Spatial variation of water quality and algal production, and the relationship between land use and nutrient loading in Delta Marsh

by

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Abstract

Delta Marsh is a 19,000-hectare freshwater coastal wetland located on the southern shore of Lake Manitoba, Canada. The ecological integrity of Delta Marsh has been declining due to changes in hydrology, nutrient loading, water quality and the introduction of invasive species. Research on variables influencing this decline is imperative for restoration. The objectives of this research were to determine which chemical and physical variable(s) drive phytoplankton chlorophyll-a concentrations and to determine if there is spatial variation of nutrient loading caused by the land use gradient across the watershed. Environmental variables were analyzed using multivariate statistical analyses and demonstrated that phytoplankton blooms were most influenced by meteorological conditions and nutrient availability. Nutrient exports to Delta Marsh were greater in areas with higher agricultural cover. This research will help inform management plans to restore the ecological function to Delta Marsh and other coastal wetlands, ultimately improving water quality, fisheries and waterfowl habitat.

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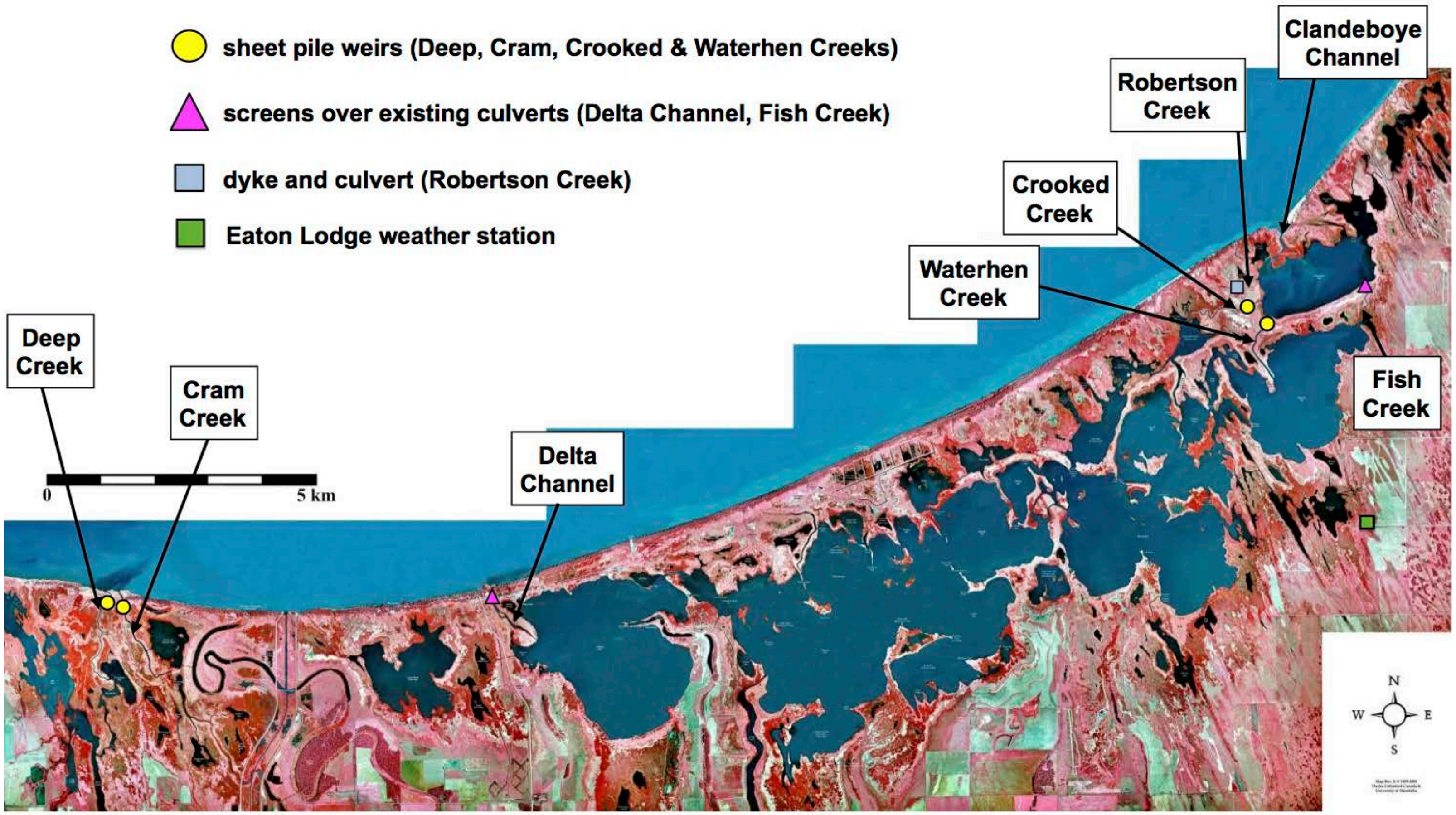
1.0 Introduction

Wetlands are important ecosystems that provide various ecological and economic benefits to society. Despite the multitude of ecological processes that occur in wetlands, including nutrient and sediment sequestration, the importance of wetlands is often overlooked. Coastal wetlands are associated with a large body of water, a lake or ocean, and perform important ecological processes for the function and health of the attached water body. Coastal wetlands filter nutrients and sediments from the land before emptying into the lake and provide crucial habitat and spawning grounds for fish and waterfowl populations (Kreiger et al, 1992, Watchorn et al, 2012). Wetland degradation as a result of anthropogenic influences is occurring globally, due to increased nutrient loading (Ansari and Gill 2014; Main et al. 2014), sedimentation (Main et al. 2014), and altered hydrology (Morrice et al. 2008). As a result, biological, chemical and physical restoration efforts have been undertaken to restore natural function of wetlands, but these can be expensive, time consuming and often with low or short-term success rates (Schindler and Vallentyne, 2008; Anttila et al, 2013; De Backer et al, 2012). Before restoration efforts can be applied to a wetland, it is important to understand the target wetland and its surrounding watershed, as each management plan should account for unique characteristics of the targeted restoration area.

Delta Marsh is a freshwater coastal wetland located at the south basin of Lake Manitoba, Manitoba, Canada (Figure 1.1). This 19,000-hectare wetland has been declining in ecological function for the last half century due to many physical, chemical and biological changes (Ducks Unlimited Canada, 2012; Watchorn et al, 2012). The “Restoring the Tradition” project being conducted at Delta Marsh, is part of an ongoing

Figure 1.1. False-colour infrared map (1997) of Delta Marsh, located at the south basin of Lake Manitoba, Manitoba, Canada overlaid with exclusion structures and the Eaton Lodge Weather Station. (50°10'N 98°17'W) (Ducks Unlimited Canada and the University of Manitoba).



research effort to understand all biotic and abiotic processes that influence the function of the marsh to help inform future management plans aimed at restoring the marsh.

1.1 Primary Production

Wetlands are among the most productive ecosystems in terms of plant biomass globally, due to available nutrients and water supply (Richardson 1995). Their plant cover consists of emergent vegetation, submersed vegetation and algae. The high primary production makes wetlands an important ecosystem in the global carbon cycle (Richardson 1995; Vitt et al. 2001). Wetland plants, primarily emergent vegetation, decay at a slower rate than upland vegetation. The high rates of productivity and low rates of decomposition that are characteristic of wetlands result in significant carbon sequestration that can potentially reduce impact of global climate change (Richardson 1995; Miller and Fujii 2009). Changes to wetland structure and dynamics, such as increased runoff and altered hydrology can affect primary production and hence carbon storage (Miller and Fujii 2009).

Primary production and biomass in wetlands is often limited by nutrient availability, and in particular by phosphorus and nitrogen (Chaudhuri et al. 2012). Phosphorus enters wetlands in many forms, organic or inorganic, and is either present in dissolved or particulate forms. The dissolved inorganic forms are considered bioavailable, where organisms can directly take up the nutrients, while particulate phosphorus is attached to or contained within suspended sediments, eroded soils, and organisms within the water column such as phytoplankton and zooplankton. Particulate and organic forms of phosphorus must undergo transformation to become bioavailable

(Carlson and Simpson 1996). The various phosphorus and nitrogen compounds found in wetlands enter from the transfer of soils and sediments from the surrounding landscapes as well as dissolved forms delivered via surface runoff and groundwater discharge. Additionally, wetland sediments and vegetation can play an important role in the internal recycling of nutrients (Verhoeven et al. 2006), resulting in changes to the stocks and forms of nutrients found within a wetland.

Primary production in wetlands varies based on changes in nutrient dynamics, water volume and hydrology, soil nutrients and species composition (Atkinson et al. 2010). There is a stronger relationship between sediments and the water column in wetlands than in lakes, causing increased nutrient recycling and high primary production (Robarts et al. 1995). Wetland sediments can bind, absorb, and release nutrients and sediments from the water column and from watershed runoff (Verhoeven et al. 2006; Watchorn et al. 2012)

Primary production in wetlands responds quickly to environmental changes, natural and anthropogenic, including ecosystem structure, global climate change, watershed runoff and human density. Furthermore, algal community composition is also often used as an indicator of aquatic health and status (Boyer et al. 2009; Chaudhuri et al. 2012). Primary productivity may be influenced by exposure. Ecosystems exposed to high wind or wave action result in enhanced sediment and nutrient re-suspension. Total depth also influences wind re-suspension of sediments and nutrients, with shallow aquatic sites with low submersed vegetation cover experiencing greater mixing caused by wind relative to deeper sites or sites with increased cover of submersed vegetation. These factors can increase suspended sediment and nutrients in the water column and ultimately primary production of algae (Eleveld 2012; Kaitaranta et al. 2013; So et al. 2013; Reardon et al. 2016).

Anthropogenic activities are driving global climate change, which shifts temperature, precipitation, hydrology, CO_2 production and many other environmental variables. These changes have negative impact on ecological function of wetlands by altering species composition, nutrient dynamics, water levels and increasing primary productivity (Reddy et al. 2010). For example, as temperature changes, phytoplankton species dominance shifts from diatoms to cyanobacteria and higher chlorophyll-a concentrations (Konopka and Brock 1978). In addition to global climate change, other anthropogenic activities (agriculture, waste water, population density and development pressures caused by economic growth) influence structure of aquatic ecosystems and ultimately primary productivity.

Temperate and permanent wetlands have high primary productivity and phytoplankton production, crucial for ecosystem function and brooding-rearing grounds for waterfowl populations. Robarts et al. (1995) studied primary productivity and phytoplankton production in three temporary wetlands in Northern Alberta. The wetlands were recorded for depth, volume, primary productivity, phytoplankton production and classification, nutrient concentrations, pH and other physical parameters. The three wetlands were all similar in nitrogen and phosphorus concentrations although varied physically. The rates of primary productivity varied greatly throughout the study due to the physical changes of the wetlands. Robarts and his colleagues determined that primary productivity was not related to the nutrient concentrations within these wetlands, but was related to changes in wetland volume and phytoplankton crops. They also determined that the changes in primary productivity were related to changes in solar radiation, macrophyte growth and decomposition, and zooplankton (Robarts et al. 1995).

Primary production in Delta Marsh has been extensively studied since the 1980s. Projects have included studying the impacts of common carp, nutrient loading and hydrology on submersed vegetation and algal production (McDougal 2001; Badiou 2005; Hnatiuk 2006; Hertam 2010; Bortoluzzi 2013). The primary production in Delta Marsh includes upland vegetation, emergent vegetation, submersed vegetation and algae (Batt 2000). Prior to the Marsh Ecology Research Program (MERP), there was no clear understanding of biotic and abiotic influences on algal production in Delta Marsh (Robinson et al. 2000). In Delta Marsh and most shallow prairie wetlands there are four different algal assemblages including phytoplankton, epipelon, epiphyton and metaphyton. The importance of algal production for total primary production in wetlands has been overlooked due to the large macrophyte cover. Although with research performed under the MERP project it was determined that algal productivity in prairie wetlands may be as significant as macrophyte productivity for total primary production due to their high productivity and turnover (Robinson et al. 2000). Algal standing biomass can account up to 34 % of primary production in an unmanipulated area, when nutrient concentration is enhanced algal standing biomass can account up to 57% of total primary production in Delta Marsh (McDougal 2001). The MERP project successfully developed models for algal production in prairie wetlands and determined the importance of algae for wetland function, overall productivity, structural habitat and support for secondary productivity (Robinson et al. 2000; Murkin et al. 2000). It was suggested that further research on algal production and nutrient cycling, trophic dynamics, and taxonomy be required to fully understand the algal dynamics in Delta Marsh (Murkin et al 2000). McDougal (2001) studied primary production of benthic algae and planktonic algae in Delta Marsh in 1995 and 1996. A mesocosm experiment revealed that benthic algae respond to increased light availability and phytoplankton algae respond to increased nutrient availability. McDougal (2001) suggested that with increased carp presence and decreased submersed vegetation cover, there would be an increase in competition between algae for light and nitrogen availability. My research focuses on the relationship between phytoplankton primary production and the biotic and abiotic variables in Delta Marsh.

1.2 Eutrophication

Freshwater ecosystems are highly valuable for the local economy; this value is found in fishing industries, recreation and provide habitat for migrating birds, fish and other organisms (Yates et al. 2012). Health of freshwater ecosystems is often degraded by excessive nutrient loading from anthropogenic influences such as, wastewater and agricultural runoff, resulting in cultural eutrophication (Schindler et al. 2012; Carlson et al. 2013). Coastal wetland health influences the stable ecological function of an attached lake. They act as a filter for nutrients and sediments before they are discharged into the lake and provide habitat for many species (Kreiger et al. 1992; Watchorn et al. 2012). Nutrient loading is a natural phenomenon arising from erosion, weathering and nutrient cycling (Bricker et al. 2008), although it is exacerbated by urban, industrial and agricultural practices (Arheimer et al. 2004). Excessive nutrient loading can have significant impact on the ecological function and water quality of wetlands. Changes to nutrient dynamics, species composition and ecological structure are a few of the consequences caused by increased nutrient loading (Verhoeven et al. 2006).

Eutrophication is a natural phenomenon that can be enhanced by anthropogenic influences; the increased duration and frequency of eutrophic conditions lower the value of the aquatic system (Hunter et al. 2012). Eutrophication has deteriorated the health of many aquatic ecosystems across the world (Faassen et al. 2012; Li et al. 2010). One of the symptoms of eutrophication is increased abundance and frequency of Cyanobacteria, also known as blue-green algae, that flourish under nutrient rich conditions and can cause serious health related risks including liver failure, respiratory and dermatological issues, as well as neurodegeneration (Faassen et al. 2012; Li et al. 2010). Cyanobacterial blooms often lead to depleted oxygen levels that cause fish populations to decline, changes to nutrient and species dynamics and compromise aesthetics of the water (Bourne et al. 2002). Observed blooms are often used as a qualitative indicator of health and trophic status (Boyer et al. 2009).

Wetlands are becoming eutrophic due to anthropogenic activities, increase in population densities, industrial practices, wastewater and agricultural intensification (Ansari and Gill 2014). Wetlands display similar responses to eutrophication as observed in shallow lakes, although the biogeochemical processes respond differently (Álvarex-Cobelas and Sánche-Andrés 2014). The increase of nutrient loading to wetlands causes shifts in sediment and water chemistry, resulting in changes in biotic and abiotic dynamics. The increase of nutrients within the water column results in a shift in wetland metabolism and an increase in primary productivity. Surface water algal blooms can cause destruction of benthic habitat for flora and fauna, macrophytic growth and submersed vegetation due to the increased algal cover resulting in limited light availability and reduced oxygen (Dorgham 2014). If excessive nutrient loading is prolonged there may be a reduction in wetland sediment nutrient retention, which can result in increased nutrient availability in the water column and productivity (Álvarex-Cobelas and Sánche-Andrés 2014). In addition to the high cover of wetland area in Canada (Campbell et al 2000), this country also has high cover of agricultural practices in its prairie region (Main et al. 2014). The increase of agriculture in the prairie pothole region of Canada is resulting in increased eutrophication, sedimentation and decreased vegetation cover, causing threat to wetland integrity (Main et al. 2014).

Changes in nitrogen and phosphorus loading can alter the N:P molar ratio within the water column, resulting in shifts of algal class communities. In the 1990's the nutrient loading to Lake Peipsi and Võrtsjärv (Estonia) were altered when nitrogen loads were significantly decreased, altering the N:P ratio and driving cyanobacterial blooms. Nõges et al. (2008) analyzed the trends and causes of cyanobacterial blooms in the two lakes in northeastern Europe, from 1985 to 2004. Lake Võrtsjärv had twice the TN concentration than in Peipsi, but both had similar TP concentrations, causing Võrtsjärv to have twice the N:P ratio relative to Peipsi. The two lakes had similar percentages of cyanobacteria, however Lake Peipsi had much higher nitrogen fixing cyanobacteria. They also noted that temperature controls algal growth, nitrogen fixation (Brauer et al. 2013) and phosphorus recycling from the sediment (Genkai-Kato and Carpenter 2005). Temperature has a crucial role in cyanobacterial nitrogen fixation, with low water temperature limiting nitrogenase activity and increasing respiratory cost for nitrogen fixation (Brauer et al. 2013).

Nutrient limitation of N and P on algal production in Delta Marsh was studied by Bortoluzzi (2013) with the use of nutrient-diffusing substrata (NDS). NDS vials were treated with four different treatments including N enrichment, P enrichment, N and P enrichment and a control. The vials were dispersed across the marsh and analyzed after three weeks of algal production. Periphyton algal growth did not respond to P enrichment, and was most stimulated by N enrichment. Bortoluzzi (2013) determined that 71% of periphyton in Delta Marsh was N-limited. In a similar study, McDougal (2001) determined that algal production was N-limited during a mesocosm experiment in Delta Marsh. McDougal found that algal standing crop biomass increased from 34 % to 57 % of all primary production with nutrient enrichment in Delta Marsh.

1.3 Land Use and Water Quality

Water quality is often reflective of land use and climate that are specific to different regions. Water flows across the land as surface water, ground water and through tributaries. The nutrients and sediments gathered in the water are dependent on the type of land over (and through) which the water flows. Change in land use affects the hydrological conditions of the land and ultimately the volume of water and nutrient loading emptying into downstream water bodies (Harbor 1994; Zedler 2003; Ehanzadeh et al. 2016). The highest water input in the Canadian Prairies occurs during the spring snowmelt, followed by summer precipitation and runoff caused by extreme precipitation (Van Der Kamp et al. 1999). Different land types are able to absorb varying levels of moisture, so each land use will produce different levels of runoff in response to precipitation (Correll et al. 1992; Van Der Kamp 1999). Different land types produce significantly different nutrient loading to downstream aquatic ecosystems, with agricultural croplands exporting more nutrients than forested and pastured areas due to fertilizer use and tillage practices (Correll et al. 1992; Zedler 2003). Quinn et al. (1997) observed that water quality degradation of natural streams in New Zealand, occurred after 2/3 of forested land was replaced by urbanization and agriculture, and that the characteristics of the streams appeared to vary as a function of land use. Additionally, these authors found that modified land often caused increased light availability, temperature, finer and more suspended solids and nutrient concentrations. These changes increased photosynthesis, algal blooms and significantly increased density of zoological benthic communities. Excessive nutrient loading can increase nitrogen and phosphorus availability in the water column, shifting species dominance and increasing algal abundance, ultimately declining aquatic ecosystem health (Verhoeven et al. 2006).

Research has shown that conservation tillage produces less suspended sediment in seasonal runoff than in conventional tillage in areas with high humidity and temperature (Tiesson et al. 2010). Tiesson et al. (2010) found that while conservation tillage decreases suspended sediment, it increases dissolved inorganic nutrients, which are considered to be bioavailable (Carlson and Simpson 1996).The prairie climate in Canada is cool and produces higher volume of snow melt than spring and summer precipitation. This phenomenon in turn can increase algal biomass in downstream aquatic ecosystems. Similarly, Timmons and Holt (1997) studied the nutrient losses from surface runoff in the prairies. They determined that N, P and K loading were higher from agricultural land than relative to native prairies due to tillage. Their results indicated that 89% and 95% of N and P runoff, respectively was attached to sediment. In order to reduce nutrient runoff to downstream ecosystems, land use management must be properly studied to keep nutrients on the land (Timmons and Holt 1997; Tiesson et al. 2010).

Carpenter et al. (1998) discussed the relationship between nutrient loading and agricultural activities. The use of fertilizers and manure production and application drive N and P export to coastal waters. There are more nutrients applied to the land than nutrients removed during harvest. This increases the accumulation of nutrients in the sediment and ultimately increases nutrients in downstream aquatic ecosystems and the atmosphere. Manure is also often applied in excess on land with high livestock densities. The concentration of nutrients in manure is comparable to raw human waste, although not monitored similarly. The increased use of fertilizers and manure heightens the concentrations of P and N in the soil. Nitrogen is highly mobile in the sediment and atmosphere, although both N and P will eventually make their way downstream by means of streams, surface and groundwater runoff and atmospheric deposition for N.

Aquatic ecosystems including wetlands, rivers and lakes interact with the watershed, and their biological, chemical and physical characteristics often reflect anthropogenic changes within the surrounding watershed (Morrice et al. 2008). Morrice et al. (2008) studied the relationship between land use and water quality/chemistry in 98 Laurentian Great Lakes coastal wetlands and determined which human induced stress variables were controlling water quality variables, including water chemistry, clarity, chlorophyll-a. They expected that wetlands strongly associated with their watershed would display strong relationships between water quality and anthropogenic activities relative to wetlands connected to lakes. Human induced stressors including agriculture, atmospheric deposition, point source pollution and human population data were analyzed for each wetland watershed area. The results of this study indicated that the water chemistry in the Great Lakes coastal wetlands varied based on the combination of the anthropogenic activities. Agriculture, atmospheric deposition, point source pollution and human population data were all significant factors in predicting water quality across the coastal wetlands (Morrice et al. 2008).

1.4 Delta Marsh

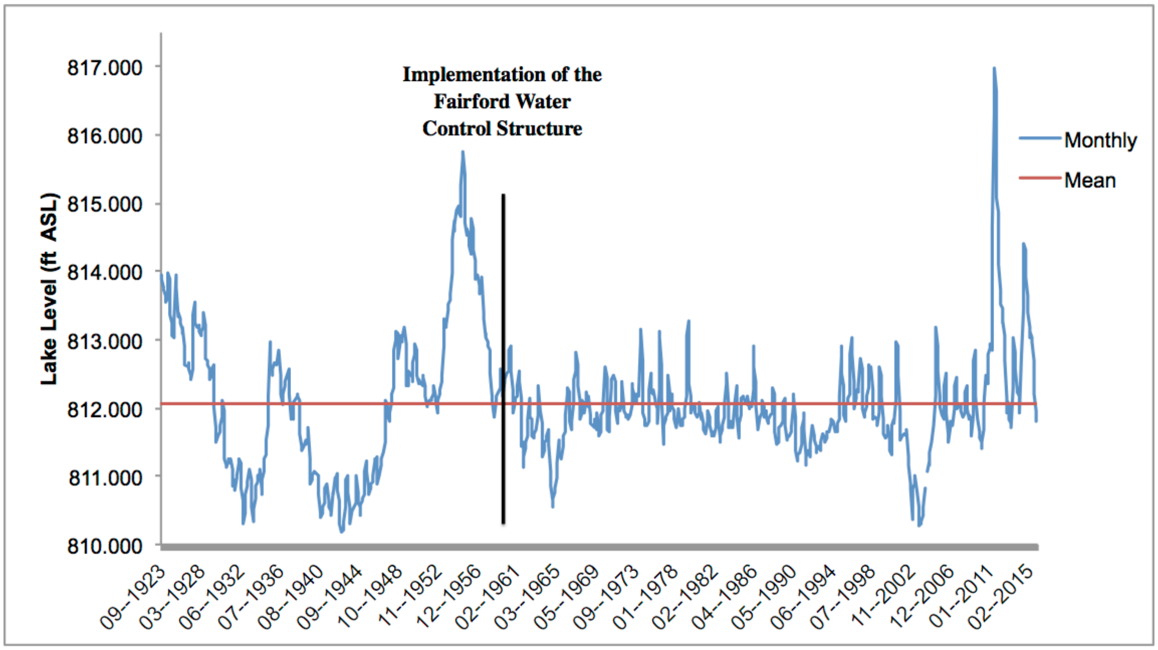
Delta Marsh is a freshwater coastal wetland located on the south basin of Lake Manitoba, Manitoba, Canada (50.10 N, 98.17 W) (Figure 1.1). There are 564km^2 (0.6km^2/km) of coastal wetlands along the shore of Lake Manitoba (Watchorn et al. 2012) Delta Marsh is approximately 19,000-hectares (190 km^2), making it one of the largest coastal wetlands in North America. This large wetland varies with large open bays to small sheltered ponds. Emergent vegetation throughout Delta Marsh is dominated by *Typha X glauca* (hybrid cattail) and previously dominated by *Phragmites australis* (giant reed). The watershed on the southern edge of Delta Marsh spans a gradient from intensive agriculture in the western and central portion of the watershed to shrubland and grassland in the eastern portion of the watershed (Appendix A).

Traditionally, Delta Marsh was used for hunting, trapping and fishing by people from all around the world (Suggett and Goldsborough 2015) and has been designated as an internationally important wetland under the 1982 Ramsar Convention (Ducks Unlimited Canada 2012; Watchorn et al. 2012). It is located at the south basin of Lake Manitoba, the third largest lake in Manitoba, Canada. Many communities surrounding Lake Manitoba rely on the lake and its watershed for agricultural activities, local tourism, hunting, recreational and commercial fisheries and some mineral extraction to support their local economies. Health of the lake is crucial to maintain these economic values (Topping et al. 2003).

Delta Marsh was formed from the connection of the Assiniboine River and Lake Manitoba approximately 2500 years ago (Teller and Last 1981; Suggett and Goldsborough 2015). The Assiniboine River naturally diverted to Lake Manitoba, causing erosion and redistribution of sand and sediment creating a beach ridge. The formation of the beach ridge to the south of the lake separated Delta Marsh from Lake Manitoba (Teller and Last 1981). Delta Marsh has been declining in ecological integrity over the last half century due to changes in hydrology, the construction of the Fairford dam, and the introduction of invasive species (Ducks Unlimited Canada 2012; Watchorn et al. 2012).

Lake Manitoba undergoes natural cycles of high and low water levels, ranging from 810.5 to 812.5 ft asl (Topping et al. 2003). This phenomenon is important for coastal marshes, as periods of high and low water are crucial to maintain wetland health. The Fairford River is the sole outlet to Lake Manitoba and in the early 1900s the connecting channel was dredged in hopes to lower lake level (Farlinger et al. 2003; Topping et al. 2003). Although this channel was unsuccessful, in 1931, a structure was created in the river to avoid low lake levels, but did not eliminate flooding of the surrounding watershed (Topping et al. 2003). In 1961, the Fairford River Water Control Structure was constructed to cause artificial stabilization of water levels in the lake, reducing natural flooding and drought periods in the marsh (Figure 1.2) (Shay 1986). The stabilization of water in the marsh reduces events of the natural wet-dry cycle. The wet-

Figure 1.2. Water surface elevation (m above sea level) of Lake Manitoba before and after the implementation of the Fairford Water Control Structure. Water surface elevation data recorded from the Steeprock Station 05LK002, Manitoba.



dry cycle is crucial for a healthy wetland ecosystem, the wetland must undergo periods of high water levels (lake phase) and periods of drought (dry phase) in order to expose the seedbed and regenerate wetland vegetation (van der Valk 2005). Long-term stabilization of water levels in a coastal marsh can lead to degradation of the ecosystem (van der Valk 2005).

Change in Lake Manitoba water quality is attributed to the creation of the Portage Diversion in 1970. The Portage Diversion is an artificial connection between the Assiniboine River and Lake Manitoba; it was created to control flooding in Winnipeg, Manitoba (Page 2011). Approximately 11% of the total annual volume input into Lake Manitoba arises from the Portage Diversion (Page 2011). With only a mean annual input of 11% to the lake, the Portage Diversion can account for up to 87% of TP loading to Lake Manitoba as observed in 2011 (Nicholson 2012). This connection is the highest input of TP to the lake, varying in levels of particulate phosphorus, total dissolved phosphorus and soluble reactive phosphorus. Diversion flow is variable annually, as is the nutrient loading to Lake Manitoba. Diversion flows accounted for 82% of TP load to the lake in 2009, 38% of TP loads in 2010, and 87% in 2011 (Nicholson 2012). The Portage Diversion is likely to have an effect on sedimentation and water quality in Delta Marsh due to the connectivity to Lake Manitoba and the fail-safe on the western section of the Marsh.

Nutrient availability in Delta Marsh is also driven by other anthropogenic activities and invasive species. Activities in the watershed, which is dominated by agriculture, can increase nutrient loading to the marsh. The Delta Marsh watershed is dominated by agricultural activities on the west, and shrubland and pastureland on the east. This land use gradient is expected to cause spatial variation of water quality across the marsh. Nutrients are also re-introduced into the water column by resuspension of wind and invasive species, such as the benthivorous fish, the common carp. The combination of nutrient loading and resuspension are expected to alter biogeochemical processes and drive algal productivity in Delta Marsh.

1.4.1 Previous Research on Delta Marsh

Previous research conducted at Delta Marsh has included studying wet-dry cycle in prairie wetlands (Murkin et al. 2000), the effect of common carp on the ecological integrity, water quality, native fish populations, algae and submersed vegetation (Badiou 2005; Hnatiuk 2006; Parks 2006; Hertam 2010), and algal production, patterns in hydrology and water chemistry across the marsh (McDougal 2001; Bortoluzzi 2013).

Badiou (2005) studied the impacts of common carp on water quality, sedimentation rate, submersed vegetation, phytoplankton biomass, zooplankton, benthic invertebrates and forage fish in large experimental wetlands and small mesocosms. The large experimental wetlands were performed in the Marsh Ecology Research Program (MERP) cells located on the northern beach ridge of Delta Marsh. The MERP project was designed to study prairie wetland ecology and the wet-dry cycle, including short-term and long-term studies over the ten years of fieldwork. The MERP research included monitoring hydrology, vegetation, water chemistry, invertebrates, vertebrates and algae (Murkin et al. 2000). It was suggested that presence of common carp in Delta Marsh enhances rate of eutrophication. In both experiments, carp increase turbidity, suspended sediment, rate of sedimentation, phytoplankton chlorophyll-a concentration, and zooplankton density, and they decrease submersed vegetation, benthic invertebrates, forage fish and light availability. Badiou (2005) determined that Delta Marsh (~ 400kg/ha carp density) increases P in the water column that is equivalent to 66% of internal loading. In order to reduce carp impact on the ecological integrity of Delta Marsh, their populations must be managed.

The relationship and effects of common carp on water quality, algal production and submersed vegetation was further studied by Hnatiuk (2006) and Hertam (2010), and the effect on native fish, invertebrates and amphibians by Parks (2006). Hnatiuk (2006) and Hertam (2010) studied the differences of water quality, algal abundance and submersed vegetation between ponds that were isolated or accessible to common carp in Delta Marsh. The goal of this research was to determine how the conditions of the ponds changed after manipulation of carp access. Hertam (2010) continued the research from Hnatiuk in 2003/04, with the additional research on the use of nutrient-diffusing substrata (NDS) to determine N and P limitation for algae in Delta Marsh. Hnatiuk (2006) and Hertam (2010) both hypothesized that ponds accessible to carp would have increased turbidity, nitrogen and phosphorus concentrations, phytoplankton and periphyton biomass and lower submersed vegetation compared to ponds isolated from carp. Hnatiuk (2006) and Hertam (2010) found that ponds with the presence of common carp experienced a loss of submersed vegetation, increase in phytoplankton, periphyton and sediment associated algae and an increase in suspended solids. There was no significant difference between N and P concentrations in the connected and isolated ponds, likely due to other factors such as autotrophs, sediment binding, denitrification, vegetation and algae.

Although there were no significant differences, connected ponds had the highest concentration of N and P. Throughout years of this study, the N and P deficiencies varied in connected and isolated ponds, likely attributed to the spatial and temporal variation of carp densities, and physical changes in the ponds.

The first comprehensive study of nitrogen and phosphorus limitation on algal production in Delta Marsh was performed by Bortoluzzi (2013). She studied the spatial and temporal patterns of hydrology, water chemistry and algal nutrient limitation of N:P from 2002-2005 in Delta Marsh. She hypothesized that with increased distance from Lake Manitoba would result in increased nutrient concentrations and algal production due to the dilution and flushing effects of the lake. Results indicated that sites closer to Lake Manitoba had the greatest degree of variation in water chemistry and clarity due to the dilution and flushing. Nutrient concentrations including TN, DIN, TP, TDP, chloride and sulfate, and conductivity decrease with increasing connectivity to Lake Manitoba. Additionally there was higher algal production as distance from the lake increased. Across the marsh, the TN:TP ratio was spatially and temporally variable with a mean ratio of 47, indicating the marsh is limited in P. When considering only the bioavailable nutrients, the marsh was limited in N. Bortoluzzi further studied the effects of N and P limitation on periphyton algal growth with the use of NDS. Periphyton growth was not stimulated by P alone, and was most stimulated by N. Bortoluzzi (2013) found that 71% of periphyton biomass was limited by N in Delta Marsh. Algal production in Delta Marsh was most stimulated by the addition of N compounds. She also determined that lake connectivity and hydrological conditions are highly influential on marsh function.

A mesocosm experiment was performed in Delta Marsh in 1995/96 to determine if the water column can shift from a clear water state to a turbid water state with the removal of submersed vegetation and addition of N and P (McDougal 2001). A clear water state is dominated with submersed vegetation and a turbid water state is dominated by phytoplankton and high concentrations of suspended solids (Scheffer 1993). The responses of phytoplankton, epiphyton, periphyton and benthic algae varied in each experimental enclosure due to light and nutrient availability. In the control experiment, epiphyton dominated with 70 % of the algal assemblage (McDougal 2001). When submersed vegetation was removed in the enclosures, there was a strong response in benthic algae, suggesting that benthic algae were light limited. Phytoplankton assemblages did not respond to macrophyte removal, but to nutrient enrichment. Both benthic and planktonic algae were stimulated with nutrient enrichment, but benthic algae were more competitive, possibly caused by the limited light due to the enclosure walls. Algal assemblages account for a high proportion of primary production in Delta Marsh. Algae account for up to 34 % of standing crop biomass in the control enclosures, and up to 57 % in the nutrient enrichment enclosures. McDougal (2001) determined that 56 % to 77 % of algae in Delta Marsh were benthic and algal production was N-limited. McDougal (2001) determined that there would be increased competition for light and N availability with the reduction of submersed vegetation and increased population of common carp.

1.4.2 Restoring the Tradition

Due to the continuing decline in the ecological integrity of Delta Marsh, in 2009 a group of stakeholders including Ducks Unlimited Canada, Delta Waterfowl Foundation, La Salle Redboine Conservation District, University of Manitoba, Manitoba Provincial Conservation & Water Stewardship, Department of Fisheries and Oceans, and Environment Canada began a multi-year project aimed at the remediation of Delta Marsh; named by Ducks Unlimited as "Restoring the Tradition" in recognition of restoring the waterfowling history of the marsh that goes back over a century. Their goal is to determine which natural or anthropogenic variable(s) have the most impact on the marsh's function in order to design a management plan to restore ecological function. The project has been designed to occur over ten years, the first five years researching carp exclusion, hydrology and nutrient dynamics as factors contributing to the marsh decline, and the last five years implementing management plans based on the research.

Cyprinus carpio (Common Carp) is an invasive fish species introduced to the North American watershed in the 1800s (Badiou 2005; Badiou and Goldsborough 2014). Their benthivorous nature increases suspended sediment and water column nutrients, changes fish, zooplankton and phytoplankton species composition and reduces submersed vegetation cover, causing wetland degradation (Badiou 2005; Hertam 2010; Ducks Unlimited Canada 2012; Badiou and Goldsborough 2014).

Carp exclusion structures were implemented along each channel connecting the marsh to Lake Manitoba in the winter of 2012/13. A sheet pile weir was placed into Deep Creek, Cram Creek, Waterhen Creek and Crooked Creek, while similar structures were also located in Delta Channel and Fish Creek (Figure 1.1). The structure was designed to allow native fish species such as *Sander vitreus* (Walleye), *Esox lucius* (Northern Pike), *Perca flavescens* (Yellow Perch), *Aplodinotus grunniens* (Freshwater Drum), *Catostomus commersonii* (White Sucker) and other fish species into the marsh to use the crucial wetland habitat for feeding and spawning while excluding Common Carp. The carp exclusion structures were designed with gates that are dropped seasonally to exclude carp from the marsh. The threshold width was determined based on monitoring of fish communities in 2009-2012. The 70 mm screen width allows an estimated 80% of walleye, 98% of northern pike, 99% of white sucker, 97% of freshwater drum and 7% of carp into the marsh (Emery and Wrubleski personal communication Wrubleski, personal communication). Carp exclusion screens were first installed in 2013, although they were unsuccessful at excluding carp because the screen installation date was too late, after the majority of carp had already entered the marsh from Lake Manitoba. A strong windstorm on July 1, 2014, combined with prevailing high lake levels, caused a breach of dikes at the Crooked Creek and Waterhen structures (Figure 1.1). Similar erosion occurred at the Deep Creek and Cram Creek structures on the west side of the marsh when the failsafe for the Portage Diversion was breached, causing overflow around the two structures (Figure 1.1) (Wrubleski, personal communication). The breach was repaired temporarily with sand bags over the winter of 2014/15, as continuing high lake levels impeded access of heavy equipment needed for proper repairs. In 2015, a breach into the marsh near the Crooked Creek structure was caused by carp activity and an intense May storm. The storm caused vegetation to accumulate on the screens, creating hydraulic pressure that causing the screens to be torn out of the structure, allowing carp entrance to the marsh. During the winter of 2015/16, repairs to the structures and dykes were performed fully and plans for weekly cleaning of the screens, to remove accumulated vegetation, were implemented by Manitoba Sustainable Development. Exclusion of large carp was mostly successful in 2016 (Wrubleski, personal communication; Emery and Wrubleski 2017).

Previous studies in Delta Marsh have shown that carp alter water quality, sedimentation, submersed vegetation abundance, phytoplankton diversity and biovolume, and fish species composition to different degrees (Badiou 2005; Parks 2006; Badiou and Goldsborough 2014). Overall, common carp presence increases phytoplankton biomass (Badiou 2005). Current research by Ducks Unlimited and the University of Manitoba, performed since 2009, has examined fish populations across the marsh before and after carp exclusion by methods of gill netting, hoop netting, electrofishing and small fish trapping (Wrubleski, personal communication). Biweekly water quality monitoring at sites across the marsh has been done by researchers from the University of Manitoba annually since 2009. Measurements of biological, chemical and physical variables has occurred between May and August at 48 sites. In 2014 and 2016, submersed vegetation was recorded by Ducks Unlimited in the shorelines of bays and along transects, respectively. Additionally, vegetation type was recorded to species or mixed stands (Emery, personal communication).

Lake Manitoba hydrology and the surrounding watershed has been studied between 2005 and 2013 and its relation to marsh hydrology in 2015 by researchers from the University of Manitoba (Berke 2011; Page 2011; Nicholson 2012; Fred 2013; Aminian 2015). The impact of the Portage Diversion on water composition and sedimentation rates has been a large concern for Lake Manitoba and Delta Marsh, but prior to 2005 there was no clear understanding of the implications of its use (Page 2011). As discussed in Section 1.4, Page (2011) noted that during years of use, 11% of the total annual volume to Lake Manitoba comes from the Portage Diversion, additionally in years of use it can release up to 87% of the TP in Lake Manitoba, as what was observed in

2011 (Nicholson 2012). The Department of Civil Engineering at the University of Manitoba began studying the impact of hydrological conditions to the function and ecological structure of Delta Marsh. Aminian (2015) researched the effect of lake hydrology, mixing and wind/wave action on eastern section of Delta Marsh. He determined that 85% of the inflow from Lake Manitoba into Delta Marsh flows through Clandeboye Channel, the most eastern channel into the marsh (Figure 1.1). This hydrological condition causes mixing to occur from east to west in the marsh. While the Portage Diversion was of concern regarding nutrient and sediment loading, it was discovered that the water from the Portage Diversion would remain in the lake unless there was wind action. He found that wind friction from the north had the largest role in the water composition and volume in Delta Marsh, followed by the Portage Diversion and the natural inflow from the lake. The inflow and outflow of Delta Marsh and Lake Manitoba contains different nutrient compositions. Aminian concluded that inflow into Cadham Bay is mainly of Portage Diversion water, assumed to have high nutrient composition, while outflow from the marsh is mainly marsh water, assumed to have low nutrient composition. This phenomenon can be driven by high winds causing inflow and Lake Manitoba draw down causing outflow. This is a concern for nutrient dynamics and health of Delta Marsh (Aminian, 2015). Not only does wind increase inflow from Lake Manitoba, but it can cause mixing and resuspension of sediment. So et al. (2013) determined that wind speeds over Lake Apopka, Florida higher than 30 kph caused re-suspension of sediments and nutrients into the shallow (< 3 m) water column, increasing algal blooms. Due to the shallow nature of Delta Marsh, it is assumed wind speeds greater than 25 kph would cause similar resuspension. The hydrological relationship between the watershed and Delta Marsh were monitored by Schellenberg in 2014 and 2015. His research addressed the hydrological dynamics of the watershed and the spatial variation of runoff that empty into the southern shore of the marsh. He developed a hydrological model that addresses the water balance across the watershed and forecasts runoff to the marsh based on multiple conditions during spring snowmelt and precipitation events. This as-yet-unpublished research will answer questions regarding the relationship between land use, nutrient loading and water quality, and where management can be implemented.

Evaluating causes of poor water quality is crucial to create management plans to improve health and ecological function in Delta Marsh and Lake Manitoba. As mentioned by Aminian (2015), wind friction, Portage Diversion flow, and natural inflow from Lake Manitoba influence wetland hydrology, nutrient dynamics and mixing. However, nutrient loading to Delta Marsh is of concern with regards to water quality and eutrophication. Although directly attached to Lake Manitoba, Delta Marsh also receives water from the land south of its border. The watershed spans a gradient of intensive agricultural practices on the western and central portions, to shrubland and grassland in the eastern portions. Water empties into Delta Marsh from streams and tributaries that span the watershed from east to west. The gradient of land use within the watershed suggests there may be a difference in nutrient loading, causing spatial variation of water quality across Delta Marsh. Cyanobacterial blooms have been observed in the marsh and recently have become more severe, increasing in size and duration. Cyanobacteria can produce toxins, can outcompete other phytoplankton species, and reduce light penetration for other algal and aquatic vegetation (Wu et al. 2010). The variability of sediment chemistry, historical nutrient accumulation and sediment sorption capacity are currently being investigated by Chris Hope (Department of Biological Sciences, University of Manitoba) to improve our understanding of nutrient dynamics in Delta Marsh.

1.4.3 Summary of Previous Research

Water quality monitoring has been ongoing since the late 1990s across Delta Marsh in the same sites monitored throughout this research. A summary table displays the means for a suite of parameters collected in 1998/99 and 2009-2014 excluding 2011 due to the flood which inhibited sampling (Table 1.1). Variables included are Secchi depth (cm), turbidity (NTU), euphotic zone depth (cm), total suspended solids (mg/L) and total chlorophyll-a and pheophytin concentration (μg/L). Data shows that water clarity increased between 1998 and 2014, the Secchi and euphotic zone depth increase and the turbidity and total suspended solids decreased. This trend may be influenced by the changing density of common carp in the marsh and other environmental changes such as flooding. Total chlorophyll-a and pheophytin concentrations are variable from 1998 to 2014, with the highest mean in 1998/99 of 155 μg/L and lowest mean in 2014 of 35 μg/L. Mean total chlorophyll increased between 2010 and 2012, from 46 μg/L to 128 μg/L (Table 1.1). The increase of phytoplankton biomass may be a result from the flood in 2011, introducing increased nutrients into the water column. The differences for the mean water quality parameters may result from changes in carp density due to the construction of the exclusion structures in 2012, the change in water chemistry, submersed vegetation cover, flooding and Portage Diversion use. The results from this research can be compared to the historical conditions in Delta Marsh to determine what conditions are influencing changes in water quality. The Portage Diversion ran in early April of 2015

Table 1.1. Historical means and ranges for water quality variables analyzed in 45 sites across Delta Marsh during the open water season in 1998/99 and 2009-2014, excluding 2011 due to a flood that inhibited sampling. Unpublished data of Goldsborough.

Variable	1998/99	2009	2010	2012	2013	2014
Secchi Depth (cm)	35 (12-135)	49 (12-160)	40 (9-181)	50 (9-172)	62 (15-142)	80 (14-183)
Turbidity (NTU)	21 (1-77)	29 (3-174)	34 (1-188)	32 (1-173)	20 (2-280)	13 (1-139)
Euphotic Zone Depth (cm)	N.A.	N.A	N.A	151 (28-472)	162 (22-601)	226 (44-941)
Total Suspended Solids (mg/L)	50 (1-314)	29 (0-130)	31 (0-165)	31 (0-223)	26 (0-214)	18 (0-166)
Total Chlorophyll-a and Pheophytin (µg/L)	155 (7-1,067)	66 (3-328)	46 (0-261)	128 (8-792)	98 (5-883)	35 (1-582)

and was not used in 2016. The marsh experienced partial exclusion in 2015 and complete exclusion for the first time in 2016. These changes may result in atypical conditions than recently observed in Delta Marsh.

1.5 Objectives and Hypotheses

The overall objectives of my thesis were to determine which chemical and physical variables in Delta Marsh and its watershed drive phytoplankton production. My first objective was to determine which chemical or physical variable(s) have the highest correlation with phytoplankton chlorophyll-a concentration in sites across the marsh. I hypothesized that sites with high light and nutrient availability would yield higher phytoplankton chlorophyll-a concentrations because primary production is often limited by nutrient supply and light penetration.

My second objective was to determine if there is a spatial gradient in nutrient loading to the marsh caused by a land use gradient in the watershed (Appendix B). I hypothesized that land use dominated with agricultural activities would yield increased nutrient loading to the marsh because manipulation of land alters watershed hydrology and fertilizer application increases the concentration of nutrient runoff. I also hypothesized that the sections of the marsh receiving the highest annual nutrient export would exhibit more eutrophic characteristics, resulting in larger phytoplankton blooms.

To test these hypotheses, I carried out a study of water quality at Delta Marsh, Manitoba, monitoring biological, chemical and physical parameters in the water column at 45 sites within Delta Marsh. Additionally, I monitored watershed nutrient runoff and total export to the marsh to analyze the relationship between land use and nutrient loading in 2015 and 2016 (Appendix B). This research occurred over two years following the start of the carp exclusion project from Delta Marsh in 2013.

2.0 Spatial Variation of Water Quality

2.1 Introduction

Delta Marsh is a large freshwater coastal wetland, spanning approximately 19,000-hectares and is divided into three sections (Ducks Unlimited Canada, 2012; Figure 2.1). The east, west and center sections are diverse in depth, aerial size, exposure and connectivity, expected to result in variation of phytoplankton blooms, fish community composition, submerged and emergent vegetation abundance, water quality, and hydrology. Water quality is expected to vary spatially due to anthropogenic influences such as agricultural activities, cabin development, introduction of common carp, and topographic factors such as connectivity to Lake Manitoba and exposure to name a few. The variability in biological, chemical and physical characteristics cause spatial and temporal distribution of phytoplankton blooms across the marsh.

This chapter focuses on the spatial variation of water quality and determines which biological, chemical or physical variable(s) have the highest influence on phytoplankton chlorophyll-a concentration across Delta Marsh. I hypothesized that sites with high light and nutrient availability would yield higher phytoplankton chlorophyll-a concentrations. I evaluated this hypothesis by examining correlations between phytoplankton abundance in 2015 and 2016 and environmental variables spanning 45 sites across the marsh.

