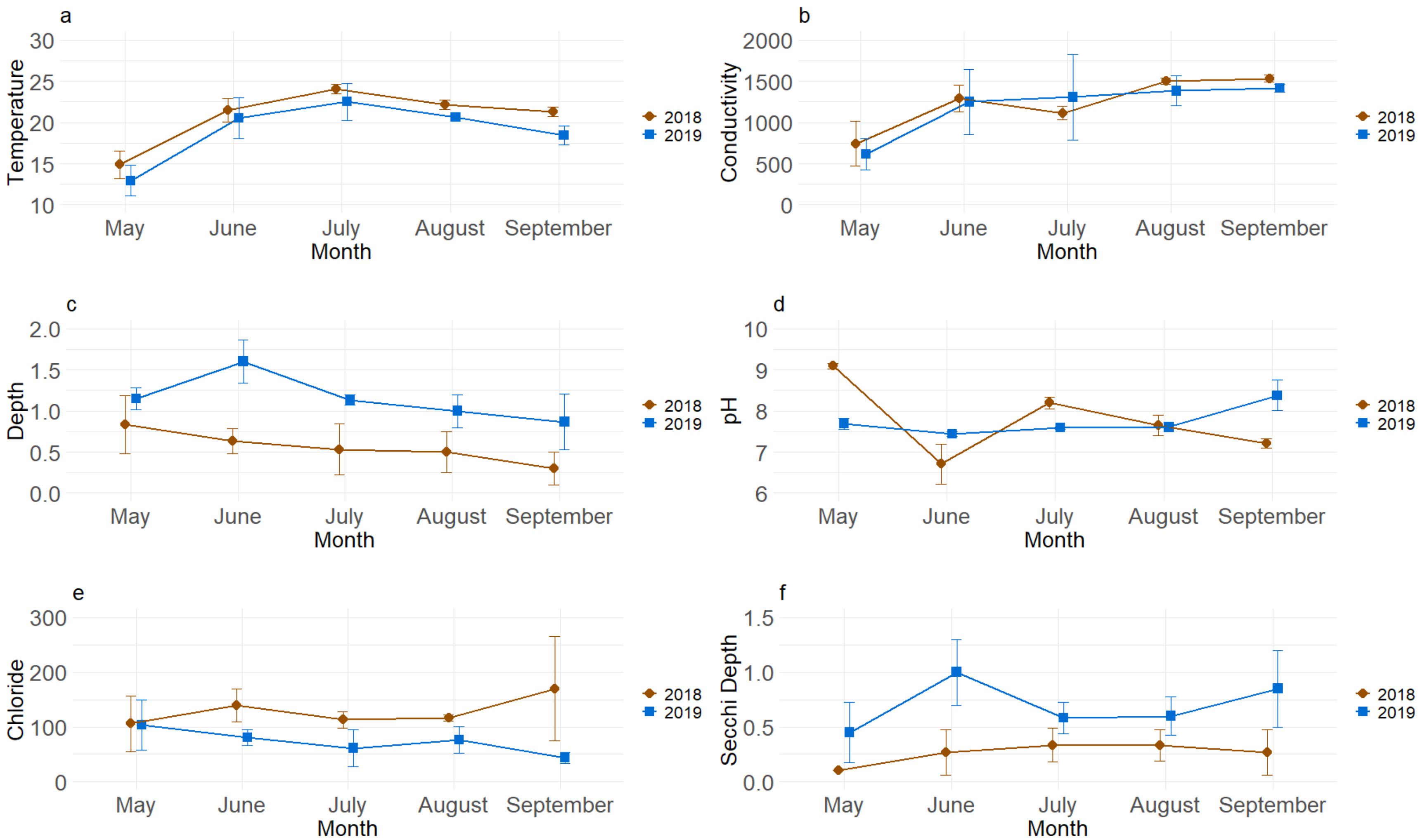

Figure A3. Trend plots showing mean and standard deviation of Lynde Marsh in (a) temperature (°C), (b) conductivity (µs cm^{-1}), (c) depth (m), (d) pH, (e) chloride (mg L^{-1}), and (f) Secchi Depth (m).

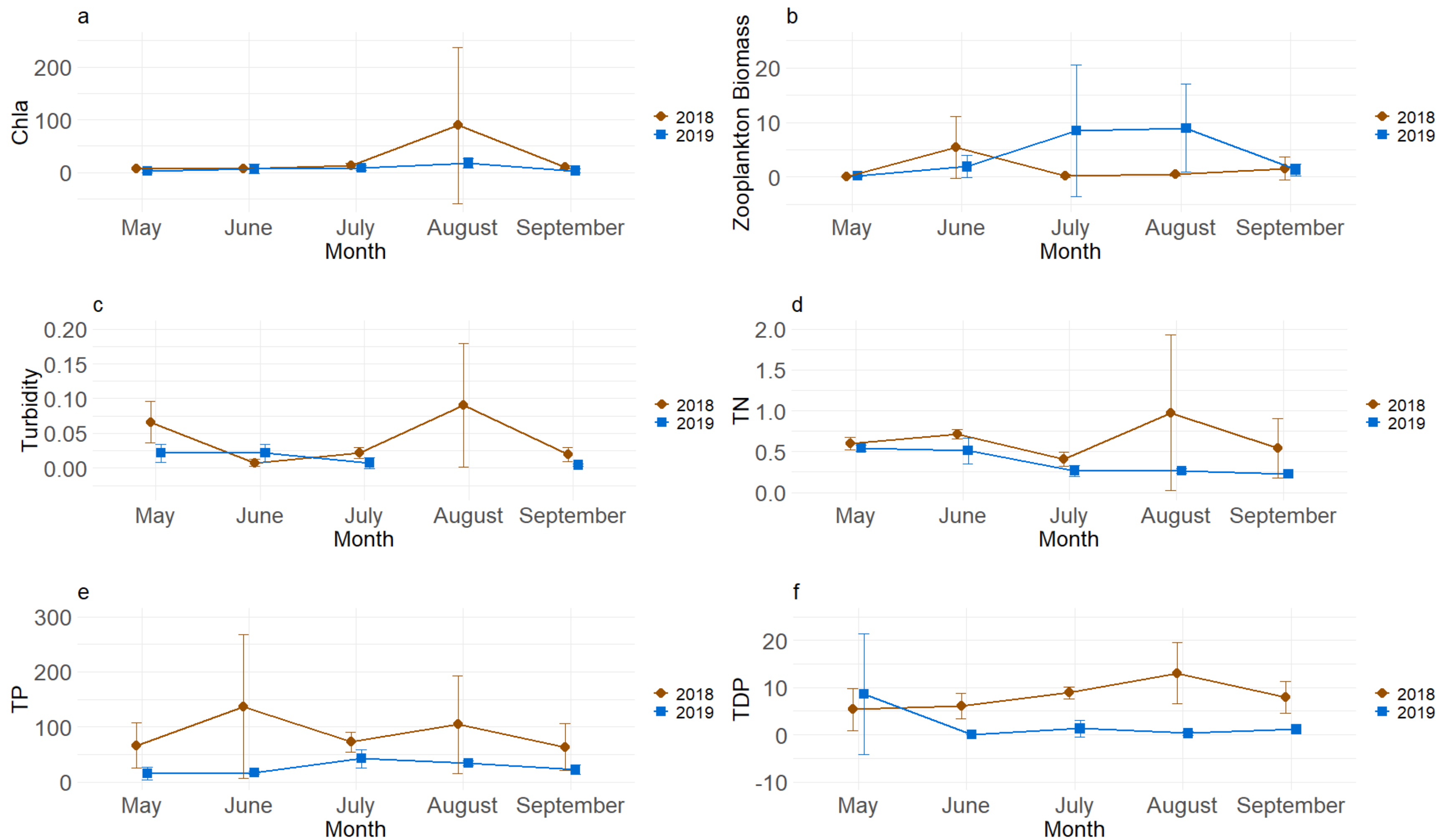

Figure A4. Trend plots showing mean and standard deviation of Lynde Marsh in (a) chlorophyll a (µg L^{-1}), (b) zooplankton biomass (mg), (c) turbidity (abs @750nm), (d) total nitrogen (mg L^{-1}), (e) total phosphorus (µg L^{-1}), and (f) total dissolved phosphorus (µg L^{-1}).