2.2 Materials and Methods

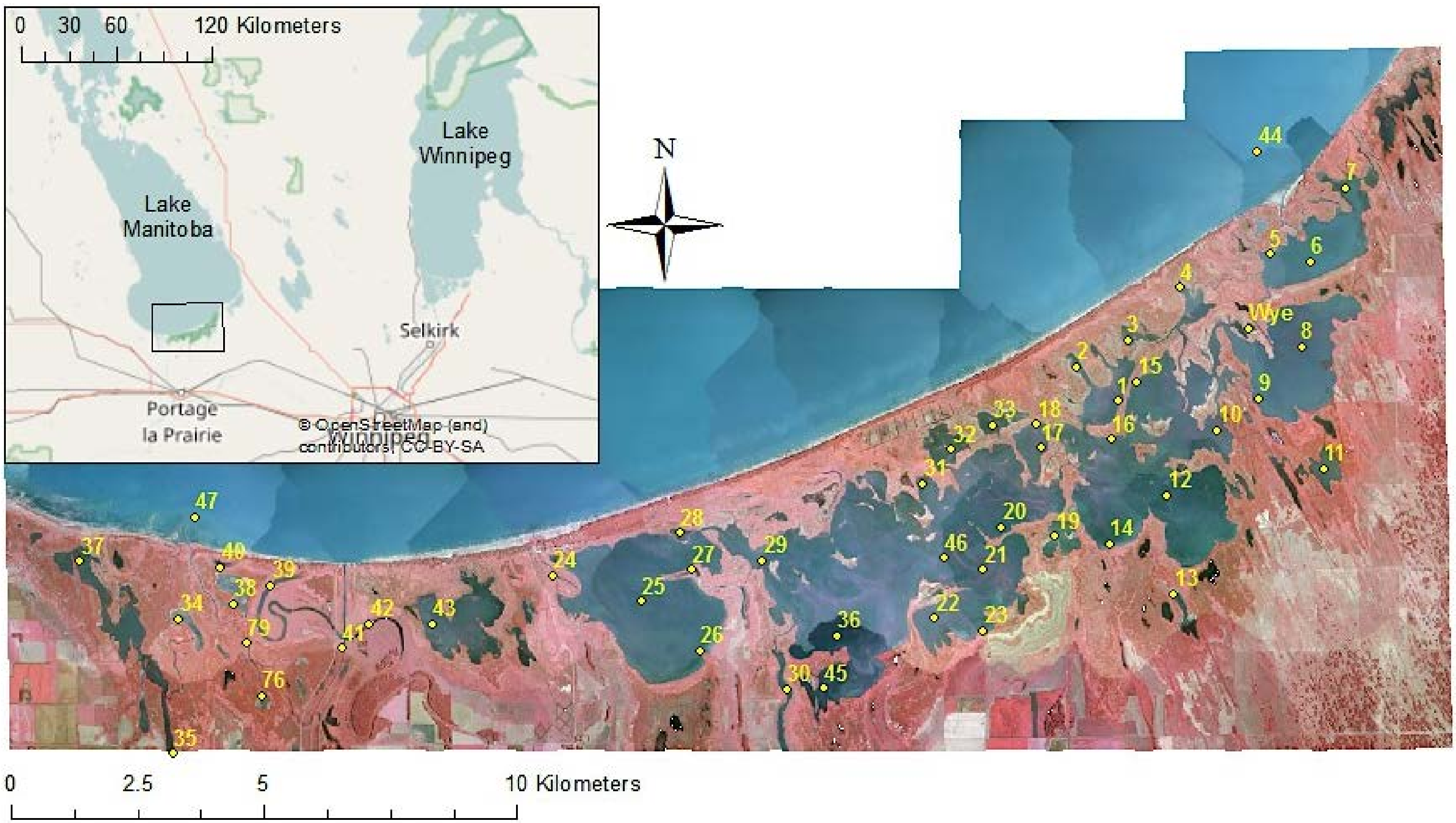
2.2.1 Site Description

Delta Marsh receives water from Lake Manitoba, its southern watershed, streams and precipitation. The marsh is connected to Lake Manitoba through six different channels, each channel is gated with a sheet pile weir structure to exclude Carp from entering the marsh (Section 1.4.1) (Figure 1.1). There are three sections of the marsh; West, Center and East. The eastern section of the marsh has a large area of open water and the western and center sections have higher cover of emergent vegetation (Figure 2.1).

Delta Marsh is characteristically shallow, approximately one meter in depth, ranging from 10 cm to over 1.8 m. As a result, its water column does not stratify thermally. The general nature of the marsh sediments are soft and highly organic (Goldsborough, personal communication). The water column of most water bodies in Delta Marsh contain submersed vegetation including *Myriophyllum* spp. (water milfoil), *Ceratophyllum* spp. (coontail), and *Stuckenia pectinata* (sago pondweed) and peripheral areas are vegetated with dense stands of the emergent macrophyte, *Typha X glauca* (cattail) (Batt 2000; Wasko 2013). Algal community composition in the water column varies based on abiotic and biotic conditions. Blue-green algal blooms, metaphyton mats and epiphyton are some of the most common algal communities in Delta Marsh.

Delta Marsh receives water from its many tributaries to the south (Appendix B), precipitation, and from six channels that connected it to Lake Manitoba: Cram Creek, Deep Creek, Delta Channel and Clandeboye Channel (Figure 1.1). Nutrient loading to

Figure 2.1. Map of Delta Marsh showing 48 of the water quality sites visited during this study (2015-2016). Site numbers correlate to a site name, latitude and longitude in Appendix C.



Delta Marsh was monitored throughout the two years of this research to determine if there was spatial variation in nutrient loading across the watershed. Delta Marsh is encroached on its south side by agricultural land (75-96% cropland cover; Appendix B), with natural lands and recreational and residential property to the north. The southern watershed area spans a gradient of agricultural intensity in the west and natural shrubland and grassland in the east. We speculated that increased cover of agriculture would result in increased nutrient loading to Delta Marsh, resulting in increased phytoplankton production. There were ten tributaries monitored for nutrient loading into Delta Marsh throughout the 2015 and 2016 field season. Results show a strong positive relationship between cropland and nutrient loading of nitrogen and phosphorus. The western and center marsh unit sub-watersheds export the most N and P to the marsh, while the eastern sub-watershed has much lower nutrient export likely due to the fact that this portion of the watershed has a higher proportion of intact grassland/shrubland landscapes and therefore lower nutrient application (Appendix B).

2.2.2 Water Quality Parameters and Sampling Protocol

Each water sampling site was visited monthly, from May to August, in 2015 and every three weeks in 2016. The 45 sites in the marsh were visited over the course of two weeks using a boat and outboard motor in 2015 and early 2016. The sampling period was reduced to one or two days in later 2016 through the use of airboats. The protocol change in 2016 was intended to reduce variability in the dataset arising from stochastic events such as wind activity and storms that occurred during the course of sampling. Each successive sampling period is hereafter described as a sampling round.

Field data included air and surface water temperature (˚C) measured with a hand-held Kestrel meter and mercury thermometer, total water depth (cm), and Secchi depth (cm measured with an 8 inch diameter disk). Photosynthetically active radiation (PAR) (µE $m^{-2}s^{-1}$) was measured from the surface to the bottom, at 10 cm intervals, with a LI-COR LI-193 spherical quantum sensor. These data were used to calculate euphotic zone depth (cm) as the depth at which 1% of the surface light value occurs from a regression of log PAR values on depth. Due to the high turbidity in Delta Marsh (mean 28 NTU), it is important to record both diffuse and incident light, as light is refracted by suspended sediment particles.

A depth-integrated water sample (surface to bottom/50-100 cm) was collected using a clear acrylic tube lowered vertically into the water column. The sample was returned to the field lab where measurements of inorganic suspended solids (mg/L), turbidity (NTU), conductivity (µS/cm) and phytoplankton chlorophyll-a concentration (µg/L) were made. Turbidity was analyzed using a Micro 100 Turbidimeter. Conductivity was measured using an YSI 85 probe. Phytoplankton in a known volume of water was collected onto Whatman GF/C filters. The filters were kept frozen until the time of analysis. Phytoplankton pigments were extracted from the filters by immersing them in 90% methanol solution in the dark for 24 hours. The absorbance of these filter extracts were measured in a spectrophotometer at 665 nm and 750 nm, then the extracts were acidified for one hour with 0.01 N hydrochloric acid. The absorbances at 665 nm and 750 nm were re-measured. Total chlorophyll-a (µg/L) and pheophytin (µg/L) concentrations were used as a proxy for phytoplankton biomass, and were calculated using the method of Marker et al. (1980). To measure suspended solids, a known volume of water was filtered under vacuum through pre-weighed Whatman GF/C filters then the filters was dried at 100˚C for 24 hours and weighed, and the filters were then incinerated at 600˚C for 1 hour and weighed again. Total suspended solids (TSS, mg/L) and inorganic suspended solids (ISS, mg/L) were calculated from the change in weight of the filter (mg) after drying at 100˚C (mg) and incineration at 600˚C (mg), respectively. Water quality variables and their full sampling protocols and analyses are listed in Appendix D.

In addition to the basic field and laboratory measurements described above, depth-integrated water samples from 30 sites collected during three of the five sampling periods in 2015 (round 1, intensive round and round 4) and 2016 (round 1, round 3, round 4 and round 6) were analyzed for nutrient concentrations. Additionally, phytoplankton species composition and biovolume was assessed at 7 of these sites (Appendix C). The rounds were timed to observe temporal changes in nutrient composition and algal communities. The sites were selected to observe spatial variation in nutrient concentrations and algal community composition and biomass across the major open-water bays of Delta Marsh as well as some of the smaller, more sheltered bays. Water chemistry and phytoplankton analyses were performed at ALS Environmental Laboratories (Winnipeg) (Appendix E). The specific parameters measured included ammonia (μg/L), nitrate, and nitrite, dissolved Kjeldahl nitrogen (DKN), dissolved inorganic nitrogen (DIN), total Kjeldahl nitrogen (TKN), total nitrogen (TN), particulate phosphorus (PP), total dissolved phosphorus (TDP), total phosphorus (TP) (mg/L), pH, dissolved organic carbon, alkalinity, and phytoplankton taxonomic composition and biovolume to the level of genus (Appendix C and E).

Based on cyanobacterial class and genera biovolumes, I was able to determine which genera had the potential ability to fix atmospheric nitrogen using literature, personal communication (Goldsborough) and Algaebase ®, I further categorized them into nitrogen-fixers and non-nitrogen-fixers. The relationship between cyanobacteria and nutrient availability was further analyzed by comparing the percent of nitrogen-fixing cyanobacteria and the DIN:TDP ratio.

A wind exposure variable was created to determine the effect of wind on sediment and nutrient resuspension in the water column and its relationship with chlorophyll-a concentration. Based on literature (So et al, 2013) and datasonde turbidity data from Delta Marsh (Page, unpublished data) and the shallow nature of the marsh, wind speeds >25 kph were determined as a threshold above which resuspension of sediments is likely to occur. Wind speed (kph) and direction (degrees) were recorded at a weather station operated by Ducks Unlimited Canada at Eaton Lodge (Figure 1.1). Wind direction, speed and gusts at a height of 3 meters were recorded every 15 minutes with a Wind Smart Sensor Set S-WSET-A. Data recorded from April to September in 2015 and 2016 were separated into wind speeds >25 kph. Wind speeds >25 kph were separated into eight 45˚ directional segments from 0˚ to 360˚. A total count of wind speeds >25 kph were recorded for each 45˚ segment to determine which direction had the highest proportion of wind >25 kph. We determined that 83% of wind above 25kph came from the north and 17% from the south. The mid-point of the four dominant directional segments were used to calculate wind exposure: 22.5˚, 157.5˚, 292.5˚ and 337.5˚. Fetch distances (m) for each water quality site was measured from the middle of each site to the edge of the emergent vegetation in Google Earth Pro © 7.1.7.2602 at each of these degrees. A model was created to calculate overall wind exposure that includes fetch distance for the four major degrees and the weight based on proportion of wind in the prevailing directions (Equation 1) (Appendix F).

Equation 1. Wind Exposure = 0.1142*(22.5˚ fetch distance) + 0.1048*(157.5˚ fetch distance) + 0.2058*(292.5 fetch distance) + 0.5753*(337.5˚ fetch distance)

2.2.3 Variable Selection

Variables that were highly inter-correlated were removed from the statistical analysis. Air and water temperature were positively correlated (r^2 of 0.82 and 0.57 in 2015 and 2016, respectively) so only water temperature was used. Secchi depth was removed because it cannot be measured at clear and shallow sites. Turbidity was correlated positively with ISS in 2016 (r^2= 0.74), and negatively with euphotic zone depth in 2015 and 2016 (r^2= -0.57 and -0.43), respectively, so it was removed.

To determine which N and P compounds to use in the models, I performed a correlation matrix with the nutrient data and phytoplankton chlorophyll-a concentration. The results indicated a positive correlation with TKN (r^2=0.57), TN (r^2=0.58), PP (r^2=0.42) and TP (r^2=0.44). There was a high positive correlation between TKN and TN (r^2=0.99), and PP and TP (r^2=0.86). TN is the sum of TKN, nitrate and nitrite. TP is the sum of PP and TDP, TDP was not correlated with chlorophyll-a (r^2=0.16) indicating the high correlation between TP and chlorophyll- a was likely caused by the PP. Due to these results, I determined that TN and PP would be most appropriate for chlorophyll-a predictor models. A stepwise regression model was performed in R using the MASS and stats software packages to determine which nutrient variables best predicted phytoplankton chlorophyll-a. Final models were determined with the stepAIC values. Chlorophyll-a was best predicted by TN and PP ($F_{2, 192}$= 48.6, p <$2.2e^{-16}$, r^2=0.3292), although for the models, TP was used because it included both compounds of P.

2.2.4 Statistical Analysis

The data were analyzed using multivariate statistics in R 3.2.3. Principal Component Analysis (PCA) and multiple stepwise regression models were used to examine correlations between biotic and abiotic variables to determine which factor(s) had the most influence on phytoplankton chlorophyll-a concentration, including TSS, ISS, conductivity, euphotic depth, air and water temperature, turbidity, Secchi depth, total water depth, TN and TP, and landscape variables including latitude, longitude and wind exposure. The water quality parameters are variable throughout the field season while the landscape variables remain constant at each site. Both water quality variables and landscape variables are included in the exploratory and confirmatory statistical analyses to determine their relationships with each other and their influence on phytoplankton chlorophyll-a concentration.

Using the Vegan software package (community ecology package) in R 3.2.3, the correlation matrix PCA provides a plane of best fit in multidimensional space, this analysis displayed which variables were correlated and uncorrelated and what sites were most similar in water quality. The PCA was used as an exploratory approach to determine the relationships and correlations between the variables and was presented as a biplot. A multiple regression analysis was performed in R using the MASS (Modern Applied Statistics with S) and Stats software packages as the confirmatory approach. This analysis was performed to create models that predict chlorophyll-a concentrations. Models were created using the stepwise approach and final models chosen based on the stepAIC values. Each water quality sampling round was analyzed with a PCA and a multiple stepwise regression. Some of the sampling rounds included additional nutrient concentration analyses at 30 of the 45 sites in Delta Marsh. Sampling rounds that included nutrient analyses have two PCAs and two stepwise regression models, one conducted across all rounds (excluding nutrient analyses), and the second for the subset of sites including nutrient analyses. Sampling round was included in a multiple stepwise regression analysis to determine the effect of temporal variation on phytoplankton chlorophyll-a concentrations. This parameter was indicated by “Round” in the models.

Chlorophyll-a concentrations that were below the detection limit of (0.4 µg/L) were given a value of 0.2 µg/L for purposes of statistical analysis. Single missing values for a variable were replaced by the mean value or, if there were multiple missing values, the site was removed from the analysis. If the distribution of the variable was non-normal, a log(x) transformation was performed or log(x+1) if there was a value of zero. An ANOVA and Tukey Kramer HSD test were performed in R to determine if phytoplankton chlorophyll-a concentrations varied across the three sections of the marsh. Shapiro-Wilk and Bartlett tests were used prior to ANOVA to test the normal distribution and homogeneity of variance of the variables.

To interpret the strength of the correlation coefficient (r) produced in some of the statistical analyses, I used scales determined by Snedecor and Cochran (1980) and Rumsey (Accessed May, 2017). Snedecor and Cochran (1980) discuss the percent variance of Y explained by X. A correlation coefficient less than 0.5 explain little variation and are considered to be weak. A correlation coefficient of 0.7 explains approximately 50% of variance, and of 0.9 explains approximately 80% of variance (Snedecor and Cochran 1980). For my research and data interpretation, a correlation coefficient from 0.75 to 1.0 are strong, from 0.5 to 0.75 are moderate, from 0.3 to 0.5 are weak, and less than 0.3 have no relationship.

2.3 Results

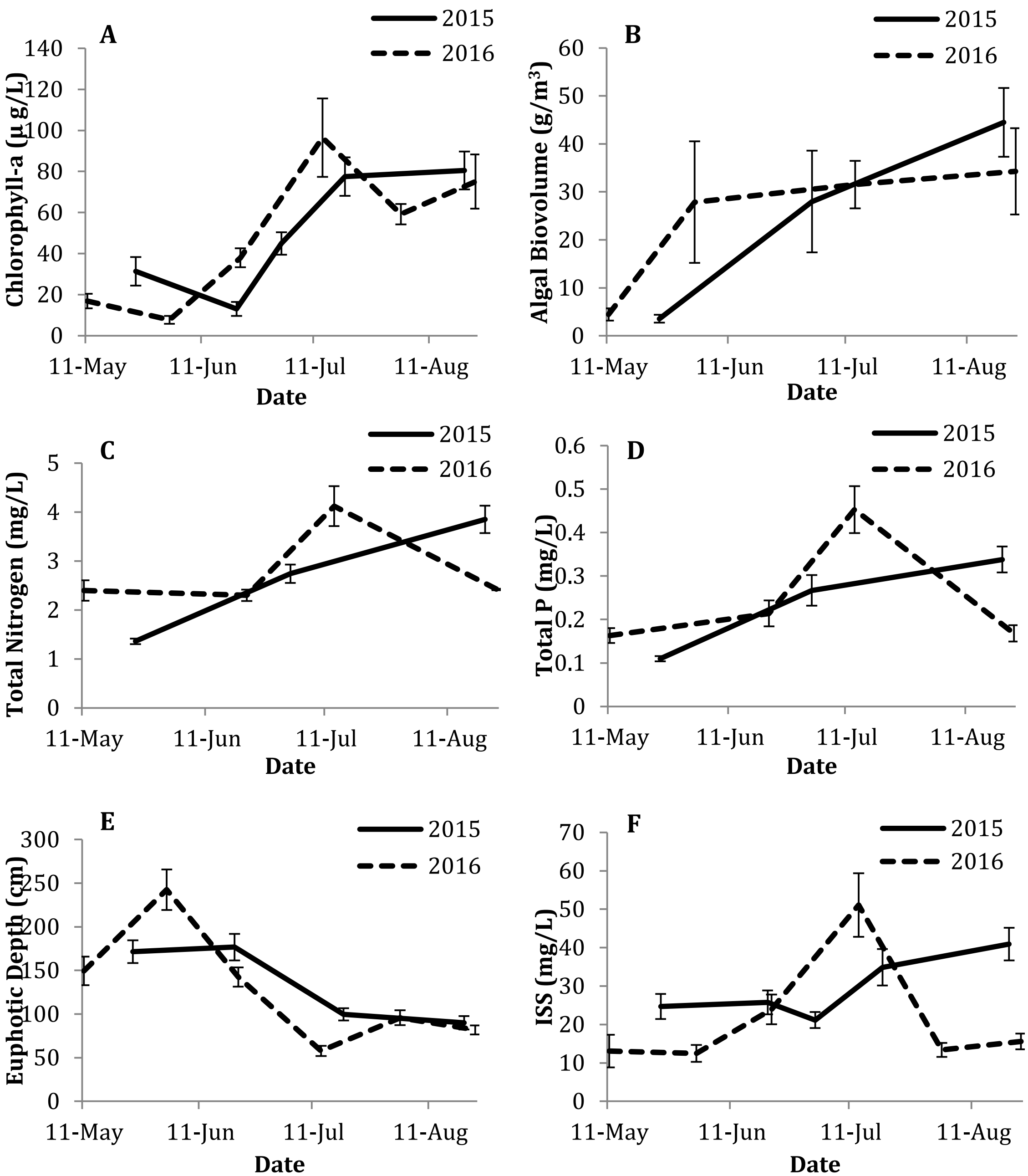
The mean, range and trends for 2015 and 2016 were similar for all variables throughout the field season, excluding TN and TP concentrations (Table 2.1, Figure 2.2). In general, each variable increased over time in 2015 and 2016, excluding euphotic depth. It appears that all variables follow a similar trend to chlorophyll-a concentration. Trends for temperature were not included in Figure 2.2, as we know that temperature increases over the field season (May to August). There was an observed peak in all variables, excluding algal biovolume on July 13, 2016, possibly caused by a windstorm on this date (minimum wind speed = 20.5 km/hr, maximum wind speed= 31.5 km/hr). TN and TP increased by approximately 2 mg/L and 0.2 mg/L, chlorophyll-a concentration increased by 60 µg/L, ISS increased by 30 mg/L and euphotic zone depth shifted by 85 cm (Figure 2.2). Phytoplankton chlorophyll-a concentration (Figure 2.2A) and euphotic depth (Figure 2.2E) experienced similar trends in 2015 and 2016. Chlorophyll-a concentration increased throughout the field season while euphotic depth decreased. There was no difference between mean chlorophyll-a concentrations in 2015 and 2016, with concentrations of means of 48 µg/L and 49 µg/L, respectively. Mean algal biovolumes were similar between 2015 and 2016, at 25.3 g/m^3 and 24.5g/m^3, respectively. Algal biovolume increased throughout the field season in 2015 and 2016 (Figure 2.2B).

Inorganic suspended solids (ISS) concentration increased throughout the field season in 2015 and remained constant in 2016 when excluding the storm in mid-July (Figure 2.2F). Mean ISS in 2015 was 31.6 mg/L and decreased in 2016 to 21.3 mg/L. Although ISS changed between years, euphotic depth did not differ significantly between years, with means of 134 and 128 cm in 2015 and 2016, respectively. Trends observed

Table 2.1. The mean and ranges for water quality variables recorded from May to August across 48 sites in Delta Marsh, Manitoba in 2015 and 2016. Note: Algal biovolume was only monitored in seven sites, and total N and total P were only monitored in thirty sites (Appendix C).

Water Quality Parameter	**2015**	**2016**
Phytoplankton Chlorophyll-a Concentration (µg/L)	48 (0.2-268)	49 (0.2-643)
Algal Biovolume (mg/L)	25 (0.87 – 90)	25 (0.76 – 108)
Water Temperature (˚C)	21 (10 – 29)	18 (12- 26)
ISS Concentration (mg/L)	29.5 (0-199)	21 (0-278)
Turbidity (NTU)	29 (2-146)	27 (0.9- 258)
Conductivity (µS/cm)	924 (456- 1128)	892 (373 – 1418)
Euphotic Zone Depth (cm)	135 (35-496)	130 (14-720)
Total N Concentration (mg/L)	2.68 (0.90-6.90)	3.37 (1.35 – 9.91)
Total P Concentration (mg/L)	0.28 (0.06-0.99)	0.38 (0.06- 5.28)

Figure 2.2. Trends of six aquatic variables recorded from May to August across 48 sites in Delta Marsh, Manitoba in 2015 and 2016. Variables included phytoplankton chlorophyll-a (A), algal biovolume (B), total N (C), total P (D), euphotic depth (E) and ISS (F). Points represent the mean (±SE) of each variable during the sampling period. (n= 48 for chlorophyll-a, euphotic depth and ISS, n=30 for total N and total P, n= 7 for algal biovolume).



for TN and TP concentrations were different between 2015 and 2016, where in 2015 concentrations increased over time and in 2016 concentrations remained relatively similar excluding the storm (Figure 2.2 C and 2.2D). Mean TN concentrations were significantly higher in 2016 than 2015 ($p<0.01$), with means increasing from 2.48 mg TN/L to 3.37 mg TN/L, respectively. Mean TP concentrations were not significantly different between 2015 and 2016 ($p=0.08$) with mean concentrations of 0.195 mg TP/L and 0.381 mg TP/L, respectively. While there is no significant difference there does appear to be higher mean TP concentration in 2016 relative to 2015. The trends indicate temporal variation and possible parameter relationships.

Round 1 – 2015 (May 14 to June 2) was reduced from nine to two principal component axes when excluding nutrient analyses. The first and second axes explained 38% and 20% of linear variation, respectively (Figure 2.3). Conductivity and water temperature were correlated positively along the first axis and correlated negatively with latitude and longitude. Phytoplankton chlorophyll-a and ISS were correlated positively between the first and second axes and with wind exposure along the second axes. Chlorophyll-a, ISS and wind exposure were correlated negatively with the euphotic zones. A multiple stepwise regression was performed to determine which variables best predict phytoplankton chlorophyll-a concentration using AIC values. The stepwise regression for round 1 – 2015 selected the model, **Chlorophyll-a = -210 + 0.93*(ISS) – 2.14*(Longitude)** ($F_{2,44}=12.4$, $r^2=0.36$, $p<0.01$).

When nutrient data were added to the analyses the first and second axes explained 33% and 19% of linear variation, respectively (Figure 2.4). In this round, chlorophyll-a was observed in warm shallow sites with high inorganic suspended solids. The euphotic

Figure 2.3. Principal Component Analysis (PCA) of water quality sampling Round 1-2015 (May 14 – June 2) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 38% and 20% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

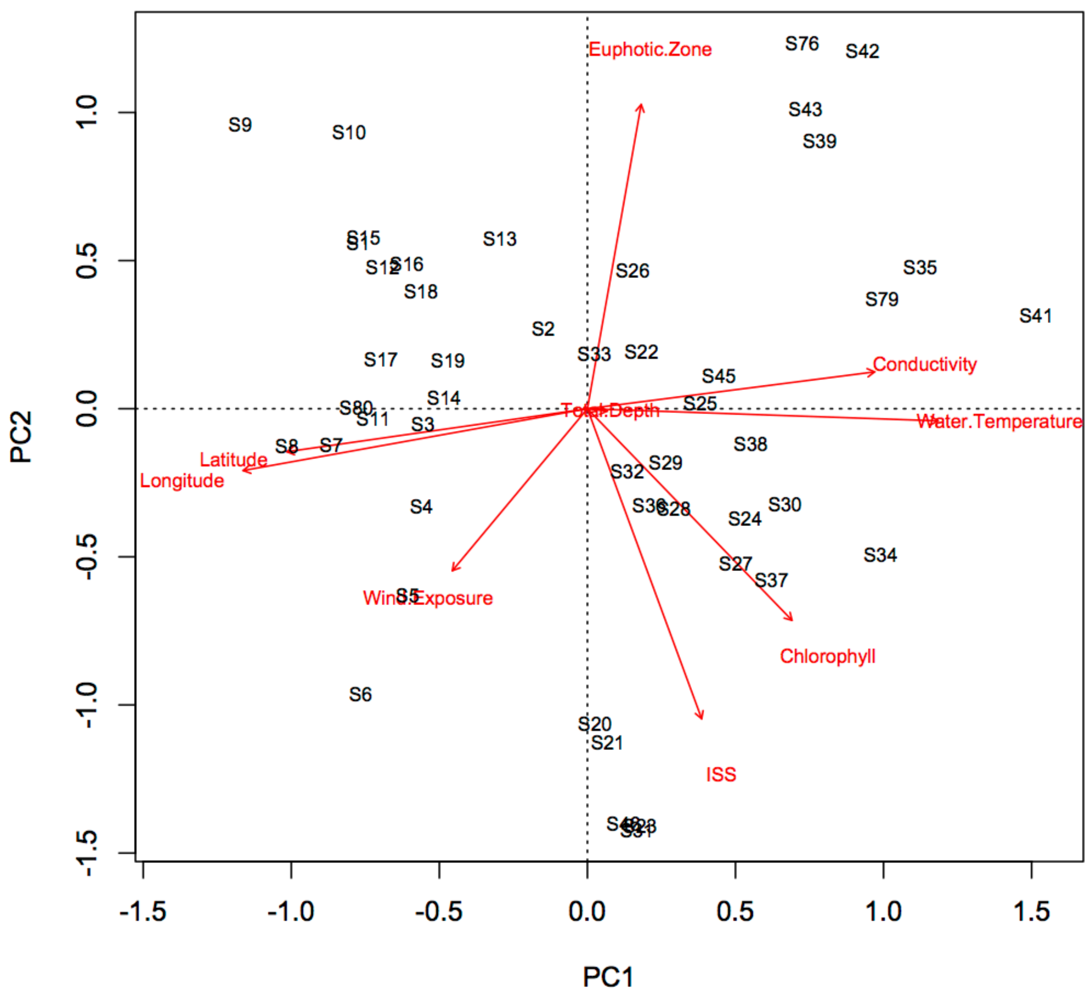
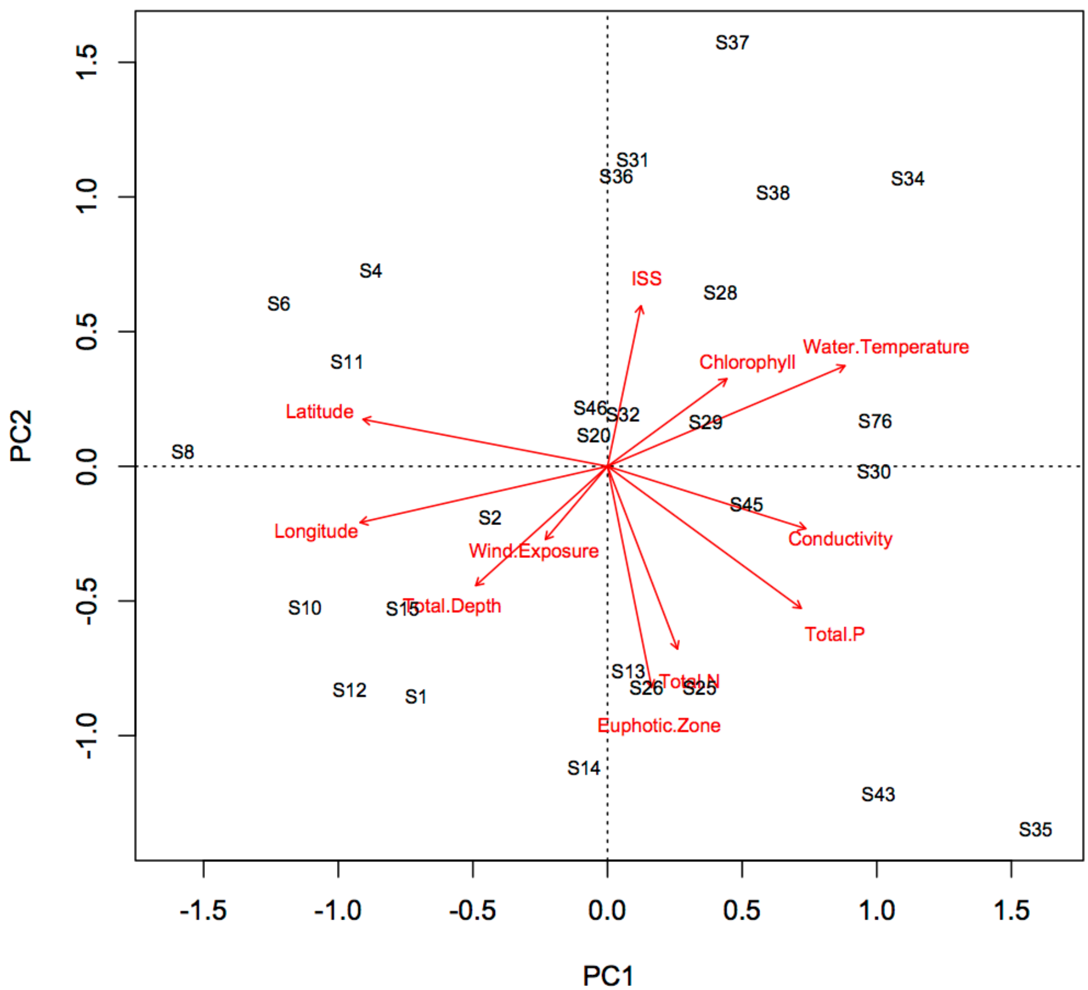


Figure 2.4. Principal Component Analysis (PCA) of water quality sampling Round 1-2015 (May 14- June 2) across 30 sites in Delta Marsh, Manitoba, including nutrient concentration data. Points on each PCA represent individual sites. The two axes account for 33% and 19% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



zone, TN and TP were correlated positively between the first and second axes and negatively with chlorophyll-a and ISS. Chlorophyll-a and water temperature were correlated negatively with wind exposure and depth. A multiple stepwise regression to predict phytoplankton chlorophyll- concentration selected the model, **Chlorophyll-a = -1.07e^{+3} – 2.58e^{-3}*(Euphotic Depth) + 6.63e^{-1}*(ISS) + 1.27e^{-1}(Latitude) – 4.37*(Longitude) + 3.24*(TN)** ($F_{5,22}$=6.78, r^2=0.61, p<0.01). When nutrients were added into the model, the correlation coefficient increased, indicating nutrients may be highly important when predicting phytoplankton chlorophyll-a concentration. Phytoplankton chlorophyll-a concentrations were best predicted by a shallow euphotic depth, high ISS and TN concentrations and spatial location.

The first and second axes for Round 2 - 2015 (June 15 to 25) explained 36% and 25% of linear variation, respectively (Figure 2.5). Similar to round 1, high chlorophyll-a concentrations were observed in shallow sites in the western section Delta Marsh. Chlorophyll-a and conductivity were correlated positively between the first and second axes, and together negatively with depth, longitude and wind exposure. Euphotic zone and water temperature were correlated negatively with ISS along the second principal component axis. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 414 + 0.91*(Euphotic Depth) – 13.7*(Latitude) – 2.78*(Longitude)** ($F_{3,43}$=8.18, r^2=0.36, p<0.01).

The first and second principal component axes for Round 3- 2015 (intensive sampling round -July 2) explained 46% and 24% of linear variation, respectively (Figure 2.6). The reduction of temporal variation caused by duration of sampling period may have increased the linear variation explained by the ordination. During the intensive

Figure 2.5 Principal Component Analysis (PCA) of water quality sampling Round 2-2015 (June 15-25) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 36% and 25% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

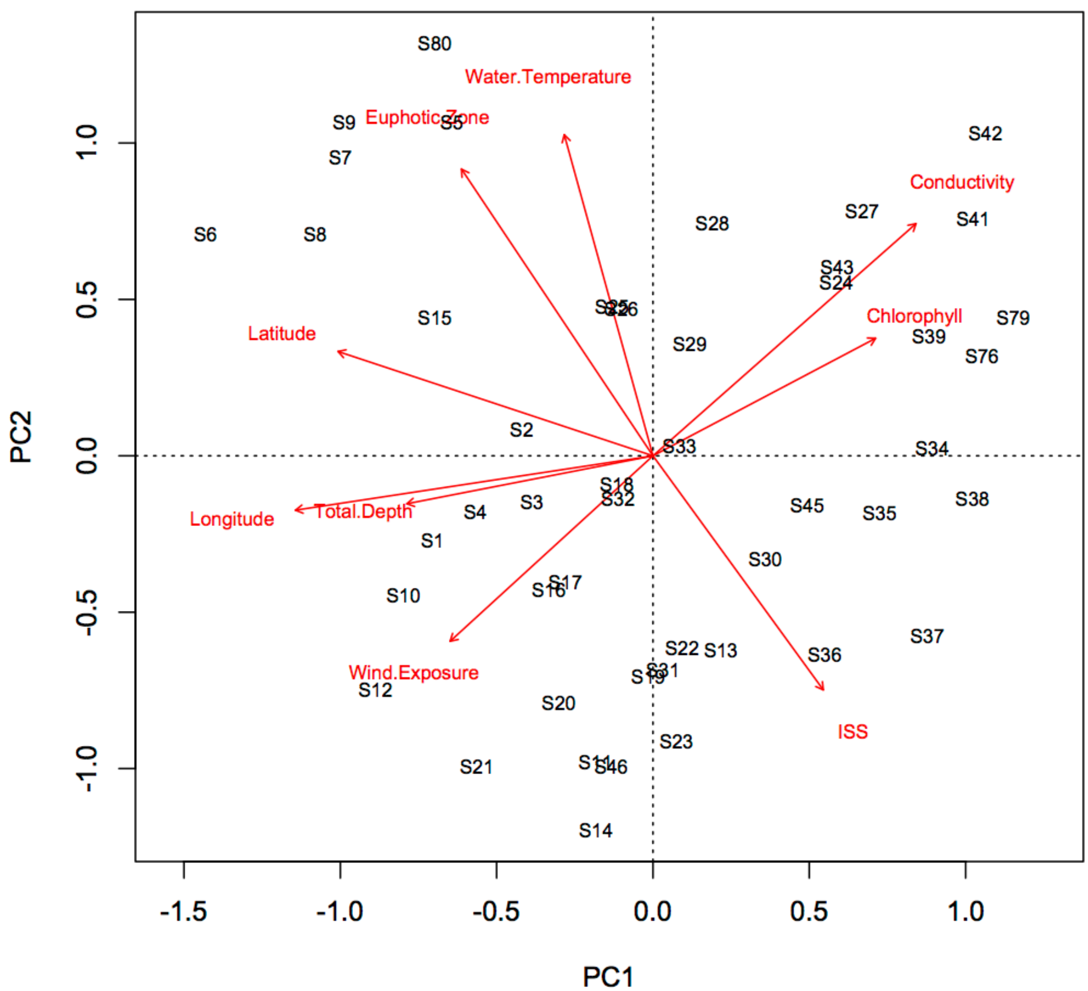
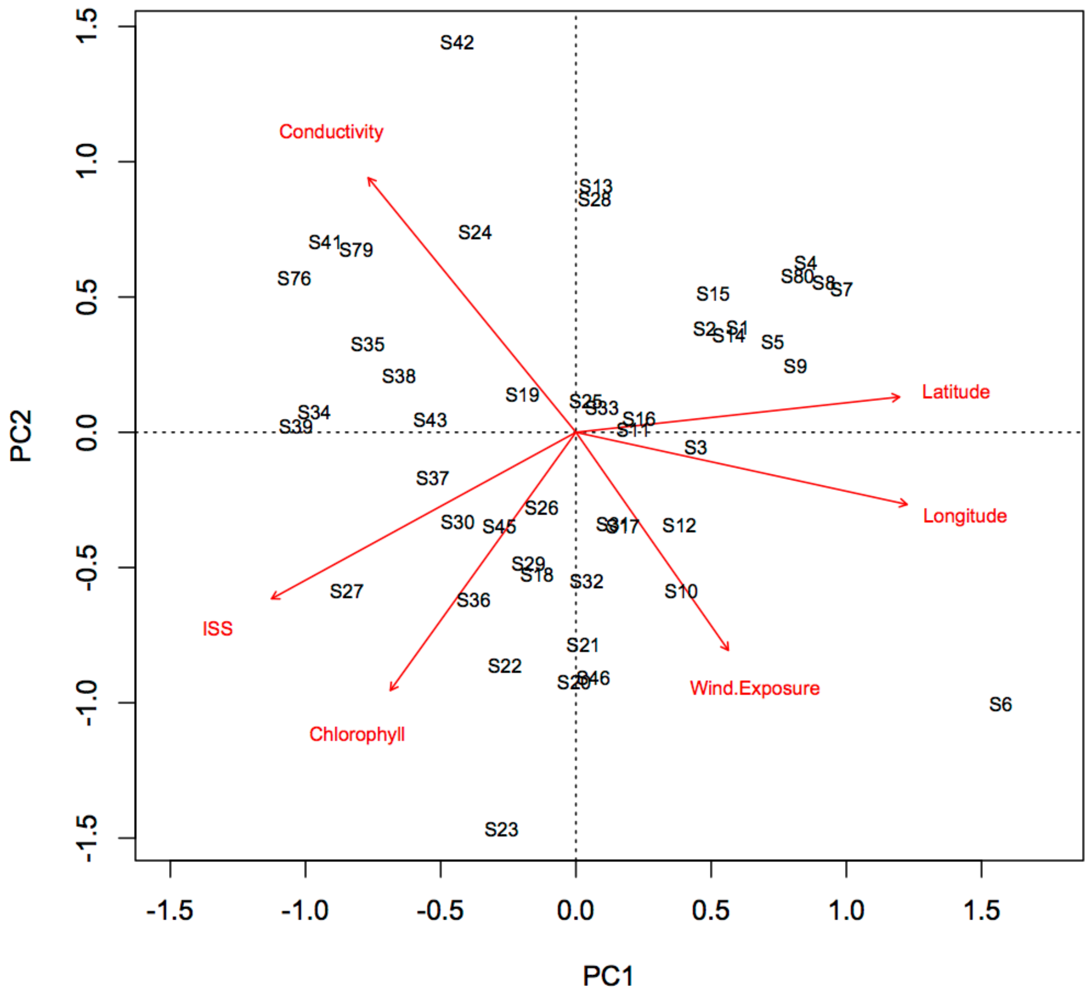


Figure 2.6. Principal Component Analysis (PCA) of water quality sampling Round 3- 2015 (the intensive sampling round -July 2) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 46% and 24% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



sampling round, chlorophyll-a was observed in sites with high concentrations of ISS and TN. Chlorophyll-a and ISS were correlated positively, while conductivity and wind exposure were correlated negatively. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = $3.25e^{2}$ – $9.55e^{-4}$*(Conductivity) + $8.08e^{-1}$*(ISS) – 4.08*(Latitude) + 1.19*(Longitude) – $1.70e^{-1}$*(Wind Exposure)** ($F_{5,41}$=7.50, r^2=0.48, p<0.01). The model indicated that phytoplankton chlorophyll-a concentrations were best predicted by high ISS concentration, low conductivity, spatial location and wind exposure. When nutrient data were included in the PCA, the first and second axes explained 43% and 23% of linear variation, respectively (Figure 2.7). Chlorophyll-a and TN were correlated positively between the first and second axes and positively with ISS along the first axis. Wind exposure was correlated negatively with TP and conductivity on the first and second axes. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 0.63 + 0.313*(TN)** ($F_{1,26}$=57.3, r^2=0.69, p<0.01). Again, when nutrients are included in the stepwise regression analysis, the coefficient of determination improves, possibly indicating that nutrients best predict chlorophyll-a concentration within the marsh.

The first and second axes for Round 4 – 2015 (July 14-23) explained 31% and 23% of linear variation, respectively (Figure 2.8), where high chlorophyll-a concentrations were observed in deep sites with increased wind exposure and ISS, extending into the eastern portions of the marsh. Chlorophyll-a and ISS were correlated positively between the first and second axes and correlated positively with wind exposure and total depth along the first axis. These four variables were correlated negatively with

Figure 2.7. Principal Component Analysis (PCA) of the water quality sampling Round 3- 2015 (the intensive sampling round -July 2) across 30 sites in Delta Marsh, Manitoba, including nutrient concentration data. Points on each PCA represent individual sites. The two axes account for 43% and 23% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

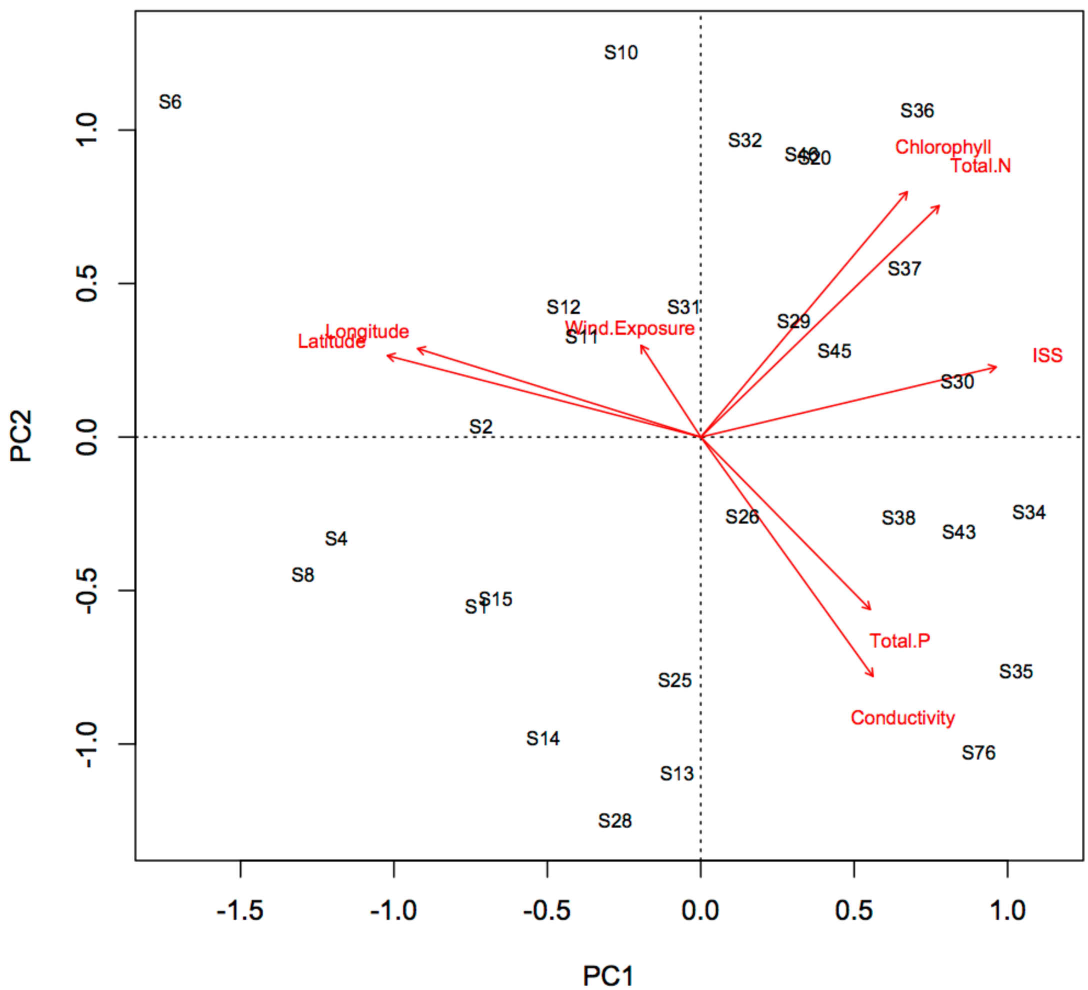
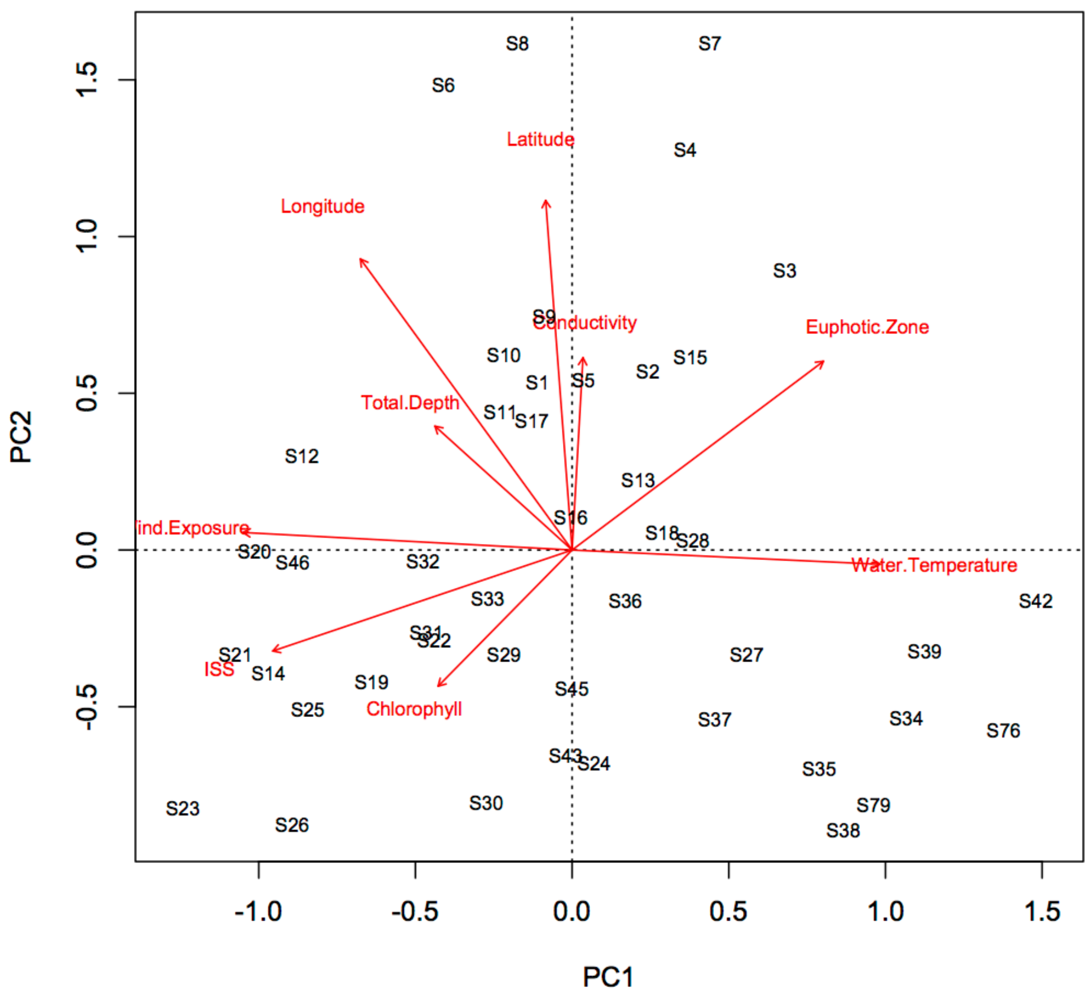


Figure 2.8. Principal Component Analysis (PCA) of water quality sampling Round 4- 2015 (July 14-23, 2015) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account 31% and 23% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



water temperature along the first axis. Conductivity was correlated positively with latitude and longitude along the second axis. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = -311 + 0.19*(Water Temperature) – 0.95*(Euphotic Depth) – 3.18*(Conductivity) + 6.34*(Latitude) + 0.35*(Wind Exposure)** ($F_{5,41}$=3.98, r^2=0.33, p<0.01). The stepwise regression model for round 4 indicated that phytoplankton chlorophyll-a concentration was best predicted by water temperature, wind exposure, a shallow euphotic depth, low conductivity and spatial location.

Round 5 – 2015 (August 11 to 19) was the final sampling round of the 2015 field season. The first and second axes explained 33% and 19% of linear variation, respectively (Figure 2.9). Chlorophyll-a and ISS were correlated on the second axis with increased wind exposure, total depth and water temperature. Euphotic zone, latitude and conductivity were negatively associated with ISS and chlorophyll-a between the first and second axes. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 0.78 – 0.006*(Depth) + 0.60*(Wind Exposure)** ($F_{5,41}$=3.98, r^2=0.11, p=0.07). The model selected to predict chlorophyll-a concentrations in round 5 was weak. This was also observed in the PCA, chlorophyll-a did not appear to have a strong association with other variables (Figure 2.9). When the nutrient data were included, the first and second axes explained 34% and 23% of linear variation (Figure 2.10), where high chlorophyll-a concentrations were found in sites with higher nutrient and ISS concentrations. Chlorophyll-a, ISS, TP and TN were correlated positively along the first axis and together were correlated negatively with euphotic zone, conductivity and latitude. TN was correlated with wind exposure, water temperature and

Figure 2.9. Principal Component Analysis (PCA) of water quality sampling Round 5- 2015 (August 11-19) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 33% and 19% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

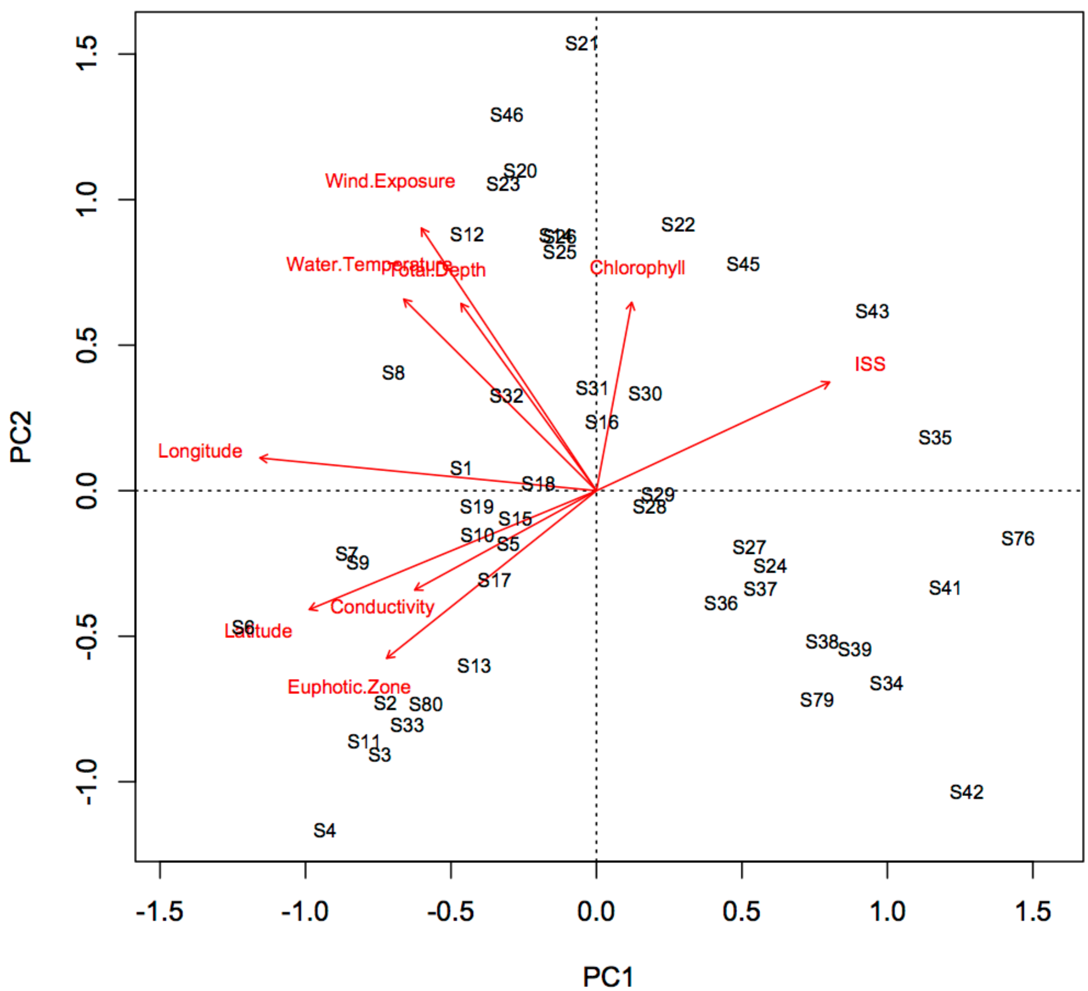
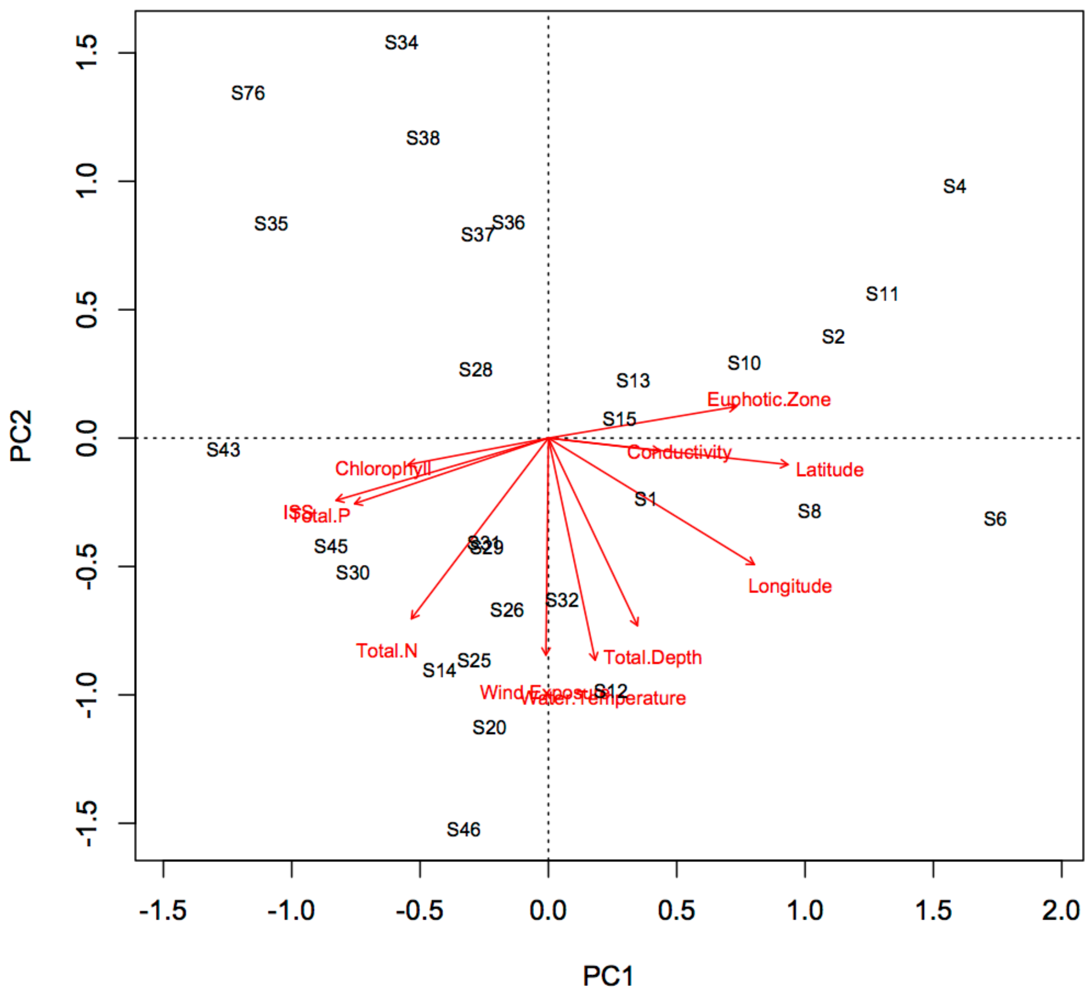


Figure 2.10. Principal Component Analysis (PCA) of water quality sampling Round 5-2015 (August 11-19) across 30 sites in Delta Marsh, Manitoba, including nutrient concentration data. Points on each PCA represent individual sites. The two axes account for 34% and 23% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



total depth along the second axis. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 10.6 – 7.52*(Water Temperature) + 2.61*(TN)** ($F_{2,25}$=2.94, r^2=0.19, p=0.07). The addition of nutrients to the stepwise regression analysis did not improve the model unlike the other rounds.

Results of an ordination that used all 2015 sampling rounds without nutrient concentration data showed that high chlorophyll-a concentrations tended to occur in deep and exposed sites with high ISS concentrations. The first and second axes explained 41% and 22% of the linear variation, respectively (Figure 2.11). Chlorophyll-a concentration was correlated positively with ISS between the first and second axes and with water temperature on the first axis. The three variables were correlated negatively with euphotic zone and latitude along the first axis. Total depth, wind exposure and chlorophyll-a were correlated negatively with conductivity along the second axis. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 4.18 – 0.002*(Conductivity) – 0.62*(Euphotic Depth) + 0.44*(ISS)** ($F_{3,43}$= 11.21, r^2=0.44, p<0.01). The model displayed similar relationships and correlations between the variables as observed in the PCA (Figure 2.11). When including the nutrient concentration data, high chlorophyll-a concentrations were found in exposed, warm sites with high nutrient and inorganic suspended sediment concentrations. The first and second axes explained 47% and 20% of linear variation, respectively (Figure 2.12). Chlorophyll-a, ISS and TN were associated along the first and second axes. Water temperature and TP were correlated positively on the first axis with chlorophyll-a, ISS and TN. The five

Figure 2.11. Mean Principal Component Analysis (PCA) of the 2015 (May-August) water quality sampling across 48 sites in Delta Marsh, Manitoba. The variables in this PCA represent a mean value for each site analyzed from May to August 2015. The two axes account 41% and 22% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

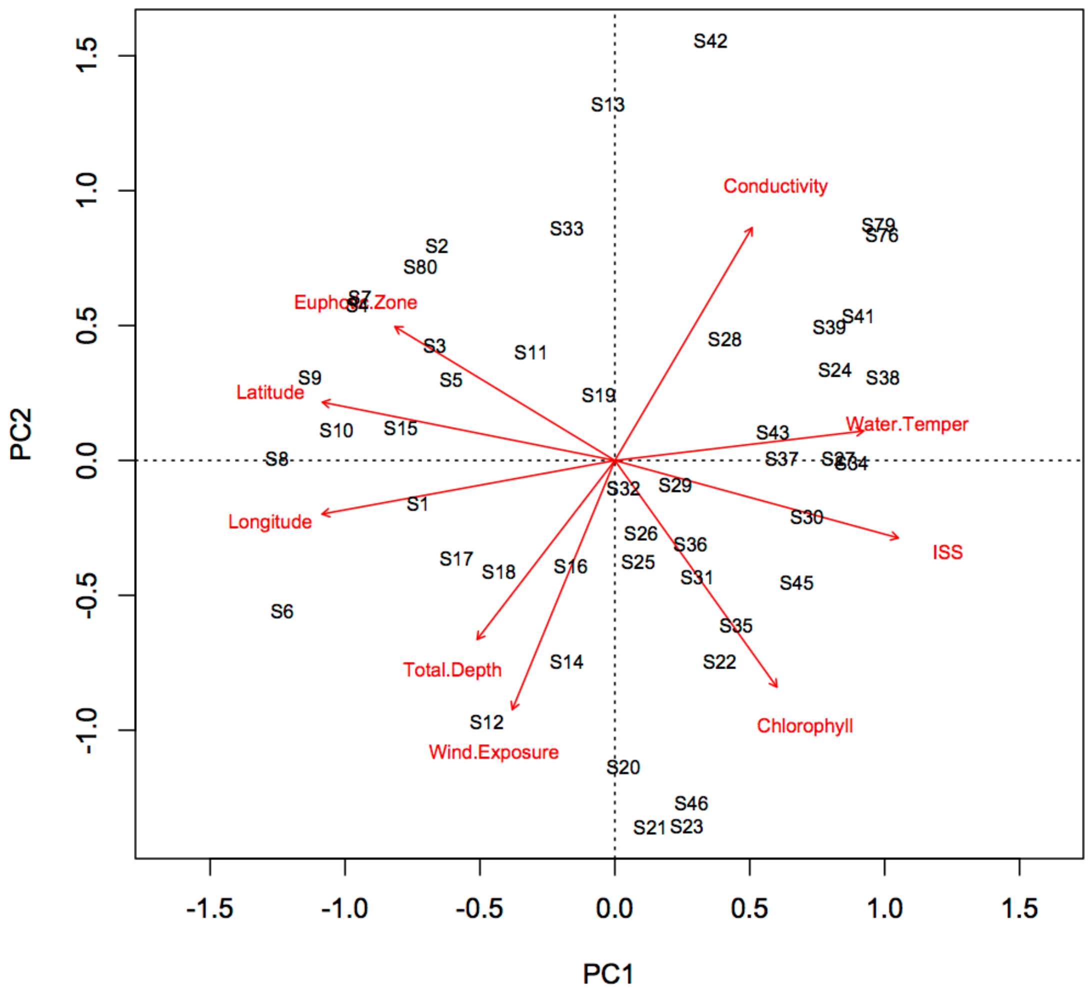
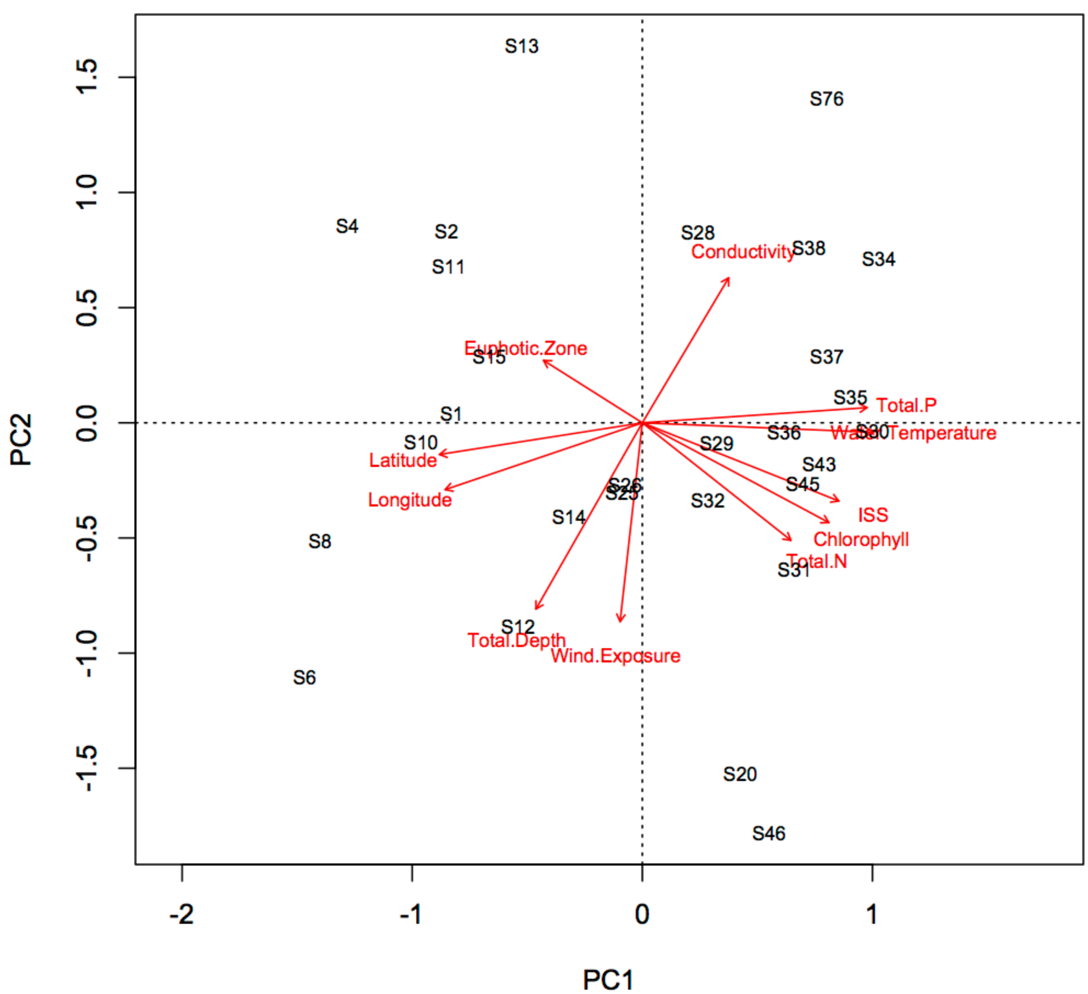


Figure 2.12. Mean Principal Component Analysis (PCA) of the 2015 water quality sampling across 30 sites in Delta Marsh, Manitoba, including nutrient concentration data. The variables in this PCA represent a mean value for each site analyzed in round 1 (May 14- June 2) and round 5 (August 11-19). The two axes account 47% and 20% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



variables were correlated negatively with latitude, longitude and euphotic zone along the first axis. Total depth and wind exposure were correlated negatively with conductivity between the first and second axes. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = -78.9 – 0.88*(Longitude) + 0.25*(TN) – 2.24*(Conductivity) + 0.28*(ISS)** ($F_{4,23}$=17.5, r^2=0.75, $p<0.01$). The model for the mean 2015 water quality data indicated that chlorophyll-a concentrations are found in sites with high TN and ISS concentration and low conductivity, similar to what was displayed in the PCA (Figure 2.12).

During Round 1 - 2016 (May 11), chlorophyll-a concentrations were not associated with any particular axis or any variable (Figure 2.13). The first and second axes explained 38% and 16% of linear variation respectively. Total depth and wind exposure were correlated positively along the first and second axes; the two were correlated negatively with chlorophyll-a and conductivity. Water temperature, ISS, longitude and latitude were correlated positively on the first axis and they were correlated negatively with euphotic zone depth. As depth and wind exposure increased, the water column had increased ISS and water temperature. There was no model created in the multiple stepwise regression analysis, this is likely due to chlorophyll-a not being associated with any variables as observed in the PCA (Figure 2.13). When the nutrient data were included in the model, high chlorophyll-a concentrations were observed at sites with higher TP and TN concentrations. The first and second axes explained 35% and 20% of linear variation respectively (Figure 2.14). The reduction of the duration of each 2016 sampling period did not display similar reduction in temporal variation as Round 3-2015 (the intensive sampling round - July 2). A multiple stepwise regression to predict

Figure 2.13. Principal Component Analysis (PCA) of water quality sampling Round 1-2016 (May 11) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 38% and 16% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

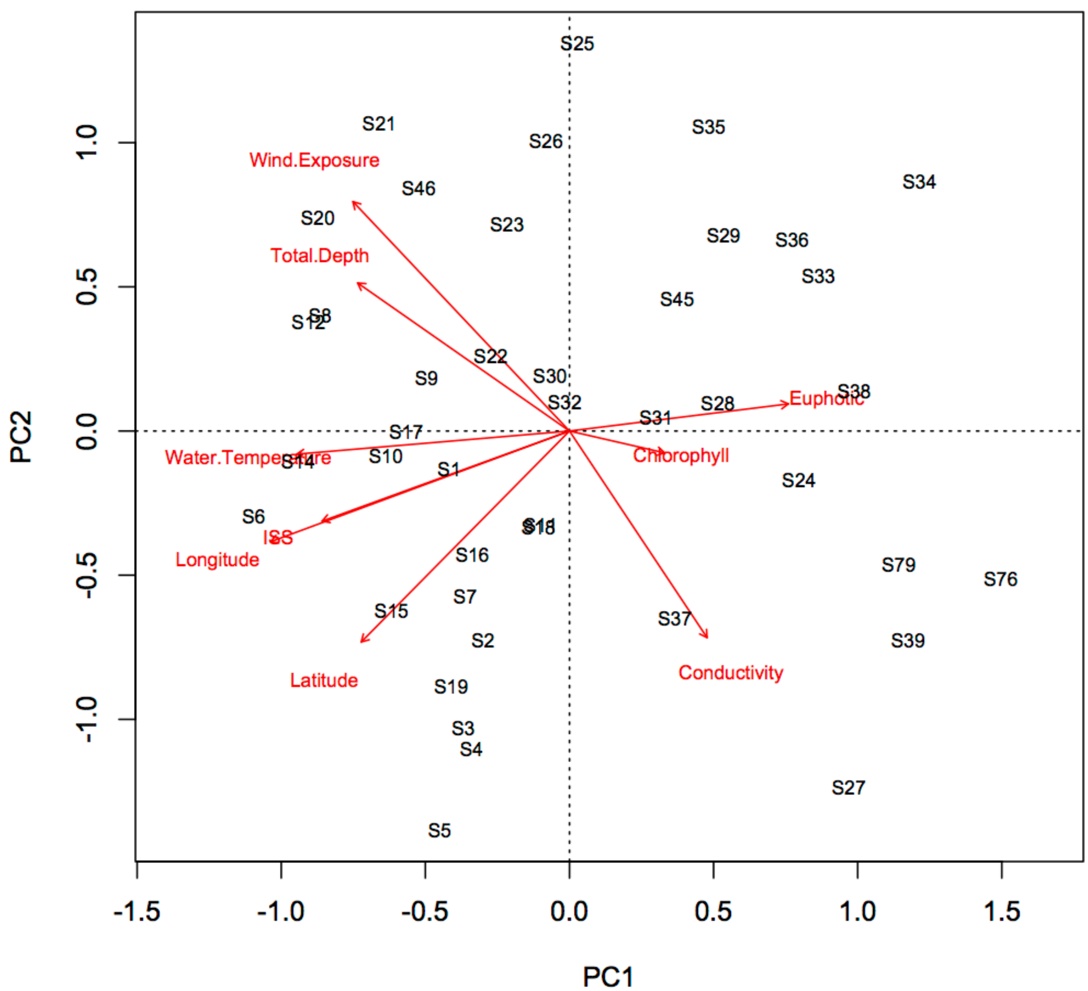
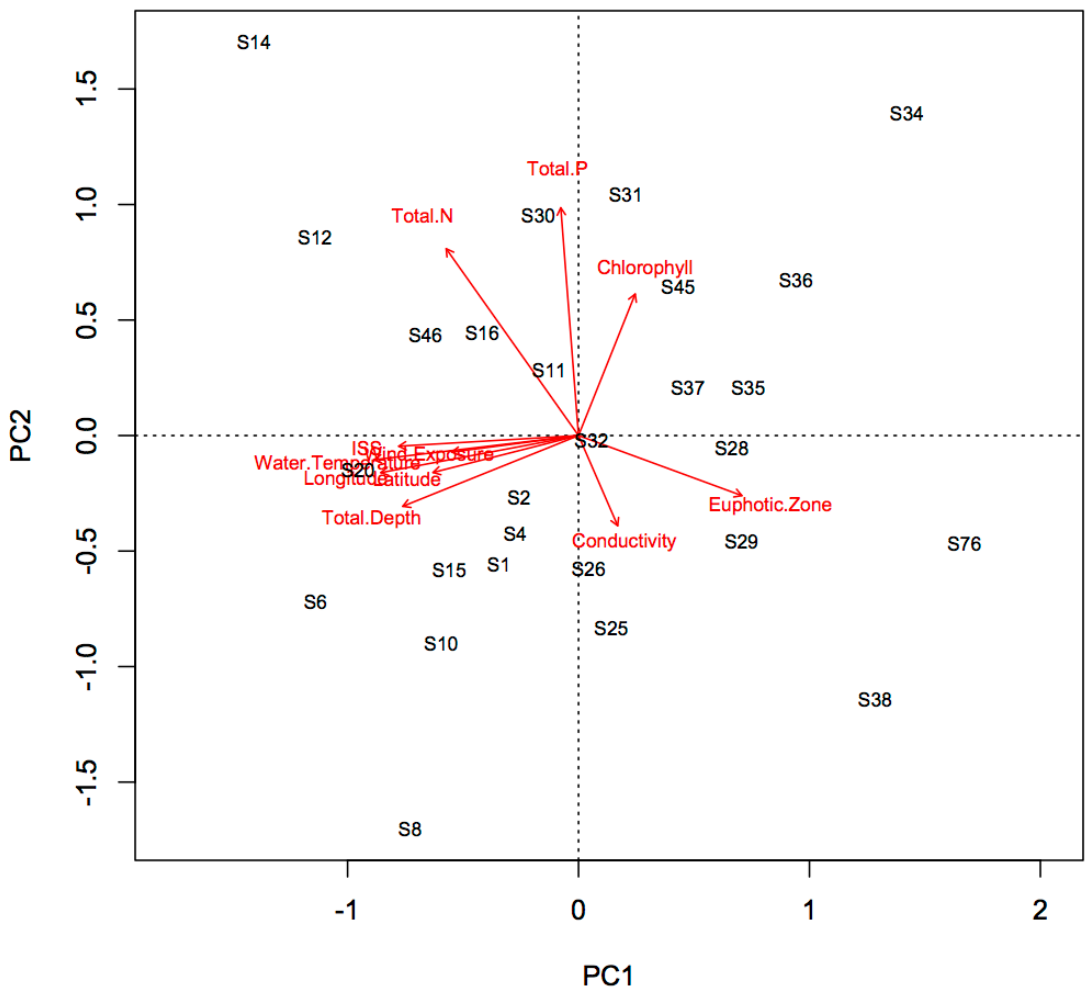


Figure 2.14. Principal Component Analysis (PCA) of water quality sampling Round 1-2016 (May 11) across 30 sites in Delta Marsh, Manitoba, including nutrient concentration data. Points on each PCA represent individual sites. The two axes account for 35% and 29% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 215 – 0.79*(ISS) + 2.16*(Longitude) + 1.80*(TP)** ($F_{3,23}$=5.57, r^2=0.42, p<0.01). TP appears to be more significant in PCA and models in 2016 than in 2015. The model indicates that in early May, chlorophyll-a concentrations were found in sites with low ISS concentrations and increased TP concentrations.

Chlorophyll-a concentrations became more important in the PCA of Round 2-2016 (June 2), where high concentrations were observed in deep, exposed sites with high ISS concentrations (Figure 2.15). The first and second axes explained 34% and 22% of linear variation, respectively (Figure 2.15). Chlorophyll-a was correlated positively with wind exposure and ISS between the first and second axes; they were correlated negatively with euphotic zone on the second axis. Chlorophyll-a, total depth, latitude, longitude and wind exposure were correlated positively and were correlated negatively with water temperature and conductivity on the first axis. A multiple stepwise regression analysis to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = -0.25 + 0.36*(Wind Exposure)** ($F_{2,42}$=3.64, r^2=0.15, p=0.03). The model for Round 2- 2016 was weak in predicting phytoplankton chlorophyll-a concentration.

The PCA for Round 3-2016 (June 21) displayed a relationship between chlorophyll-a, wind exposure, ISS, and longitude, all of which were correlated negatively with water temperature and euphotic depth (Figure 2.16). The first and second axes explained 31% and 23% of linear variation respectively. Chlorophyll-a and latitude were correlated negatively along the second axis, indicating that as you move south, chlorophyll-a concentrations increase. A multiple stepwise regression to predict

Figure 2.15. Principal Component Analysis (PCA) of water quality sampling Round 2-2016 (June 2) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 34% and 22% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

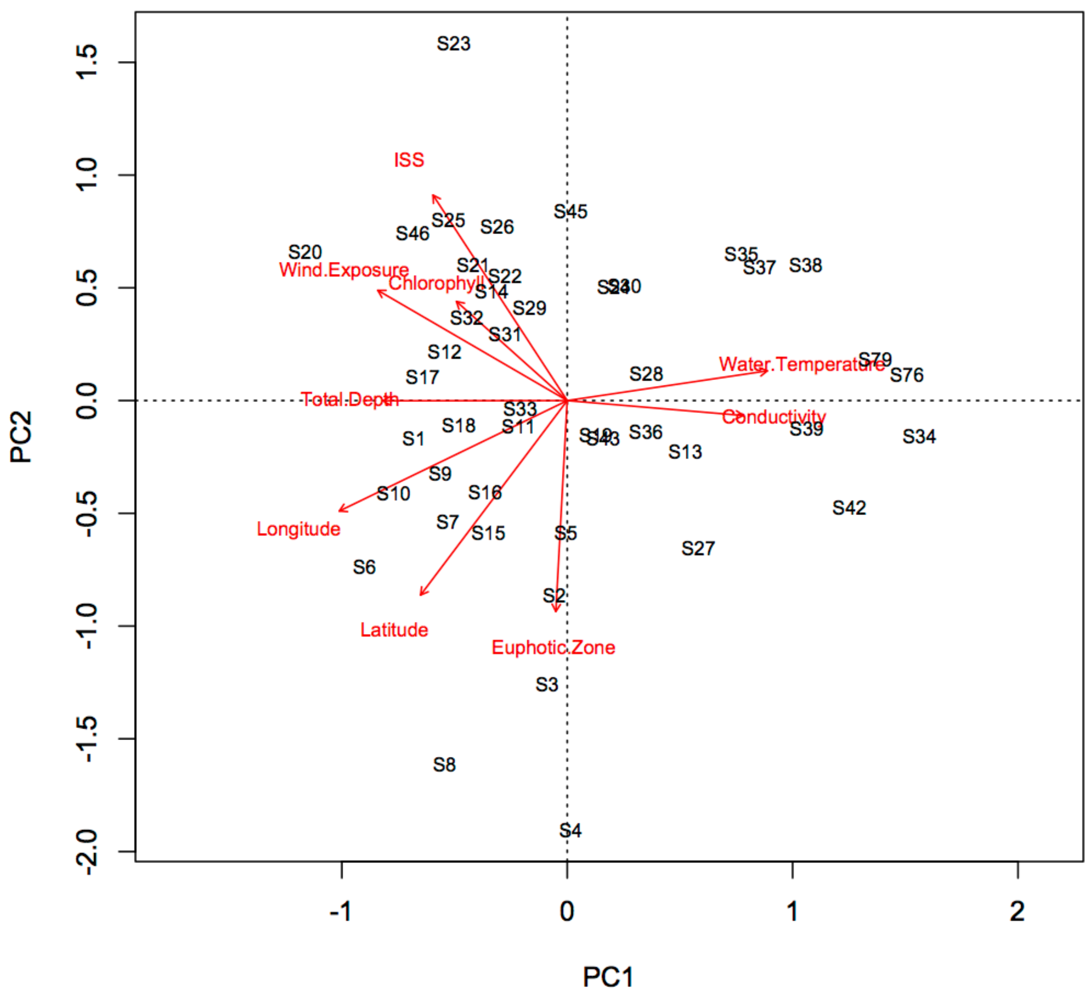
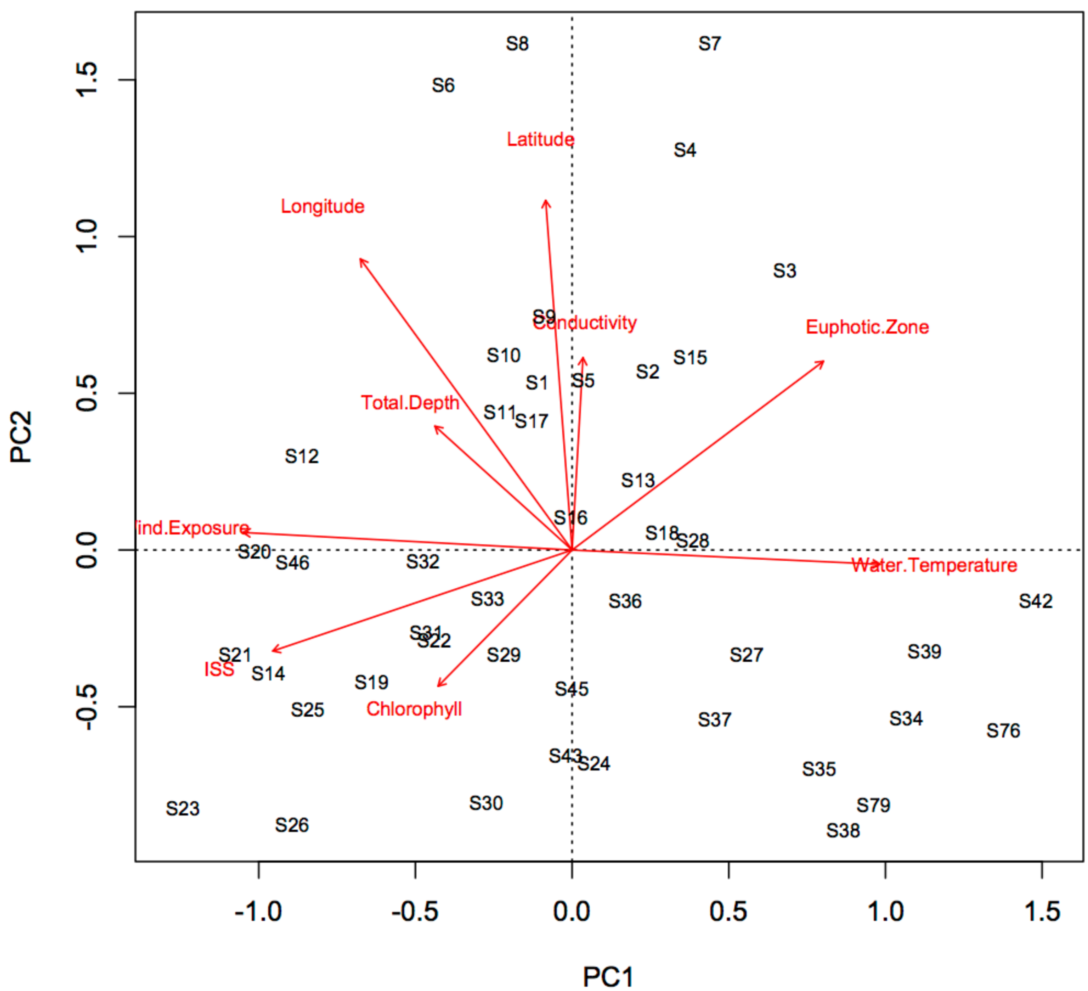


Figure 2.16. Principal Component Analysis (PCA) of water quality sampling Round 3-2016 (June 21) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 31% and 23% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 301 – 5.98*(Latitude) + 0.41*(Wind Exposure)** ($F_{2,42}$=3.64, r^2=0.15, p=0.03). When nutrient data were included, chlorophyll-a concentrations were associated with deep, exposed sites with high ISS and nutrient concentrations. The first and second axes explained 35% and 27% of linear variation, respectively (Figure 2.17). Chlorophyll-a was correlated positively with TN and TP between the first and second axes, and correlated negatively with euphotic zone, conductivity, longitude and latitude on the first axis. Total depth, wind exposure, ISS, chlorophyll-a and TN were correlated negatively with water temperature and euphotic zone on the first axis. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = -10.5 + 0.26*(Water Temperature) + 0.002*(Conductivity) + 0.003*(Euphotic Depth) – 0.73*(ISS) + 1.70*(Wind Exposure) + 4.27*(TN) + 0.95*(TP)** ($F_{7,20}$=3.61, r^2=0.56, p=0.01). When nutrients were included in the stepwise regression analysis, the coefficient of determination increased, again supporting nutrient concentration may be important in predicting phytoplankton chlorophyll-a concentration in the water column. TN was the most important variable included in the selected model for Round 3-2016. Phytoplankton chlorophyll-a concentrations were found in sites with high TN and TP concentrations and wind exposure.

The PCA for Round 4-2016 (July 13) indicated that chlorophyll-a was correlated positively with ISS, wind exposure, conductivity on the first and second axes, and correlated negatively with euphotic depth (Figure 2.18). Total depth, longitude and latitude were correlated negatively with water temperature between the first and second axes. The first and second axes explain 33% and 24% of linear variation, respectively. A

Figure 2.17. Principal Component Analysis (PCA) of water quality sampling Round 3-2016 (June 21) across 30 sites in Delta Marsh, Manitoba, including nutrient concentration data. Points on each PCA represent individual sites. The two axes account for 35% and 27% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

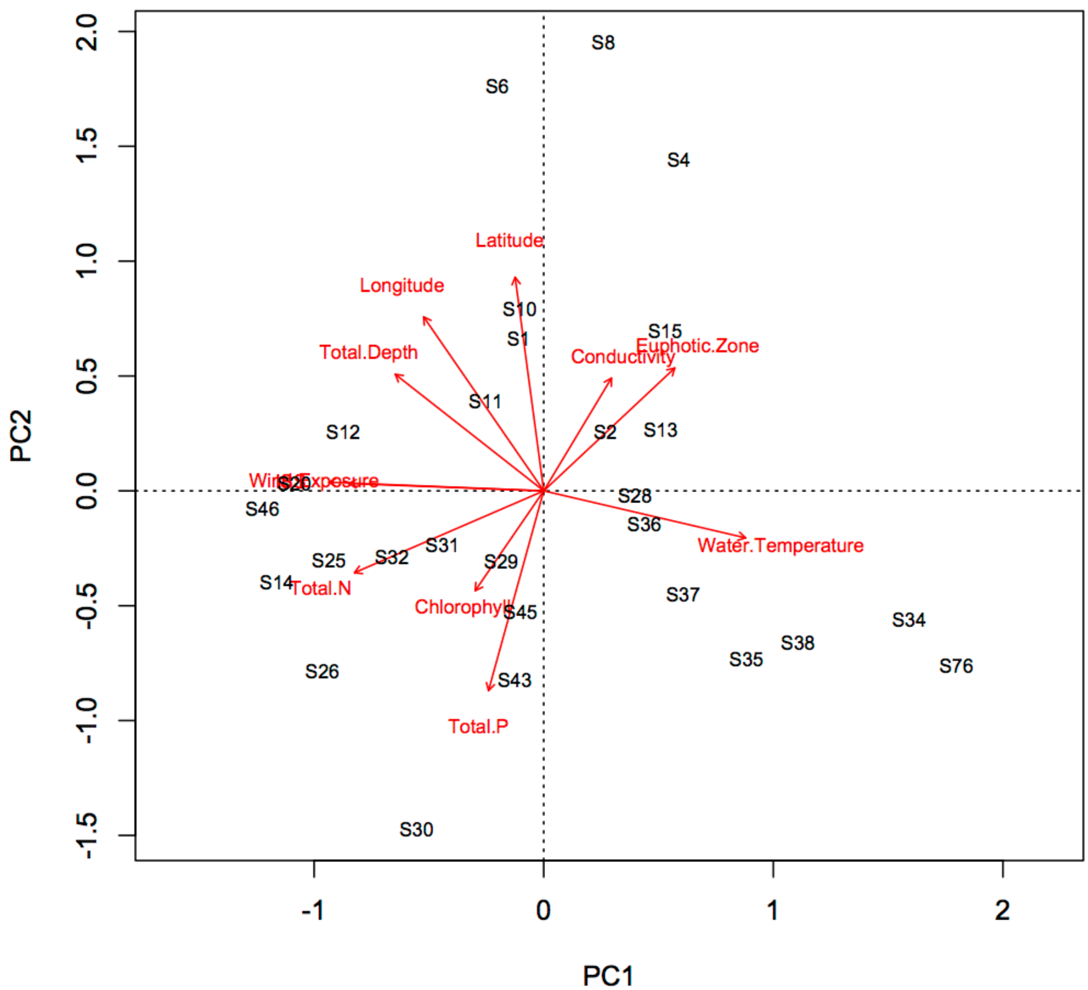
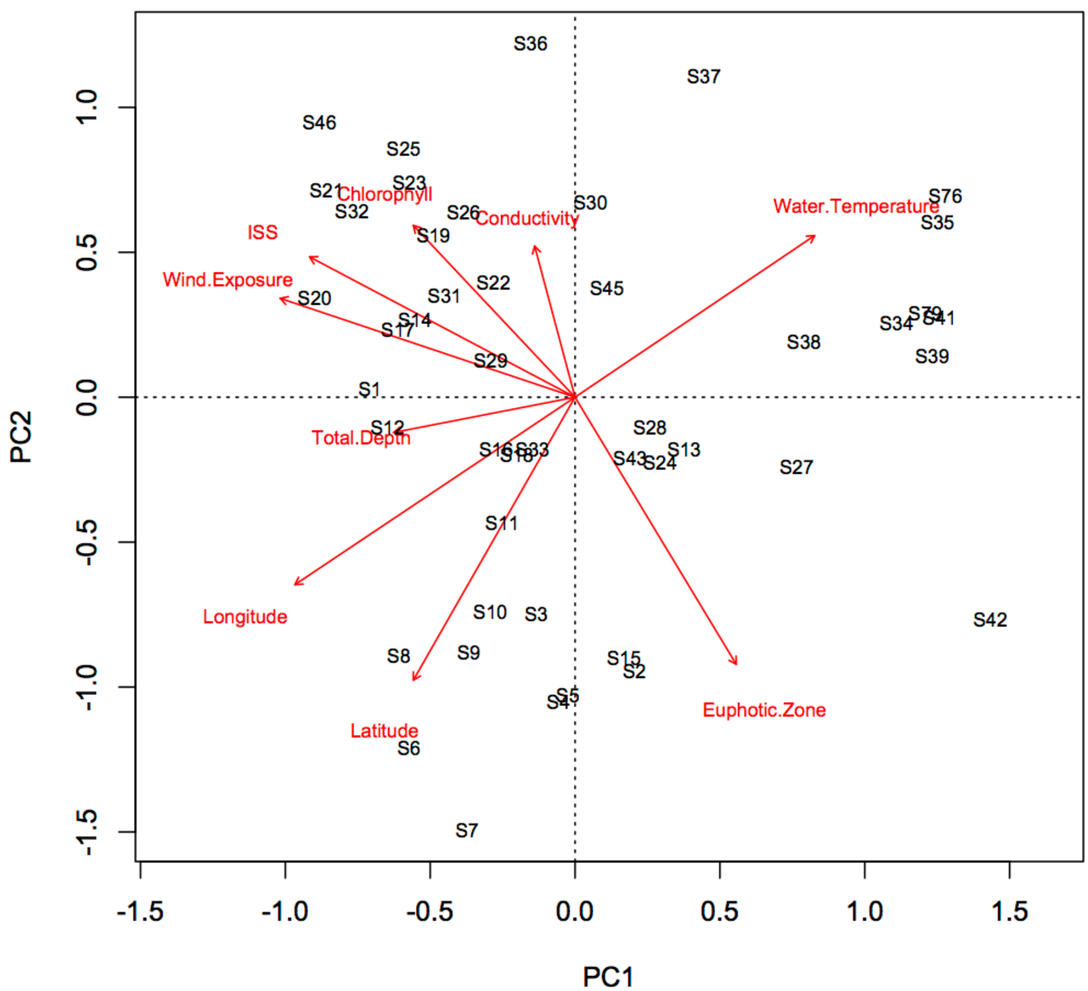


Figure 2.18. Principal Component Analysis (PCA) of water quality sampling Round 4-2016 (July 13) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 33% and 24% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 1.89 – 0.34*(Euphotic Depth) + 0.20*(Wind Exposure)** ($F_{2,43}$=3.61, r^2= 0.22, p<0.01).When nutrient data were included, high chlorophyll-a concentrations were associated with deep, exposed sites with high conductivity and nutrient and ISS concentrations. The first and second axes explained 35% and 28% of linear variation, respectively (Figure 2.19). Chlorophyll-a was correlated positively with TP, conductivity, TN, ISS and wind exposure, and negatively with euphotic zone depth on the first axis. Latitude, longitude, total depth and wind exposure were correlated negatively with water temperature on the second axis. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a= -0.45 + 0.001*(Depth) + 0.14*(Water Temperature) – 0.002*(Conductivity) – 0.13*(ISS) + 0.22*(Euphotic Depth) + 2.54*(TN)** ($F_{6,21}$=27.9, r^2=0.88, p<0.01). Phytoplankton chlorophyll-a concentration was best predicted by high TN concentration, water temperature, low conductivity and ISS concentration and a shallow euphotic depth.

The PCA for Round 5- 2016 (August 3) indicated that chlorophyll-a concentrations were associated with exposed deep sites (Figure 2.20). The first and second axes explained 27% and 22% of linear variation, respectively. Chlorophyll-a was correlated positively with longitude, euphotic zone, total depth and wind on the first axis. Conductivity, ISS and wind exposure were correlated positively on the second axis, and negatively with water temperature. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 213 – 5.68*(Water Temperature) – 24.5*(ISS)** ($F_{2,43}$=2.75, r^2=0.11, p=0.07).