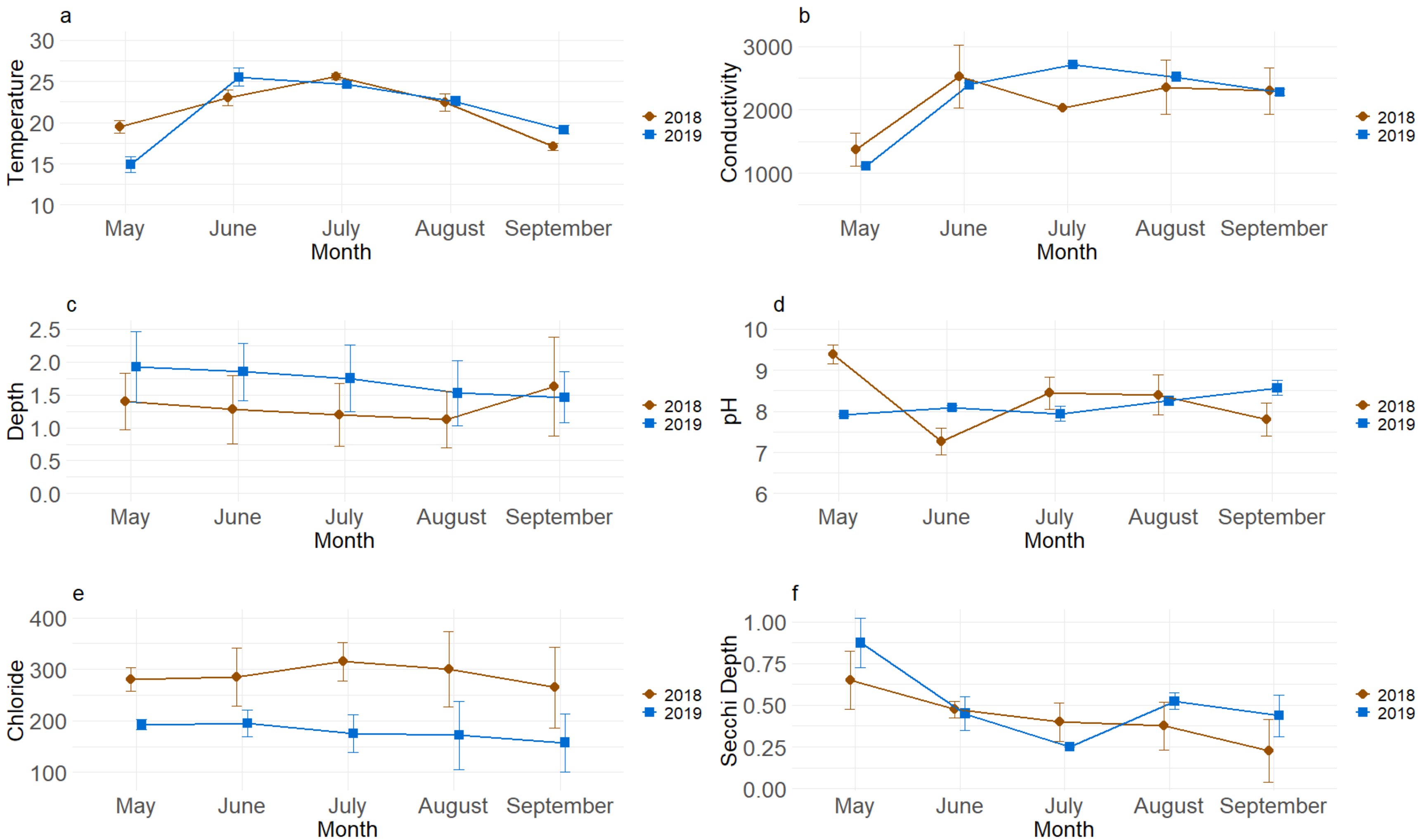

Figure A5. Trend plots showing mean and standard deviation of McLaughlin Bay in (a) temperature (°C), (b) conductivity (μs cm^{-1}), (c) depth (m), (d) pH, (e) chloride (mg L^{-1}), and (f) Secchi Depth (m).