Figure 2.19. Principal Component Analysis (PCA) of water quality sampling Round 4-2016 (July 13) across 30 sites in Delta Marsh, Manitoba, including nutrient concentration data. Points on each PCA represent individual sites. Together, the two axes account for 35% and 28% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

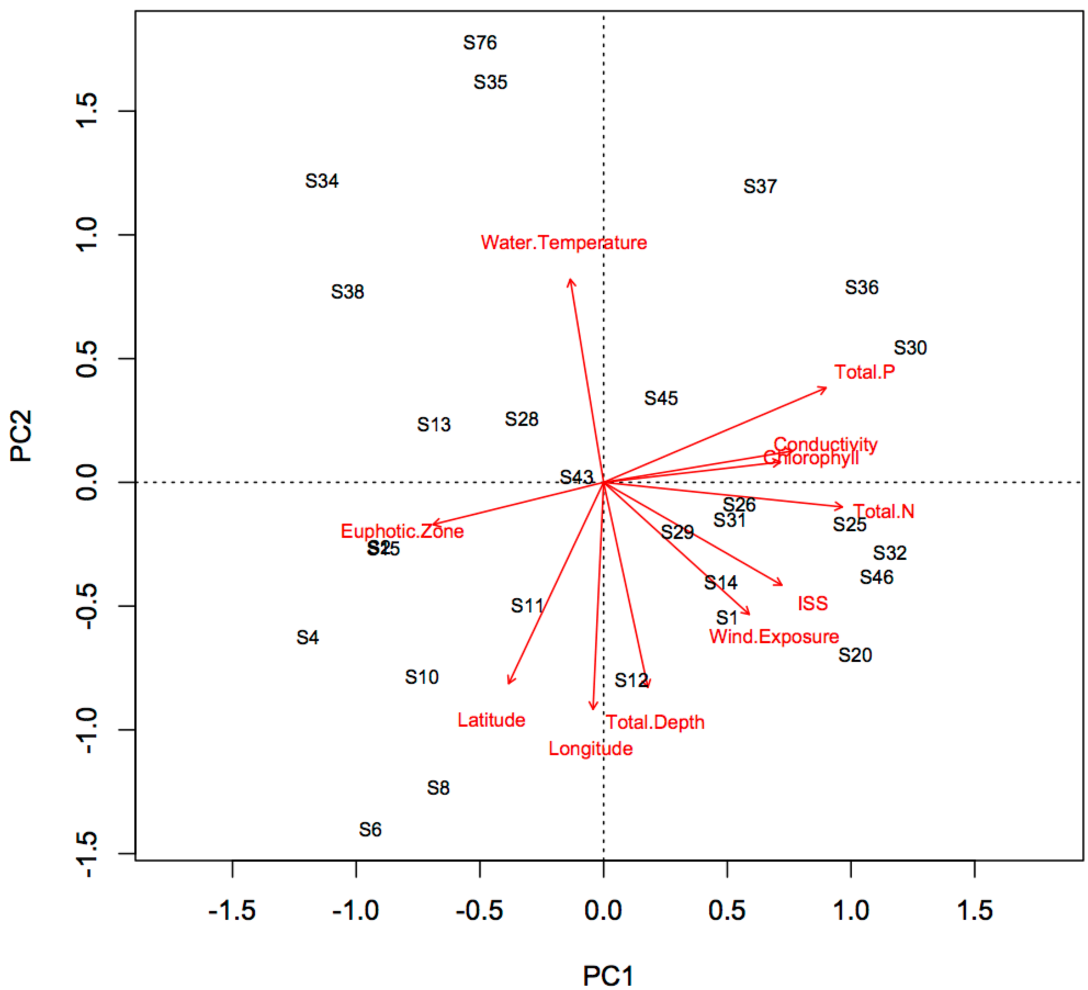
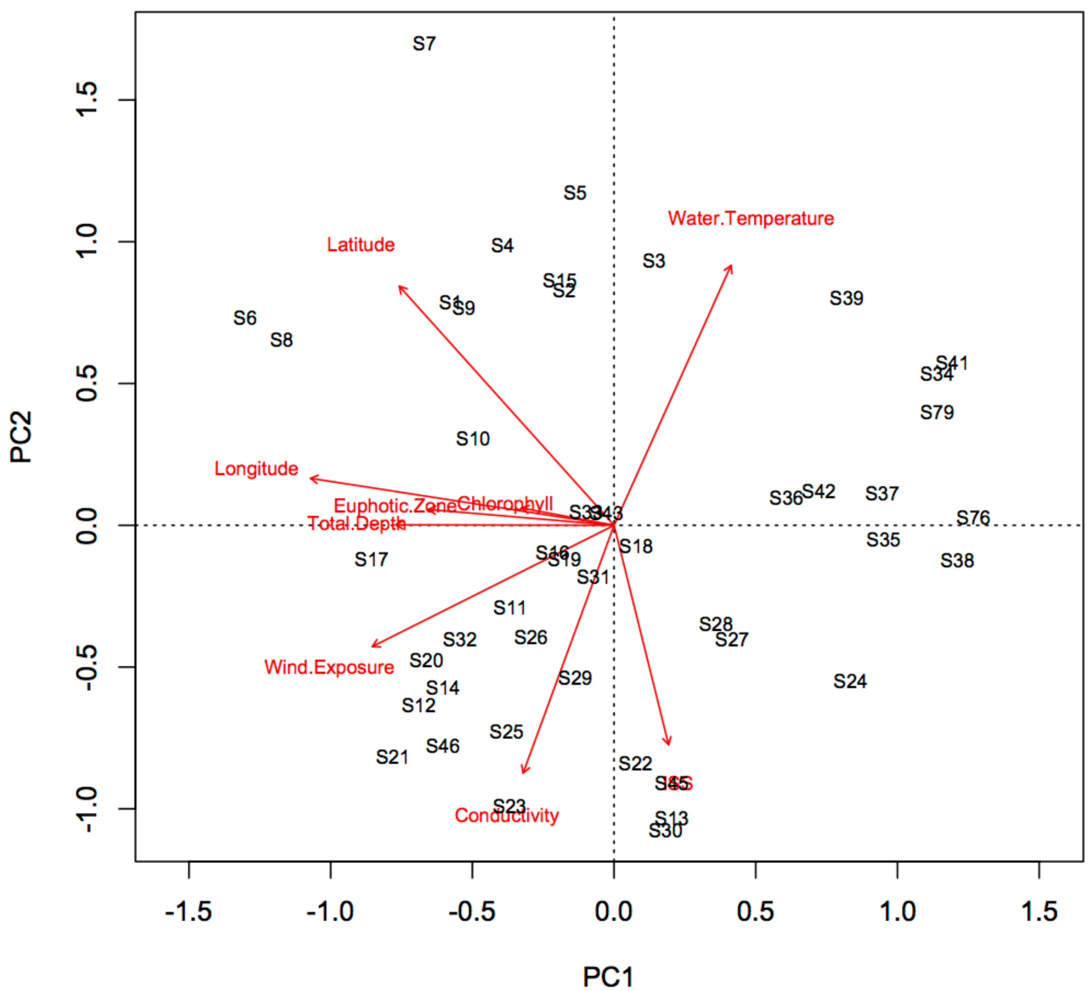


Figure 2.20. Principal Component Analysis (PCA) of water quality sampling Round 5-2016 (August 3) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 27% and 22% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



During Round 6-2016 (August 23), chlorophyll-a concentrations were correlated positively with total depth, wind exposure, ISS and longitude on the first axis, and all five variables were correlated negatively with water temperature. Euphotic zone and latitude were correlated negatively with conductivity along the second axis. The first and second axes explained 31% and 19% of linear variation, respectively (Figure 2.21). A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a= -3.16e^{-2} – 1.00e^{-1}*(Water Temperature) – 5.05e^{-3}*(Euphotic Depth) + 6.37*(Latitude)** ($F_{3,42}$=3.37, r^2=0.19, p=0.03). Phytoplankton chlorophyll-a concentration was best predicted by euphotic zone depth, water temperature and spatial location in Round 6-2016. When nutrient data were included in the model, high chlorophyll-a concentrations were associated with exposed, deep-water sites with high concentrations of TN and ISS. The first and second axes explained 34% and 21% of linear variation, respectively (Figure 2.22). Chlorophyll-a was correlated positively with TN and conductivity, and negatively with euphotic zone depth on the second axis. Chlorophyll-a, wind exposure, total depth, longitude and ISS were correlated positively on the first axis, and all correlated negatively with water temperature. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = -280 – 0.21*(Water Temperature) + 0.32*(ISS) – 3.98*(Conductivity) + 5.89*(Latitude) + 2.97*(TN) + 0.35*(TP)** ($F_{6,21}$=6.92, r^2=0.66, $p < 0.01$). Phytoplankton chlorophyll-a concentrations in late August were best predicted by high concentrations of TN, TP and ISS and low conductivity.

Results of an ordination that used all 2016 sampling rounds without nutrient concentration data indicated that high chlorophyll-a concentrations were associated with

Figure 2.21. Principal Component Analysis (PCA) of water quality sampling Round 6-2016 (August 23) across 48 sites in Delta Marsh, Manitoba. Points on each PCA represent individual sites. The two axes account for 31% and 19% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

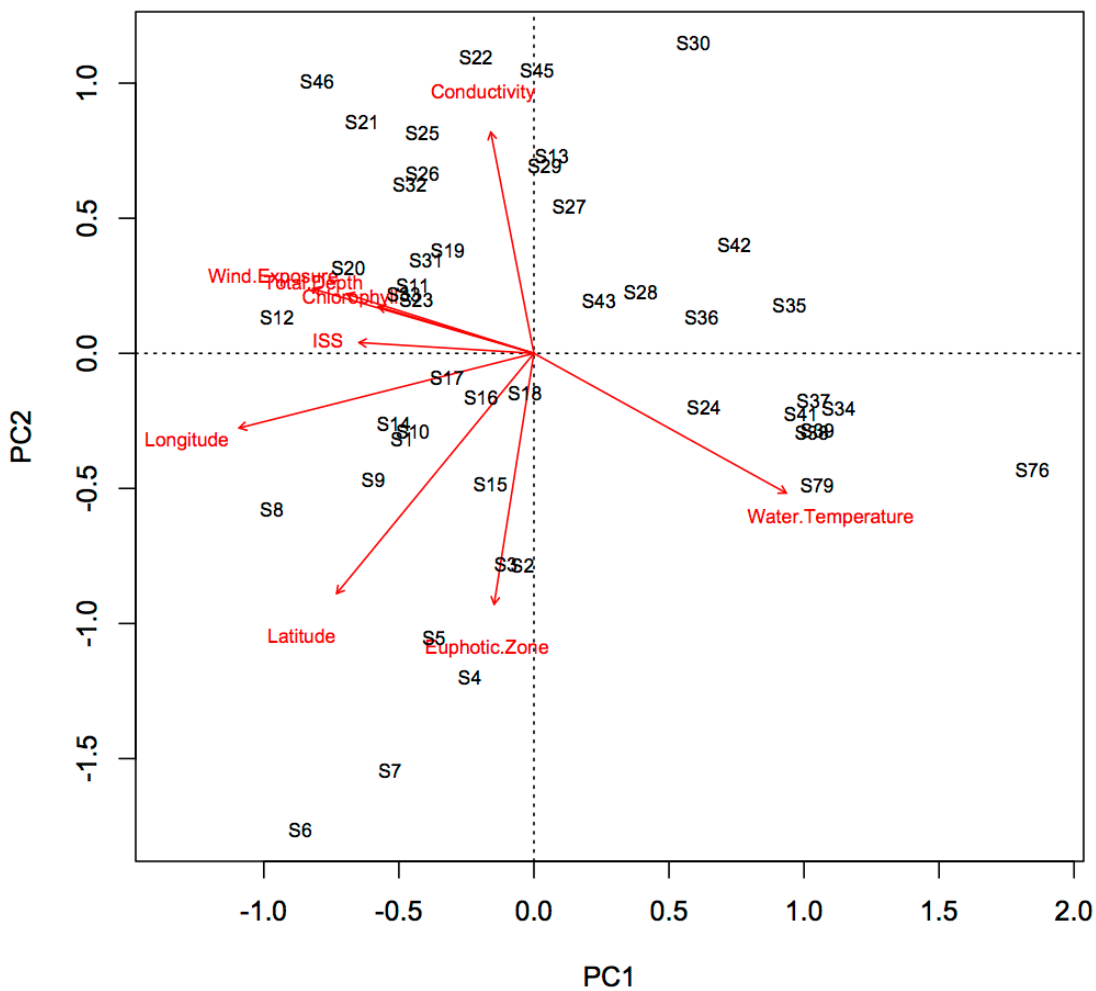
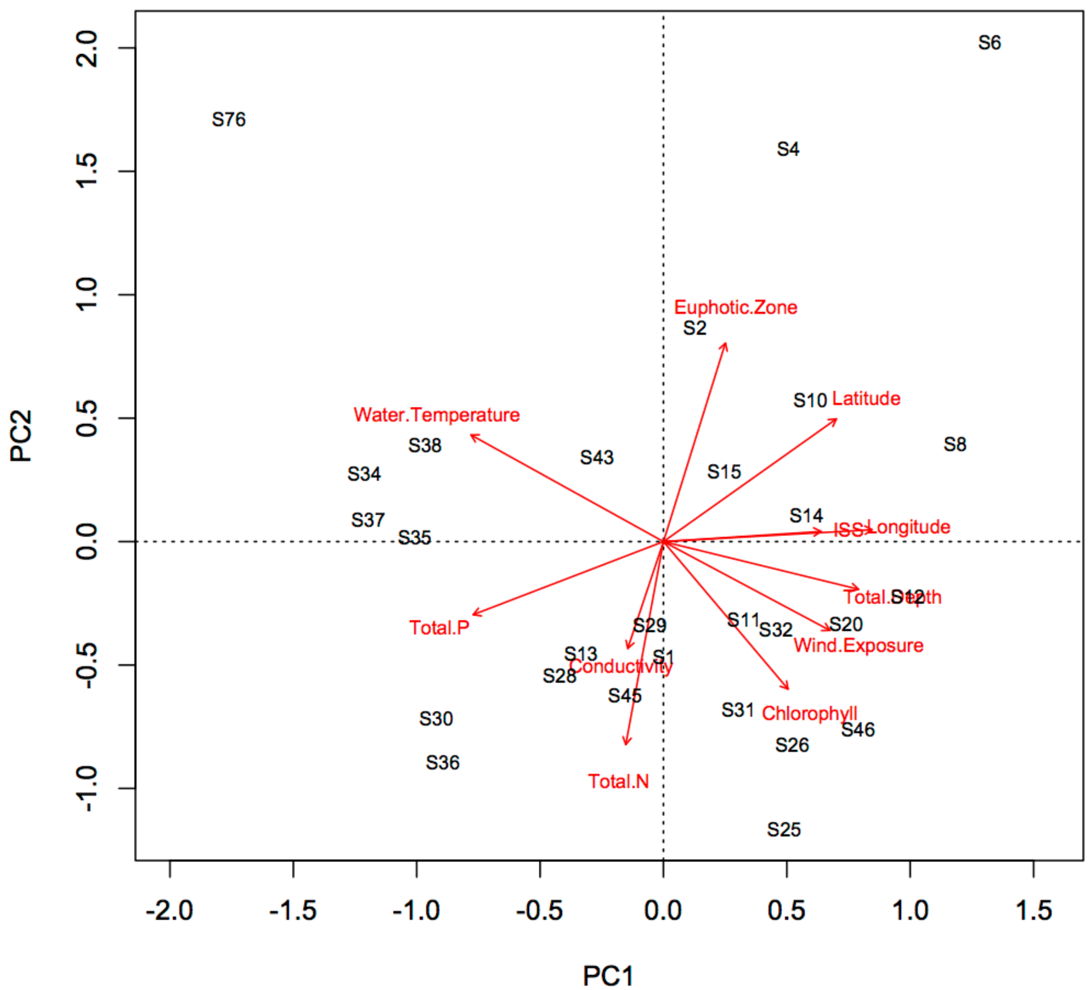


Figure 2.22. Principal Component Analysis (PCA) of water quality sampling Round 6-2016 (August 23) across 30 sites in Delta Marsh, Manitoba, including nutrient concentration data. Points on each PCA represent individual sites. The two axes account for 34% and 21% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



deep, exposed sites with high ISS concentrations. The first and second axes explained 34% and 21% of linear variation respectively (Figure 2.23). Chlorophyll-a was correlated positively with ISS, conductivity and wind exposure between the first and second axes. Chlorophyll-a, ISS, wind exposure, total depth, and longitude were negatively correlated with water temperature along the first axis. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = 1.33 + 0.26*(ISS)** ($F_{1,44}$=8.54, r^2=0.16, p<0.01). This model is a weak predictor of phytoplankton chlorophyll-a concentration for the mean water quality data in 2016. When nutrient data were included, the mean model for 2016 indicated that high chlorophyll-a concentrations are found in exposed, deep-water sites to the south with high conductivity and concentrations of ISS, TN and TP. It also showed that spatial distribution may influence chlorophyll-a concentrations. The first and second axes explained 38% and 26% of linear variation respectively (Figure 2.24). Chlorophyll-a was correlated positively with conductivity, TN and TP on the second axis. Wind exposure, ISS, total depth, longitude, chlorophyll-a and TN were correlated negatively with water temperature and euphotic zone on the first axis. A multiple stepwise regression to predict phytoplankton chlorophyll-a concentration selected the model, **Chlorophyll-a = -286 + 0.25*(TN) + 3.96*(Latitude) – 0.96*(Longitude) – 2.58*(Conductivity) + 0.40*(Depth) + 0.29*(TP)** ($F_{6,21}$= 13.6, r^2=0.79, p<0.01). When nutrients were included in the mean model for 2016 data, the coefficient of determination increased to strong, indicating that nutrients may be important for predicting phytoplankton chlorophyll-a concentrations throughout the field season at Delta Marsh. The 2016 stepwise regression model indicates that conductivity, TN, TP, depth and spatial distribution of the sites best

Figure 2.23. Mean Principal Component Analysis (PCA) of the 2016 (May-August) water quality sampling across 48 sites in Delta Marsh, Manitoba. The variables in this PCA represent a mean value for each site analyzed from May to August 2016. The two axes account 34% and 21% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.

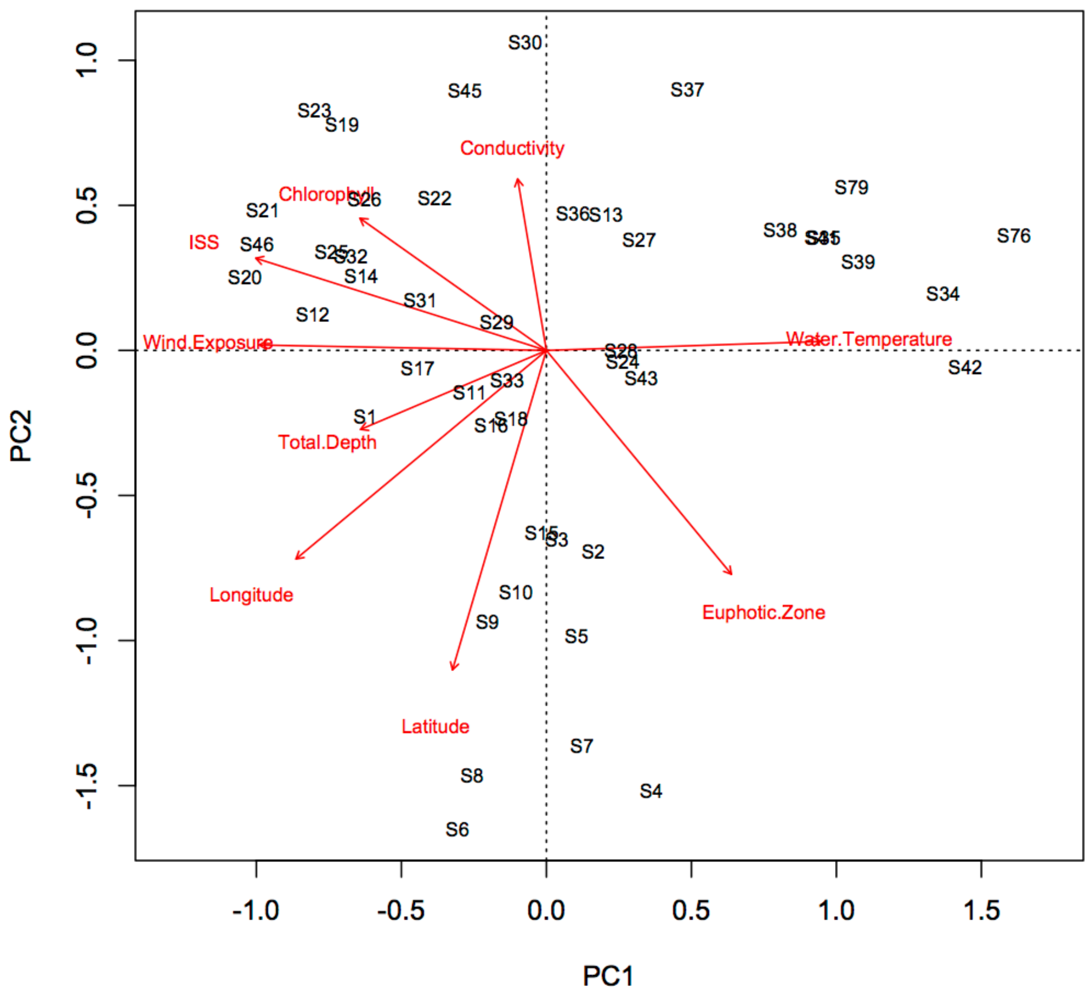
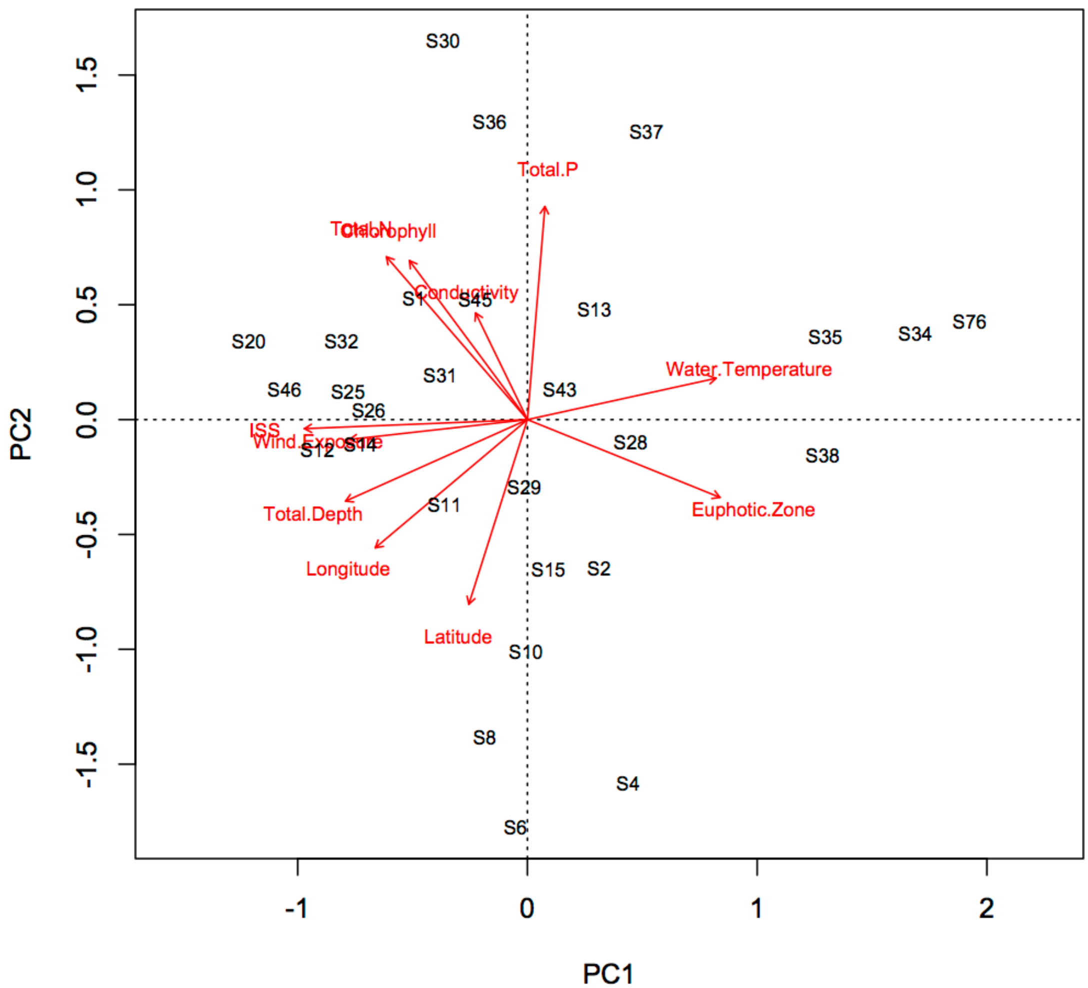


Figure 2.24. Mean Principal Component Analysis (PCA) of the 2016 water quality sampling across 30 sites in Delta Marsh, Manitoba, including nutrient concentration data. The variables in this PCA represent a mean value for each site analyzed in round 1 (May 11), round 3 (June 21), round 4 (July 13) and round 6 (August 23). The two axes account 38% and 26% of linear variation, respectively. All variables that displayed non-normal distribution were log(x) and log (x+1) transformed.



predict chlorophyll-a concentrations.

A multiple stepwise regression analysis that included round number was performed to determine how temporal variation influenced phytoplankton chlorophyll-a concentrations in Delta Marsh during the 2015 and 2016 field season. The model selected for the 2015 data was, **Chlorophyll-a = -212 + 0.19*(Round) – 0.62*(Euphotic Depth) – 2.18*(Longitude) + 0.20*(Wind Exposure)** ($F_{4,178}$=14.5, r^2=0.25, p<0.01). The model selected for the 2016 data was, **Chlorophyll-a= 1.61 + 0.20*(Round) – 0.48*(Euphotic Depth)** ($F_{2,243}$=84.9, r^2=0.41, p<0.01). The models selected for both 2015 and 2016 indicated that chlorophyll-a concentration was predicted by sampling date and euphotic zone depth. The models indicate that chlorophyll-a concentrations increased throughout the field season and when euphotic zone depth is shallow. When nutrient concentrations were included in the stepwise regression for the 2015 field season data, the model selected was, **Chlorophyll-a = -153 - 1.55*(Longitude) + 1.18*(TN) + 0.74*(ISS)** ($F_{3,52}$=10.1, r^2=0.37, p<0.01). Spatial location, TN and ISS concentrations were more important than temporal variation (sampling round) in predicting phytoplankton chlorophyll-a concentration in the 2015 field season. The 2016 selected model that included nutrient concentrations was, **Chlorophyll-a = 307 + 0.004*(Depth) – 6.18*(Latitude) + 1.74*(TN) + 2.36*(Water Temperature)** ($F_{4,99}$=18.77, r^2=0.43, p<0.01). Similar to the 2015 model, temporal variation (sampling round) was not important in predicting phytoplankton chlorophyll-a concentration. Chlorophyll-a concentration was best predicted by spatial location, TN concentration, water temperature and total water depth.

Throughout the 2015 and 2016 sampling rounds, phytoplankton chlorophyll-a concentrations were most influenced by TN and TP concentrations. Most models without nutrient concentration data did not have high predictive power for phytoplankton chlorophyll-a concentrations. Other important variables for predicting phytoplankton chlorophyll-a concentrations included in the PCA and the multiple stepwise regression models were ISS concentration, water temperature, spatial location, conductivity, and wind exposure.

Phytoplankton chlorophyll-a concentration varied spatially and temporally across the marsh and throughout the field season in 2015 and 2016. Throughout the 2015 field season, there was a significant difference in chlorophyll-a concentrations temporally, Rounds 3 to 5- 2015 (July 2- August 19) had significantly higher concentrations relative to Rounds 1 and 2 – 2015 (May 14- June 25) (F_4=24.2, p<0.001) (Figure 2.25). Chlorophyll-a concentrations decreased from Round 1-2015 to Round 2- 2015, with a mean (±S.E.) phytoplankton chlorophyll-a concentration of 31.4 ± 6.99 µg/L and 13.3 ± 3.51 µg/L, respectively. Throughout the rest of the season the mean chlorophyll-a concentrations increased from Round 3- 2015 (the intensive sampling round in early July) to the final sampling round in late August. In 2016, Round 2-2016 had a lower mean chlorophyll-a concentration than in Round 1-2016, with mean concentrations (± SE) of 7.5 ± 1.80 µg/L and 16.9 ± 3.58 µg/L respectively (Figure 2.26). A similar trend to 2015 was observed in 2016, where chlorophyll-a concentrations increased throughout the field season, although Round 4-2016 (July 13) had the highest mean chlorophyll-a concentration of 96.8 ± 18.3 µg/L. Rounds 4 to 6 (July 13- August 23) were significantly higher than rounds 1 to 3 (May 11- June 21) (F_5=46.27, $p<2e^{-16}$). The spatial variation

Figure 2.25. Temporal Variation of phytoplankton chlorophyll-a concentrations (µg/L) (± S.E., N=45) throughout the 2015 field season.

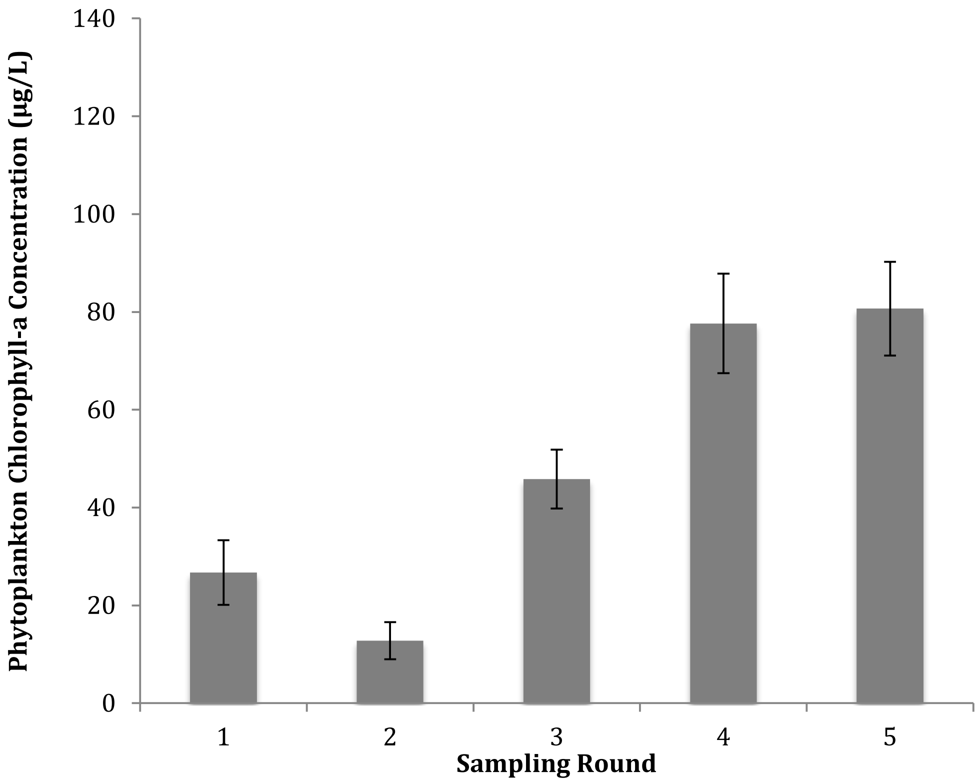
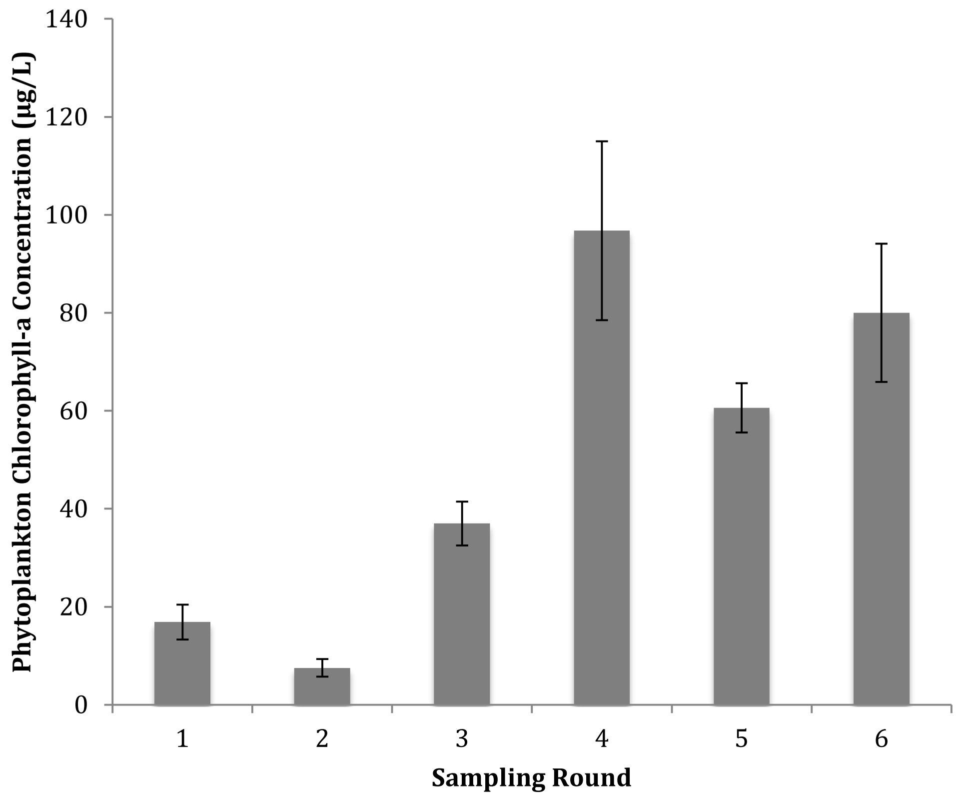


Figure 2.26. Temporal variation of phytoplankton chlorophyll-a concentrations (µg/L) (± S.E., N=45) throughout the 2016 field season.



results in 2015 and 2016 were similar, indicating there was no significant spatial variation of mean chlorophyll-a across the marsh, (F_2=0.117, p=0.89) and (F_2=0.199, p=0.82) for 2015 and 2016, respectively (Figure 2.27 and Figure 2.28).

Seven major classes of algae were observed in the marsh in 2015 and 2016, including Chlorophyceae (green algae), Chrysophyceae (golden algae), Cryptophyceae (cryptomonads), Cyanophyceae (cyanobacteria), Dinophyceae (dinoflagellates), Euglenophyceae (euglenids), and Bacillariophyceae (diatoms). Round 1 2015 (May 14 to June 2, 2015) had all algal classes present in the west and east sections of the marsh with a mean total biovolume (± S.E., N=7) of 3.55 ± 0.808 mg/L (Figure 2.29). NW Eaglenest, Mid Cadham and Mid Simpson had the lowest total biovolume of all sites (1.38 mg/L, 0.89 mg/L and 1.39 mg/L, respectively). The intensive sampling round in 2015 (July 2, 2015) had nearly ten times more algal biovolume than Round 1, with a mean biovolume of 27.9 mg/L (Figure 2.30). The west section of the marsh had the highest number of algal classes, while Cyanophyceae dominated in the center and eastern section of the marsh (40 – 98% excluding Mid Waterhen). The mean algal biovolume (± S.E.) was 27.9± 10.6 mg/L for the deep-water sites, while Mid Simpson had three times the mean biovolume of 89.9 mg/L, 98% attributed to Cyanophyceae. Round 4 (August 11 to 19, 2015) had the highest mean algal biovolume of 44.5 ±7.16 mg/L (Figure 2.31). Cyanophyceae was the most common algal class observed across the marsh with the highest biovolume in each of the seven sites, ranging with 73% to 99.9% of total biovolume in the eastern sites, and 39% to 49% in the west.

Figure 2.27. Analysis of variance of log phytoplankton chlorophyll-a (µg/L ± S.E., N=45) across three main bays of Delta Marsh, Center and East 1, East 2 and West. Sampling was performed from May 14 to August 19, 2015. O represents outliers.

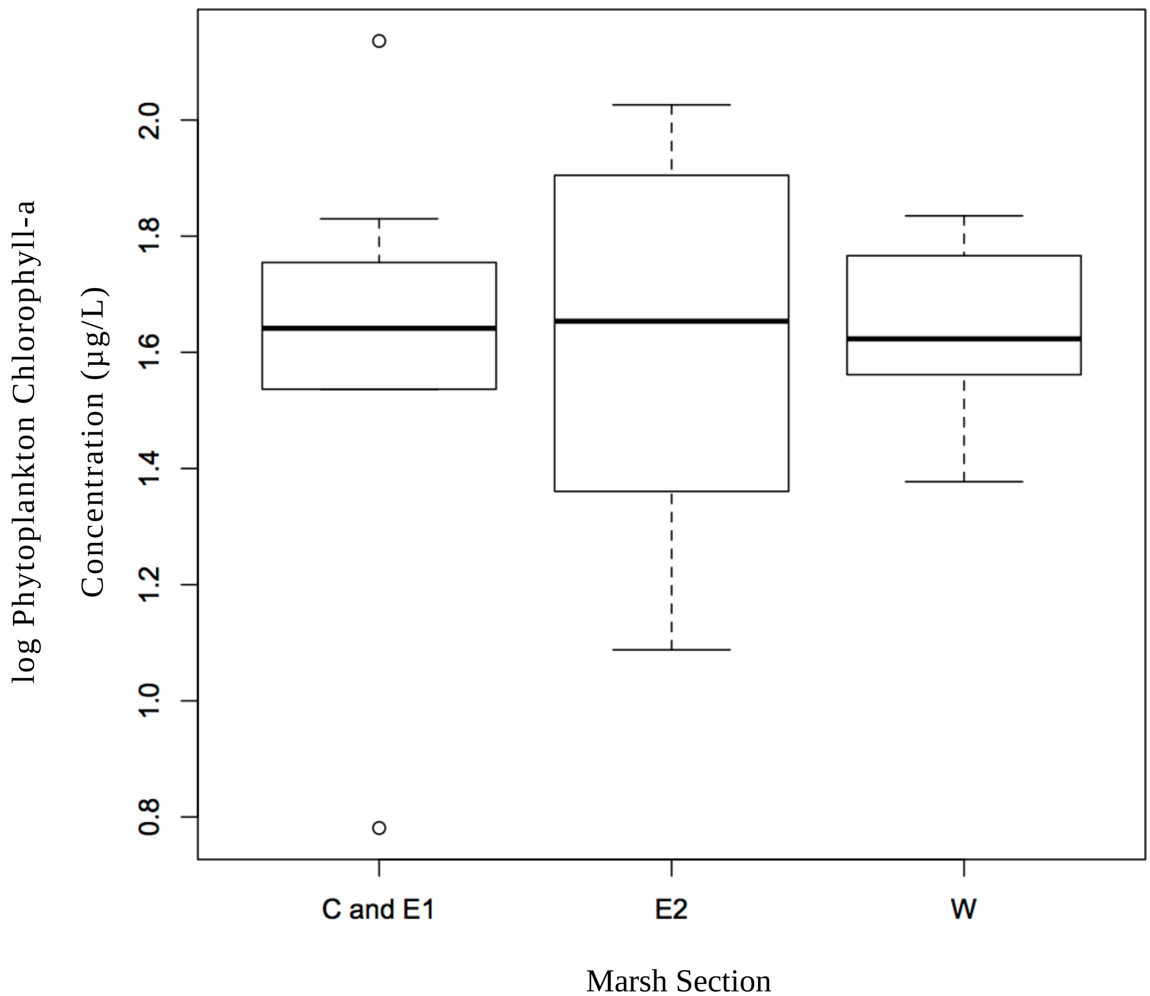


Figure 2.28. Analysis of variance of log phytoplankton chlorophyll-a (µg/L ± S.E., N=45) across three main bays of Delta Marsh, Center and East 1, East 2 and West. Sampling was performed from May 11 to August 23, 2016. O represents outliers.

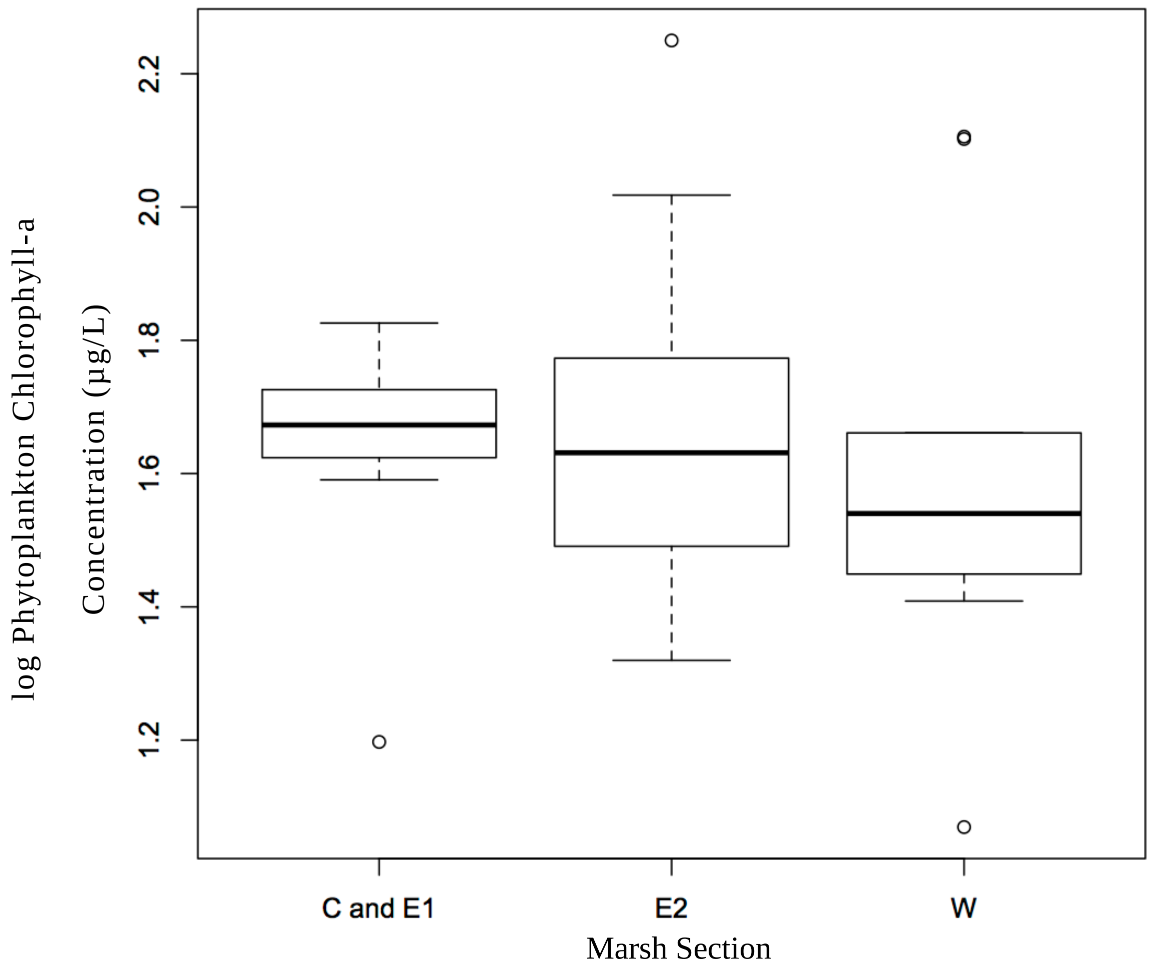


Figure 2.29. Total algal biovolume (mg/L) of seven open-water sites at Delta Marsh Manitoba, in Round 1 2015 (May 14 to June 2, 2015). The mean algal biovolume (± S.E. n=7) was 3.55 ± 0.808 mg/L.

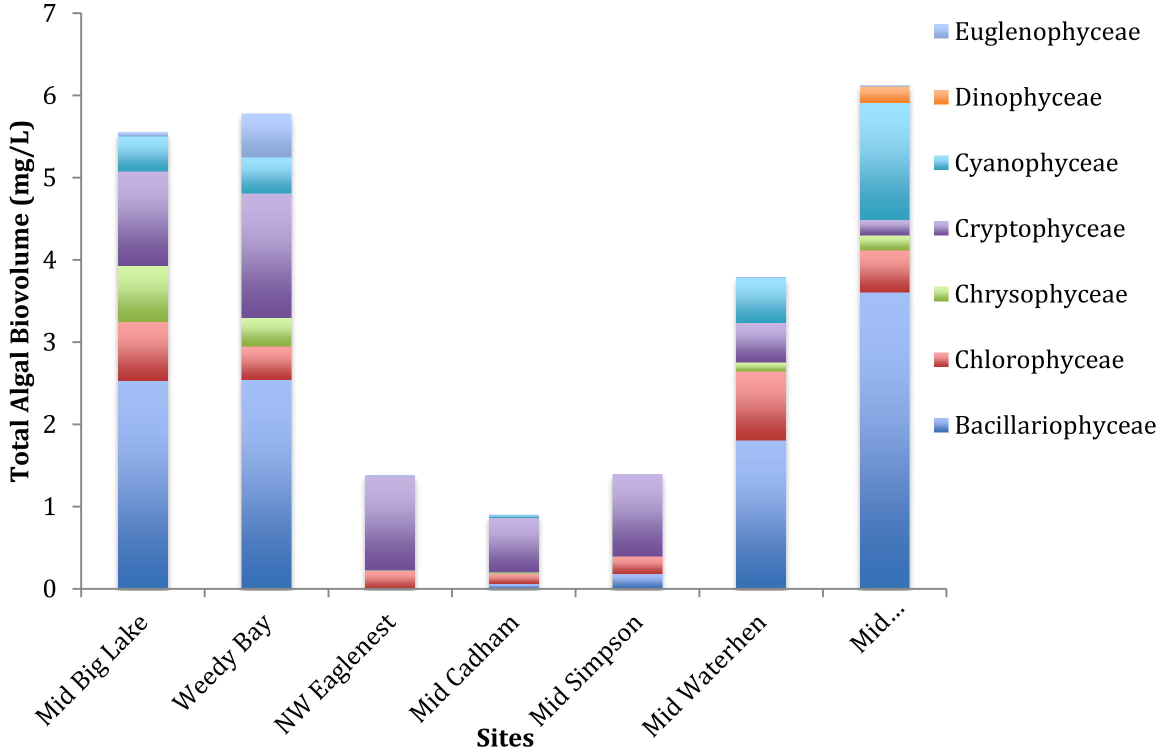


Figure 2.30 Total algal biovolume (mg/L) of seven open-water sites at Delta Marsh Manitoba, in the 2015 intensive sampling round (July 2, 2015). The mean algal biovolume (± S.E. n=7) was 27.9 ± 10.6 mg/L).

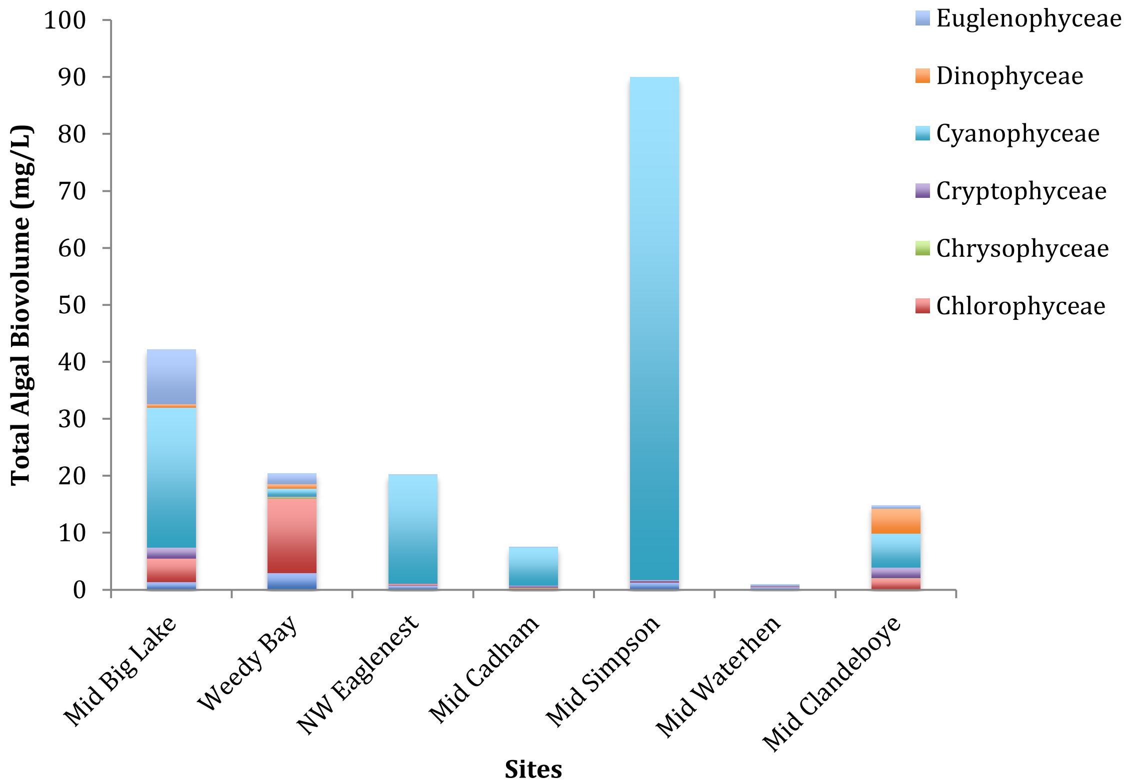
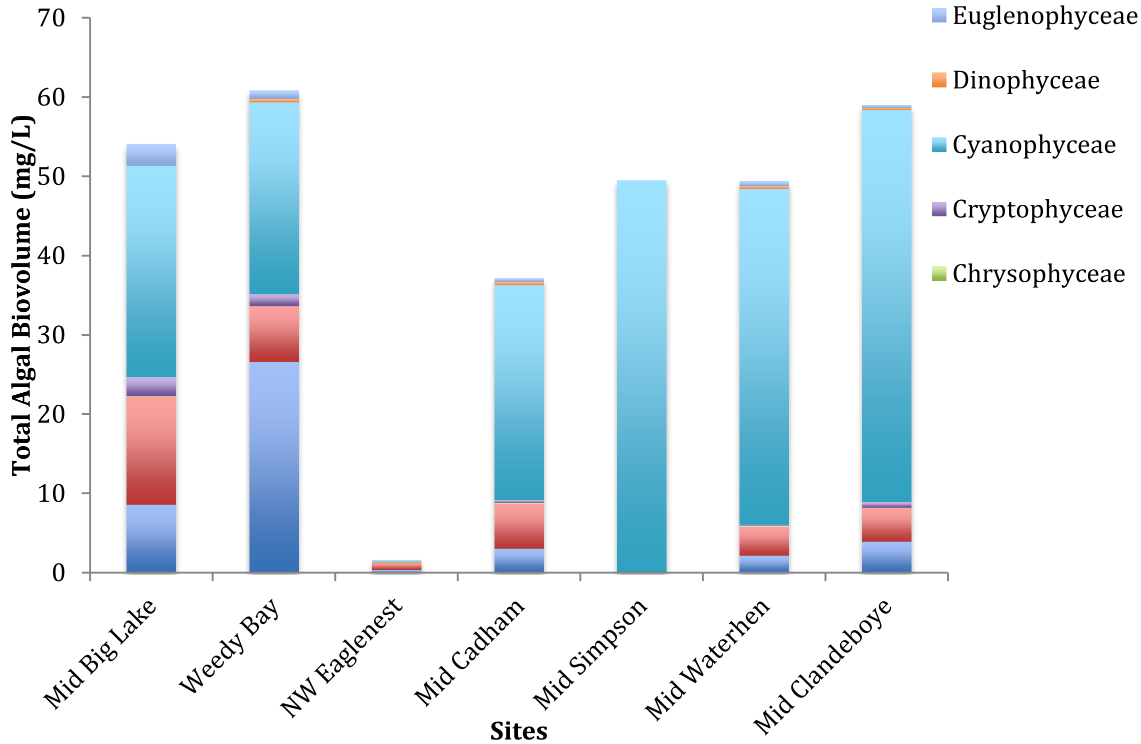


Figure 2.31. Total algal biovolume (mg/L) of seven open-water sites at Delta Marsh Manitoba, in Round 4 2015 (August 11 to 19, 2015). The mean algal biovolume (± S.E. n=7) was 44.5 ±7.16 mg/L.