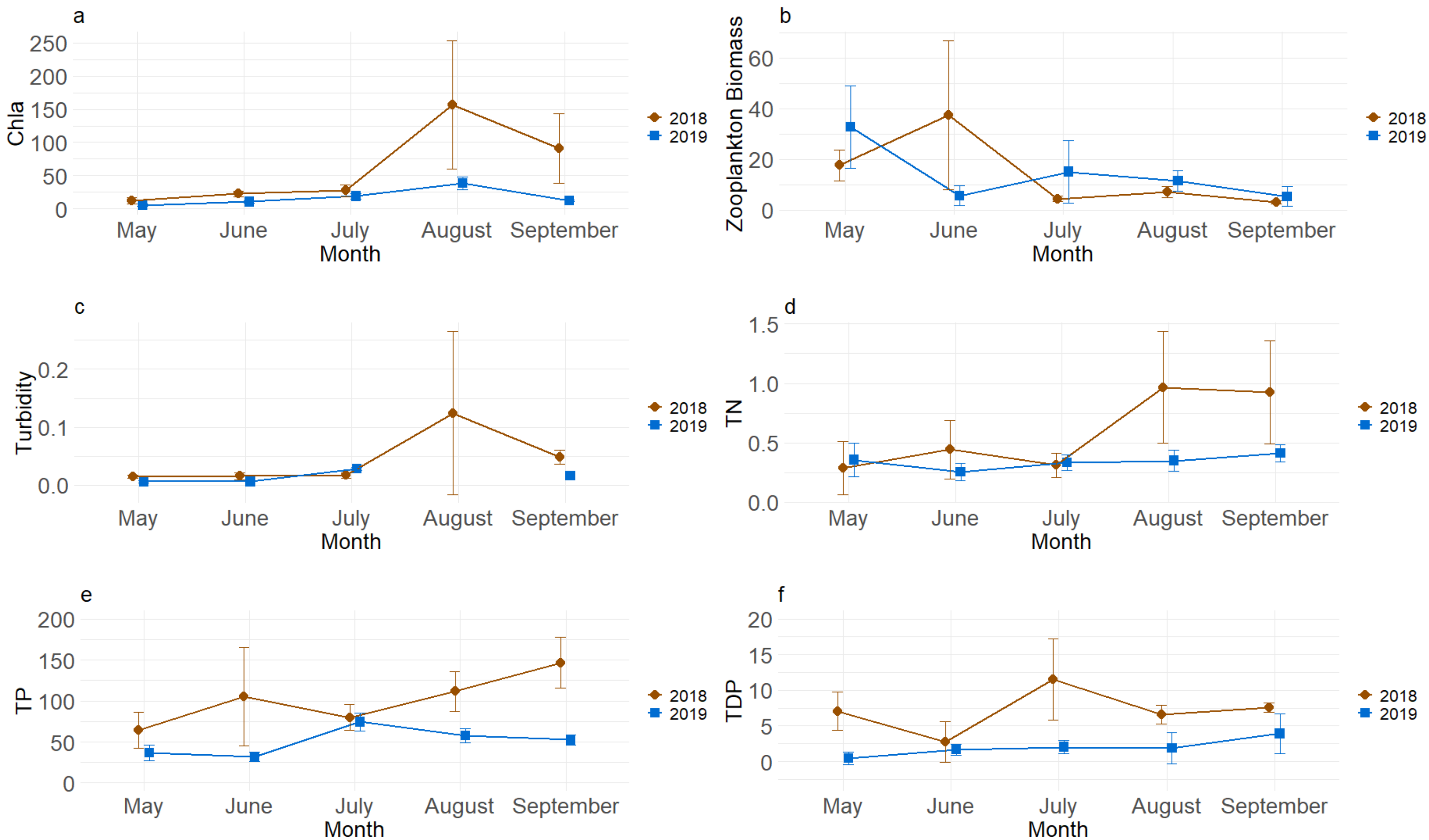

Figure A6. Trend plots showing mean and standard deviation of McLaughlin Bay in (a) chlorophyll a (μg L^{-1}), (b) zooplankton biomass (mg), (c) turbidity (abs @750nm), (d) total nitrogen (mg L^{-1}), (e) total phosphorus (μg L^{-1}), and (f) total dissolved phosphorus (μg L^{-1}).

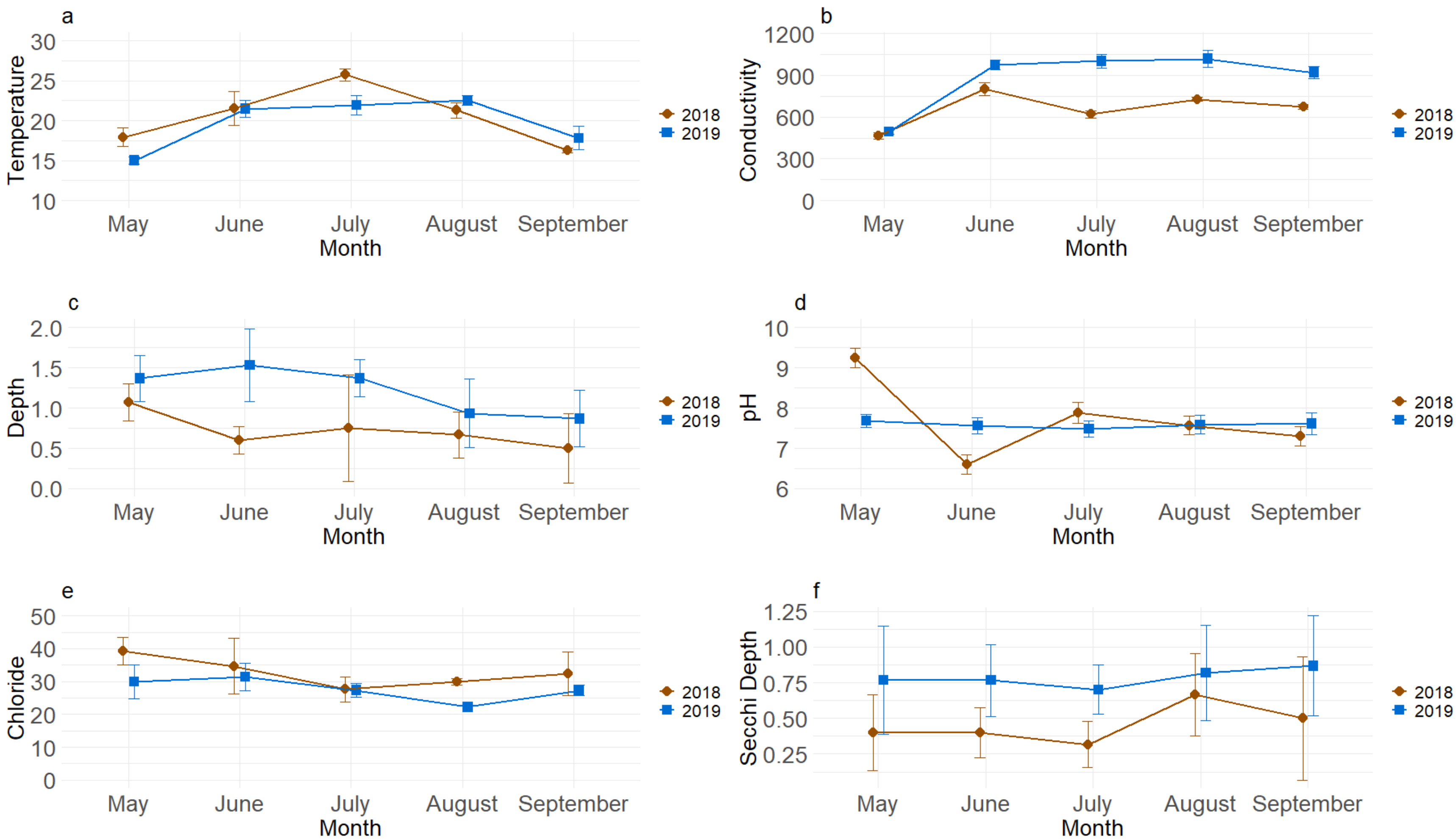

Figure A7. Trend plots showing mean and standard deviation of Bowmanville Marsh in (a) temperature (^{o}C), (b) conductivity (μs cm^{-1}), (c) depth (m), (d) pH, (e) chloride (mg L^{-1}), and (f) Secchi Depth (m).