Round 1 2016 (May 11, 2016) had the most algal classes observed across all seven open water sites of Delta Marsh with a mean biovolume of 4.46 ± 1.27 mg/L (Figure 2.32). Mid Simpson and Mid Clandeboye had the highest total biovolume, approximately two times the mean. Round 3 2016 (June 21, 2016) had multiple algal classes across all deep water sites excluding NW Eaglenest, with a mean algal biovolume of 27.8 ± 12.7 mg/L (Figure 2.33). All sites had multiple algal classes from west to east excluding NW Eaglenest, which was dominated by Chlorophyceae with 81% of the total algal biovolume. NW Eaglenest had the highest algal biovolume of 107.9 mg/L, four times higher than the mean. Round 4 2016 (July 13, 2016) observed an increase in total algal biovolume with a mean of 31.5 ± 4.94 mg/L (Figure 2.34). Cyanophyceae increased in dominance relative to what was observed in round 3, it accounted for 98% of total biovolume in Mid Big Lake and approximately 60% of total biovolume in both Mid Cadham and Mid Simpson. Bacillariophyceae and Chlorophyceae were two other important algal classes observed in all deep-water sites in center and east sections of the marsh. Round 6 2016 (August 23, 2016) observed similar trends as in Round 4, with a mean algal biovolume of 34.3 ± 8.98 mg/L, approximately 10 mg/L less than observed in 2015 (Figure 2.35). Mid Simpson and Mid Waterhen had the highest algal biovolume of 69.6 mg/L and 66.2 mg/L respectively, while Mid Big Lake had the lowest total biovolume of 2.44 mg/L.

Total algal biovolume was temporally variable during each field season and between 2015 and 2016 (Figure 2.36). Round 1 in 2015 had a mean (± S.E., N=7) biovolume of 3.55 ± 0.81 mg/L, increasing to 27.9 ± 10.6 mg/L in the intensive sampling round on July 2, 2015 and to 44.5 ± 7.16 mg/L in round 4, late August. Total algal

Figure 2.32. Total algal biovolume (mg/L) of seven open-water sites at Delta Marsh Manitoba, in Round 1, 2016 (May 11, 2016). The mean algal biovolume (± S.E. n=7) was 4.46 ± 1.27 mg/L.

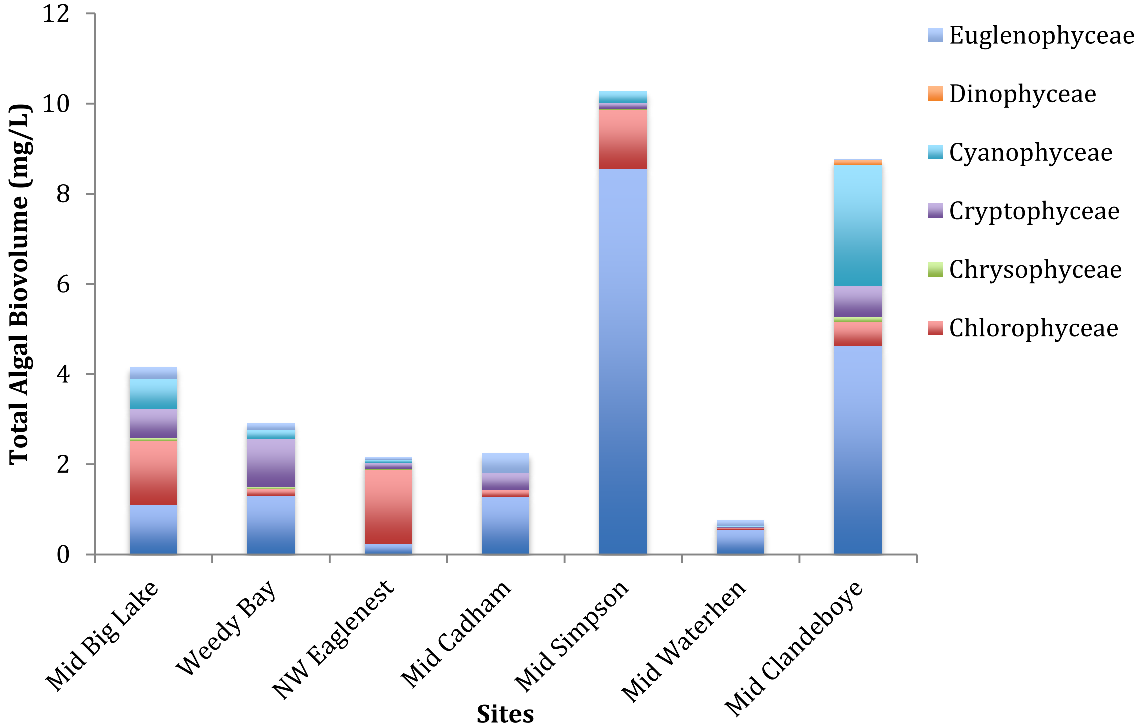


Figure. 2.33 Total algal biovolume (mg/L) of seven open-water sites at Delta Marsh Manitoba, in Round 3, 2016 (June 21, 2016). The mean algal biovolume (± S.E. n=7) was 27.8 ± 12.7 mg/L.

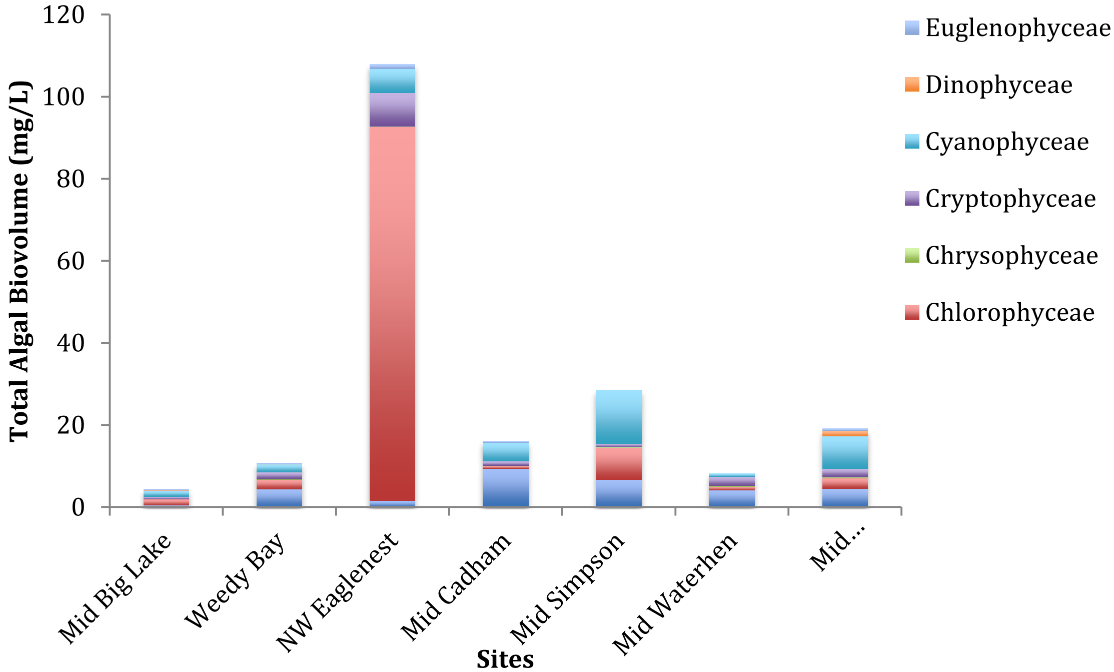


Figure 2.34. Total algal biovolume (mg/L) of seven open-water sites at Delta Marsh Manitoba, in Round 4, 2016 (July 13, 2016). The mean algal biovolume (± S.E. n=7) was 31.5 ± 4.94mg/L.

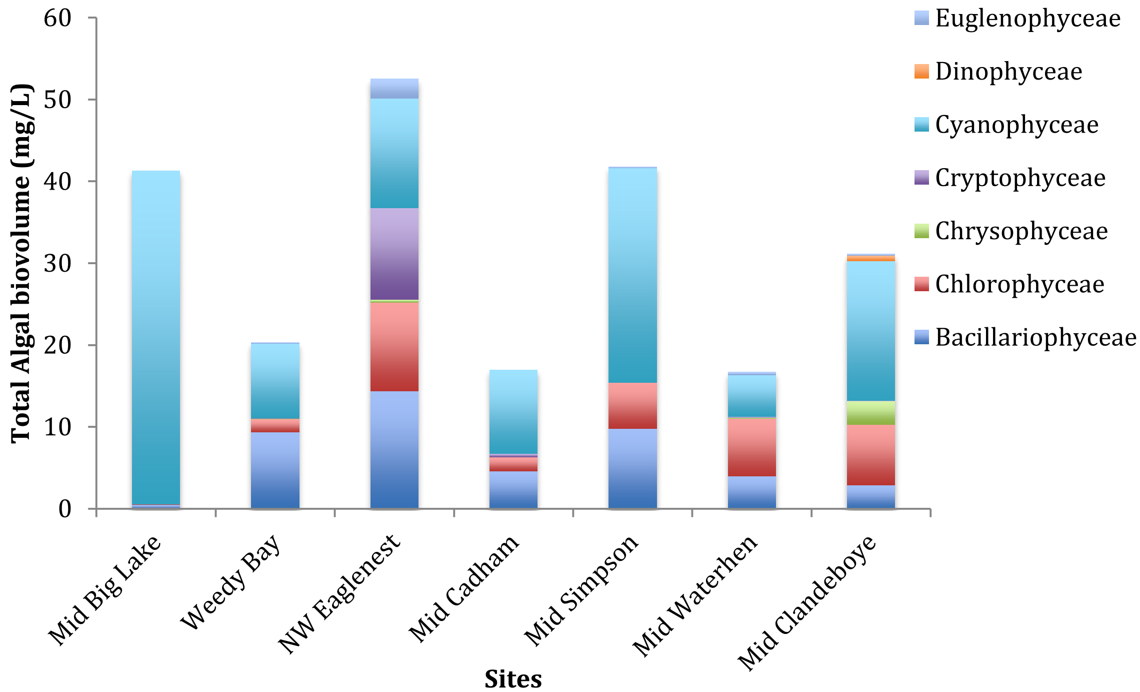


Figure 2.35. Total algal biovolume (mg/L) of seven open-water sites at Delta Marsh Manitoba, in Round 6, 2016 (August 23, 2016). The mean algal biovolume (± S.E. n=7) was 34.3 ± 8.98 mg/L.

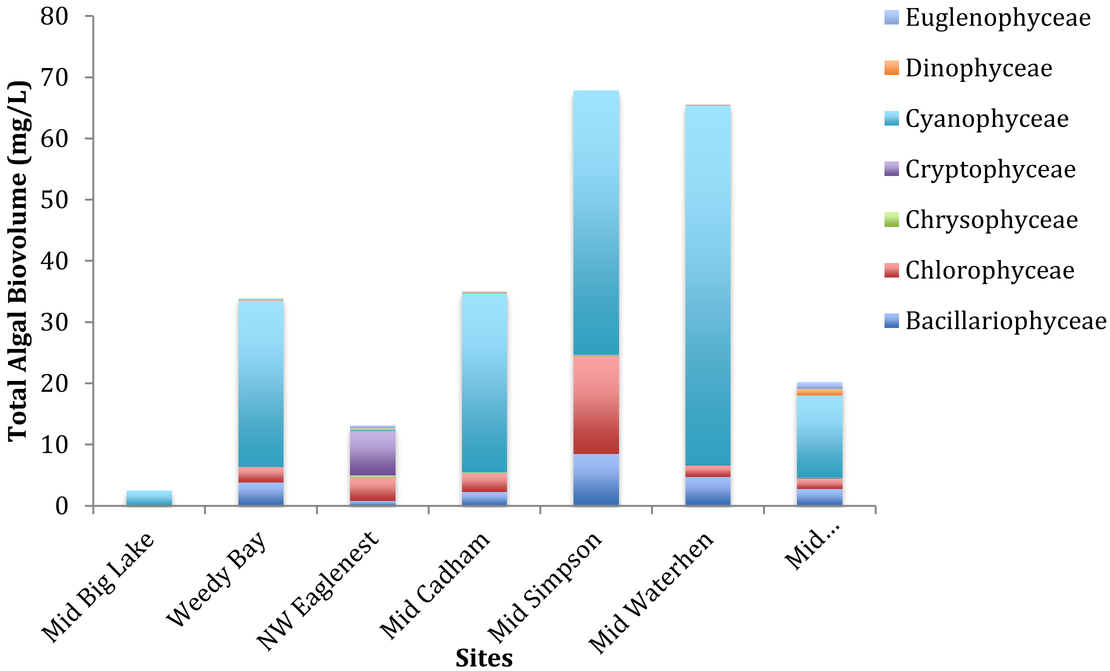
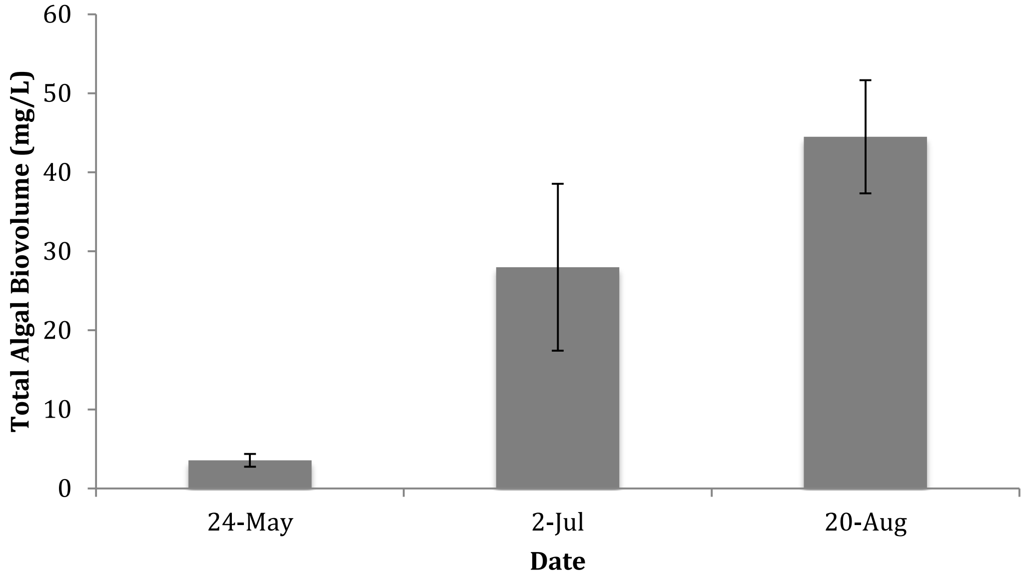
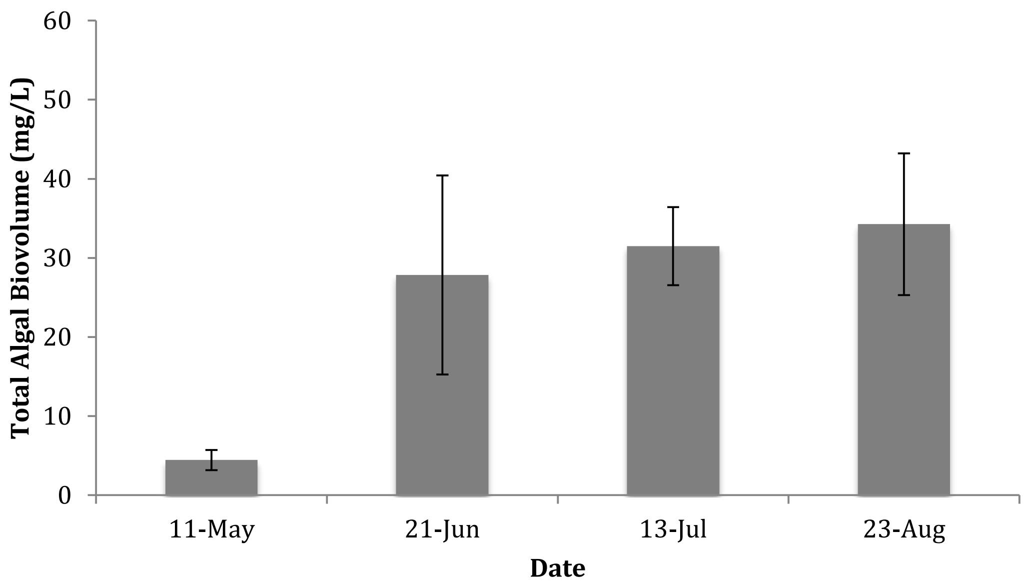


Figure 2.36. Temporal variation of total algal biovolume (mg/L) (±S.E., N=7) throughout the field season in 2015(A) and 2016 (B).

(A)



(B)



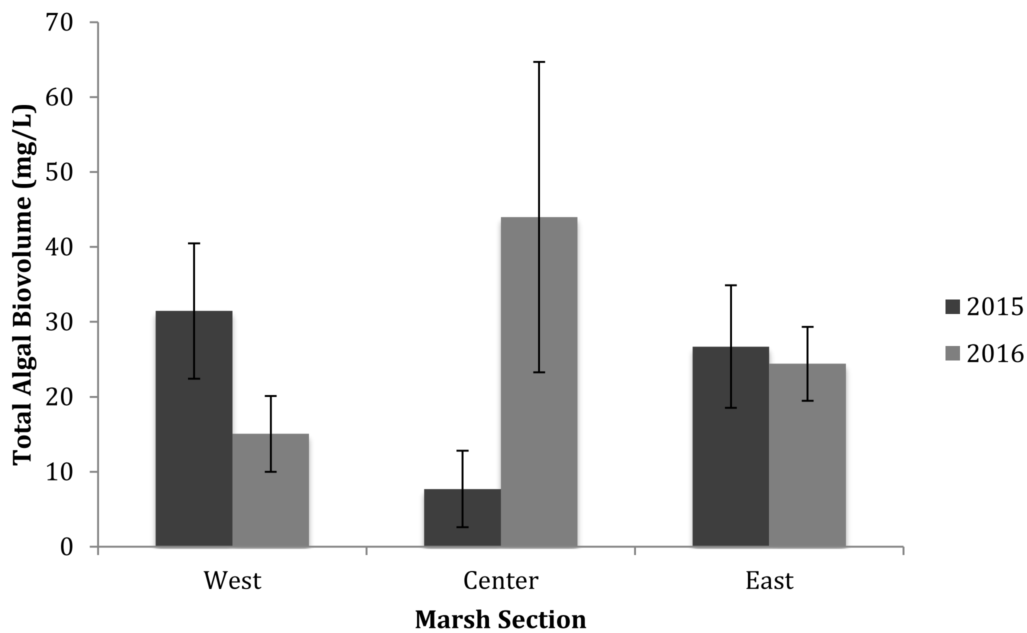
biovolume in round 1 (May) remained similar to 2015 with a mean biovolume of 4.46 ± 1.27 mg/L. Round 3 reflected similar biovolumes as the intensive round in 2015, with a mean of 27.9 ± 12.6 mg/L, although round 4 and round 6 did not increase with mean biovolumes of 31.5 ± 4.49 mg/L and 34.3 ± 8.97 mg/L, respectively. The west and east had similar mean (± S.E.) total algal biovolumes in 2015 with means of 31.5 ± 9.03 mg/L and 26.7 ± 8.17 mg/L, respectively (Figure 2.37). The center marsh had a much lower mean total algal biovolume of 7.70 ± 5.09 mg/L. The mean total algal biovolume in the west decreased in 2016 to 15.1 ± 5.06 mg/L, increased in the center to 43.9 ± 20.7 mg/L and stayed the same in the east with a mean of 24.4 ± 4.95 mg/L.

There were three nitrogen-fixing algal genera recorded in 2015 and 2016, including *Anabaena*, *Aphanizomenon* and *Oscillatoria*. The percent of nitrogen-fixing algae was compared between the west, center and east sections of Delta Marsh in 2015 and 2016 (Figure 2.38). The west and center sections had the highest percent of nitrogen fixers in 2015 with means of 38.2 ± 8.34 % and 54.0 ± 22.7 %, respectively. The east had the lowest percent of nitrogen fixers with a mean of 19.5 ± 9.85 %. In 2016, the west section had the highest mean percent of nitrogen fixers of 42.2 ± 10.1 %, while center and east had means of 6.29 ± 4.14 % and 12.8 ± 4.49 %, respectively. The percent of nitrogen fixers remained similar in 2015 and 2016 for the west and east section, excluding center marsh (Figure 2.38).

The percentage of nitrogen-fixing algae in each site was compared to the N:P ratio (Figure 2.39). The percent of nitrogen fixing algae and DIN:TDP appear to have a negative curvilinear relationship. The relationship may not be well defined because the DIN:TDP was greater than 16 once during the 2015 and 2016 field seasons. As nitrogen

Figure 2.37. Spatial variation of mean total algal biovolume (A) and mean cyanobacteria biovolume (B) (mg/L) (±S.E.) in the west (N=6 and 8 in 2015 and 2016), center (N=3 and 4 in 2015 and 2015) and eastern (N=12 and 16 in 2015 and 2016) sections of Delta Marsh in 2015 and 2016.

(A)



(B)

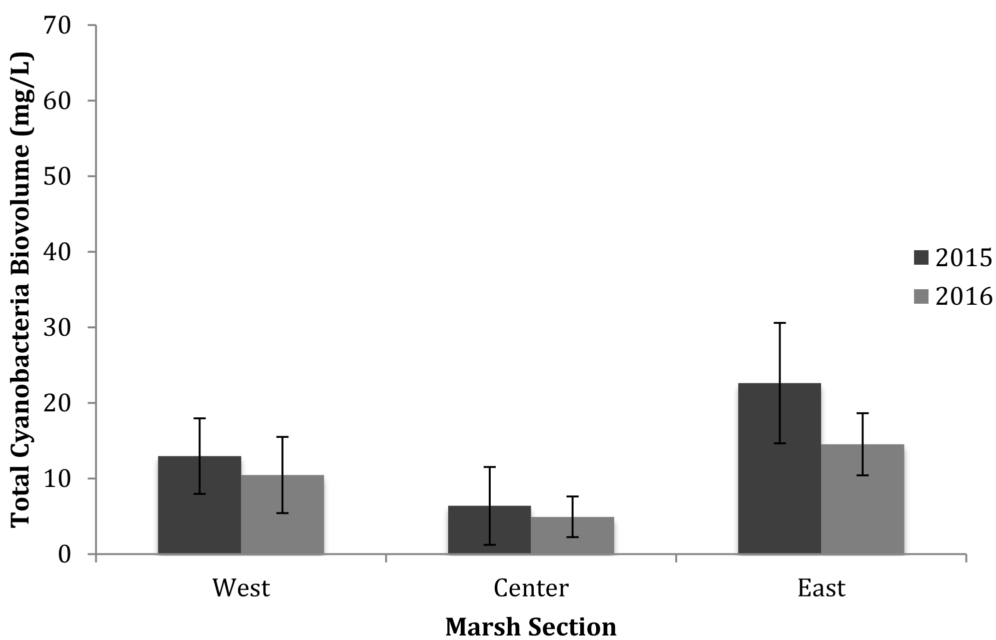


Figure 2.38. Spatial variation of the mean percent of nitrogen-fixing algae of total algal biovolume (%)(±S.E.) in the west (N=6 and 8 in 2015 and 2016), center (N=3 and 4 in 2015 and 2015) and eastern (N=12 and 16 in 2015 and 2016) sections of Delta Marsh in 2015 and 2016.

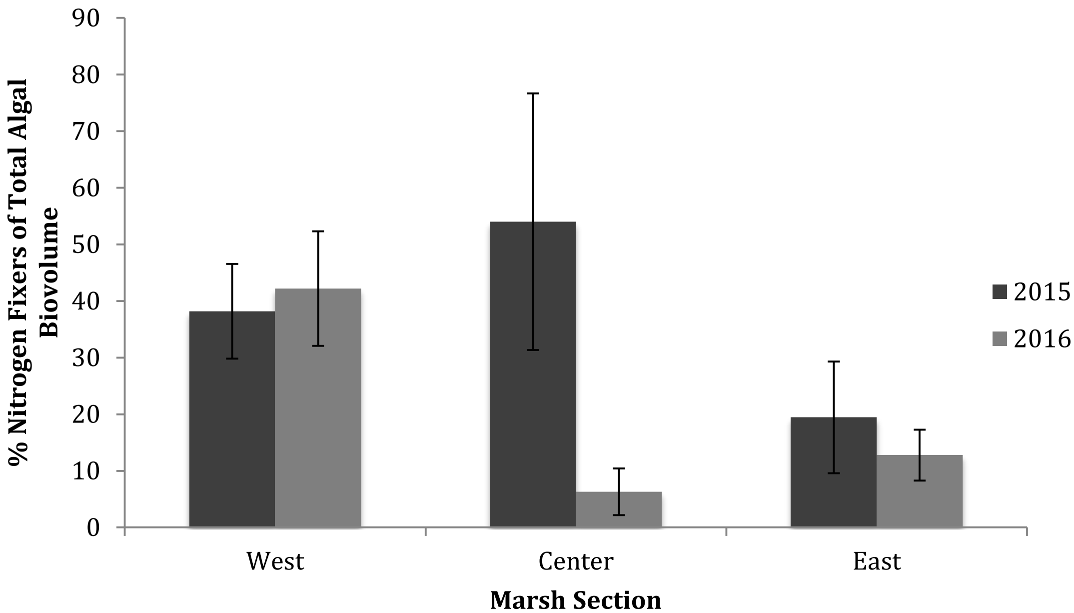
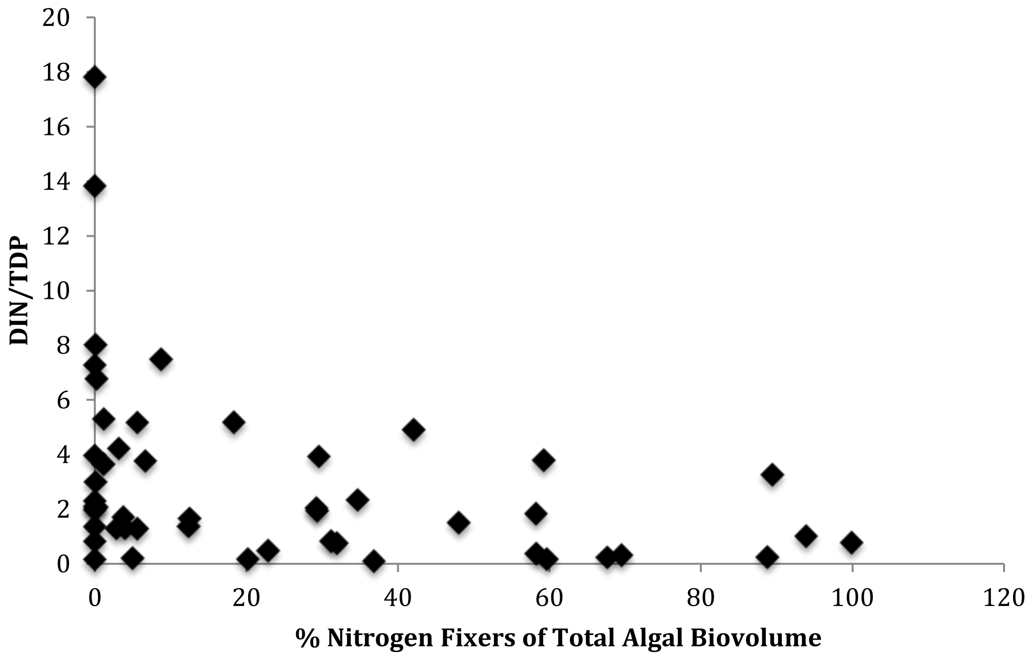


Figure 2.39. The relationship between percent of nitrogen fixing algae and DIN/TDP in seven open-water sites in Delta Marsh, Manitoba. Sites include Mid Big Lake, Weedy Bay, NW Eaglenest, Mid Cadham, Mid Simpson, Mid Waterhen and Mid Clandeboye. Samples were collected seven times between 2015 and 2016.



became increasingly limited (low DIN:TDP) the percentage of nitrogen-fixing algae increased.

2.4 Discussion

Spatial and temporal variation in phytoplankton chlorophyll-a concentrations in Delta Marsh were driven by temperature, high ISS, low euphotic zone depth, high nutrient availability, and latitude/longitude. Algal class dominance was influenced by the ratio of N to P, shift in temperature over the season, water clarity influenced by wind events, and spatial location, likely influencing exposure to wind and nutrient loading, driving blooms. Based on the analyses, it appeared that algal blooms were most predicted by meteorological conditions (wind, temperature and precipitation) and nutrient availability from land (Appendix B), water and sediments. Wind increases resuspension of sediment, liberating nutrients to the overlying water column that drive production of phytoplankton. Higher water temperature increases production of phytoplankton, mainly of cyanobacteria during July and August. Precipitation increases the volume of nutrient runoff (Appendix B), ultimately increasing nutrient concentration in the water column, leading to phytoplankton blooms.

Water temperature is one of the variables that drive cyanobacteria blooms in Delta Marsh. Other studies have shown that, as water temperature increases, algal dominance shifts from diatoms to cyanobacteria (Konopka and Brock 1978). This same sequence was observed across Delta Marsh. Diatoms were dominant in the early season of 2015 and 2016, and as temperature increased in late June and early July, dominance shifted to cyanobacteria where they represented 40% to 99% of total algal biovolume. In addition to temperature, some algal classes such as cyanophyceae are able to thrive when light and nutrients are limiting for most other algal groups (Reynolds et al. 1987; Scheffer et al. 1997; Michalak et al. 2013).

I found that high ISS in Delta Marsh decreases light penetration and drives blooms dominated by cyanobacteria, increasing concentration of phytoplankton chlorophyll-a in the marsh water column. I hypothesized that increased light availability would drive phytoplankton production in Delta Marsh; although my results indicate that phytoplankton chlorophyll-a concentration was highly positively correlated to ISS concentration, indicating limited light did not inhibit algal production. High ISS concentrations is likely attributed to the shallow nature of the marsh, which along with wind events (>25 kph), wave action and presence of common carp, leads to suspension of sediment particles and benthic algae, resulting in reduction of light penetration, expected to cause competition between phytoplankton for light. Cyanophyceae was the most common algal class observed in Delta Marsh, ranging from 40 to 99 % of total algal biovolume in July and August. This type of algae can outcompete other species under low-light conditions (Scheffer et al. 1997). Some cyanobacteria are able to regulate their buoyancy with gas vacuoles that move them up and down the water column until ideal light is reached (Reynolds et al. 1987); this characteristic is very favorable in ecosystems with low light availability, such as Delta Marsh. They also can respond quickly during periods of increased sediment loading. This phenomenon may be observed in Delta Marsh from increased sediment and nutrient loads from the watershed, presence of common carp and wind events causing re-suspension. Previous research at Delta Marsh found that the presence of common carp increases water column suspended sediment and nutrient availability, thus increasing phytoplankton biomass (Badiou 2005). Additionally, Hnatiuk (2006) and Hertam (2010) both found that the increased density of common carp and pond connectivity at Delta Marsh increased suspended sediment, turbidity, nutrient concentration, and periphyton and phytoplankton biomass. Similar relationships were observed here in that phytoplankton chlorophyll-a concentrations and ISS were highly correlated. The increase of suspended sediment may also increase nutrient resuspension, driving algal production in the water column. Phytoplankton chlorophyll-a and ISS were positively associated throughout the field season likely due to cyanobacteria dominating during periods of high turbidity. In addition, high ISS was often associated with increased nutrient concentration in Delta Marsh.

Nutrient availability is another important parameter controlling phytoplankton production in Delta Marsh. Results from this research indicate that nutrient availability, mostly of TN, was the most significant variable in predicting phytoplankton chlorophyll-a concentration. This was observed with the strong relationship between TN and phytoplankton chlorophyll-a concentrations in the PCA, and the increase in the correlation coefficient (r) when TN and TP were added to the multiple stepwise regression models. Previous research conducted at Delta Marsh in the mid 1990's and early 2000's found that benthic algal production was most stimulated by N enrichment due to the N-limitation in the Delta Marsh water column (McDougal 2001; Bortoluzzi 2013). Nitrogen and phosphorus are noted to be two main drivers of algal growth and production in aquatic ecosystems (Anderson et al. 2002). The negative curvilinear relationship between percent of nitrogen-fixing cyanobacteria and DIN:TDP indicates there was a shift in algal taxonomic composition when nitrogen became limiting. In Delta Marsh, the TN:TP ratio was often above 16 with a mean molar ratio of 29, indicating the marsh is replete with total nitrogen and deficient in total phosphorus, and the DIN:TDP ratio was often below 16 with a mean molar ratio of 2.9, indicating DIN is limiting, resulting in dominance of cyanobacteria blooms. In Delta Marsh, DIN appears to limit algal production as dissolved inorganic forms of nutrients are considered bioavailable. Whereas TN or TP can include many forms of the nutrient and some forms must undergo transformation to become bioavailable for algal production (Carlson and Simpson 1996). When nitrogen is limiting, nitrogen-fixing cyanobacteria flourish and can outcompete other algal species within Delta Marsh, similar to what was observed in Lake Erie in 2011 (Michalak et al. 2013). The spatial variation of nitrogen fixers in Delta Marsh may be reflective of nutrient loading and biogeochemical processes. DIN was limited across the marsh in both years of this study, possibly causing nitrogen fixation to occur by nitrogen-fixing cyanobacteria. This process increases the nitrogen concentration within the algal communities, thus increasing the TN in the water column. The high correlation between phytoplankton chlorophyll-a concentrations and TN in Delta Marsh may be reflecting the nitrogen fixation process by cyanobacteria. This phenomenon was discussed by Kosten et al. (2012), they mentioned that it is difficult to determine the true relationships between nutrients and phytoplankton due to the nutrients contained within phytoplankton cells. The nutrient analysis in the water column may include the nutrients present within the phytoplankton community. The strong relationship between TN and phytoplankton chlorophyll-a concentration in this research is likely resulting from the nitrogen within the algal cells. However, nutrient availability and eutrophic conditions do enhance algal production. Similar results were found the research performed by Kosten et al. (2012), it was found that the percentage of cyanobacteria increased with eutrophic conditions and warm water temperatures in shallow lakes in South America and Europe.

Bortoluzzi (2015) studied algal nutrient limitation in Delta Marsh with the use of nutrient diffusion substrata (NDS). NDS vials were treated with different nutrient enhancements including N, P, N and P, and a control, which were dispersed across the marsh to determine which nutrients limit periphyton growth. The findings indicated that P alone caused no stimulation of periphyton growth and that N enhancement caused the most stimulation. She found that 71 % of periphyton samples were limited by N. Similar to previous research, my findings indicate that the bioavailable N is limited in the Delta Marsh water column, and phytoplankton chlorophyll-a concentrations are highly correlated with TN concentration. Due to the limited N for primary production, when the water column is enhanced with N compounds, it is expected that phytoplankton biomass will flourish. Nutrient availability and ratios fluctuate temporally and spatially in Delta Marsh possibly due to changes in nutrient loading from the watershed that is dominated by cropland (76-96%) (Appendix B), use of the Portage Diversion (Aminian 2015), connection to Lake Manitoba, wind events causing resuspension of sediments, common carp presence, biogeochemical processes such as nitrogen and phosphorus cycling, and cover of submersed vegetation. The temporal and spatial fluctuation of nutrients may influence the size, duration and type of algae present within Delta Marsh.

Additionally, latitude was an important predictor of phytoplankton chlorophyll-a concentrations in the multiple stepwise regression models, and it was correlated negatively with phytoplankton chlorophyll-a concentrations in every principal component analysis. Higher concentrations of phytoplankton chlorophyll-a were found in southern bays of Delta Marsh relative to the northern bays near Lake Manitoba. This may suggest that algal production increases when closer to sources of nutrient input from the watershed (Appendix B), in areas with increased wind exposure and with decreased lake connectivity. Bortoluzzi (2013) found that algal production and nutrient availability in Delta Marsh increased with distance from Lake Manitoba, due to the dilution and flushing effects on nutrient concentration from lake connectivity.

Due to the differences in environmental conditions between the two years of this research it was difficult to interpret some of the data and determine the causes of algal production in Delta Marsh. In 2015, the Portage Diversion ran in early April and there was partial carp exclusion to the marsh. In 2016, the Portage Diversion did not run and there was complete carp exclusion to the marsh. As discussed, the connectivity to the lake causes fluctuation in water quality and chemistry due to the dilution and flushing effect (Bortoluzzi 2013). Additionally, the presence of common carp alters the submersed vegetation, water quality, nutrient and sediment resuspension and algal production in Delta Marsh (Badiou 2005; Hnatiuk 2006; Hertam 2010). These combined effects were potentially highly influential on water quality and phytoplankton biomass in Delta Marsh during the years of this study and historically. As mentioned in Section 1.4.3, water quality monitoring has been ongoing since the late 1990’s. Variables consistently monitored included Secchi depth, turbidity, euphotic zone depth, TSS, and chlorophyll-a and pheophytin concentration (Table 1.1). Water clarity decreased during the two years of this research, there was a decline in Secchi depth and euphotic zone depth, and an increase in turbidity and TSS (Table 2.2). This may be attributed to the complete exclusion of carp in 2016. Carp removal caused a response in submersed vegetation and increased light availability. This, paired with eutrophic conditions enhanced phytoplankton algal production, which then decreased light availability, recorded as

Table 2.2. Historical means and ranges for water quality variables analyzed in 45 sites across Delta Marsh during the open water season in 1998/99 and 2009-2016, excluding 2011 due to a flood that inhibited sampling. This research was conducted in 2015 and 2016. Unpublished data provided by Gordon Goldsborough (personal communication). (This table is an extension of Table 1.1).

Variable	1998–99	2009	2010	2012	2013	2014	2015	2016
Secchi Depth (cm)	35 (12–135)	49 (12–160)	40 (9–181)	50 (9–172)	62 (15–142)	80 (14–183)	47 (10–143)	40 (8-154)
Turbidity (NTU)	21 (1–77)	29 (3–174)	34 (1–188)	32 (1–173)	20 (2–280)	13 (1–139)	29 (2–146)	27 (0.9-258)
Euphotic Zone Depth (cm)	–	–	–	151 (28–472)	162 (22–601)	226 (44–941)	135 (35–496)	130 (14-720)
Total Suspended Solids (mg/L)	50 (1–314)	29 (0–130)	31 (0–165)	31 (0–223)	26 (0–214)	18 (0–166)	29 (0–199)	34 (0-457)
Total Chlorophyll-a and Pheophytin (µg/L)	155 (7–1,067)	66 (3–328)	46 (0.2–261)	128 (8–792)	98 (5–883)	35 (1–582)	87 (3–503)	75 (0.2-1120)

Secchi depth and euphotic zone depth (Table 2.2). There was an overall increase in chlorophyll-a and pheophytin concentrations in 2015 and 2016, with mean concentrations of 87 µg/L and 75 µg/L, respectively. Mean 2015 total chlorophyll-a and pheophytin concentration increased from 2014, from 35 µg/L to 87 µg/L. There was a flood in early May of 2015 which resulted in carp entry to the marsh, possibly causing increased sediment and nutrient resuspension resulting in phytoplankton blooms. Additionally, there was an extreme flood in July of 2014, which caused a breach in the failsafe of the Portage Diversion causing entry of water into the west side of the marsh. As mentioned, the water from the Portage Diversion can account up to 87% of TP input into Lake Manitoba annually (Nicholson 2012). The Diversion water is replete in nutrients and may have caused increased nutrient availability in the water column for algal growth in 2015, causing the increase in mean chlorophyll-a and pheophytin concentration and the decreased water clarity. Complete carp exclusion was successful in 2016, and there was an observed decrease in phytoplankton chlorophyll-a and pheophytin concentration, possibly due to lower nutrient availability arising from the decreased carp density (Table 2.2). There were no drastic differences between 2015 and 2016 after complete carp exclusion. This could have been influenced by the increased precipitation and nutrient loading to the marsh in 2016 (Appendix B), changes to the physical structure of the marsh (ex. depth), wind velocity and exposure, and there may be a lag in the water quality response after carp exclusion (Hertam 2010) for reestablishment of submersed vegetation.

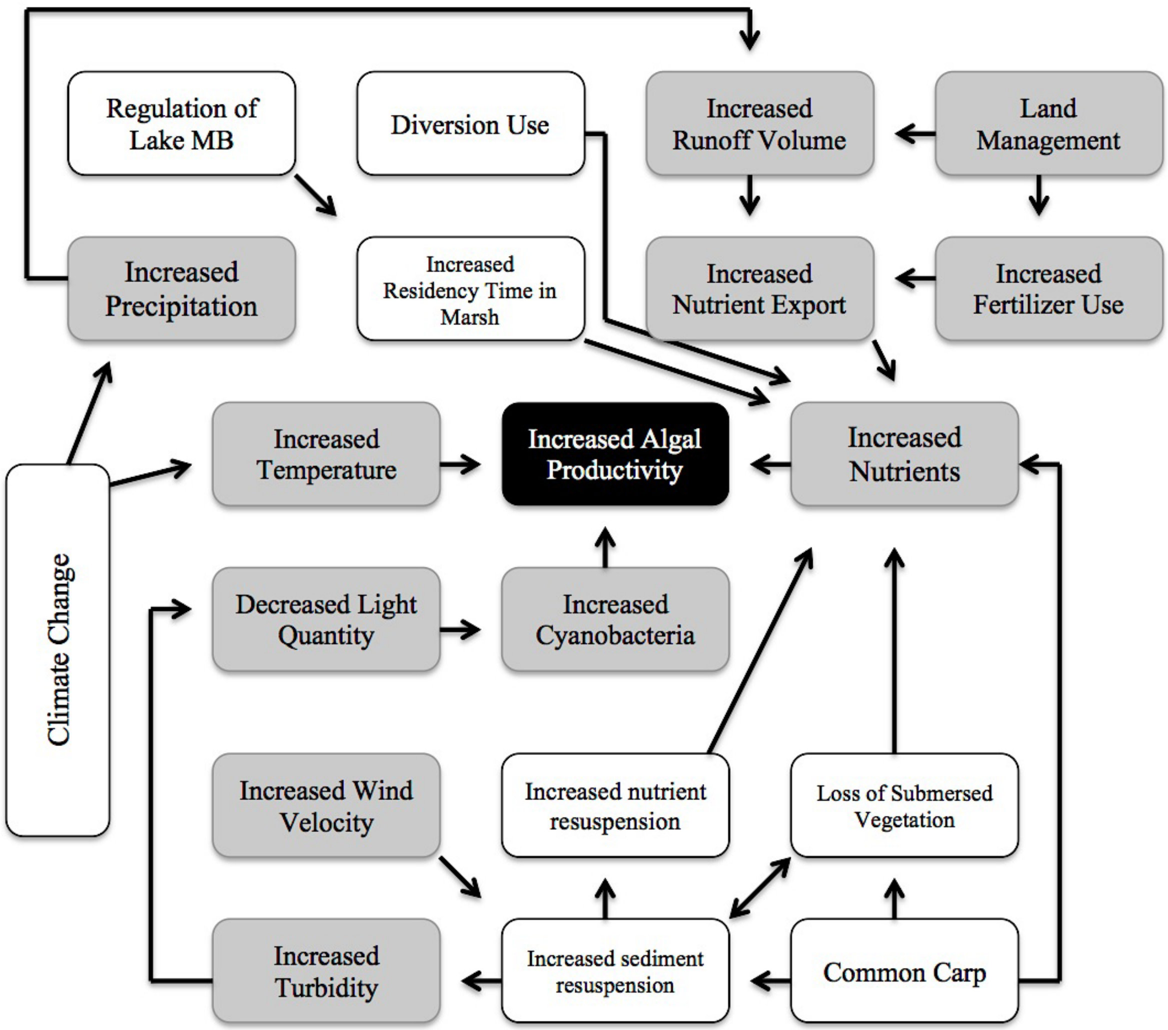
Phytoplankton blooms were spatially variable based on chemical and physical characteristics of each site in Delta Marsh. I speculated that phytoplankton chlorophyll-a concentrations would vary between the west, center and east sections of Delta Marsh, although results from this research have indicated that phytoplankton blooms may be more related to the specific characteristics in each site, such as exposure to wind or nutrient runoff. The highest phytoplankton chlorophyll-a concentrations were found in exposed sites with high ISS concentrations, temperature and nutrient availability.

2.6 Synthesis

My thesis demonstrates that water quality in Delta Marsh was controlled by meteorological factors such as wind and temperature, and by nutrient supply from land (Appendix B), water and sediments. Wind increases the sediment and nutrient resuspension in the water column, causing increased suspended sediment and nutrient availability for phytoplankton growth, mainly of cyanobacteria (Scheffer et al. 1997; Anderson et al 2002). Precipitation drives nutrient loading to the marsh, increasing nutrient availability in the water column. Temperature drives all forms of primary production, although is more favourable for cyanobacteria (Konopka and Brock 1978). A conceptual diagram shows how climate, land use management, invasive species and Lake Manitoba can regulate algal productivity in Delta Marsh, Manitoba, and what aspects of those relationships were addressed by my research (Figure 2.40).

Wind exposure at any given site in the marsh increases the suspended sediments and nutrients at that site, resulting in a high-nutrient, turbid water column in which algae tolerant of low-light environments such as cyanobacteria may thrive. Increases in surface water temperature throughout the field season, can cause shifts to the algal communities, from diatoms to cyanobacteria (Konopka and Brock 1978). As shown throughout this

Figure. 2.40 Conceptual diagram of factors considered to be influences the increase of algal productivity in Delta Marsh, Manitoba. Grey boxes represent those discussed throughout this thesis.



thesis, meteorological conditions and nutrient availability were significant predictors of phytoplankton production in the marsh. Intense precipitation events can increase nutrient loading from the surrounding agricultural watershed, leading to increased nutrient availability and possibly algal blooms in the marsh (Appendix B). This input varies over time. For example, my results show that spring snowmelt, spring and summer precipitation were higher in 2016 than 2015 and normal annual precipitation, resulting in four to six times higher nutrient loading in 2016 than 2015 (Appendix B). Although precipitation and temperature cannot be managed, alteration in landscape management practices can alter the impact of precipitation on nutrient runoff (Figure 2.40), possibly by implementing management plans to reduce total water yield from the watershed.

I found that water from the western and central watersheds entering Delta Marsh had higher loads of TN, DIN, TP and TDP than that from the eastern watershed due to the spatial gradient of land use within the watershed. This spatial difference was likely due to the fact that land use was dominated by cropland in the western watershed whereas the eastern watershed, while still dominated by cropland, had large areas of grassland and shrubland. These differences would result in different fertilizer use, as well as differences in water yield. This combination causes a multiplicative relationship between percent crop cover and nutrient export to Delta Marsh (Appendix B). Donahue (2013) found similar results showing patterns of increased nutrient loading with increased agricultural intensity. He mentions that this relationship may be a function of fertilizer application, grazing, or manure application (Donahue 2013). The nature of land use, coupled with sandy soils in the western watershed area (Experimental Farm Service 1958), should tend to drive increased N and P loads to the marsh ecosystem (Appendix B). Interestingly, although the nutrient exports to the west and center marsh were higher than the east, there was no significant difference in phytoplankton chlorophyll-a concentrations across the marsh. Additionally, there was no difference in mean total algal biovolume between the west, center and eastern bays, although the results may be influenced by the small sample size of seven. It was hypothesized that sections of the marsh receiving the highest nutrient export would yield increased phytoplankton chlorophyll-a concentrations because of the N-limitation in Delta Marsh and the strong correlation between chlorophyll-a and nutrient concentrations. However, this was not supported by the results, there was no spatial variation in phytoplankton chlorophyll-a production across the marsh in either year of this study.

Nutrients from the surrounding land may remain in the water column of the marsh, be sequestered in its sediments, or consumed by plants and animals. A companion study to my project addressed the chemistry and sorption capacity of sediments across Delta Marsh, by collecting sediment cores from various marsh sites and analyzing discrete sediment strata for their phosphorus concentration and sorptive capacity for phosphorus. Chris Hope (Department of Biological Sciences, University of Manitoba) has collected 5-cm and 50-cm sediment cores from seven open-water, emergent zone, and wet meadow sites across Delta Marsh during the summers of 2015 and 2016. The sites were Mid Big Lake, Canvasback Bay, Weedy Bay, Eaglenest Bay, Cadham Bay, Lyttle Bay, and Waterhen Bay (Figure 2.1).

I obtained preliminary data from Chris to investigate the possible relationship between water chemistry, measured in my study, with his measurements of sediment nitrogen and phosphorus concentrations. The water quality variables included mean chlorophyll-a (µg/L), TN (mg/L), DIN (mg/L), TP (mg/L), and TDP (mg/L). Water chemistry variables from Mid Simpson, site 46, were used for Lyttle Bay (Figure 2.1), as for water chemistry for the latter was not measured. Sediment chemistry variables included TN (mg/kg), TP (mg/kg). A correlation matrix between sediment chemistry and water quality variables (Table 2.3) showed that sediment TN was moderately positively correlated ($r^2 = 0.61$) with phytoplankton chlorophyll-a and TN in the overlying water. There was no relationship between sediment TN and water DIN (r= -0.17), suggesting that DIN in the water column may be arising primarily from atmospheric nitrogen fixation. There was a higher percentage of nitrogen-fixing cyanobacteria in the water column and a low DIN:TDP ratio, indicating bioavailable nitrogen was limited. The DIN in the water column was not associated with the TN in the sediment, also indicating the nitrogen was likely being fixed from the atmosphere. Sediment TP was strongly positively correlated with phytoplankton chlorophyll-a (r= 0.73) and P in the water column (r=0.50). Sediment chemistry had a stronger correlation with chlorophyll-a concentrations than water chemistry, sediment TN and TP with r-values of 0.61 and 0.73, respectively (Table 2.3). This may indicate that sediment nutrients may influence phytoplankton production more than water chemistry. The Delta Marsh sediment chemistry results in the ongoing project by Hope may help determine why there was no spatial difference in phytoplankton chlorophyll-a across the marsh. For example, there may be spatial variation in sediment nutrient sorption capacity, altering the nutrient availability in the water column for phytoplankton production.

The western and central sections of the marsh have higher cover of vegetation and the eastern section of the marsh has higher area of open water (Figure 1.1 & 2.1). The

Table 2.3 Correlation matrix of sediment chemistry (Hope, personal communication) and water quality variables of seven open-water sites (Big Lake, Canvasback Bay, Weedy Bay, Eaglenest, Cadham Bay, Lyttle Bay and Waterhen Bay) at Delta Marsh, Manitoba during 2015 and 2016. Water quality variables including chlorophyll-a concentrations (Chl-a), TN, DIN, TP, and TDP concentrations were means for 2015 and 2016.

	Chl-a	**TN Water**	**DIN Water**	**TP Water**	**TDP Water**	**TN Sediment**	**TP Sediment**
Chl-a	1.00	0.57	0.45	0.36	0.20	0.61	0.73
TN Water		1.00	0.18	0.65	0.50	0.68	0.19
DIN Water			1.00	-0.32	-0.52	-0.17	-0.08
TP Water				1.00	0.97	0.46	0.50
TDP Water					1.00	0.44	0.50
TN Sediment						1.00	0.53
TP Sediment							1.00

increased nutrient loading to the west and center sections may result in different forms of primary production, such as emergent vegetation, submersed vegetation, epiphytic algae or benthic algae. McDougal (2001) determined that 56-77 % of all algae in Delta Marsh was benthic in the late 1990's, perhaps there was higher cover of benthic algae in the western section that outcompete phytoplankton for available nutrients. Additionally, the eastern section of the marsh has greater open water area, causing an increased fetch distance for wind disturbance. This increases the wind influence of water column mixing and sediment and nutrient resuspension (So et al. 2013), resulting in increased nutrient availability in the water column and a turbid water state, ideal conditions for cyanobacterial blooms (Scheffer et al. 1997). Additionally nutrients can be up-taken by autotrophs, microbacteria, biogeochemical processes such as denitrification, and bound in sediment particles (Hertam 2010). The variability in sediment chemistry, different forms of primary production, nutrient loading, wind exposure, open water area and biogeochemical processes may have reduced the spatial variation in phytoplankton chlorophyll-a production in Delta Marsh during the years of this study. Although the nutrient loading does not appear to have a direct influence on phytoplankton production, it may increase the nutrient concentration in the sediments and nutrient availability in the water column (Verhoeven et al. 2006).

Common carp density may have had a large influence on phytoplankton production during the two years of this research. Full exclusion of carp from Delta Marsh was largely achieved in 2016 and extensive submersed vegetation was observed across the marsh that summer. Submersed vegetation cover may reduce nutrient availability from the water column and sediments (Figure 2.40), and decrease influence of wind and wave action, indirectly decreasing sediment and nutrient resuspension (Sullivan et al. 2014) and ultimately decreasing phytoplankton blooms and increasing other types of algae (benthic/epiphytic). However, during years of this study, the presence of submersed vegetation increased light availability, paired with the continuing eutrophic water column causes ideal conditions for phytoplankton production (Table 2.2).

The Portage Diversion contributes excessive nutrient loading to Lake Manitoba during years of use (Page 2011; Nicholson 2012). When the failsafe is breached on the western section of the marsh, the Portage Diversion can introduce excessive nutrient and sediment loads to the water column, possibly driving phytoplankton production (Figure 2.40) (Aminian 2015) Most of the water arising from the Portage Diversion enters through Delta Channel into Cadham Bay (Figure 1.1), however the connectivity to Lake Manitoba at Clandeboye Channel controls most of the hydrology in the marsh.

The results from my research have shown there are strong relationships between algal abundance, nutrient availability, temperature and suspended sediments (Figure 2.40). The first hypothesis of my project, which stated that sites with increased light and nutrient availability would yield higher phytoplankton chlorophyll-a concentrations, was supported in part by the results of this study. I found that phytoplankton blooms were correlated with high nutrient availability. TN concentrations were the most significant variable in models predicting phytoplankton chlorophyll-a concentrations. TN is likely auto-correlated with phytoplankton chlorophyll-a as a result of enhanced nitrogen fixation due to the limited bioavailable N in the water column of Delta Marsh. However, reduced light availability did not impede the occurrence of phytoplankton blooms, in fact it resulted in increased cyanobacteria biomass. Further investigation on the relationship between light availability and algal class dominance in Delta Marsh should be monitored throughout carp exclusion, as suspended sediment is expected to decrease in the water column and submersed vegetation cover to increase.

My second hypothesis, that land use dominated with agricultural activities would yield increased nutrient loading to the marsh, was supported (Appendix B). Seasonal nutrient export recorded in 2015 and 2016 were higher in sub-watershed areas that had higher cover of cropland and water yield. Additionally, nutrient export was lower in sub-watershed areas that had natural land cover, such as shrubland and grasslands. The results displayed a spatial gradient of high nutrient export in the west marsh, to lower nutrient export in the east.

My third hypothesis, that the sections of the marsh receiving the highest annual nutrient export would yield more eutrophic characteristics, was not supported. Phytoplankton chlorophyll-a concentrations did not vary between the western, center and eastern portions of the marsh. The results from the research discussed in Appendix B indicated that there was spatial variation of nutrient loading likely caused by the land use gradient across the watershed. The western and central portions of the marsh receive higher nutrient loading than the eastern portion, however there was no difference in phytoplankton production possibly due to physical differences in each marsh section. With the reduction of carp presence and an increase of submersed vegetation, I expect to see a response in different forms of algal blooms across the marsh, such as an increase of epiphytic and benthic algal biomass. Further investigation on other forms of primary productivity and the biogeochemical processes occurring in the sediment of the marsh and the tributaries may give insight into this result.

2.6 Recommendations

2.6.1 Future Research

1. My findings that runoff events can contribute significant nutrient loading to the marsh (Appendix B) support the need for ongoing flow and nutrient monitoring, especially during the spring snowmelt period. Further data on the variability of nutrient export and water yield across the watershed will give insight to how landscape management practices can be informed and should be explored further in detail to inform such management decisions.

2. My findings indicate that there is a strong relationship between land use, water yield and nutrient loading to Delta Marsh, which could potentially increase nutrient availability and enhance phytoplankton production. I predicted that increased nutrient loading would drive phytoplankton chlorophyll-a concentrations, however, there was no significant differences in chlorophyll-a concentrations between the western, central and eastern portions of the marsh. Sediment chemistry data (Hope, personal communication) may reveal variation in sediment sorption capacity or fractionation causing the lack of phytoplankton spatial variation. Further research into the relationships between land use, nutrient runoff, water quality and sediment biogeochemical processes will clarify the causes of marsh degradation and stimulation of algal production.

3. The results from this research indicate that complete carp exclusion may reduce sediment and nutrient resuspension, and thereby reduce phytoplankton blooms. Although phytoplankton blooms may be reduced, other types of algae may flourish, such as metaphyton, epiphyton or benthic algae. A reduction of carp presence and an increase in

submersed vegetation may cause a shift in light and nutrient availability in the water column causing changes in algal competition (McDougal 2001). Further investigation of the relationship between water quality, carp exclusion and submersed vegetation cover should be performed after multiple years of complete exclusion, to observe how algal types may shift under varying conditions. This analysis will combine two major portions of the Restoring the Tradition project, carp exclusion and water quality, and will inform the causes of changes in marsh health. Water quality sampling should continue by monitoring the 45 sites within one to two days across Delta Marsh every three to four weeks. The short sampling period will reduce temporal variation of the dataset and allow analysis of the spatial differences of biological, chemical and physical variables across the marsh.

2.6.2 Implications for management

My thesis has shown that phytoplankton blooms, water quality, sediment chemistry, and land use around Delta Marsh are inter-related. Continuing the monitoring of phytoplankton chlorophyll-a, water clarity, nutrient concentration, temperature within the marsh, and nutrient export, precipitation and water yield within the watershed, during the management portion (post-2017) of the Restoring the Tradition project will provide a stronger basis for management actions to be taken to reduce nutrient export from the agricultural landscape for the long-term benefit of the marsh ecosystem. Once management is implemented, it is important to understand if these relationships change and if the management is successful in restoring health to Delta Marsh.

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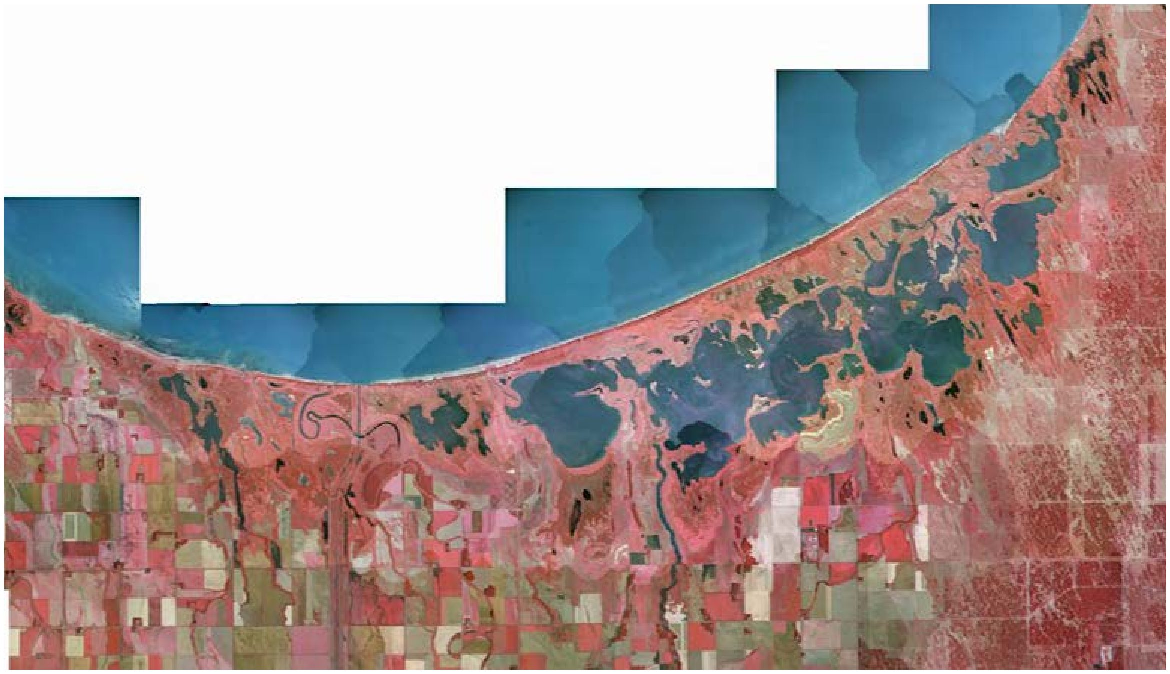
Appendix

Appendix A. False-colour infrared map (1997) of Delta Marsh and its watershed displaying the land use gradient from west to east (Ducks Unlimited Canada and the University of Manitoba).



Appendix B. Nutrient Loading and Land Use

Introduction

The landscape on the south side of Delta Marsh is dominated by agriculture of varying intensities, ranging from natural cover and pastureland on the southeast, to cropland on the southwest. Nutrient loading into the marsh arises from spring runoff, followed by spring and summer precipitation, ground water and through tributaries that empty into the marsh. The variability in land cover and land use across the watershed may cause spatial differences in nutrient loading to the marsh. Nutrient loading will also vary temporally due to seasonal patterns in precipitation. This study focused on the relationship between land use and nutrient loading for the sub-watersheds emptying into the southern edge of Delta Marsh.

The objective of this appendix was to determine if there is a spatial difference in nutrient loading caused by differences in land use within the watershed. I hypothesized that land use dominated (highest percentage of cover) with agricultural activities will yield higher nutrient loading to the marsh compared to areas with lower levels of agricultural intensity. Agricultural land will yield increased nutrient loading because manipulation of land alters the watershed hydrology and fertilizer application increases the concentration of nutrient runoff.

Materials and Methods

Site Description

The Delta Marsh watershed is 20,762 hectares and contains a number of tributaries that deliver surface water runoff to the marsh. Nutrient loading from ten of these tributaries emptying into Delta Marsh was monitored at culverts along Highway 227 and off roads leading from the highway (Figure B.1) (Table B.1). These sub-watersheds of the greater Delta Marsh watershed vary in area and land use. Two tributaries west of the Portage Diversion were sampled, along with eight tributaries to the east (Figure B.1). Watershed boundaries could only be delineated for nine of the ten tributaries and used for calculating nutrient export from various land use types. Tributary E1 could not be delineated, as it is located within the watershed area for E2. Nutrient loading (kg) from E1 in 2015 and 2016 are discussed later in this section. Percent of land use types were calculated for each of the eight sub-watershed area based on the Agriculture and Agri-food Canada Annual Crop Inventory (2015). Sub-watershed areas for E1 and E9 could not be divided into crop types. Crop types and area were identified using satellite images; images were made with multiple sensors during important developmental stages of the crops including reproduction, seed development and senescence. During years of this study, Landsat-8 and RADARSAT-8 were used for optical and radar imagery for determining crop type, with a resolution of 30 m. The percent of accuracy for crop type in Manitoba was 90.3% and 90.0% in 2014 and 2015, respectively. The percent accuracy for non-agricultural land in Manitoba for 2015 was 62.9% (Agriculture and Agri-food Canada 2015). The area monitored for land use types in this area was not ground-truthed, although in the future this may improve accuracy and confidence in land and crop types in the Delta Marsh watershed area.

Land use cover was divided into major categories, including areas of open water, exposed land, urban developed land, shrubland, grassland, wetland, pasture, too wet to be seeded, cropland and forested land. The major land use type was cropland, spatially

Figure B.1. Watershed area (km^2) of ten tributaries located perpendicular to Highway 227 that empty into Delta Marsh, each analyzed for nutrient loading. Sample site locations are noted with a black box. Prepared by Greg Schellenberg, 2015. Tributary E1 could not be delineated, its site locaation is labelled below. (Paired with Table B.1).

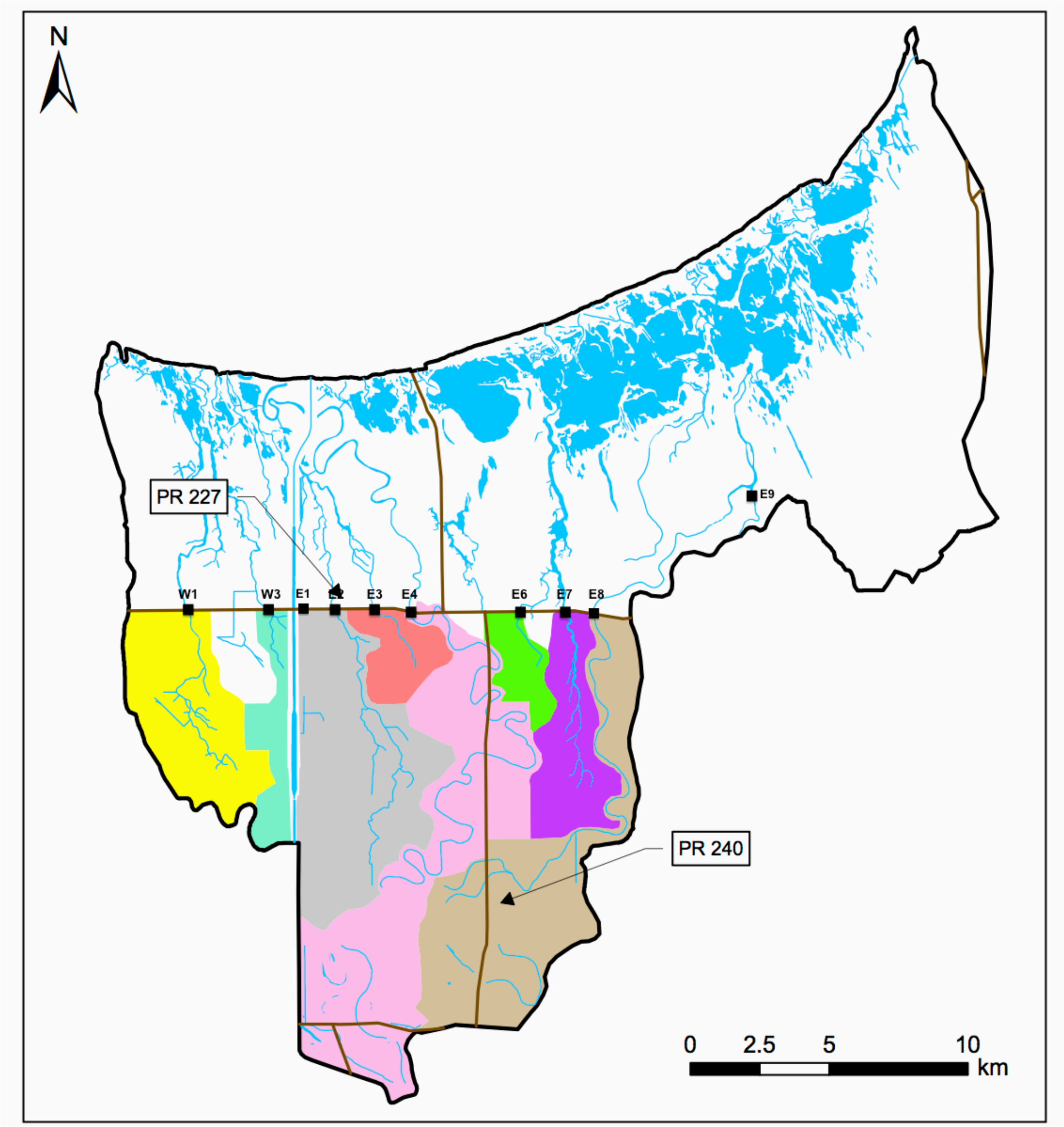


Table B.1 Nutrient loading sampling sites with their designated coordinates. (Paired with Figure 3.1)

Site	Latitude	Longitude
W1	50.063325 N	98.255763 W
W3	50.063233 N	98.232333 W
E1	50.063189 N	98.223343 W
E2	50.062769 N	98.213276 W
E3	50.062809 N	98.201969 W
E4	50.062839 N	98.191891 W
E6	50.062925 N	98.154074 W
E7	50.062894 N	98.143834 W
E8	50.062902 N	98.134644 W
E9	50.081607 N	98.084604 W

ranging between 77 to 96% cover in each of the sub-watersheds in 2014 and 2015. Crop cover was further categorized in crop type, including, barley, oats, rye, winter wheat, spring wheat, corn, canola, flaxseed, sunflower, soybeans, beans and potatoes. The methods for determining crop classes are the same as listed above; ground-truthing for future studies may improve accuracy and confidence in the crop classes identified for the Delta Marsh watershed. Spring wheat, canola, soybeans and beans were the dominant crop types accounting for 22%, 35%, 12% and 19% of land cover across the watershed, respectively.

Nutrient Loading Sampling Protocol

Water flow (m^3/s) and nutrient concentrations of tributaries that empty into the marsh were monitored during the spring snowmelt period and at all significant precipitation events (>10mm) throughout the field season in 2015 and 2016. Spring runoff sampling began on March 18, 2015 and March 14, 2016 and monitoring continued until August 25, 2015 and August 23, 2016. In 2015, flow velocity (m^3/s) at the centre of each culvert was recorded using a FP111 Global Water Flow Probe; the velocity (m/s) was recorded around the culvert and an average velocity was calculated by the probe. These data, along with measurements of water depth and culvert dimensions, were used to calculate the water volume entering the marsh from each of these tributaries. In 2016, flow velocity (m^3/s) was recorded using a HACH FH950 Portable Velocity Meter. Surface water samples were collected from flowing water in the middle of the culvert or stream channel. These samples were analyzed for TN, DIN, TP and TDP concentrations (mg/L) at ALS Environmental Laboratories (Winnipeg), following Standard Methods (2017) (Appendix E).

Nutrient loading (kg) at each tributary was calculated by multiplying flow velocity (L/s) and nutrient concentration (mg/L). The daily nutrient loads (kg) were used to calculate the field seasonal nutrient loading from March 18 to August 26, 2015 and March 14 to August 23, 2016. Data for dates on which flow or nutrient concentrations were not recorded were extrapolated using precipitation data and a linear trend series in Microsoft Excel. Precipitation data were used to indicate when flows began and ended for each tributary. Precipitation data were recorded at the Portage Southport Station, Manitoba (Environment Canada, 49.913944 N 98.272118 W). Nutrient export (kg/hectare) was calculated by dividing total nutrient load (kg) by the area (hectare) of each sub-watershed calculated by Schellenberg (Figure B.1). Site E1 was excluded from the analysis, as it is included within the E2 sub-watershed area and cannot be separately delineated.

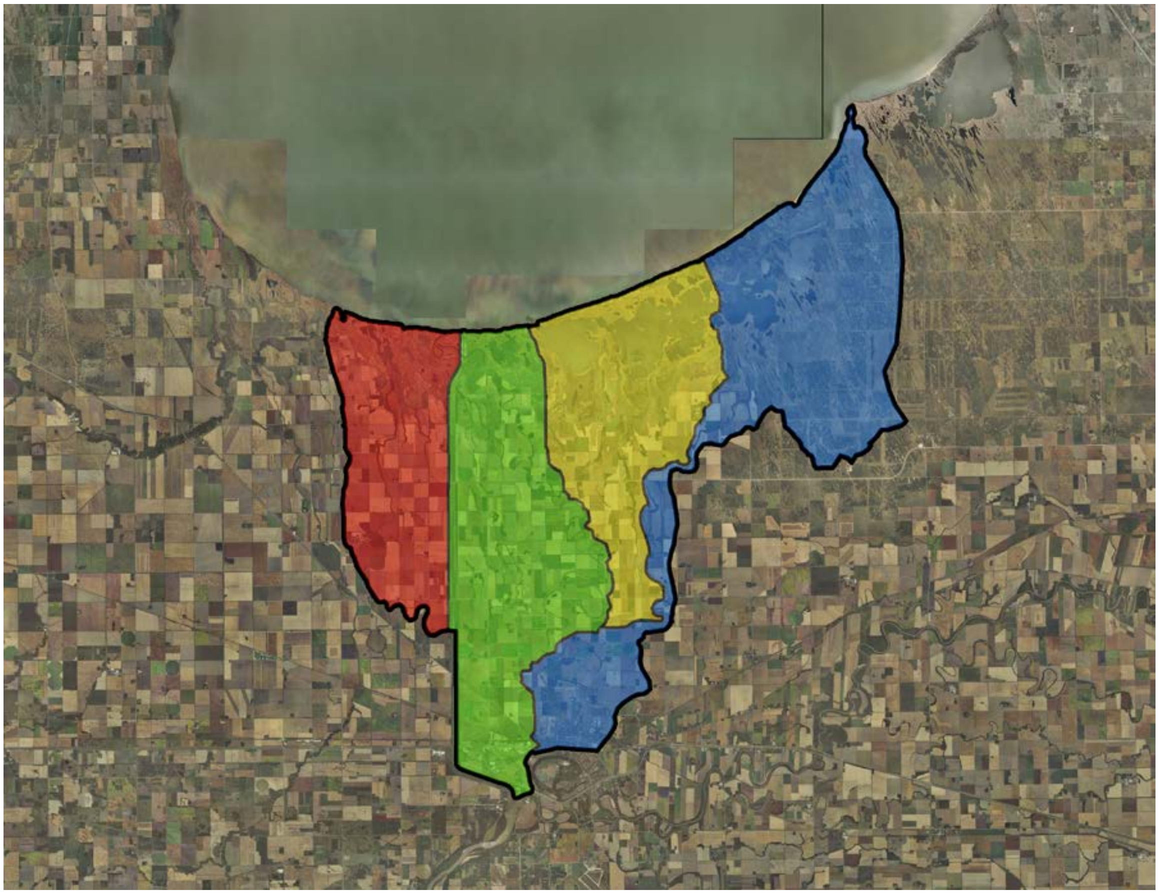
The relationships between land use in 2014 and 2015 were compared with nutrient export for 2015 and 2016, as land use from prior years influences nutrient export the following spring. This analysis did not take into account such variables as tillage and drainage practice, nutrient application type and timing, soil type and other factor influencing nutrient runoff, although some of these can be implied based on the type of crop being grown. Tillage, drainage and soil type influence the hydrology of the watershed and ultimately the water yield, altering nutrient runoff (Correll et al 1992; Howarth et al. 1996; Carpenter et al. 1998; Liu et al. 2008). Nutrient application type and timing influences the concentration of nutrients released in runoff. If nutrient application occurs prior to a precipitation event, much of the nutrient will be lost in the runoff (Correll et al 1992; Zedler 2003).

The marsh was divided into four major units to determine the distribution of nutrient loading per wetland area; West, Center, East 1 and East 2 (Figure B.2). The delineation was based on elevation of the watershed and wetland area and hydrological conditions. The nutrient export was calculated for each watershed area draining into the four sections of the marsh. The nutrient export (kg/hectares) was multiplied by the drainage area to obtain total nutrient load (kg) into each section of the marsh. Total nutrient load (kg) to each section was then divided by wetland area for each subsection (hectare) to determine how the nutrients exported were distributed per wetland area (Figure B.2).

Statistical Analysis

Nutrient export from tributaries emptying into Delta Marsh was compared spatially by observing overall differences of seasonal export for each sub-watershed. Land use and nutrient export were analyzed with a Pearson's Product Moment Correlation Matrix to determine which land use types were correlated with TN and TP export. The strength of the correlation coefficient (r) will be based off the scale discussed in Section 2.2.4. The scale indicates relationships as strong (0.75-1.0), moderate (0.5- 0.75), weak (0.3-0.5) and no relationship (0.0-0.3) (Snedecor and Cochran 1980; Rumsey [accessed May 2017]).

Figure B.2. Map of the drainage areas for the four major sections of Delta Marsh; West (Red), Center (Yellow), East 1 (Green) and East 2 (Blue). Delineation was prepared using ArcMap ® 10.4.

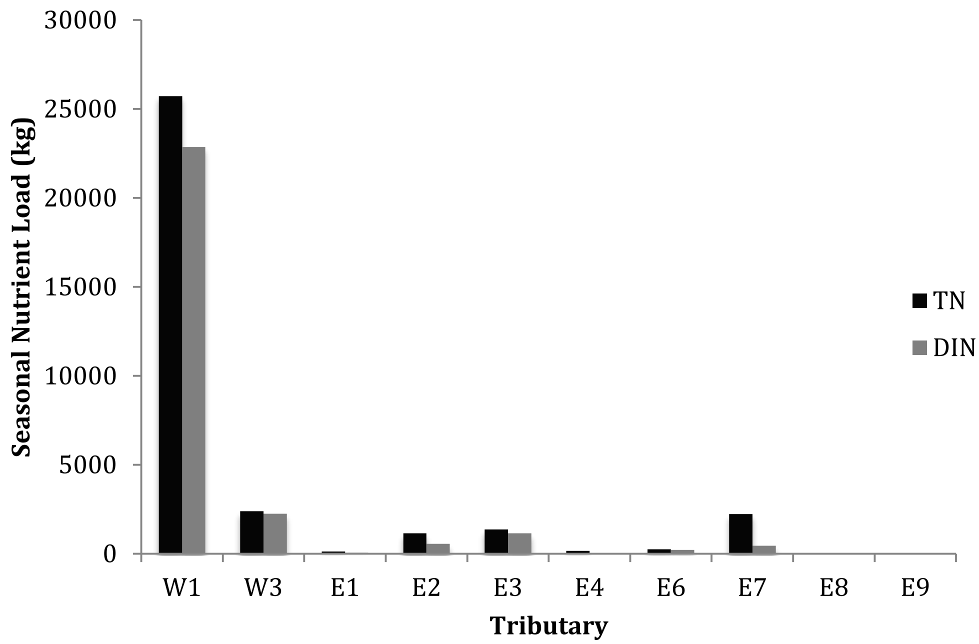


Results

In 2015, tributary W1 had the highest TN load to the marsh of 25,711 kg TN, with 10 to 195 times higher TN (kg) than the other tributaries emptying into the marsh and the highest TP load to the marsh of 2306 kg TP (Figure B.3). W3 (2390 kg TN), E2 (1142 kg TN), E3 (1358 kg TN) and E7 (2231 kg TN) followed W1 with similar TN loads, however they had variable TP loads of 60.4 kg, 678 km, 151 kg and 621 kg in W3, E2, E3 and E7, respectively. Sites E4, E6 and E9 had very little nutrient load to the marsh, with a combined mean of 142 kg TN, 74 kg DIN, 36 kg TP and 31 kg TDP. The combined TN and TP load from all tributaries in 2015 was 33,394 kg TN and 3957 kg TP. The mean for all tributaries recorded in 2015 was 3339 kg TN, 2751 kg DIN, 395 kg TP and 291 kg TDP (Figure B.3). W1 had the highest nutrient export to the marsh, 10.1 kg TN/hectare, 9.00 kg DIN/hectare, 0.91 kg TP/hectare and 0.76 kg TDP/hectare (Figure D.4), approximately six times higher than the mean TN and DIN export (± SE, n=9) of 1.85 ± 0.96 kg TN/hectare and 1.52 ± 0.87 kg DIN/hectare respectively (Figure B.4). Site W3 had the second highest TN and DIN export of 2.70 kg TN/hectare and 2.54 kg DIN/hectare, followed by E3 and E7. TP and TDP loads and exports displayed similar trends to the N compounds. W1 had the highest TP and TDP export of 0.91 kg/hectare and 0.76 kg/hectare respectively, five times higher than the mean of 0.19 kg TP/hectare and 0.15 kg TDP/hectare. E7 had the second highest TP and TDP export of 0.38 kg TP/hectare and 0.21 kg TDP/hectare, followed by E3, E2 and W3. Overall the nutrient export appeared low for all tributaries when compared to site W1 in 2015. The western section of the marsh received the highest nutrient load (kg) for TN and DIN, of 28,101 kg TN and 25,093 kg DIN, followed by Center, East 1 and East 2, with a mean of 8,315 kg

Figure B.3. Seasonal nutrient load (kg) of TN and DIN (A) and TP and TDP (B) in ten tributaries that empty into Delta Marsh, Manitoba, from March 31 to August 28, 2015.

(A)



(B)

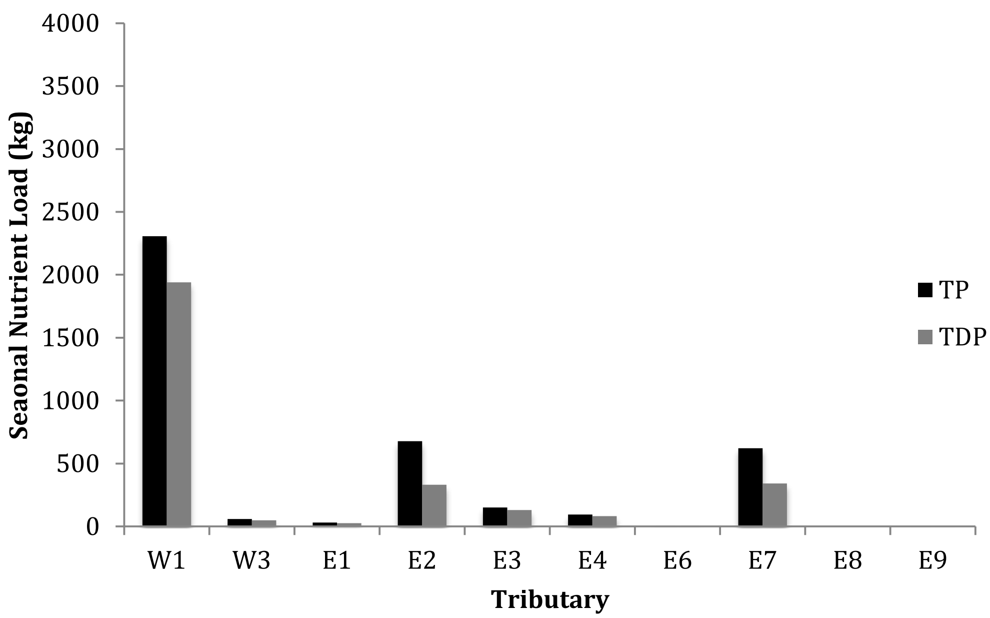
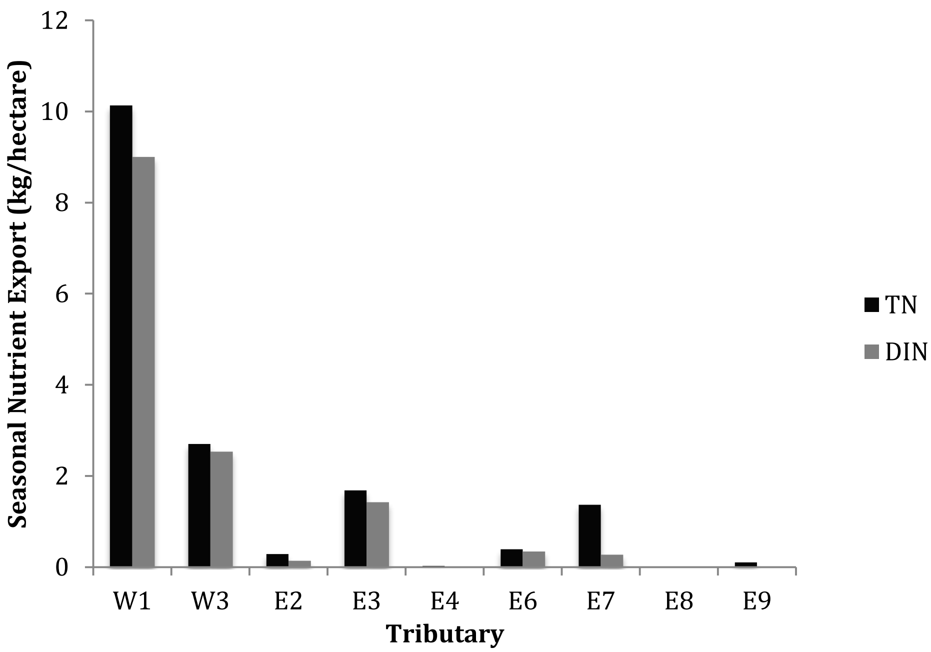
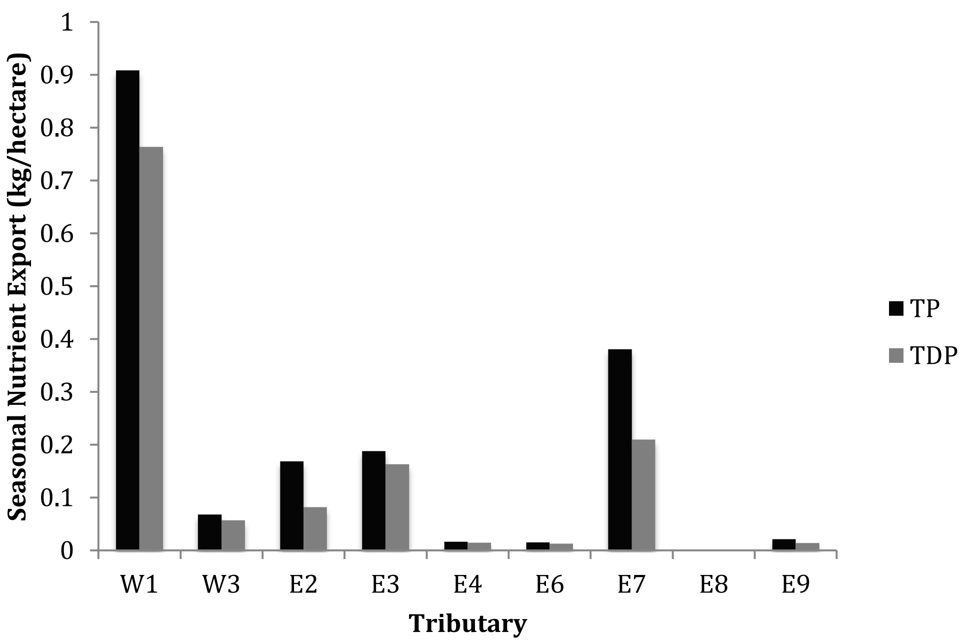


Figure B.4. Seasonal nutrient export (kg/hectare) of TN and DIN (A), TP and TDP (B) for nine tributaries that emptied into Delta Marsh, Manitoba from March 31 to August 28, 2015. The E1 site was removed because its area (hectares) could not be delineated.

(A)



(B)

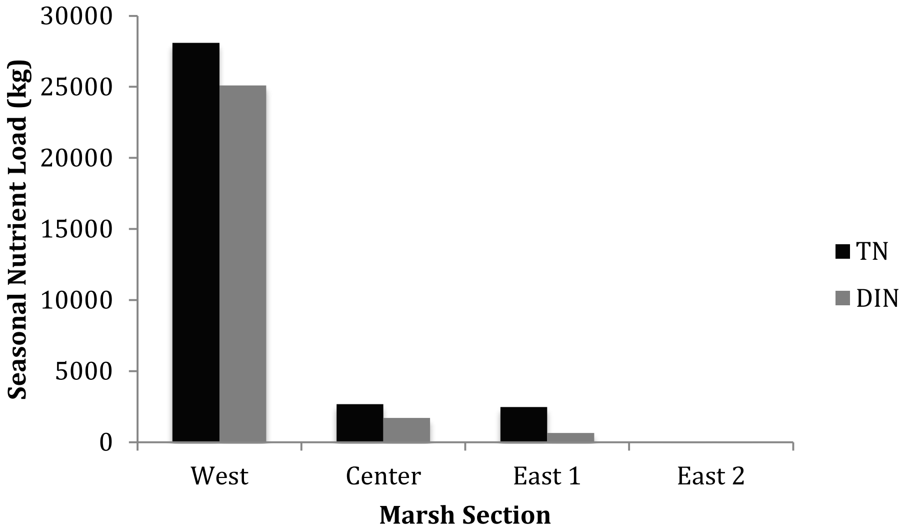


TN and 6,865 kg DIN (Figure B.5). East 2 had very little nutrient loading of 17.7 kg TN, 0.23 kg DIN, 3.66 kg TP and 2.49 kg TDP, as E8 did not flow during 2015. Center and East 1 had similar nutrient loads in nitrogen and phosphorus compounds. The West watershed exports the highest TN and DIN of 8.21 kg TN/hectare and 7.33 kg DIN/hectare, respectively, 32 and 45 times higher than observed in the center and eastern watersheds (Figure B.6). The mean export for 2015 was 2.39 kg TN/hectare, 1.94 kg DIN/hectare, 0.26 kg TP/hectare and 0.19 kg TDP/hectare. The TP and TDP exported in the west were 11 and 8 times higher than in Center and East with exports of 0.69 kg TP/hectare and 0.58 kg TDP/hectare, respectively.

Winter precipitation, which is the driver of spring runoff, was 23 mm greater in 2016 than 2015 with a total winter precipitation of 75 mm and 52 mm, respectively. Additionally, the field season precipitation was 33 mm greater in 2016 than in 2015 with a total field season precipitation of 416 mm and 383 mm, respectively. Overall the annual 2016 precipitation was 106 mm greater than in 2015, which resulted in greater runoff in 2016. As a result nutrient (TN, TP, DIN and TDP) loading was higher in 2016 relative to 2015 (Figure B.7). The combined TN and TP load from all tributaries in 2016 was 103.903 kg TN and 12,102 kg TP. Among all tributaries, W1 and E3 had the highest TN loads of 23,019 kg TN and 20,621 kg TN respectively, nearly two times greater than the mean TN load of 10,432 ± 2416 kg TN. These two tributaries also had the highest TP, DIN and TDP loads. W1 released 3,405 kg TP, 18,091 kg DIN, and 3086 kg TDP throughout the 2016 field season, and E3 released 2068 kg TP, 17,255 kg DIN, and 2,068 kg TDP throughout the 2016 field season, approximately two times greater than the mean (±SE, N=9) nutrient loads of 1,211 ± 337 kg TP, 8,316 ± 1,975 kg DIN, and 1,108 ± 307

Figure B.5. Seasonal nutrient load (kg) of TN and DIN (A), TP and TDP (B) to the four main sections of Delta Marsh, Manitoba, from March 31 to August 28, 2015.

(A)



(B)

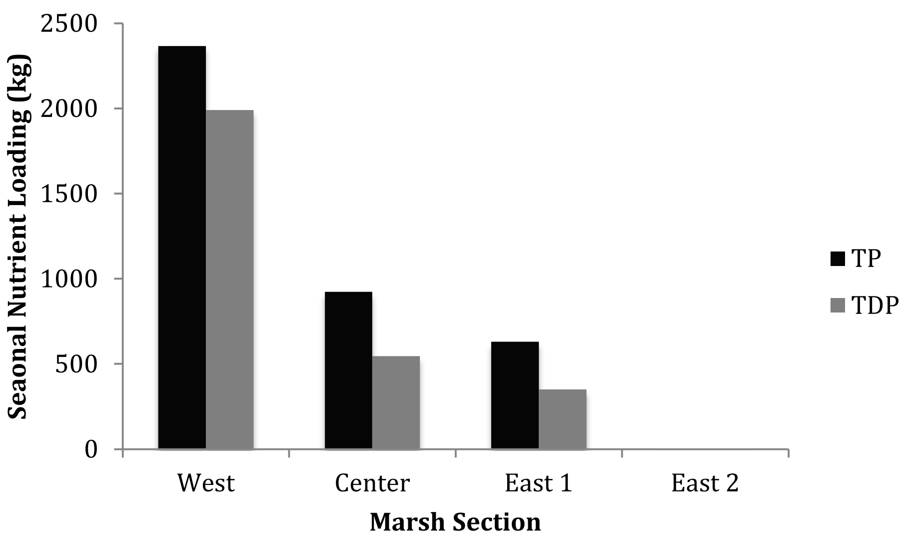
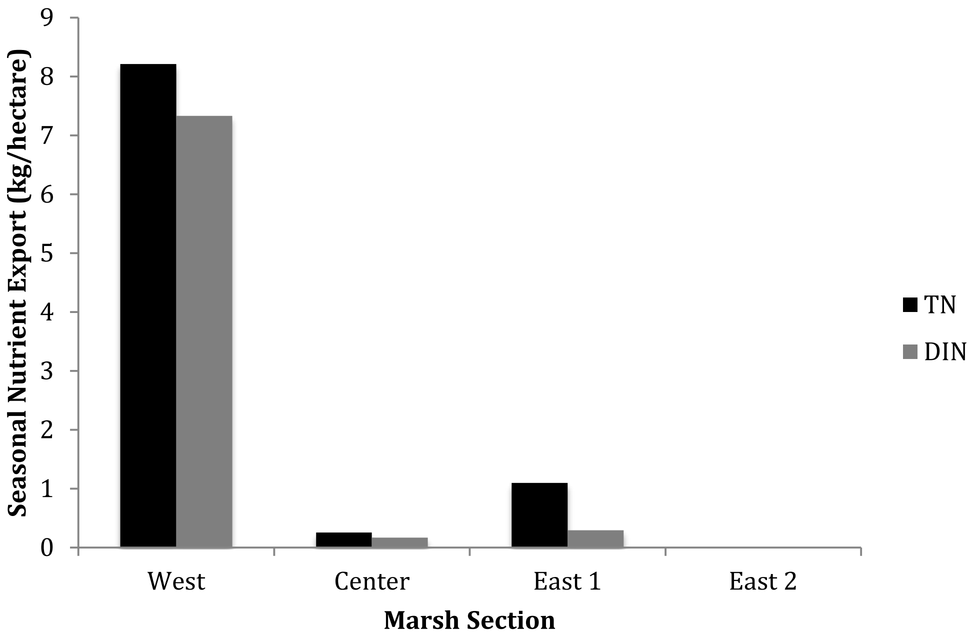


Figure B.6. Seasonal nutrient export (kg/hectare) of TN and DIN (A) and, TP and TDP (B) to the four main sections of Delta Marsh, Manitoba, from March 31 to August 28, 2015.