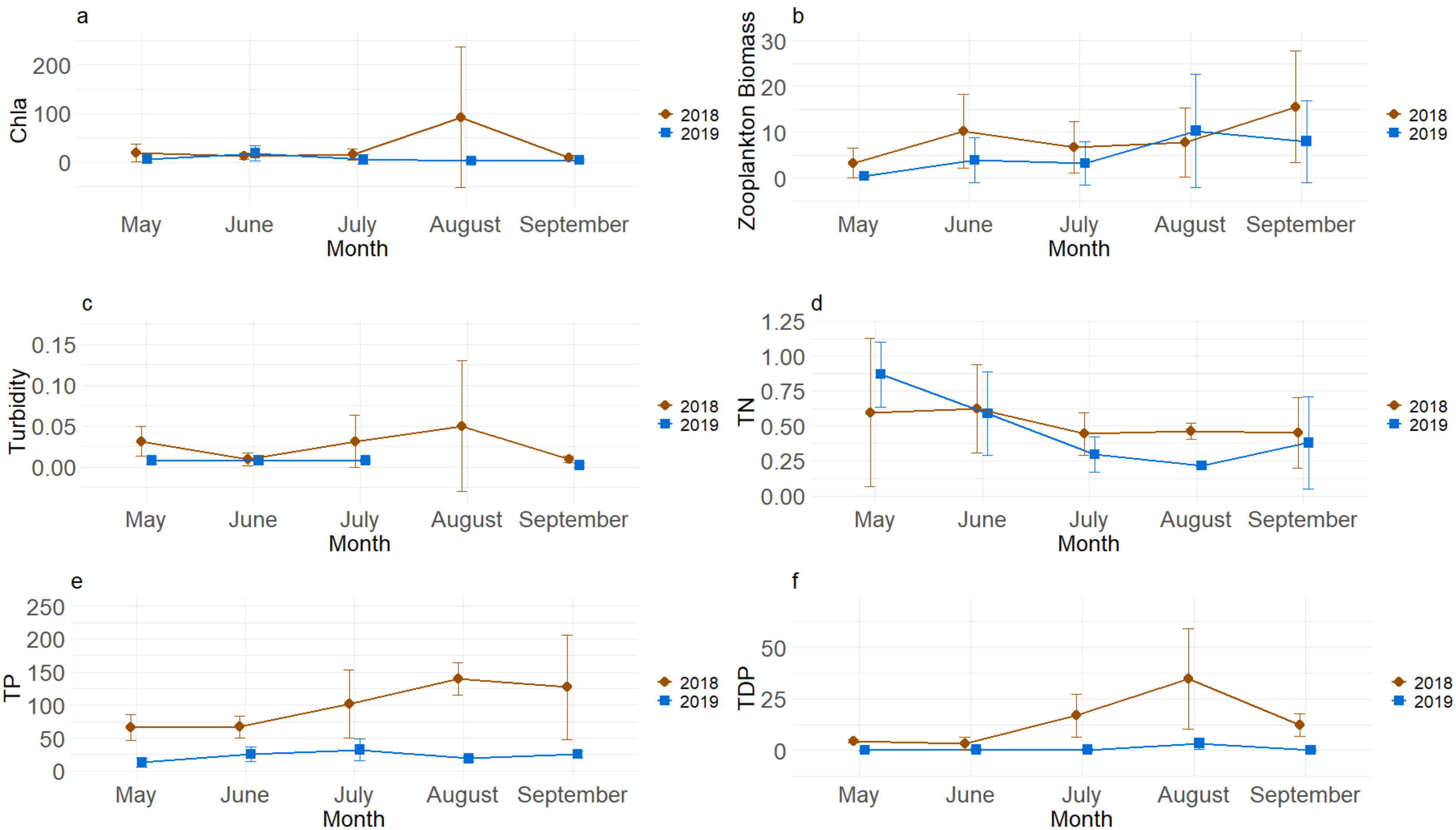

Figure A8. Trend plots showing mean and standard deviation of Bowmanville Marsh in (a) chlorophyll a (µg L^{-1}), (b) zooplankton biomass (mg), (c) turbidity (abs @750nm), (d) total nitrogen (mg L^{-1}), (e) total phosphorus (µg L^{-1}), and (f) total dissolved phosphorus (µg L^{-1}).

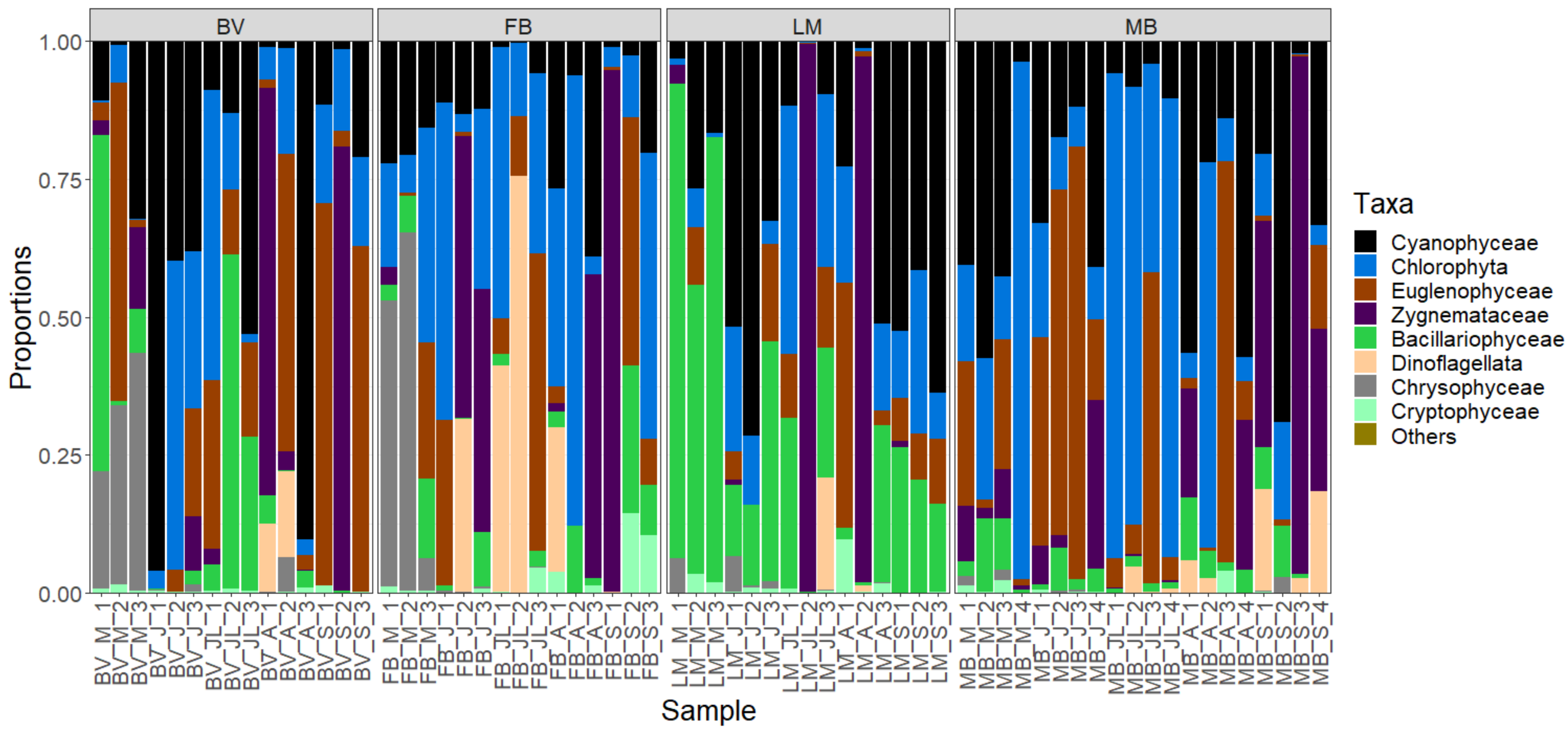

Figure A9. Relative abundance plot of phytoplankton groups over May-September in 2018. Each bar represents the relative abundances in each site of overall phytoplankton, grouped by wetlands.