(A)



(B)

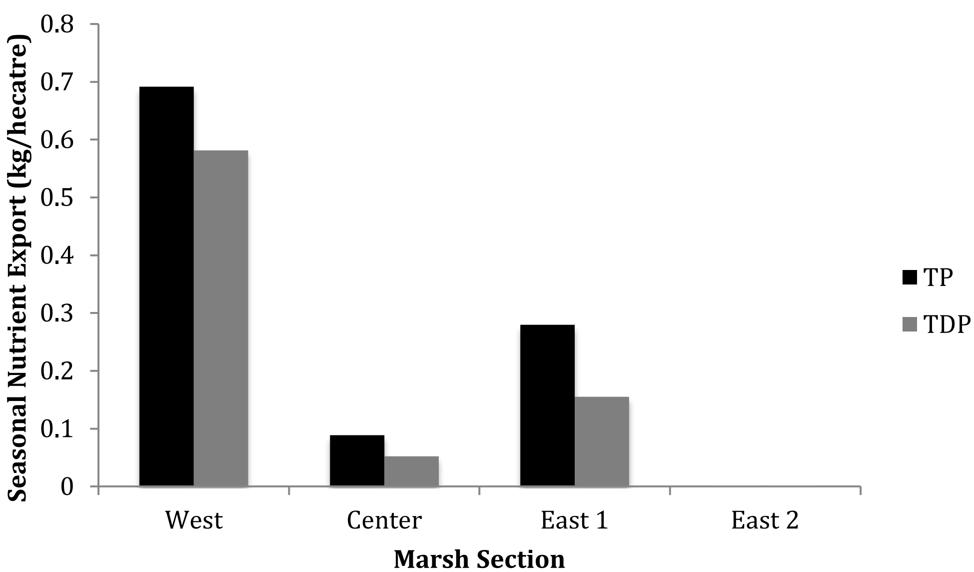
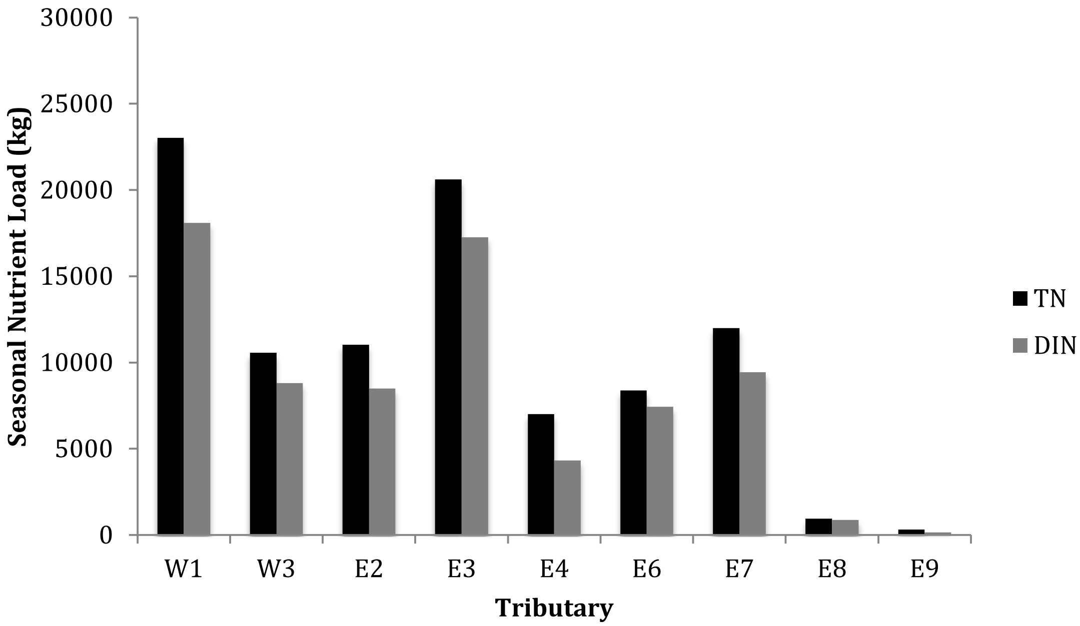
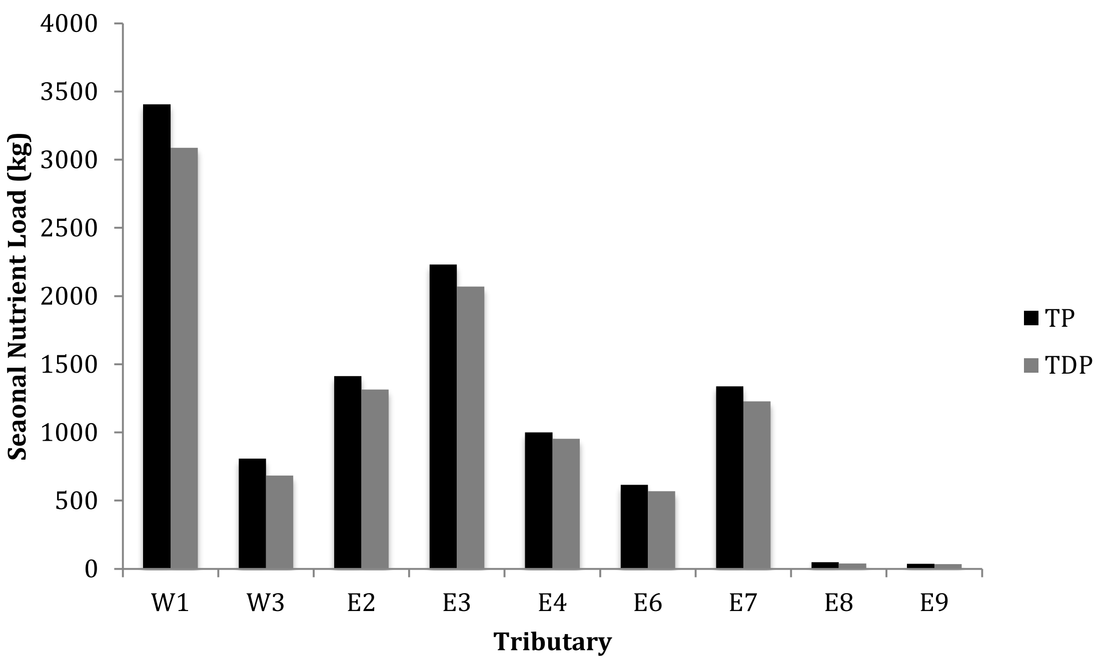


Figure B.7. Seasonal nutrient load (kg) of TN and DIN (A), TP and TDP (B) in nine tributaries that empty into Delta Marsh, Manitoba, from March 14 to August 27, 2016.

(A)



(B)



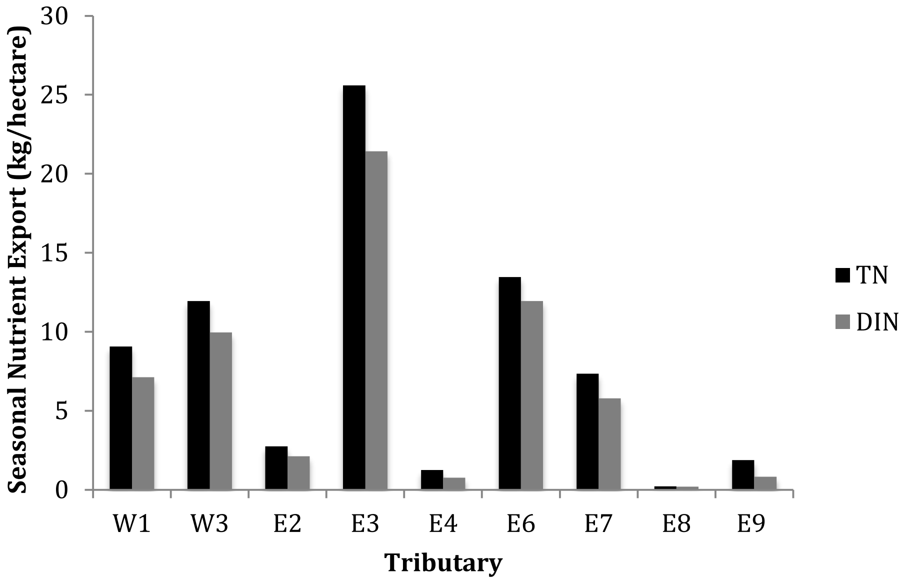
kg TDP.

Nutrient export in 2016 was higher and more spatially variable than what was recorded in 2015 (Figure B.8). The mean nutrient export for all tributaries in 2016 for TN, DIN, TP and TDP were 8.48 kg TN/hectare, 6.82 kg DIN/hectare, 0.88 kg TP/hectare and 0.80 kg TDP/hectare respectively (Figure B.7), increasing from the mean nutrient export in 2015 of 1.85 kg TN/hectare, 1.52 kg DIN/hectare, 0.19 kg TP/hectare and 0.15 kg TDP/hectare. Tributary W1 had the highest nutrient load (kg) of all nine tributaries in 2016 (Figure B.7) although had an average export of 9.06 kg TN/hectare, 7.13 kg DIN/hectare respectively (Figure 3.8). Tributary E3 had the highest TN and DIN export from March to August of 25.6 kg/hectare and 21.42 kg/hectare (Figure D.8). Tributaries E2, E4 and E8 had the lowest TN and DIN exports of all tributaries, with 34%, 15% and 2.6% of the mean TN export respectively, and 31%, 12% and 2.9% of the mean DIN export respectively. TP and TDP exports in 2016 varied across the watershed, with the highest export in E3 with 2.77kg TP/hectare and 2.57 kg TDP/hectare respectively; nearly three times the mean of 0.8 kg TP/hectare and 0.8 kg TDP/hectare). W1, W3, E6 and E7 had similar TP and TDP exports in 2016. Tributary E8 had the lowest TP and TDP export of 0.011kg/hectare and 0.0088kg/hectare respectively 84 and 96 times lower than the mean exports of 0.92 kg TP/hectare and 0.84 kg TDP/hectare. Tributary E9 had exports of 1.87 kg TN/hectare, 0.83 kg DIN/hectare, 0.22 kg TP/hectare and 0.20 kg TDP/hectare (Figure B.8).

Water yield (mm) for each tributary varied spatially and temporally (Figure B.9). Tributary W1, E3 and E7 had the highest water yield in 2015 with yields of 47.5 mm, 50.6 mm, and 44.6 mm, respectively. Water yields increased in every tributary in 2016,

Figure B.8. Seasonal nutrient export (kg/hectare) of TN and DIN (A), TP and TDP (B) for nine tributaries that empty into Delta Marsh, Manitoba from March 14 to August 27, 2016. Note that E1 was removed because area (hectares) could not be delineated.

(A)



(B)

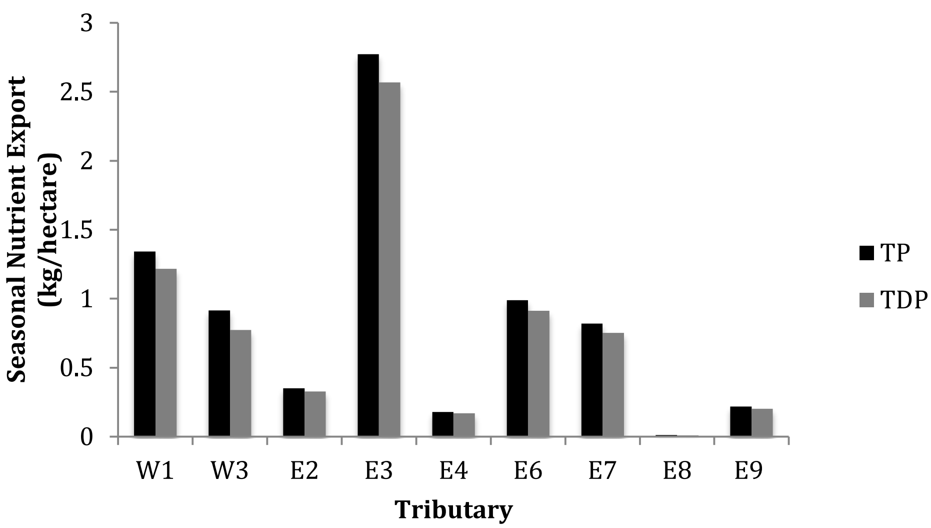
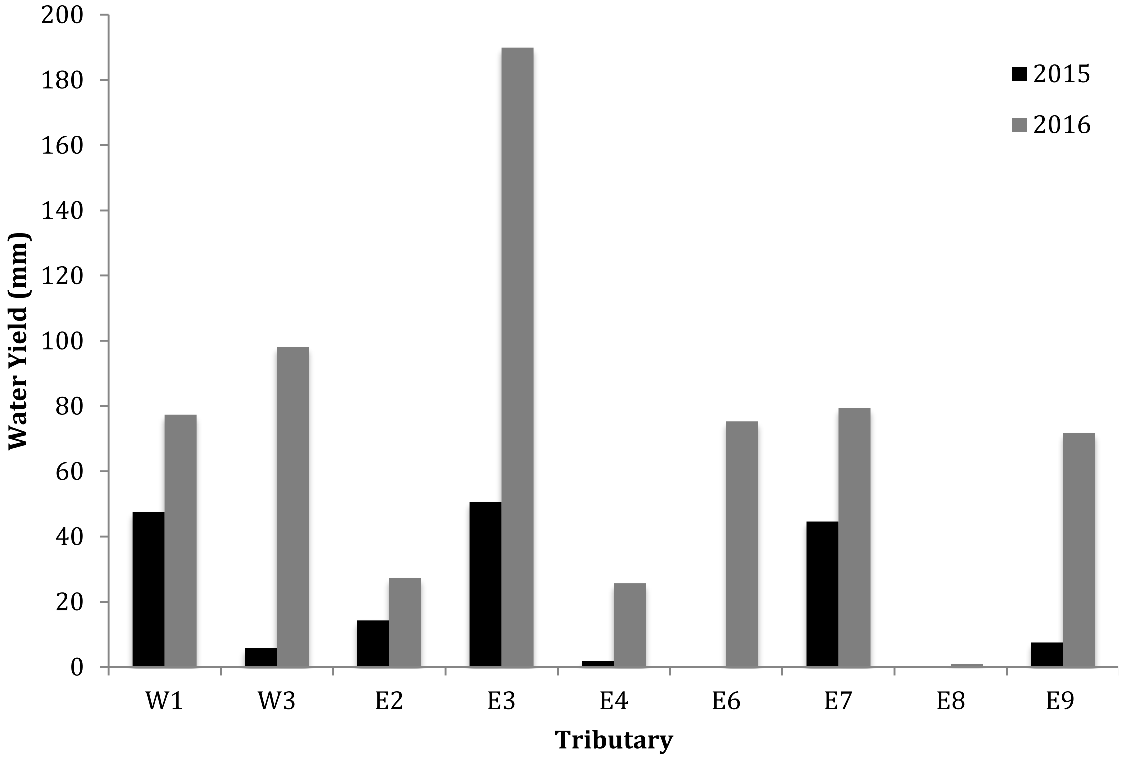


Figure B.9. Total water yield (mm) for nine tributaries that empty into Delta Marsh in 2015 and 2016.



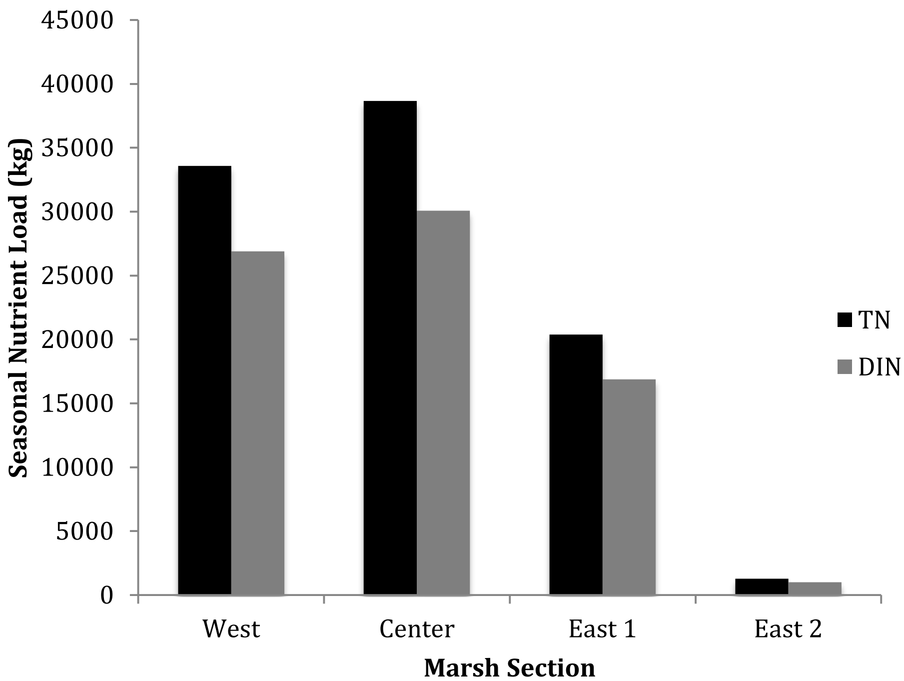
and was the highest in tributary E3 (189.9mm), also with the highest nutrient export. Water yield had a high positive correlation between TN, DIN, TP and TDP exports with an r^2 of 0.82, 0.77, 0.85 and 0.85, respectively.

In 2016, the West and Center units delivered roughly equal nutrient loads to Delta Marsh. West and center deliver approximately two times more N than East 1 and 28 times more N than East 2, and two times more P than East 2 and 52 times more P than East 2 (Figure B.10). Similar to 2015, East 2 had minimal nutrient loading compared to the other watersheds (1277 kg TN, 1009 kg DIN, 87 kg TP, and 74 kg TDP). The West and Center had nearly twice the TP and TDP than the eastern section of the marsh. West marsh had the highest TN and DIN export of 9.8 kg TN/hectare and 7.9 kg DIN/hectare, nearly three times the export from Center and East units (Figure B.11). The West marsh had the highest TP and TDP export of 1.2 kg TP/hectare and 1.1 kg TDP/hectare, twice the export observed in Center and East units.

The West subsection of the marsh received the highest nutrient load per wetland area from the watershed in 2015; with 37.26 kg TN/hectare and 33.27 kg DIN/hectare (Table B.2), approximately ten and thirty times higher than the center and east sections respectively. The TN:TP ratio of the nutrient export indicated that the west received excessive TN, while the Center and East sections received limited TN and were excessive in TP in 2015.The P levels in west were 3.14 kg TP/hectare and 2.64 kg TDP/hectare in 2015. Nutrient concentrations increased from 2015 to 2016, Center marsh had the highest nutrients per wetland area for each N and P compound. It had 56.9 kg TN/hectare and 44.3 kg DIN/hectare, which was slightly higher than what was observed in the West marsh (44.5 kg TN/hectare and 35.6 kg DIN/hectare). Center marsh had a six times

Figure B.10. Seasonal nutrient load (kg) of TN and DIN (A), TP and TDP (B) to the four main sections of Delta Marsh, Manitoba, from March 14 to August 27, 2016.

(A)



(B)

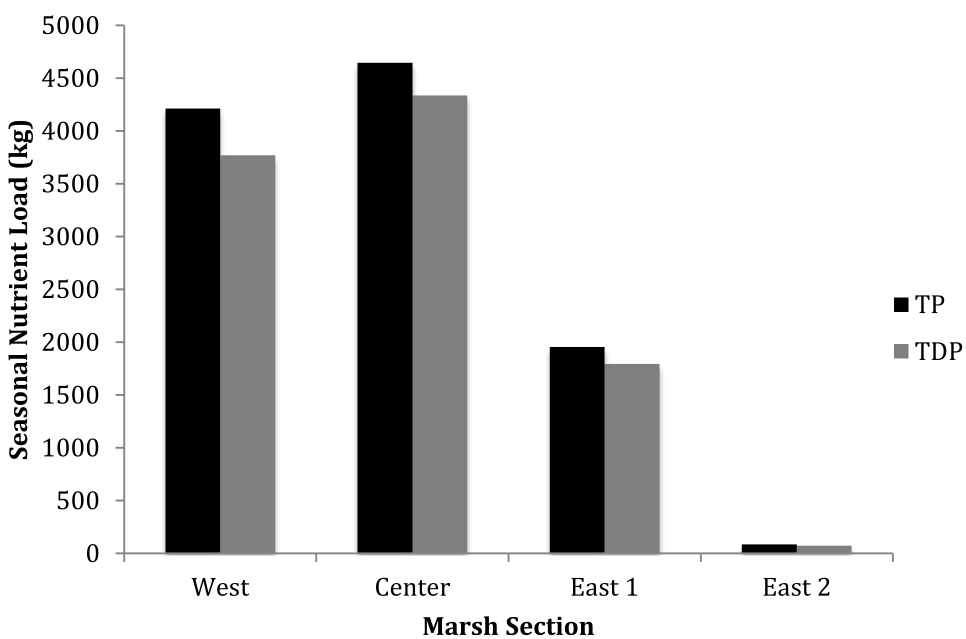
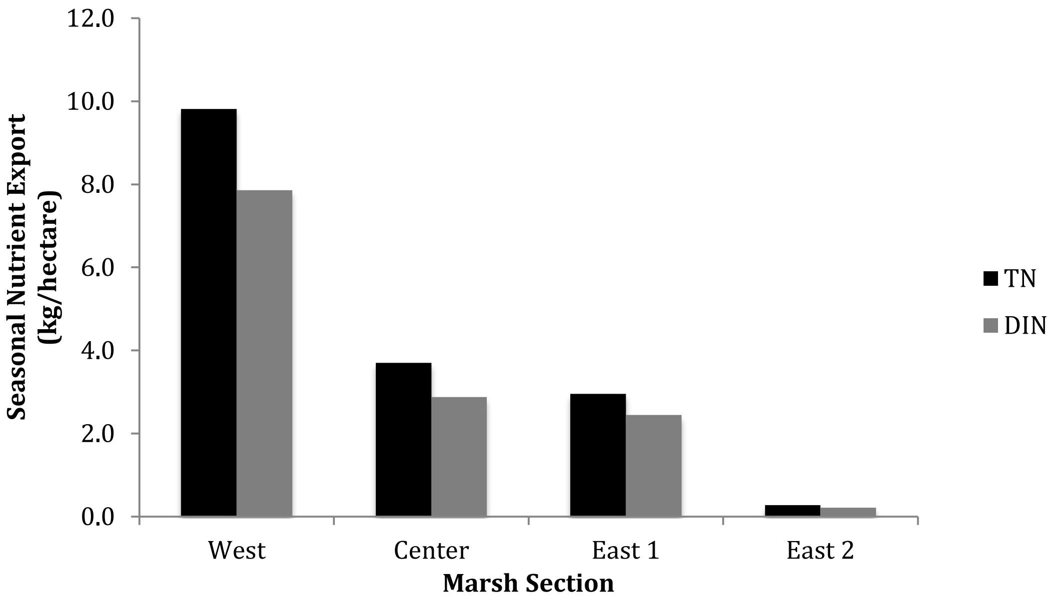


Figure B.11. Seasonal nutrient export (kg/hectare) of TN and DIN (A) and TP and TDP (B) to the four main sections of Delta Marsh, Manitoba, from March 14 to August 27, 2016.

(A)



(B)

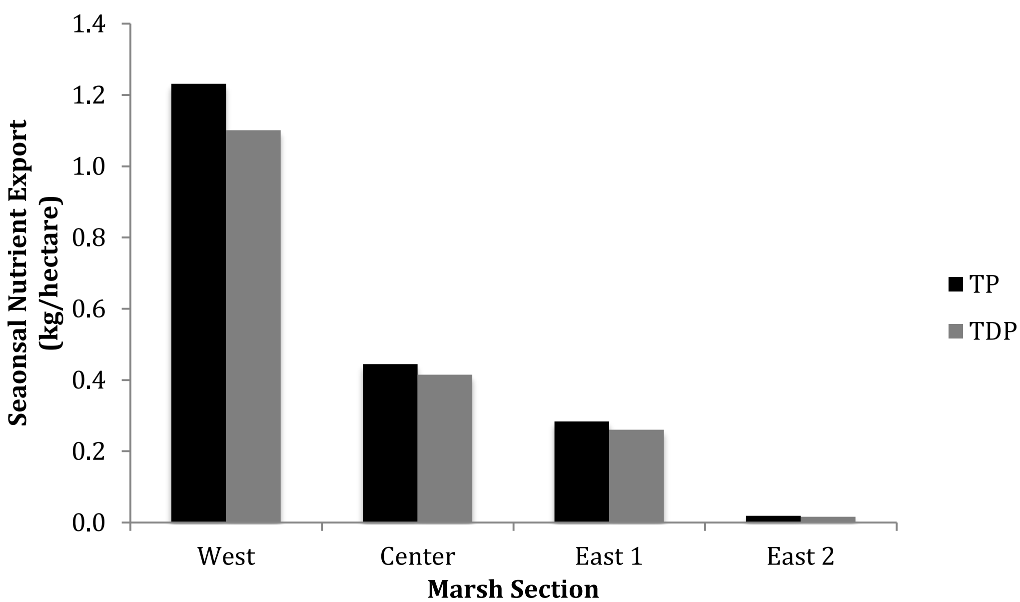


Table B.2. Total nutrient export into four subsections of Delta Marsh, recorded in nutrient per wetland area (kg/hectare) for TN, DIN, TP and TDP, including the TN:TP and DIN:TDP. Nutrient exports to Delta Marsh were recorded from March to September in 2015 and 2016.

		Nutrient per Wetland Area (kg/hectare)					
Year	Marsh Section	TN	DIN	TP	TDP	TN:TP	DIN:TDP
2015	West	37.3	33.3	3.14	2.64	26.3	27.9
2015	Center	3.93	2.53	1.36	0.80	6.39	6.95
2015	East 1	1.54	0.41	0.39	0.22	8.68	4.13
2015	East 2	<0.01	<0.01	<0.01	<0.01	10.7	0.21
2015	Delta Marsh	5.28	4.36	0.63	0.45	18.7	21.0
2016	West	44.5	35.6	5.59	5.00	17.6	15.8
2016	Center	56.9	44.3	6.84	6.39	18.4	15.3
2016	East 1	12.7	10.5	1.22	1.12	23.1	20.8
2016	East 2	0.70	0.56	0.05	0.04	32.5	30.0
2016	Delta Marsh	14.9	11,8	1.73	1.58	19.0	16.6

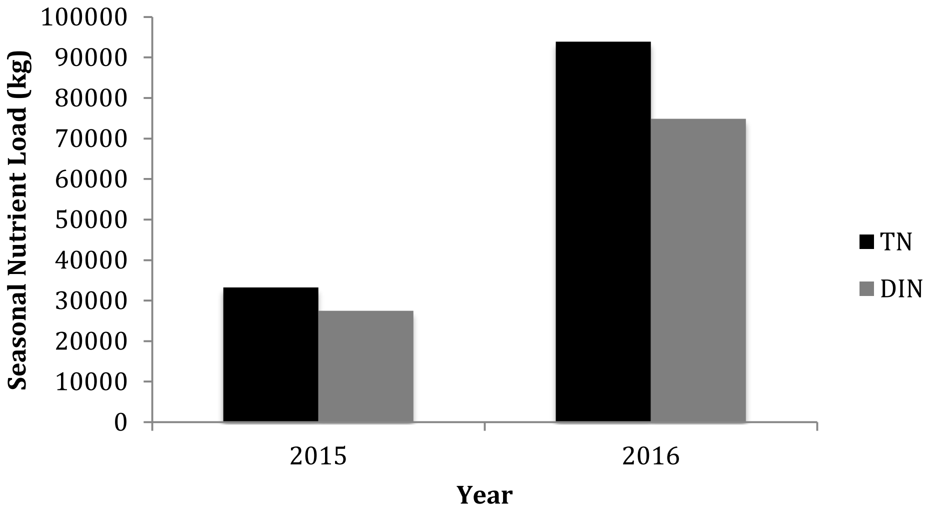
higher export of P between 2015 and 2016, with 6.84 kg TP/hectare and 6.39 kg TDP/hectare respectively. Section East 1 displayed approximately ten times higher nitrogen in 2016 than in 2015, with 12.7 kg TN/hectare and 10.52 kg DIN/hectare.

Total nutrient loads of TN, DIN, TP and TDP (kg) to Delta Marsh in 2016 were approximately two to three times higher than in 2015 (Figure B.12). The nutrient loads were 33,263 kg TN, 27,463 kg DIN, 3,924 kg TP and 2,287 kg TDP in 2015. The loads increased in 2016 to 93,892 kg TN, 74,852 kg DIN, 10,902 kg TP and 9,977 kg TDP. Nutrient export values for the watershed were 1.60 kg TN/hectare, 1.32 kg DIN/hectare, 0.19 kg TP/hectare and 0.14 kg TDP/hectare in 2015 (Figure B.13). The total nutrient export increased in 2016 to 4.52 kg TN/hectare, 3.61 kg DIN/hectare, 0.525 kg TP/hectare and 0.481 kg TDP/hectare.

The N:P molar ratios in sites W1, W3, E3 and E7, the four major flow contributors in 2015, were consistently over 16, the Redfield ratio, indicating the stream water contained excess N compared to P (Figure B.14). The other tributaries were all noted to be N limited. In 2016, all tributaries excluding W1 were excessive in TN, while W1, E2, E4 and E9 were limited in DIN.

The nutrient concentrations in the tributaries relative to the nutrients in Delta Marsh were two to three times greater. The 2015 mean TN concentration in tributaries was 5.87 ± 2.03 mg/L (1.05 mg/L – 61.18 mg/L), and the mean TN concentration in the Delta Marsh water column was 2.68 ± 0.14 mg/L (0.90 mg/L – 6.90 mg/L). The 2015 mean TP concentration in tributaries was 1.61 ± 0.43 mg/L (0.14 mg/L – 16.6 mg/L), and the mean TP concentration in the Delta Marsh water column was 0.28 ± 0.02 mg/L (0.06

Figure B.12. Seasonal Nutrient Load (kg) of TN, DIN, TP and TDP to Delta Marsh, Manitoba in 2015 and 2016.



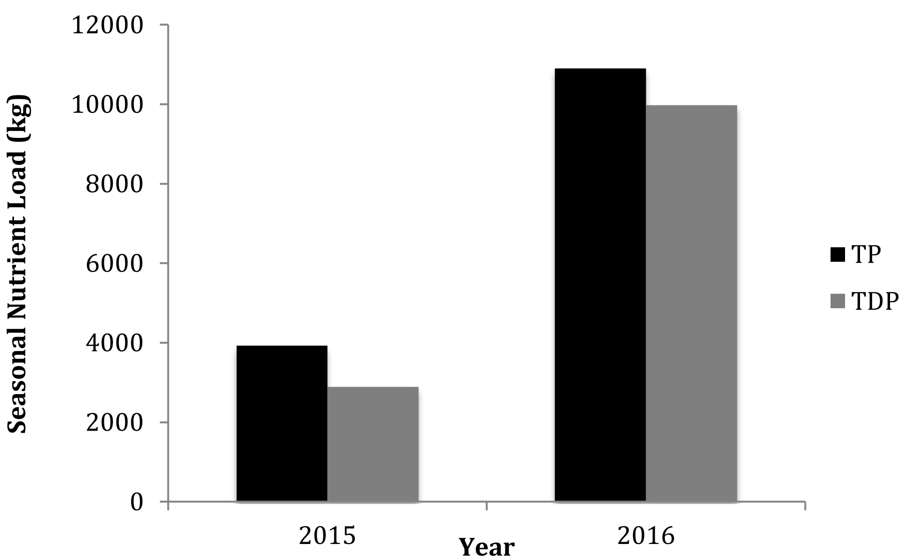
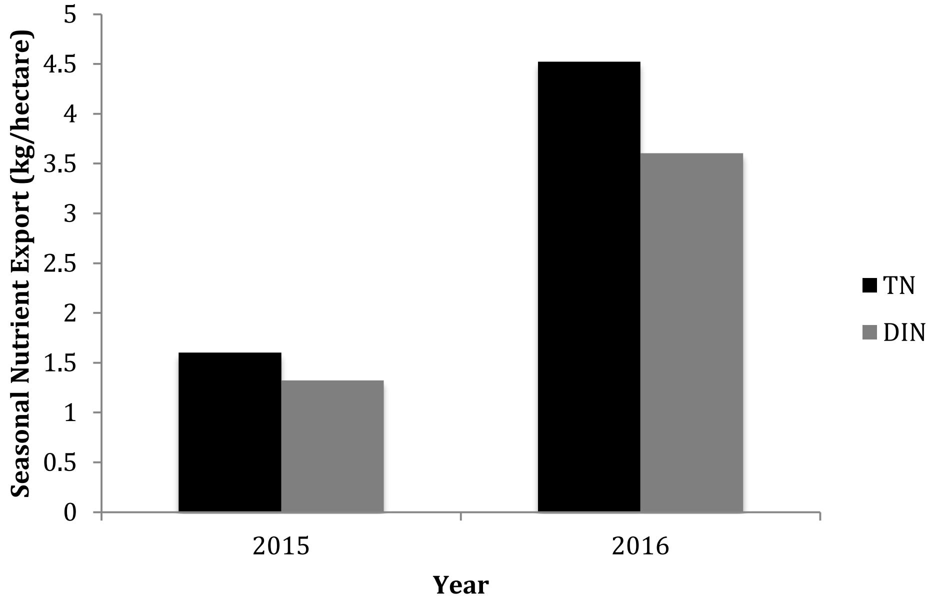


Figure B.13. Seasonal nutrient export (kg/hectare) of TN and DIN (A) and TP and TDP (B) to Delta Marsh, Manitoba for 2015 and 2016.

(A)



(B)

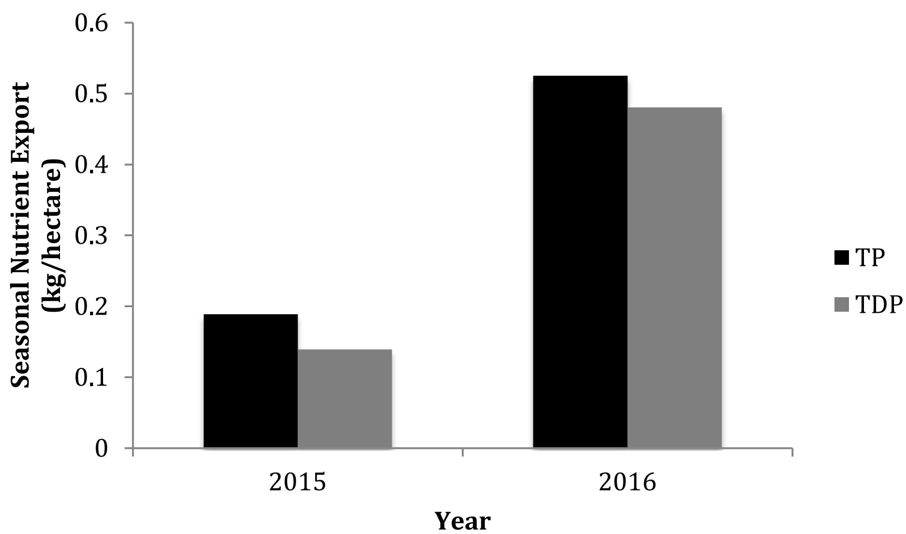
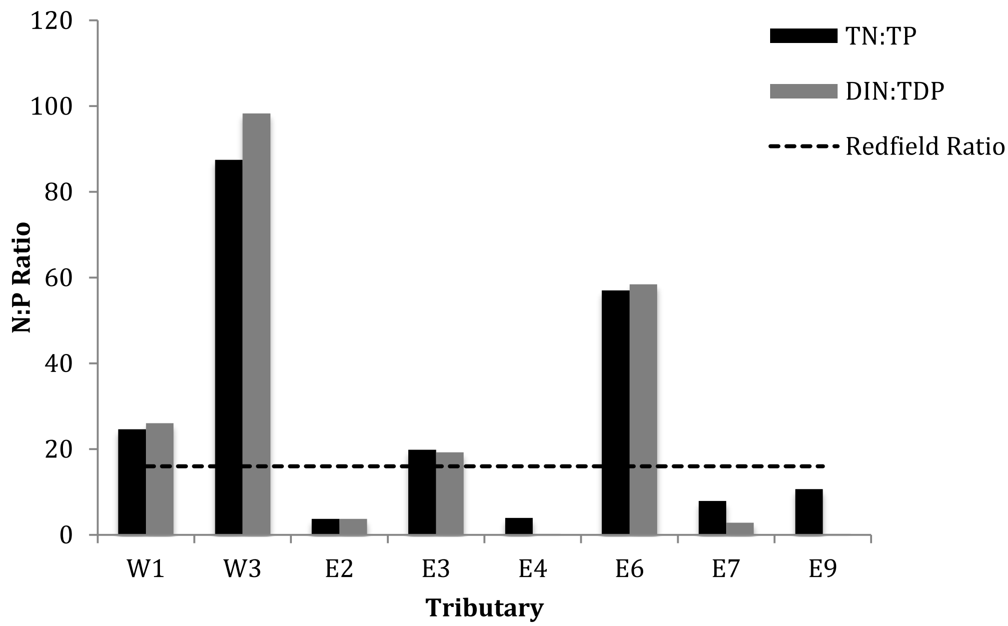
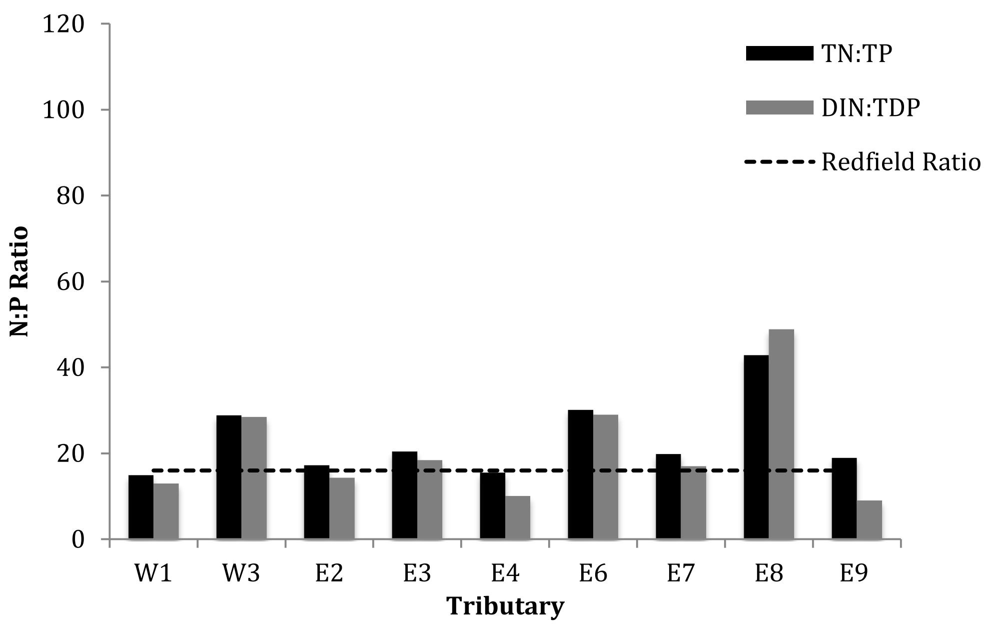


Figure B.14. The N:P molar ratios of nine tributaries that empty into Delta Marsh Manitoba in 2015 (A) and 2016 (B). Note that E8 did not flow in 2015 (A).

(A)



(B)



mg/L - 0.99 mg/L). The 2016 mean TN concentration in tributaries was 8.98 ± 0.66 mg/L (0.90 mg/L – 27.6 mg/L), and the mean TN concentration in the Delta Marsh water column was 3.37 ± 0.16 mg/L (1.35 mg/L – 9.90 mg/L). The 2016 mean TP concentration in tributaries was 0.98 ± 3.18 mg/L (0.02 mg/L – 3.18 mg/L), and the mean TP concentration in the Delta Marsh water column was 0.38 ± 0.05 mg/L (0.06 – 5.28 mg/L).

In 2014, cropland cover in the eight tributaries ranged from 79% to 96%. E3 had the highest overall cropland cover and E8 had the lowest (Figure B.15). Natural lands, including water, exposed land, shrubland, grassland, wetland, pasture, land too wet to seed and forested land, increased in cover from west to east. E4 and E8 had the highest natural land cover in 2014 of 8.6% and 7.5%, respectively. E4 and E8 also had the highest cover of urban developed land of 7% to 13%, respectively. Due to the large contribution of cropland in each sub-watershed, this land use was divided further into crop type to explore how nutrient loading may be influenced by crop type.

The dominant crop types in 2014 included spring wheat, beans and canola with a mean cover of 29%, 26% and 25% for all eight-sub-watersheds (Figure B.16). Sub-watersheds E3 and E7 had the highest spring wheat cover of 47% each. The lowest spring wheat cover was located in the E2 sub-watershed with 16%. Sub-watershed E2 was dominated by 30% canola and 37% beans. Sub-watershed W3 had the highest cover of canola and soybean at 36% and 16% respectively. E3 and E6 had two and five times higher cover of oats than the mean cover across the watershed. Rye, winter wheat, barley, corn, flaxseed, sunflowers and potatoes contributed between 0% and 14% cover across the sub-watersheds.

Figure B.15. Percent of 2014 land use in each sub-watershed of the eight tributaries flowing into Delta Marsh, Manitoba.

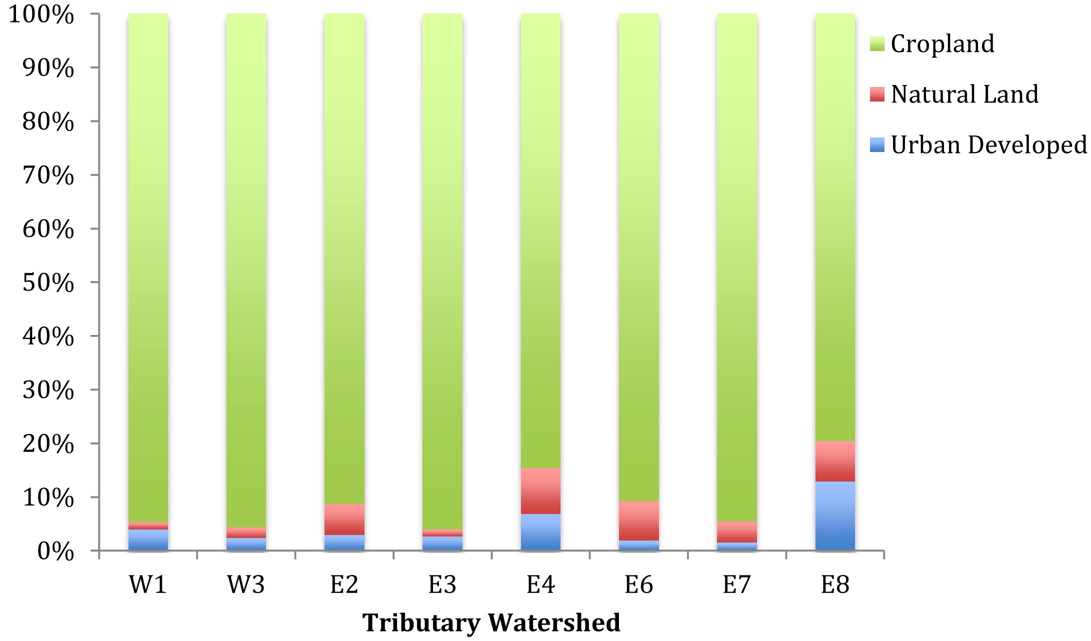
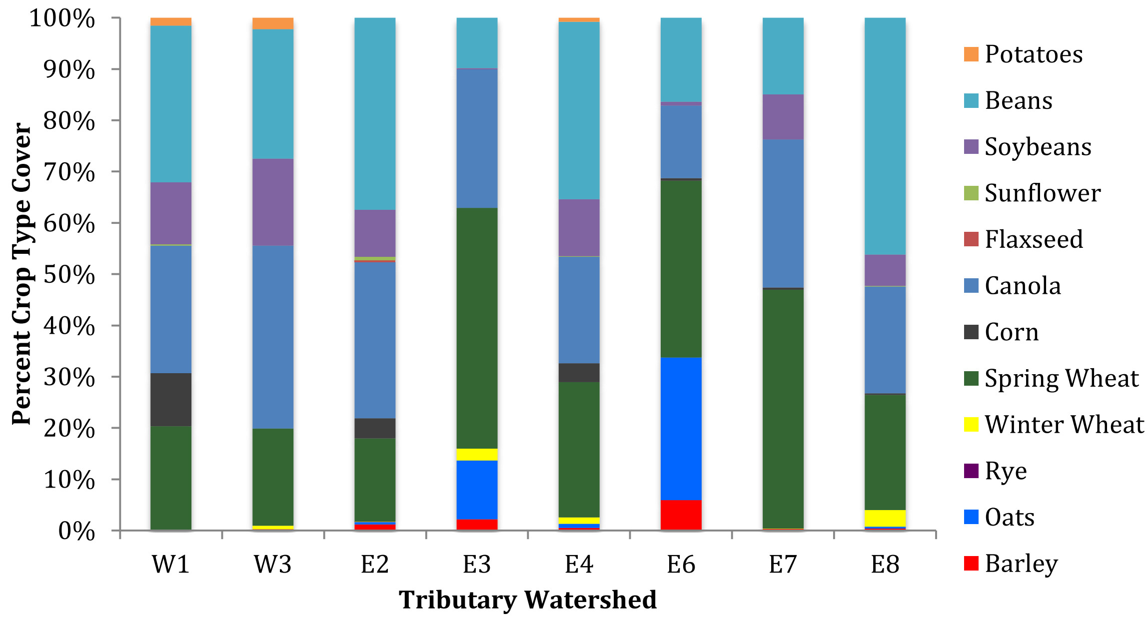


Figure B.16. Percent of 2014 crop type in each sub-watershed of the eight tributaries flowing into Delta Marsh, Manitoba.



Land use cover in 2015 was dominated by cropland in all eight sub-watershed areas (Figure B.17). E3 and W1 had the highest percentage of cropland in the sub-watershed of 95 and 94% and E8 had the lowest percentage of cropland of 77% in 2015. Forested, pastured, wetland, shrubland, exposed land and water total cover accounted for less than 10% of watershed area except for E2, E4 and E8 which had approximately 11 to 14% area. Urban developed land increased from west to east with the highest percentage of 9.6% in E8. Similar to 2014, the 2015 cropland area was dominant in all sub-watersheds and crop type was analyzed further. The dominant crop type for 2015 was Spring Wheat and Canola in each sub-watershed area (Figure B.18). Soybeans and beans appeared to be dominant in all sub-watershed areas except for E3. Sub-watershed E3, had the highest spring wheat cover of 38%, 23% canola 12% corn, and 14% barley. W1 had 11% corn cover and 46% canola, with approximately 23% of soybean and bean cover. W3 was the only other sub-watershed with 14% Barley cover and similarly dominated by spring wheat. The total cover of corn and barley increased between 2014 and 2015.

A Pearson product-moment correlation analysis between land use and nutrient export revealed that forested land had the moderate negative correlation with TN, TP, export between 2014 and 2015 land use and 2015 TN export (r= -0.47 and -0.54) (Table B.3 and Table B.4). DIN and TDP exports were not included in the correlation matrix due to their strong positive correlations with TN and TP, respectively (r=0.99 and 0.99). Cropland had the highest positive correlation with all nutrient exports in both matrices, with mean correlation of 0.47 for 2014 land use and 0.49 for 2015 land use (Table B.3 and Table B.4). The other land types in 2014 and 2015 were correlated negatively with the nutrient exports in 2015. Forest and shrubland were strongly correlated with

Figure B.17. Percent of 2015 land use in each sub-watershed of the eight tributaries flowing into Delta Marsh, Manitoba.