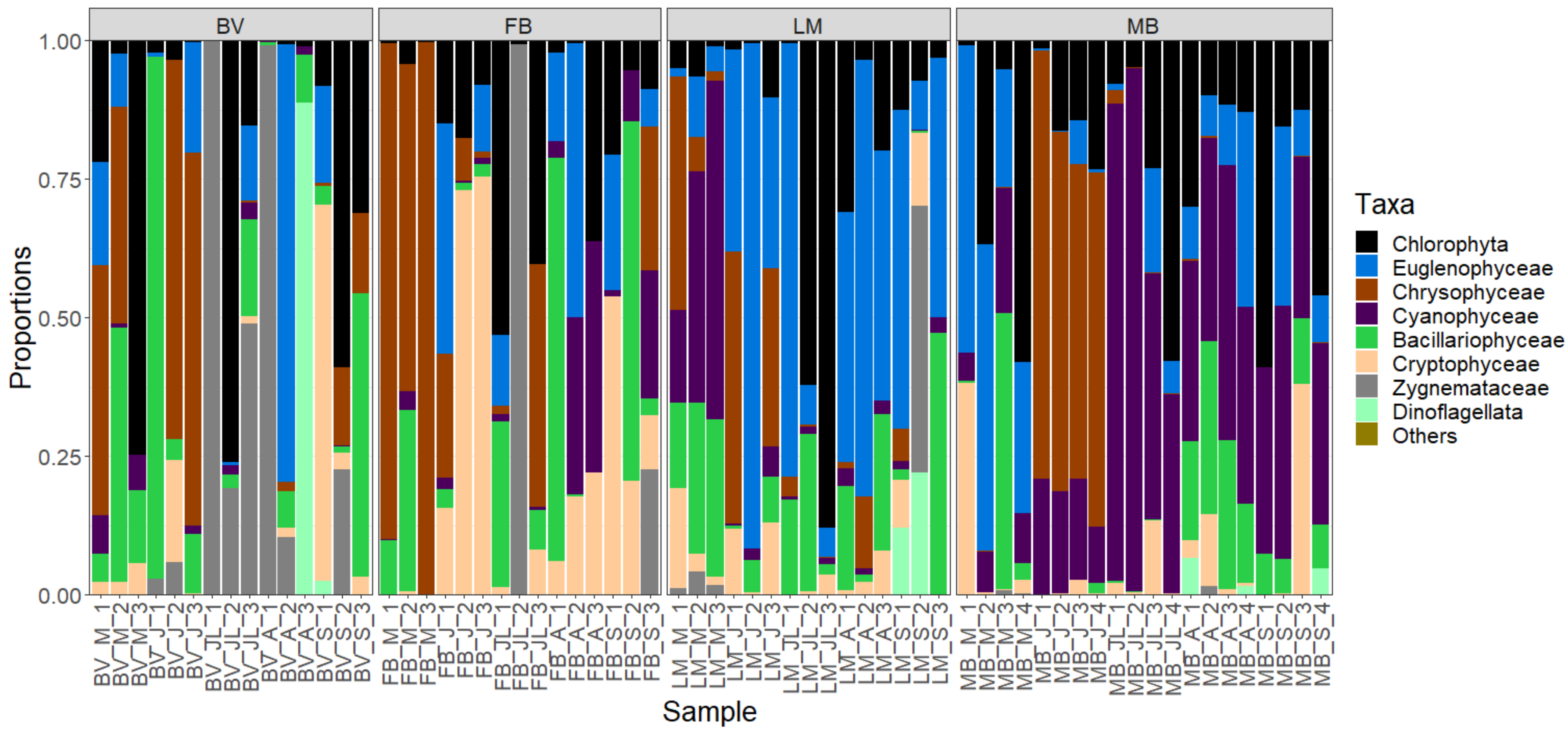

Figure A10. Relative abundance plot of phytoplankton groups over May-September in 2019. Each bar represents the relative abundances in each site of overall phytoplankton, grouped by wetlands.

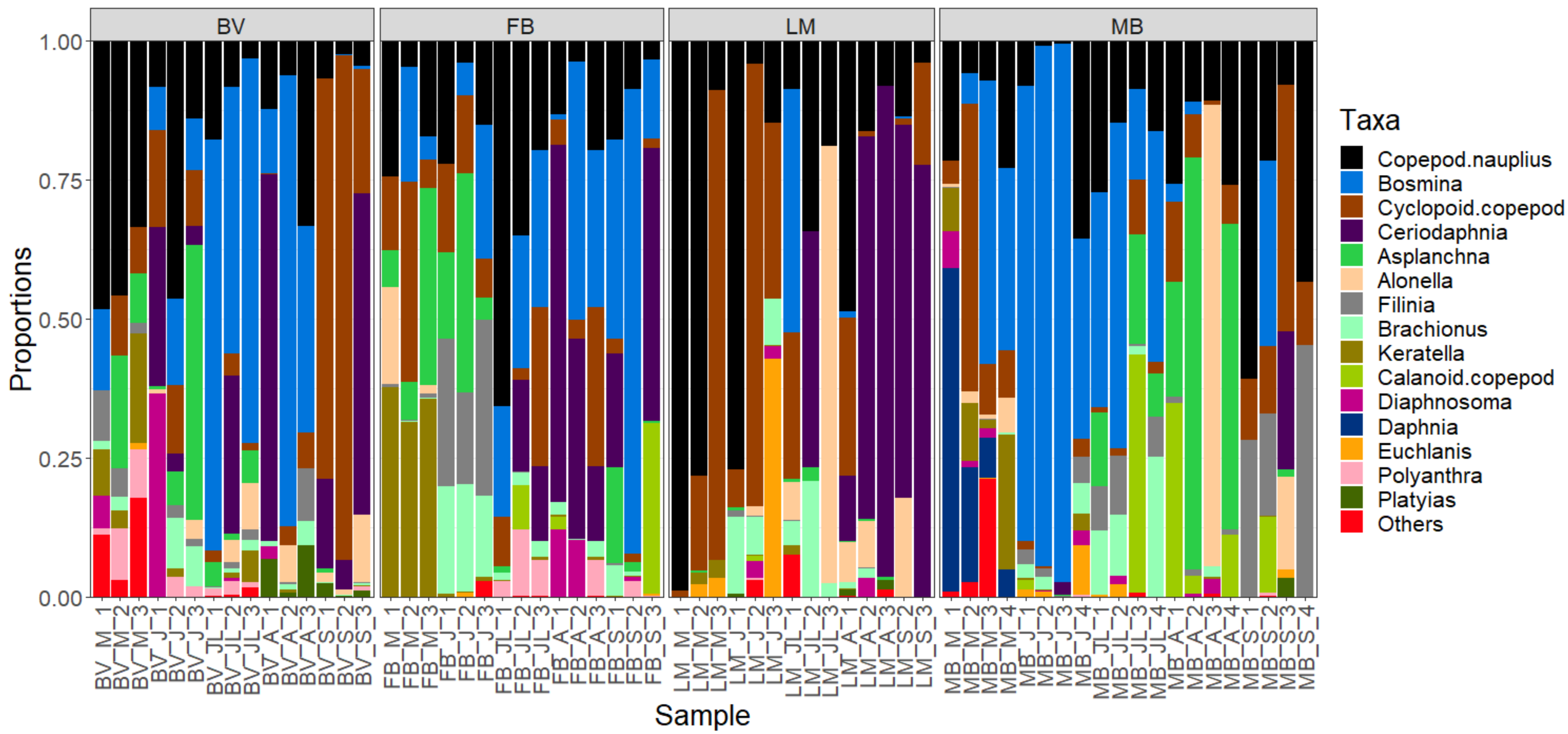

Figure A11. Relative abundance plot of zooplankton groups over May-September in 2018. Each bar represents the relative abundances in each site of overall zooplankton, grouped by wetlands.

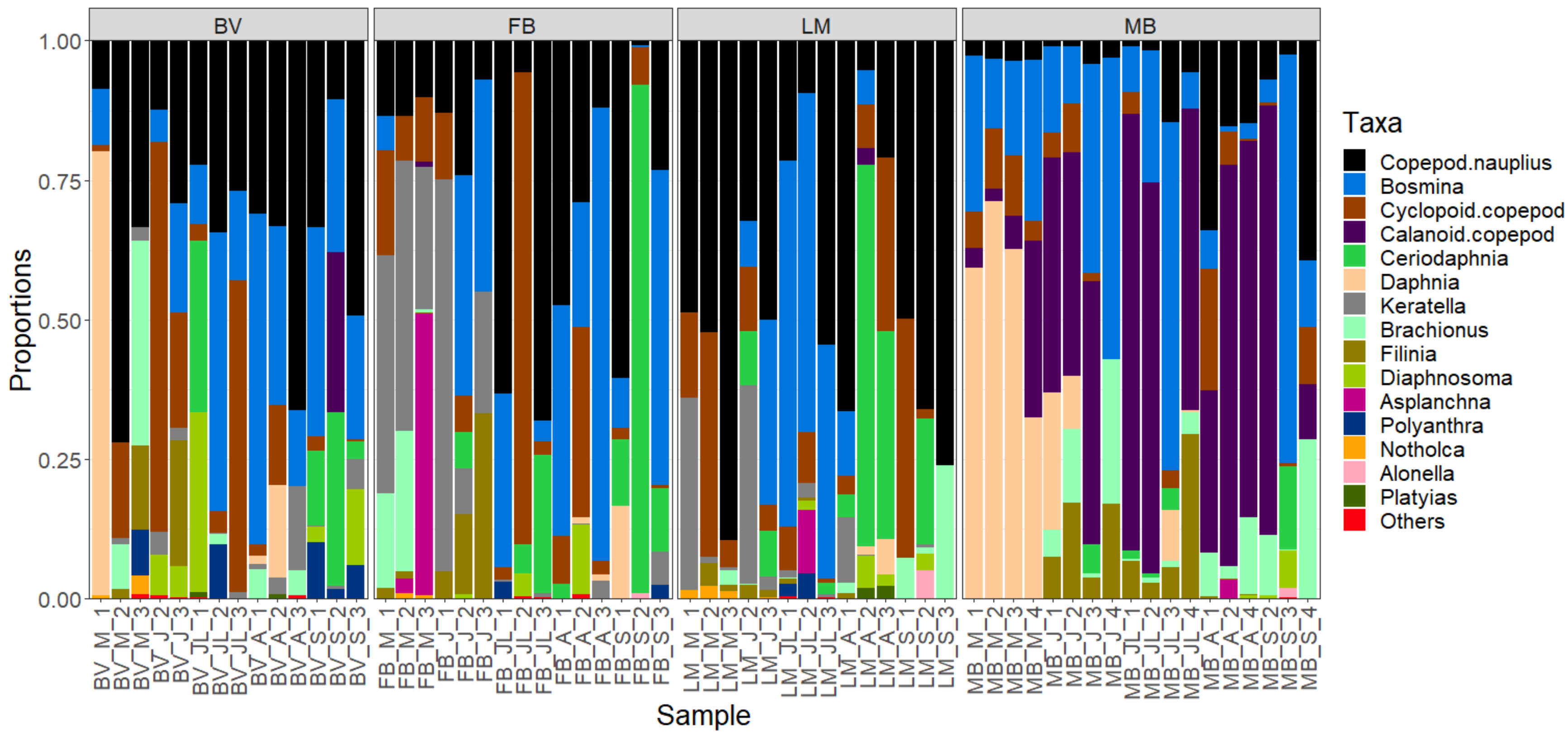

Figure A12. Relative abundance plot of zooplankton groups over May-September in 2019. Each bar represents the relative abundances in each site of overall zooplankton, grouped by wetlands.

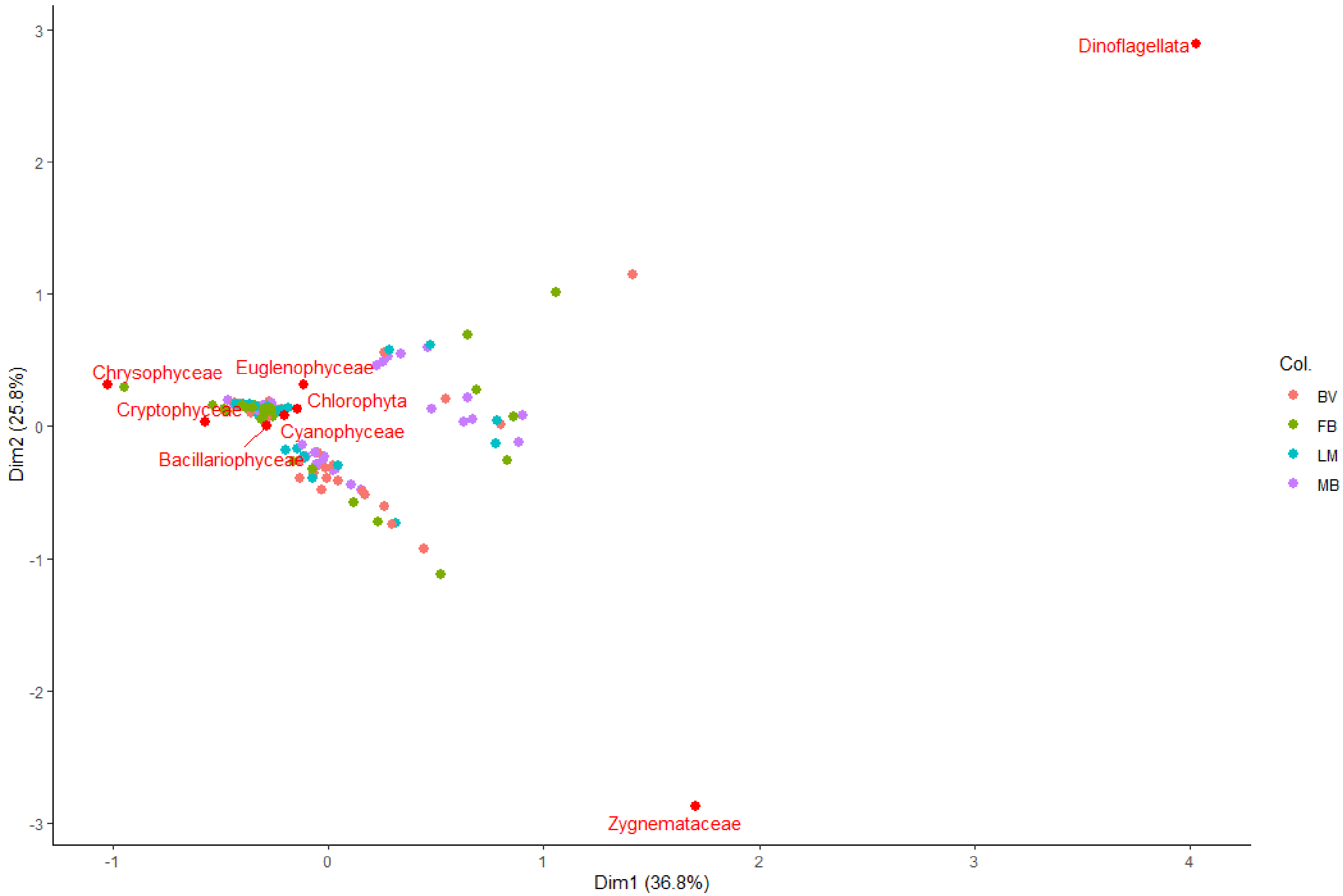

Figure A13. Correspondence analysis of phytoplankton community sorted by group.

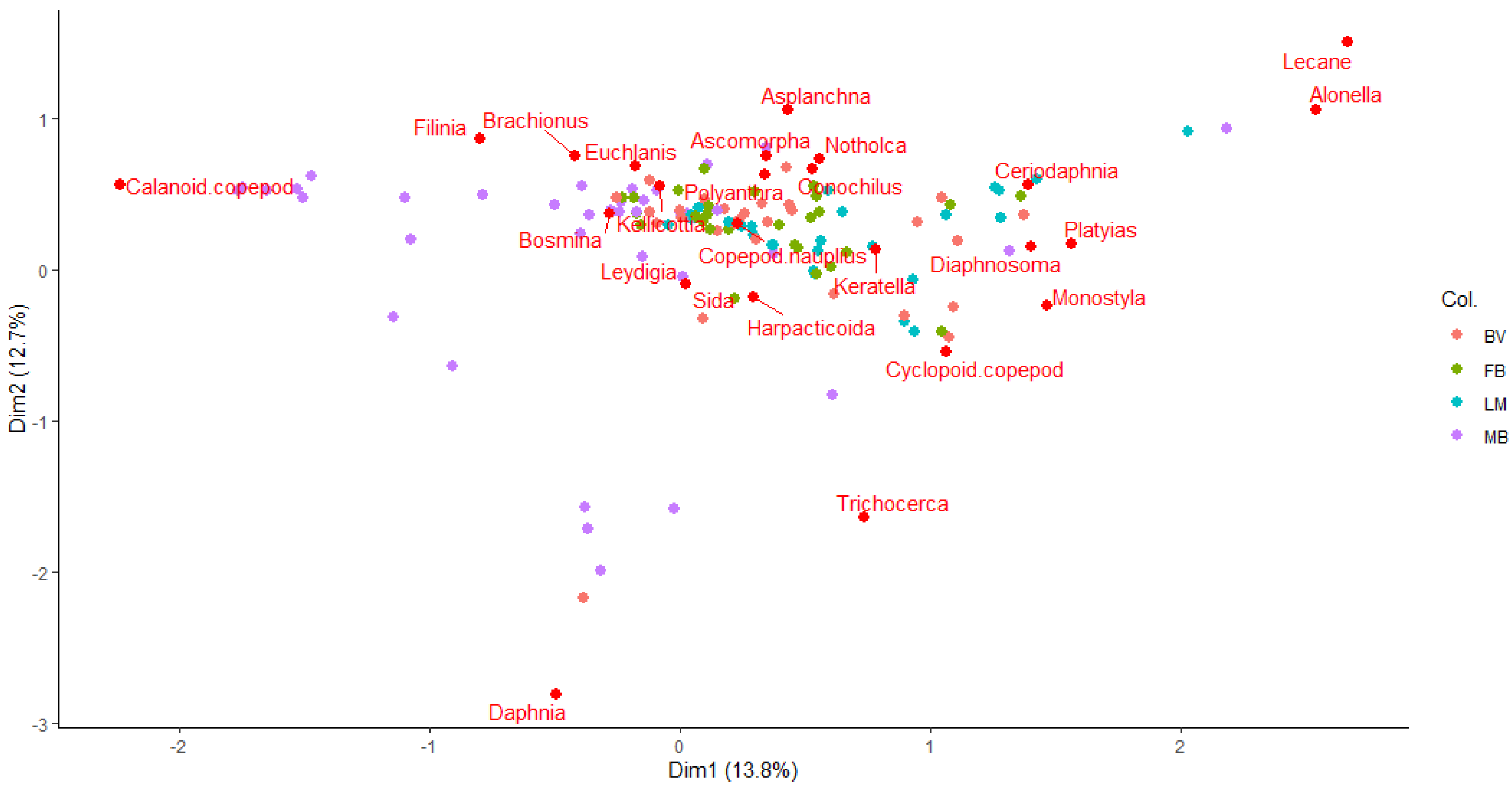

Figure A14. Correspondence analysis of zooplankton community by group.

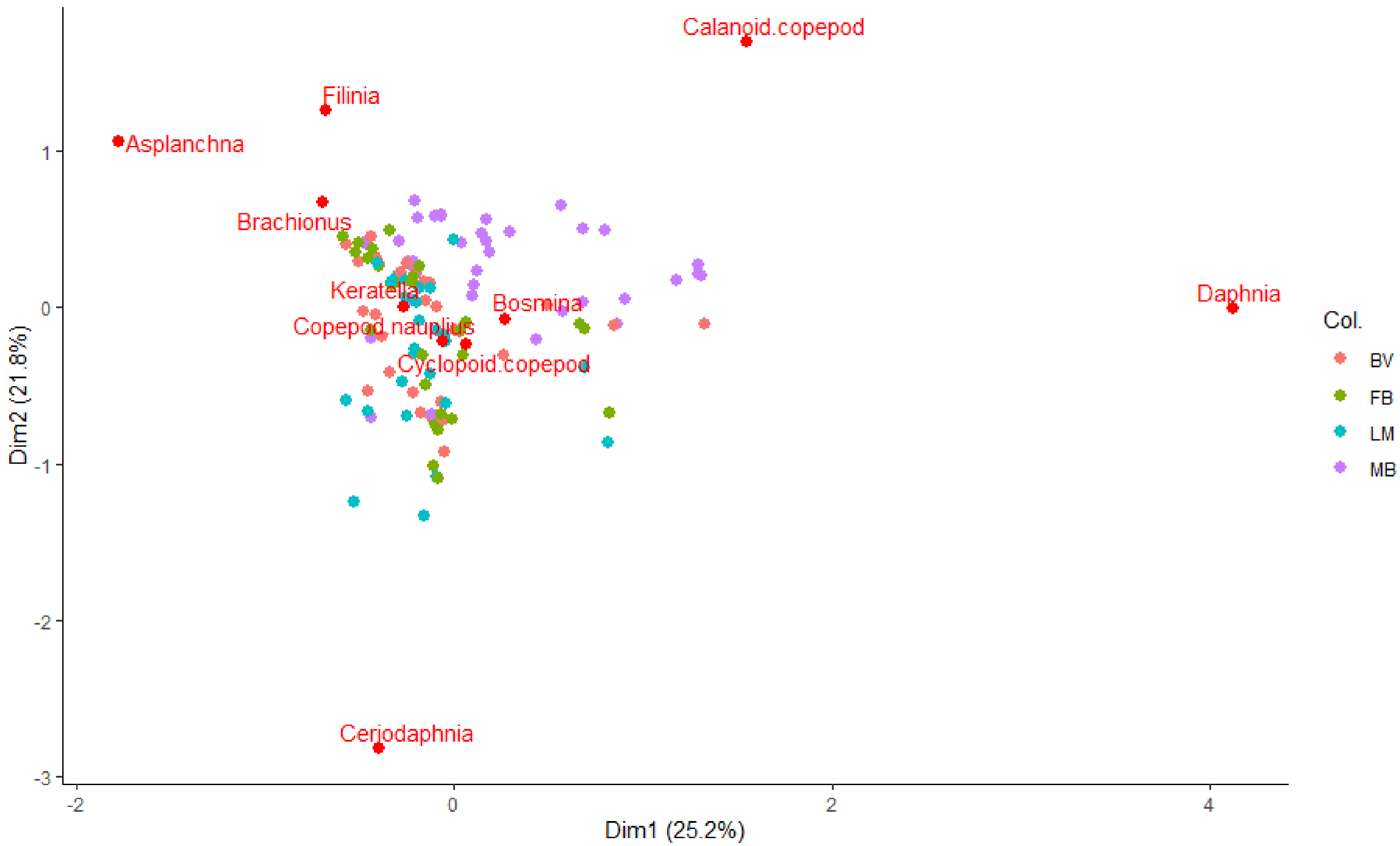

Figure A15. Correspondence analysis of top ten zooplankton community by group.

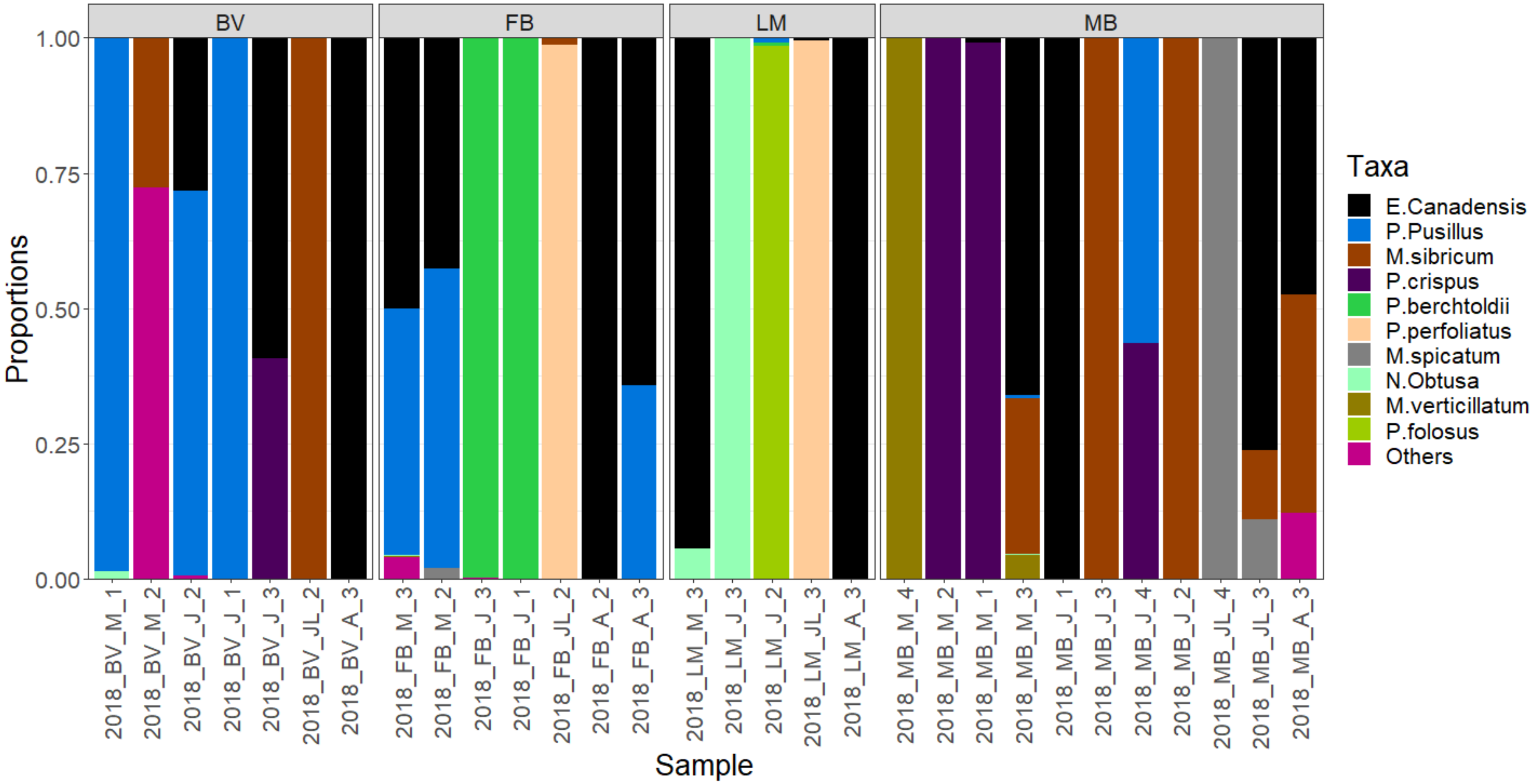

Figure A16. Relative abundance of macrophyte communities where plants were present from May to August in 2018. Top ten most abundant taxa are included, any less than top ten are included in others.

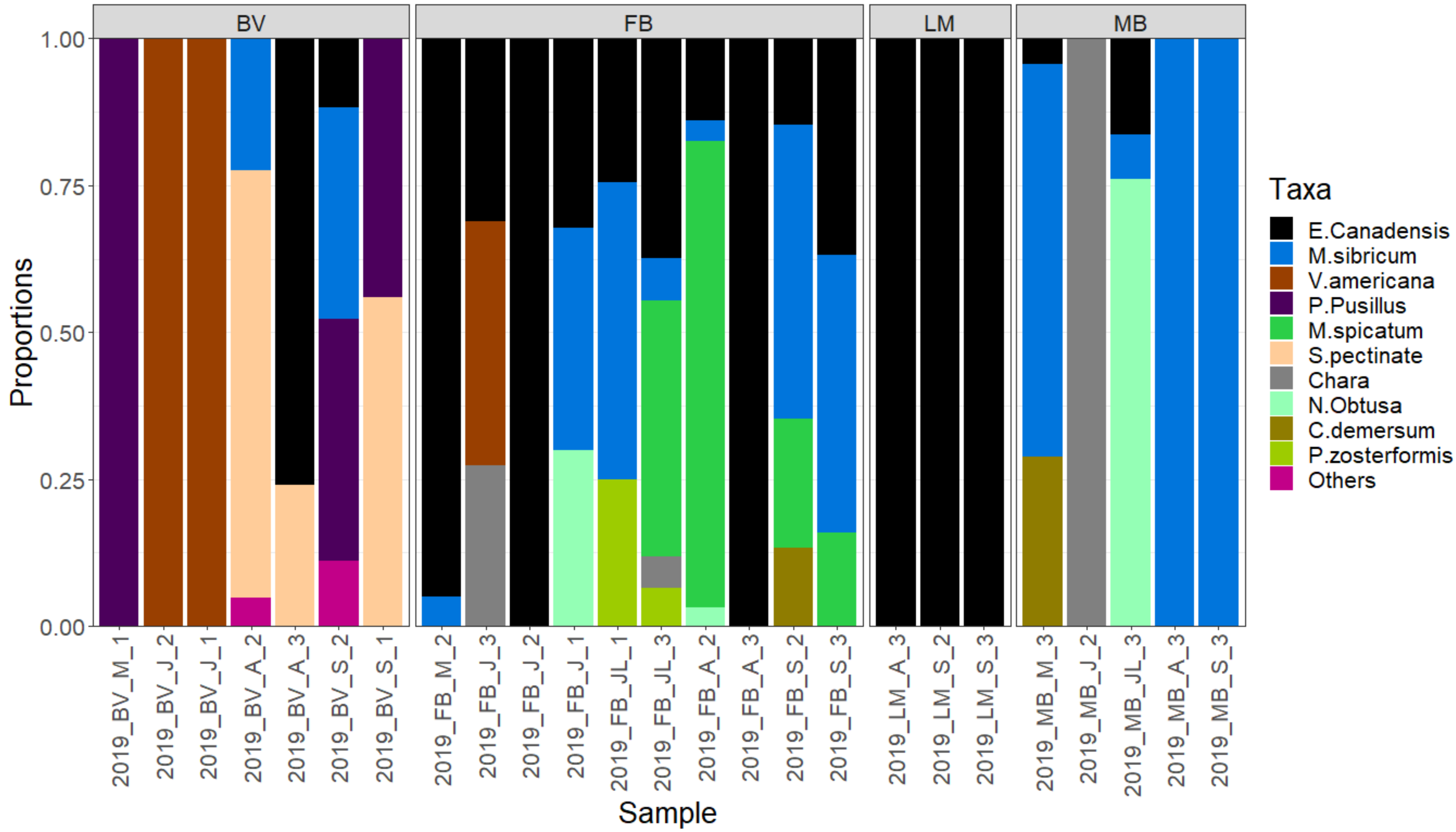

Figure A17. Relative abundance of macrophyte communities where plants were present from May to September in 2019. Top ten most abundant taxa are included, any less than top ten are included in others.