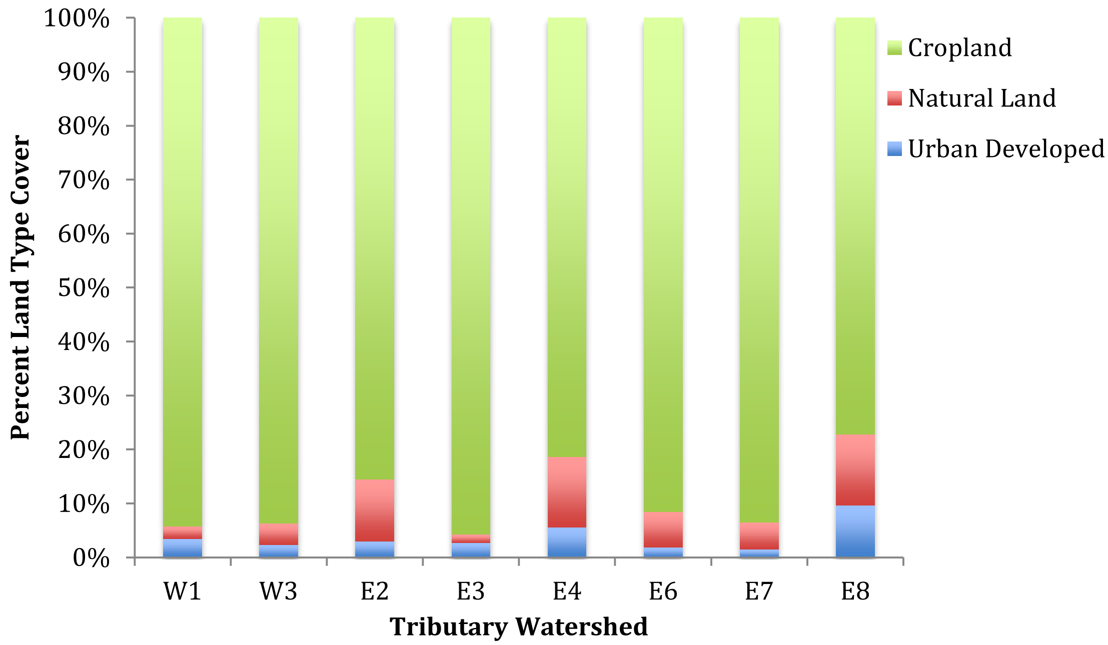


Figure B.18. Percent of 2015 crop type in each sub-watershed of the eight tributaries flowing into Delta Marsh, Manitoba.

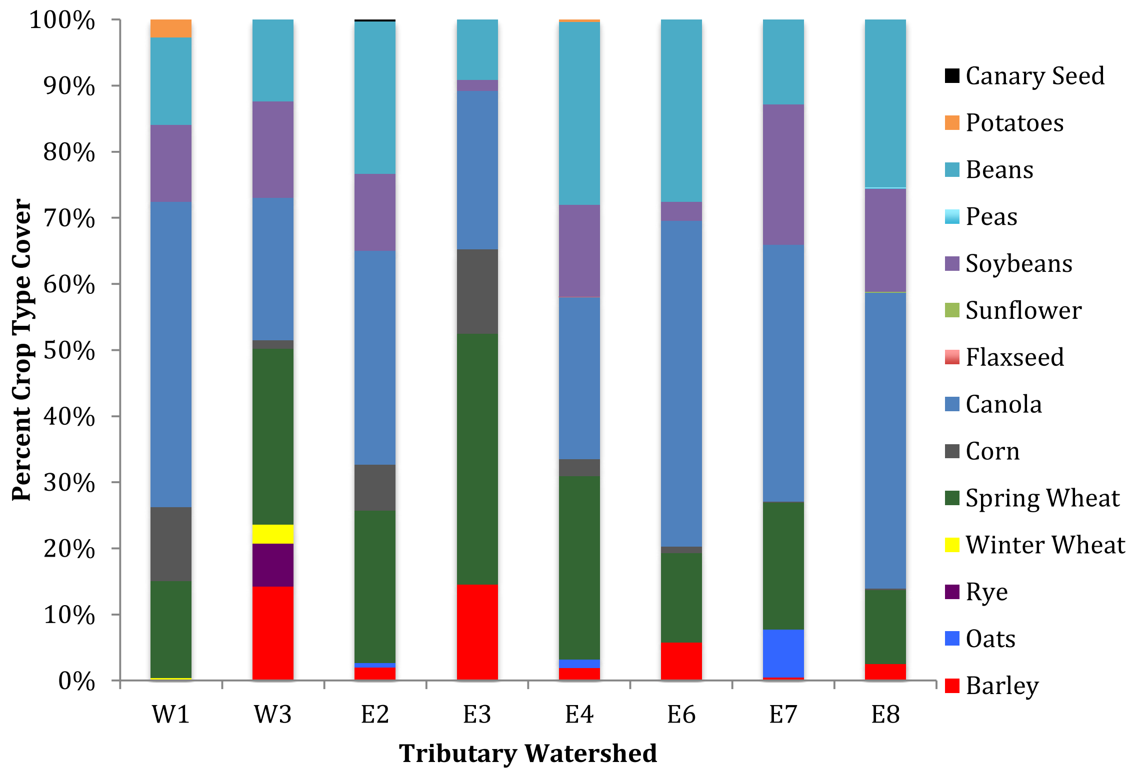


Table B.3. Pearson Product Moment Correlation of 2015 nutrient export from tributaries emptying to Delta Marsh and 2014 land use in the surrounding sub-watershed areas.

	TN	TP	Water	Exposed	Urban	Shrubland	Grassland	Wetland	Pasture	Too Wet	Cropland	Forest
TN	1.00	0.92	-0.01	-0.17	-0.20	-0.42	-0.43	-0.16	-0.32	-0.29	0.46	-0.47
TP		1.00	0.27	-0.23	-0.27	-0.45	-0.45	0.14	-0.18	-0.32	0.48	-0.37
Water			1.00	-0.24	-0.31	-0.22	-0.25	0.98	-0.19	-0.20	0.25	0.16
Exposed Land				1.00	0.95	-0.13	0.88	-0.15	0.05	0.79	-0.91	-0.28
Urban Dev.					1.00	-0.17	0.78	-0.25	0.14	0.93	-0.90	-0.33
Shrubland						1.00	0.10	-0.14	0.11	-0.11	-0.21	0.90
Grassland							1.00	-0.12	0.33	0.59	-0.91	-0.09
Wetland								1.00	-0.21	-0.18	0.14	0.25
Pasture									1.00	0.17	-0.25	-0.05
Too Wet										1.00	-0.82	-0.21
Cropland											1.00	-0.05
Forest												1.00

Table B.4. Pearson Product Moment Correlation of 2015 nutrient export from tributaries emptying to Delta Marsh and 2015 land use in the surrounding sub-watershed areas.

	TN	TP	Water	Exposed	Urban Dev.	Shrubland	Grassland	Wetland	Pastureland	Cropland	Forest
TN	1.00	0.92	-0.15	-0.21	-0.20	-0.54	-0.40	-0.18	-0.27	0.50	-0.54
TP		1.00	-0.05	-0.26	-0.27	-0.49	-0.42	0.12	-0.18	0.49	-0.45
Water			1.00	-0.15	-0.10	-0.53	0.02	0.58	0.10	0.07	-0.41
Exposed Land				1.00	0.76	0.04	0.93	-0.04	0.00	-0.79	-0.13
Urban Dev.					1.00	-0.13	0.86	-0.11	0.21	-0.86	-0.27
Shrubland						1.00	0.06	-0.01	0.03	-0.20	0.97
Grassland							1.00	0.02	0.28	-0.95	-0.10
Wetland								1.00	-0.25	0.04	0.15
Pastureland									1.00	-0.48	-0.10
Cropland										1.00	-0.03
Forest											1.00

eachother in 2014 and 2015 land uses with values of 0.90 and 0.97 respectively. Nutrient export in 2016 and 2015 cropland had a stronger correlation (r=0.78 and 0.79) than observed with the 2015 exports. The mean correlation between 2015 cropland and 2016 nutrient exports was 0.78. (Table B.5). Similarly, the other land types were all negatively correlated with the 2016 nutrient exports. Land use data for 2016 has not yet been published so I could not investigate their correlations with nutrient exports.

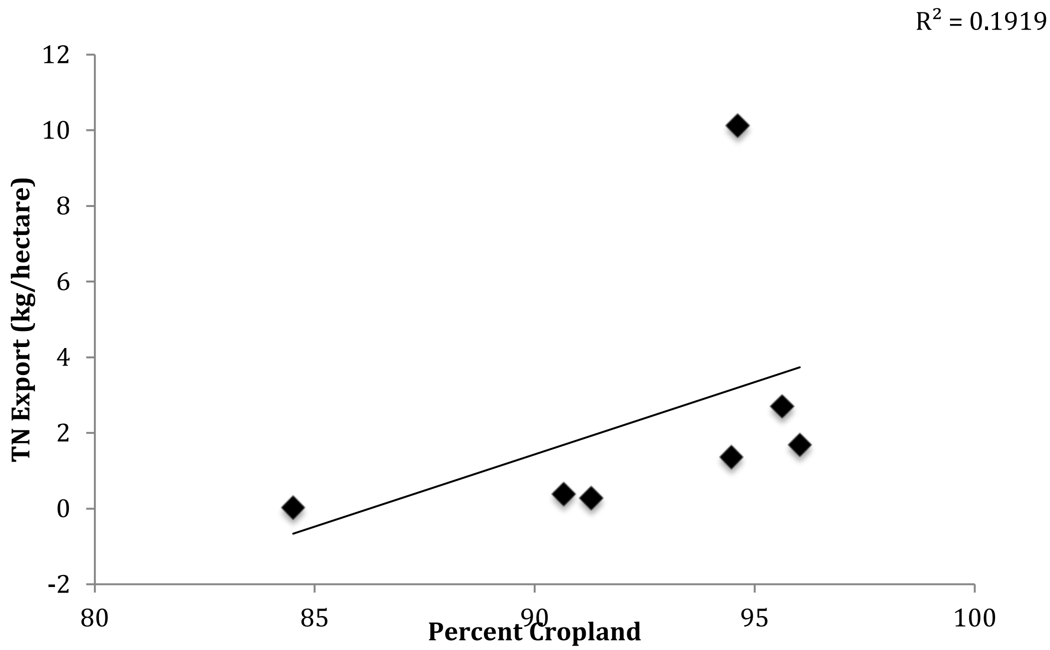
A Pearson Product Moment Correlation assumes a linear relationship between the X and Y variables. A simple linear regression of TN and TP export with percent cropland revealed that the relationship was not linear, but multiplicative. As percent cropland increases in the watershed, there is a multiplicative increase of nutrient export. After a log(x) transformation to the 2015 TN export, the relationship with 2014 land use improves, from weak (r=0.44) to strong (r=0.91) (Figure B.19). The relationship between 2015 TP export and 2014 land use also improves after a log(x) transformation (Figure B.20). After a log(x) transformation to the 2015 TP export, the relationship with 2014 land use improves, from weak (r=0.42) to moderate (r=0.68). Similarly, the relationship between 2015 TN export and 2015 land use improves after log(x) transformation, from weak (r=0.47) to strong (r=0.89) (Figure B.21), and the 2015 TP export and 2015 land use improves slightly after log(x) transformation, from r=0.42 to r=0.52 (Figure B.22). The relationship between nutrient export in 2016 and land use in 2015 observed a much stronger improvement after log(x) transformation. The relationship between 2016 TN export and the 2015 land use improved after log(x) transformation, from r= 0.79 to r= 0.96 (Figure B.23). Similarly, the 2016 TP export and the 2015 land use improved after log(x) transformation, from r=0.78 to r=0.94 (Figure B.24).

Table B.5. Pearson Product Moment Correlation of 2016 nutrient export from tributaries emptying to Delta Marsh and 2015 land use in the surrounding sub-watershed areas.

	TN	TP	Water	Exposed	Urban Dev.	Shrubland	Grassland	Wetland	Pastureland	Cropland	Forest
TN	1.00	0.96	-0.14	-0.60	-0.56	-0.13	-0.77	-0.24	-0.51	0.79	0.01
TP		1.00	-0.13	-0.55	-0.50	-0.30	-0.74	-0.20	-0.48	0.78	-0.17
Water			1.00	-0.15	-0.10	-0.53	0.02	0.58	0.10	0.07	-0.41
Exposed Land				1.00	0.76	0.04	0.93	-0.04	0.00	-0.79	-0.13
Urban Dev.					1.00	-0.13	0.86	-0.11	0.21	-0.86	-0.27
Shrubland						1.00	0.06	-0.01	0.03	-0.20	0.97
Grassland							1.00	0.02	0.28	-0.95	-0.10
Wetland								1.00	-0.25	0.04	0.15
Pastureland									1.00	-0.48	-0.10
Cropland										1.00	-0.03
Forest											1.00

Figure B.19. Linear regression of 2015 TN Export (kg/hectare) and 2014 percent cropland (A) and linear regression of 2015 log TN Export (kg/hectare) and 2014 percent cropland (B).

A



B

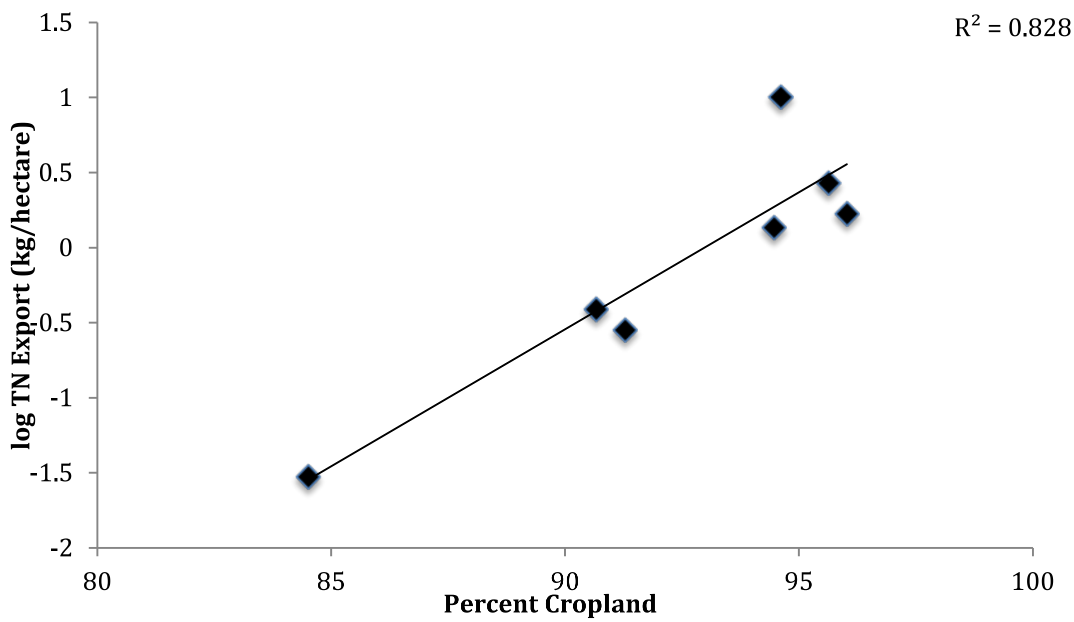
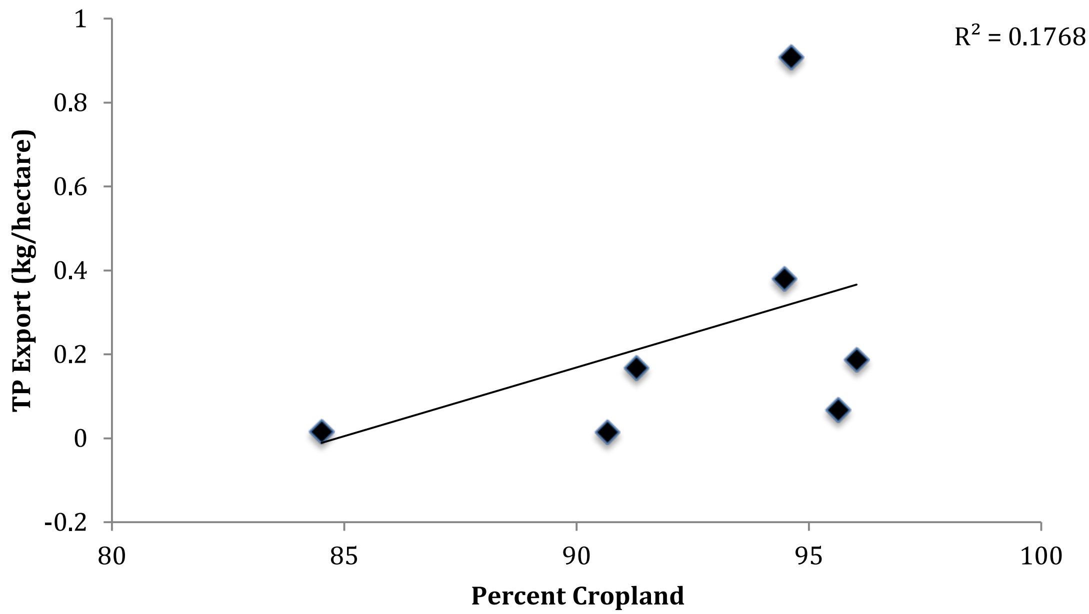


Figure B.20. Linear regression of 2015 TP Export (kg/hectare) and 2014 percent cropland (A) and linear regression of 2015 log TP Export (kg/hectare) and 2014 percent cropland (B).

A



B

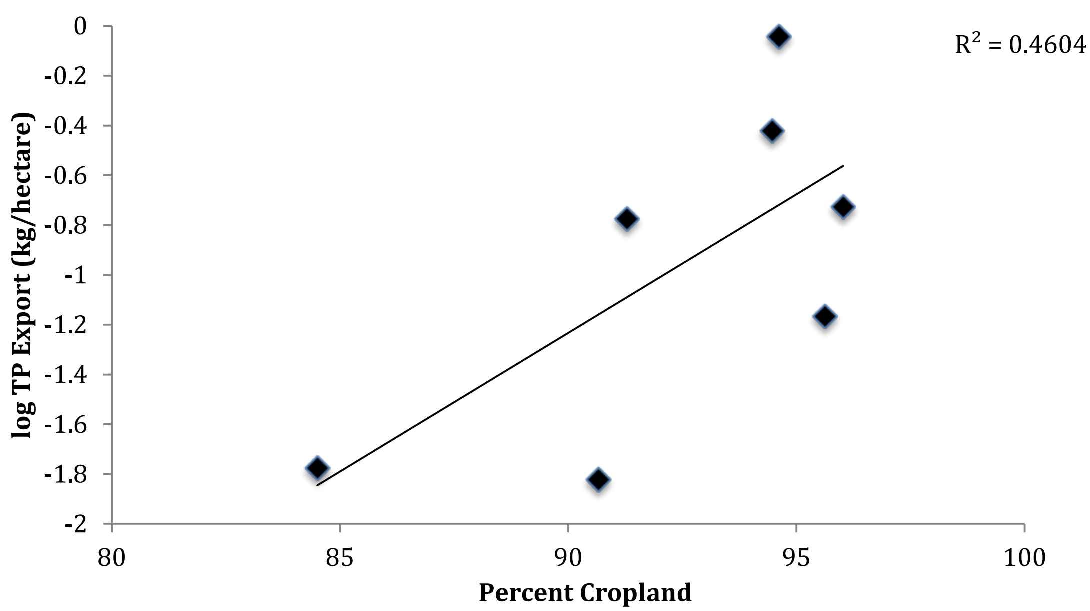
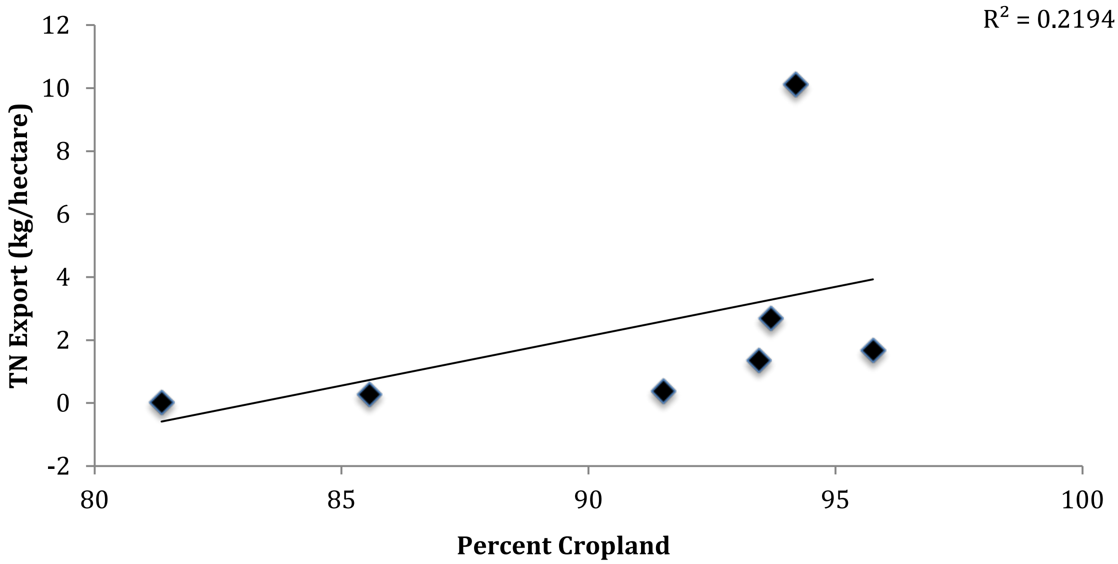


Figure B.21. Linear regression of 2015 TN Export (kg/hectare) and 2015 percent cropland (A) and linear regression of 2015 log TN Export (kg/hectare) and 2015 percent cropland (B).

A



B

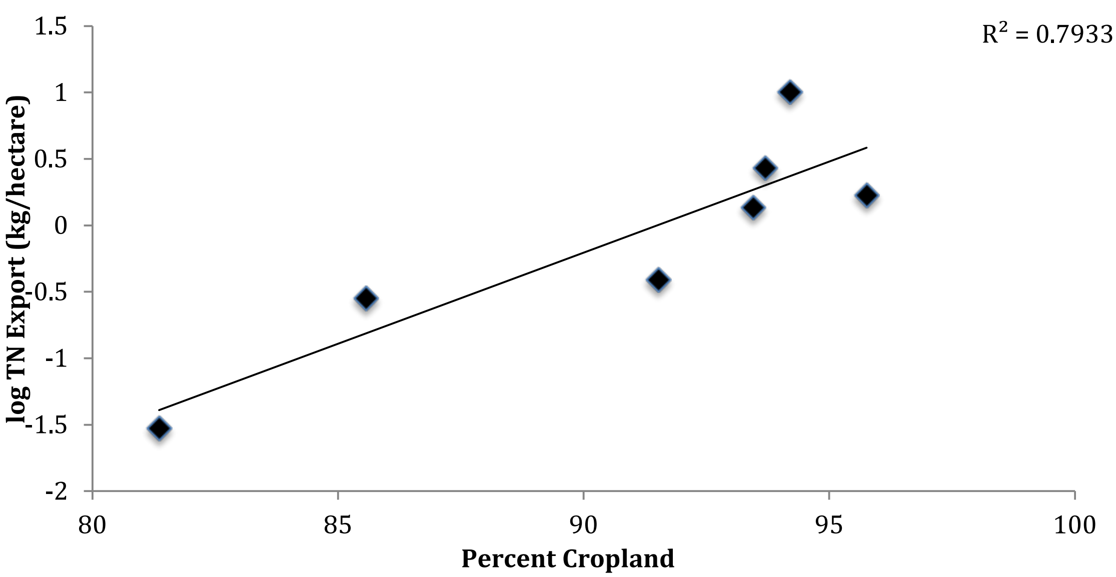
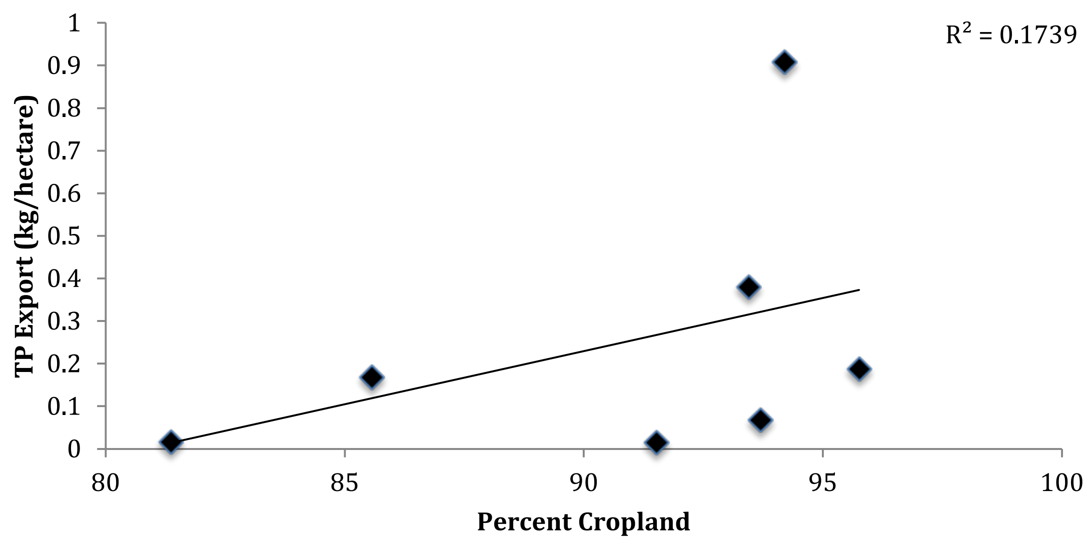


Figure B.22. Linear regression of 2015 TP Export (kg/hectare) and 2015 percent cropland (A) and linear regression of 2015 log TP Export (kg/hectare) and 2015 percent cropland (B).

A



B

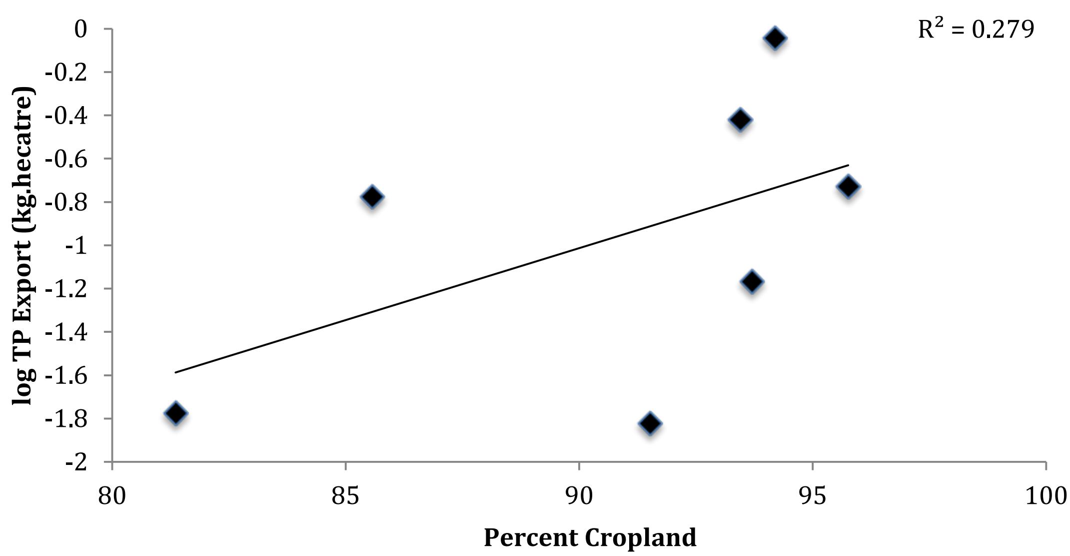
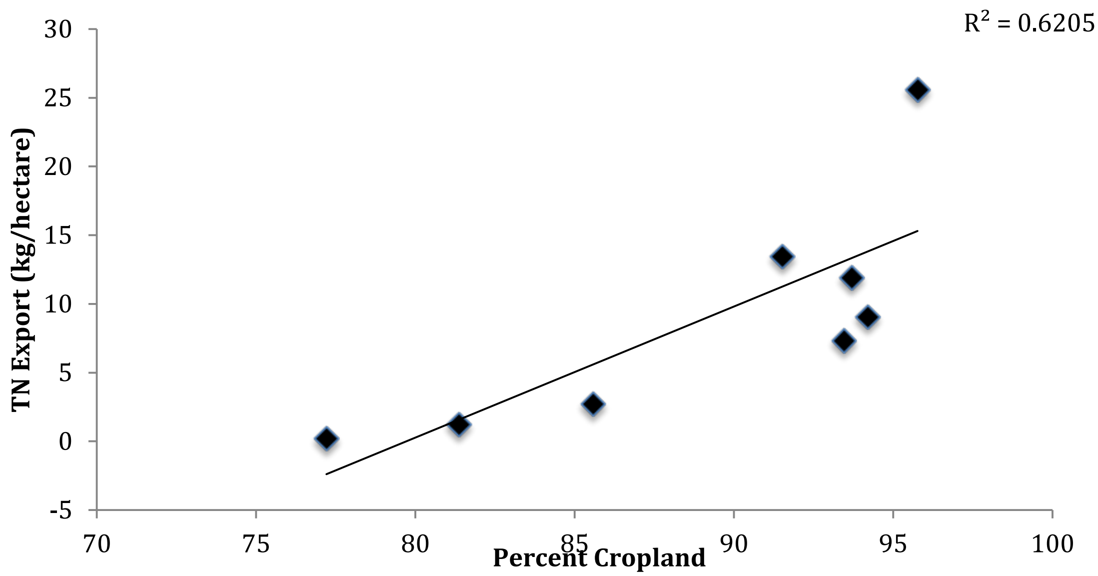


Figure B.23. Linear regression of 2016 TN Export (kg/hectare) and 2015 percent cropland (A) and linear regression of 2016 log TN Export (kg/hectare) and 2015 percent cropland (B).

A



B

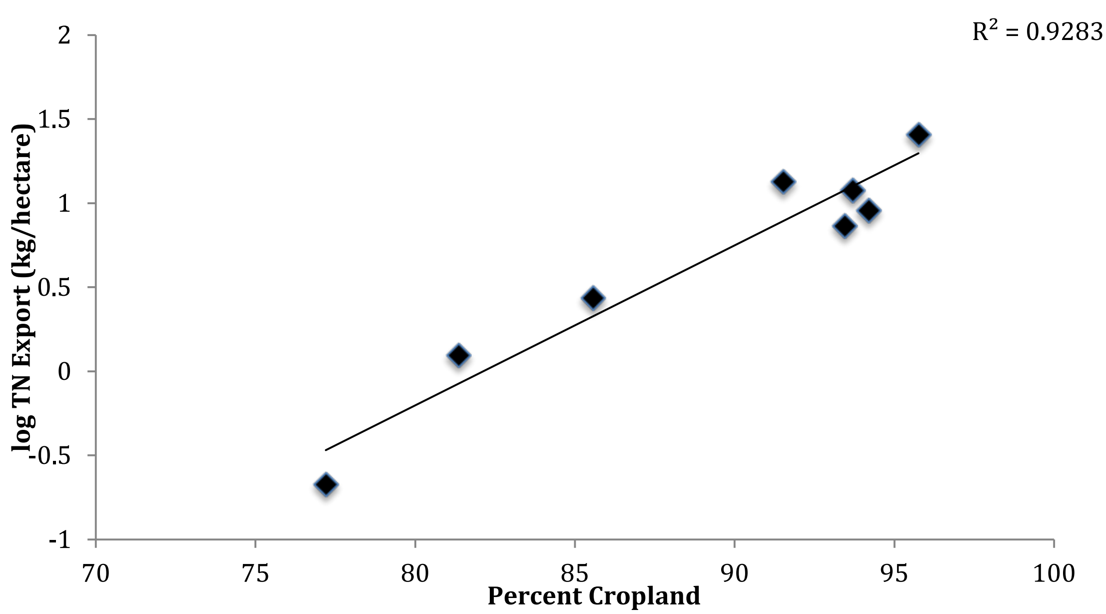
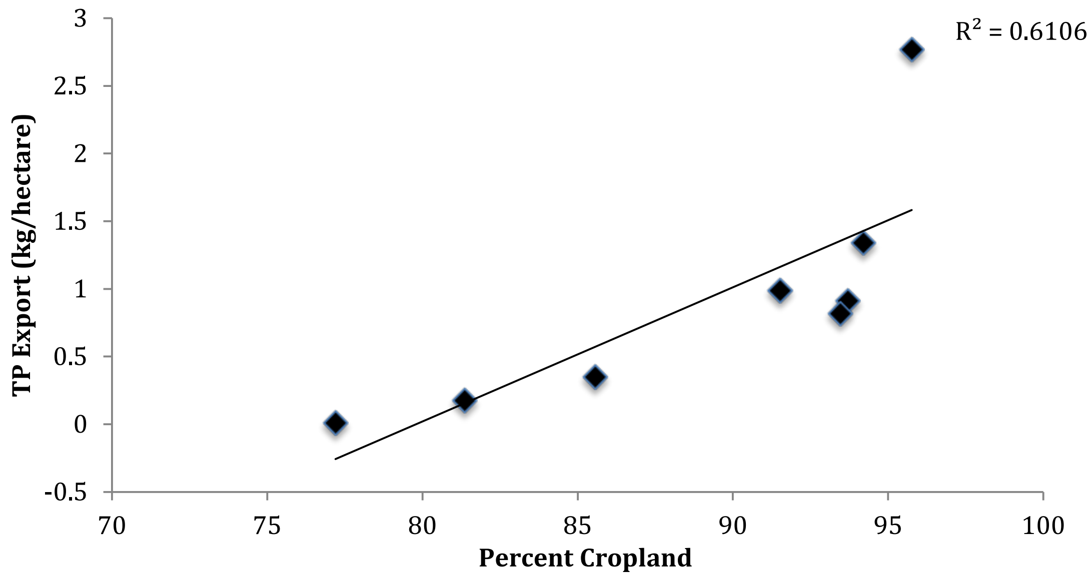
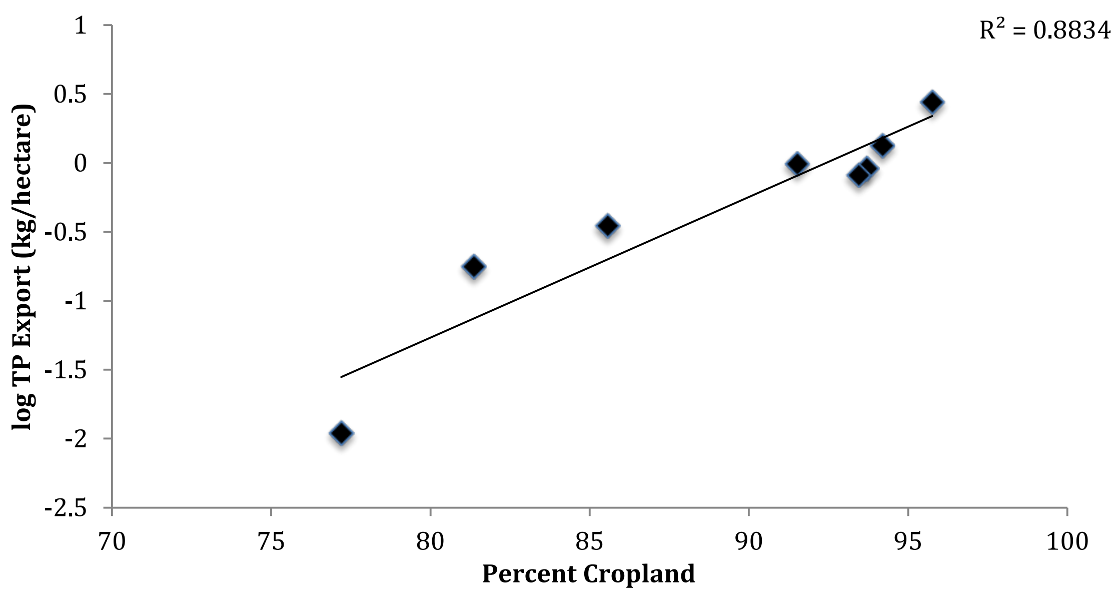


Figure B.24. Linear regression of 2016 TP Export (kg/hectare) and 2015 percent cropland (A) and linear regression of 2016 log TP Export (kg/hectare) and 2015 percent cropland (B).

A



B



Discussion

Nutrient export into Delta Marsh was directly related to volume of precipitation, water yield and land use practices. Tributary W1 had the highest nutrient export in 2015 and tributary E3 had the highest export in 2016, over twice the highest export in 2015. Most tributaries in 2016 displayed similar exports to W1, which had the highest nutrient export in 2015 but remained the same in 2016. The difference in nutrient export between 2015 and 2016 was likely caused by the difference in precipitation events during the winter and spring, as land use does not vary between years. Sub-watersheds with high cropland cover release the highest nutrient loading, causing spatial variation of nutrient export across the Delta Marsh watershed.

Volume of precipitation influences total nutrient export into Delta Marsh. Wet years result in higher nutrient loading from the watershed. My findings reveal there was temporal variation in nutrient export likely due to difference in precipitation between 2015 and 2016. As mentioned in the Results section, precipitation was 106 mm more in 2016 than that in 2015, and 95 mm more (approximately 17% higher) than the normal annual precipitation for Portage La Prairie of 532.5 mm (Environment Canada, accessed March 2017). Winter precipitation in the Delta Marsh watershed increased from 52.5 mm in 2015 to 75.9 mm in 2016, resulting in higher nutrient export from the landscape. High precipitation preceding a dry year may result in increased nutrient loading due to nutrients that remained on the landscape from the prior year (Lobb, personal communication). The correlation between nutrient export and cropland cover increased from moderate to a strong relationship between 2015 (r=0.50) and 2016 (r=0.79), likely due to the increased volume of precipitation in 2016. This indicates that years of increased precipitation may result in nutrient loading that is highly reflective of land use practices. The increase of precipitation in 2016 results in higher TN and TP exports than in 2015, excluding tributary W1, which had the same nutrient export in 2015 and 2016. This may be attributed to the timing of precipitation, flow paths in the watershed and the spatial distribution of nutrient application or land types within the W1 sub-watershed area. If nutrients application occurred prior to a precipitation event, much of the nutrients would be exported downstream. Additionally, if nutrient application occurred close to the flow path, this may result in increased nutrient loading to the tributaries. Precipitation events that occur during agricultural or land use management activities cannot be predicted or controlled. A precipitation event during the spring when fertilizers have been applied to land may cause extreme nutrient loading if the crop or vegetation has not yet taken up the nutrients. Although nutrient application or land management in the Delta Marsh watershed was not directly monitored, my data suggests that the croplands release higher volumes of water and export greater amounts of nutrients relative to more natural landscapes. Additionally, the relationship between cropland cover and nutrient export is not linear. The increased agricultural intensity results in a multiplicative increase of nutrient loading to Delta Marsh. For example, the correlation coefficient between 2016 TN export and 2015 cropland improved after log(x) transformation of TN export, from r^2=0.62 to r^2=0.93. The western and central sub-watershed areas had the highest nutrient export and the highest cover of cropland, likely due to this multiplicative relationship.

Nutrient loading to Delta Marsh was highly correlated to land use within the watershed. My findings indicate that manipulated land, such as cropland, had higher total nutrient export to Delta Marsh than less intensely manipulated land. Croplands covered

76% to 96% of sub-watershed areas, and had a very strong multiplicative relationship with nutrient export. Water yield (mm) also had a strong relationship with TN and TP export, with r^2 of 0.82 and 0.85, respectively. The combinations of land use and water yield appear to drive nutrient loading to the marsh. Agricultural croplands release higher nutrient loads due to hydrological manipulation of land, such as tillage practices, drainage ditches, and fertilizer use (Correll et al. 1992).

Crop type may also influence nutrient export based on varying nutrient application across the watershed. The six major crop types in Delta Marsh were canola, spring wheat, soybeans, beans, corn and barley. The suggested fertilizer application is crop-specific and varies with soil nutrient composition and method of application (Manitoba Provincial Government, accessed December 2016). The spatial variation of crop type, nutrient application and land management likely causes spatial variation in nutrient loading into Delta Marsh. The eastern sections of Delta Marsh have higher cover of natural lands including shrubland, grassland, pastureland and forest. These types of landscapes are not intensively manipulated nor do they require extensive nutrient application. These three land types have a strong negative correlation with TN, TP, DIN and TDP exports in 2015 and 2016. Natural land does not require tillage or nutrient application, ultimately reducing volume of runoff and concentrations of nutrients (Beaulac and Reckhow 1982). The further east on the watershed, the lower the total nutrient exports. Land use manipulation increases water yield (Liu et al. 2008), and ultimately nutrient export in the Delta Marsh watershed. As mentioned, the western sub-watersheds have higher cover of manipulated land (cropland) and the east has higher natural land, resulting in varying water yields and nutrient exports. Additionally, the spatial variation of crop type within each sub-watershed area may result in differences in the areas of nutrient application, tillage and flow paths, resulting in varying nutrient loading. The difference in land use variation from west to east has led to a spatial gradient in nutrient loading in the Delta Marsh watershed.

Soil type influences the release or sorption of nutrients within the Delta Marsh watershed. Loss of N after fertilizer application depends on soil type in the watershed. Soils formed of clay/loam lose 10%-40% of N and sandy soils can lose between 25%-80% of N to the atmosphere and from runoff (Howarth et al. 1996; Carpenter et al. 1998). The western watershed is comprised of three different soil types, sandy loam and silty clay in large proportion, followed by Red River clay and small patches of Almasippi loamy sands (Experimental Farm Service 1958). The eastern portion of the watershed is strictly loam and clay loam, with the ability to bind P compounds and decrease N leaching (Experimental Farm Service 1958). The entire watershed is mostly made up of clay type soils, although the west has higher concentrations of sand (Experimental Farm Service 1958). The increase of nutrient export of N from the western section of the watershed may be attributed to the change in soil type.

This preliminary research indicates that there was temporal and spatial variation in nutrient loading to Delta Marsh. These differences were likely attributed to the timing of precipitation, land use differences and water yield. The varying levels of winter, spring and summer precipitation result in differences in spring snowmelt, and spring/summer precipitation, altering nutrient loading to Delta Marsh. The land use gradient within the watershed resulted in spatial variation of nutrient loading. The western and center sub-watershed area had the highest cover of cropland and the highest nutrient export to the marsh; the eastern sub-watershed area had the highest cover of natural land and the lowest nutrient export to the marsh. The relationship between cropland and nutrient export indicates that with increasing percent cover of cropland there is a multiplicative increase of nutrient export to Delta Marsh.

Recommendations

My preliminary findings indicate that total nutrient loading is strongly related to land use cover in each sub-watershed area. Cropland had the strongest positive correlation with TN and TP export to the marsh. An increased cover of cropland resulted in a multiplicative increase in nutrient runoff. Further investigation on land management practices, such as tillage, nutrient application, drainage and overall land management will provide further detail to the major influences on nutrient loading. This research will inform what land practices cause the greatest nutrient loading to the marsh and may be used to create land management plans to reduce nutrient loading to Delta Marsh.

Appendix C. Water quality points of interest in Delta Marsh, matched with their designated site number and coordinates. (Paired with Figure 2.1)

Site #	Site Name	Latitude	Longitude
1	South Clair Lake*	50.21195	-98.15620
2	Second Lead*	50.21814	-98.16740
3	Riley Bay	50.22284	-98.15322
4	Small Bluebill Bay*	50.23203	-98.13832
5	North Clandeboye Bay	50.23779	-98.11351
6	Mid Clandeboye Bay***	50.23639	-98.10210
7	Souix Pass	50.24946	-98.09215
8	Mid Waterhen Bay***	50.22122	-98.10503
9	South Waterhen Bay	50.21213	-98.11691
10	Southeast Bluebill Bay*	50.20658	-98.12863
11	St. Mark's Lake**	50.19949	-98.09927
12	Mid-east Bluebill Bay**	50.19499	-98.14267
13	Johnson Lake*	50.17749	-98.14128
14	Southwest Bluebill Bay*	50.18659	-98.15896
15	Gadwell Bay*	50.21522	-98.15077
16	Northwest Gadwell Bay	50.20511	-98.15809
17	East Wilson Lake	50.20372	-98.17746
18	North Wilson Lake	50.20790	-98.17875
19	High Point Lake	50.18797	-98.17392
20	South Blackfox Lake*	50.18968	-98.18862
21	Lyttle Bay	50.18212	-98.19394
22	Home Bay	50.17359	-98.20754
23	Tin Town	50.17126	-98.19407
24	Delta Channel	50.18194	-98.31329
25	Mid Cadham Bay	50.17714	-98.28871
26	South Cadham Bay *	50.16819	-98.27273
27	Pitblado's Channel	50.18265	-98.27468
28	Bell Bay**	50.18921	-98.27799
Site #	Site Name	Latitude	Longitude
29	Naegele Island**	50.18426	-98.25508

30	Portage Creek**	50.16133	-98.24880
31	Division Bay*	50.19749	-98.21037
32	22 Bay**	50.20365	-98.20242
33	Twin Lakes	50.20782	-98.19106
34	Canvasback Bay**	50.17472	-98.41682
35	South Big Lake*	50.15074	-98.41879
36	Mid Big Lake***	50.17056	-98.23456
37	North Big Lake*	50.18512	-98.44404
38	Weedy Bay***	50.17729	-98.40175
39	West Blind Channel	50.18053	-98.39144
40	Crescent Pond	50.18393	-98.40512
41	Portage Diversion	50.16927	-98.37184
42	East Blind Channel	50.17354	-98.36408
43	Northwest Eaglenest Bay***	50.17345	-98.34672
44	Lake Manitoba	50.25613	-98.11666
45	Portage Creek Bay*	50.16130	-98.23848
46	Mid Simpson Bay***	50.18431	-98.20473
47	Lake Manitoba	50.19288	-98.41235
76	Thompson Bay**	50.16084	-98.39410
79	Crossroads Bay	50.17030	-98.39817
Wye	Wye's Pond	50.22452	-98.11945

*Nutrients

** Nutrients, metals, pH, DOC and alkalinity

*** Nutrients, metals, pH, DOC, alkalinity and phytoplankton species biovolume

Appendix D. Water Quality Variables and Method of Analysis Protocol.

Variable	**Method of Analysis**
Euphotic Zone Depth	Depth (cm) at 1% surface light, calculated by the regression of logPAR. Photosynthetic Active Radiation (PAR) recorded every 10 cm with a LI-COR-LI-193 Spherical Quantum Sensor. Detection Limit: 0 cm
Secchi Depth	Mean depth of disappearance and reappearance of 8-inch diameter disk (cm).
Surface Water Temperature	Mercury Thermometer (˚C)
Total/ Inorganic Suspended Solids Concentration	Suspended Solids Drying Method for Total Suspended Solids (Standard Methods 1995). Inorganic Suspended Solids is recorded after incineration at 600˚C. Detection Limit: 0 mg/L
Phytoplankton Chlorophyll-a Concentration	Spectrophotometric Determination of Chlorophyll-a (Marker et al. 1980; Standard Methods 1995). Detection Limit: 0.4 µg/L
Turbidity	Micro 100 Turbidimeter Detection Limit: 0 NTU
Conductivity	YSI 85 Probe Detection Limit: 0 µS/cm

Appendix E. Methods for N and P analyses for water quality at ALS Environmental Laboratories (Winnipeg) following Standard Methods (2017).

Chemical Compound	Method of Analysis
DKN	APHA 4500 NorgD (Modified)
TKN	APHA 4500 NorgD (Modified)
NH_3 – Ammonia	APHA 4500 NH3 F
NO_2-L-IC-N-WP (Low Range Nitrite)	EPA 300.1 (Modified)
NO_3-L-IC-N-WP (Low Range Nitrate)	EPA 300.1 (Modified)
TN	TN= TKN + NO_3 + NO_2
P-T-COL-WP (Total Phosphorus)	APHA 4500 P Phosphorus
P-TD-COL-WP (Total Dissolved Phosphorus)	APHA 4500 P Phosphorus
P-TR-COL-WP (Total Reactive Phosphorus)	APHA 4500 P Phosphorus

Appendix F. Wind exposure (no units) for each site across Delta Marsh, Manitoba. Exposure was calculated using fetch distance (m) in four major wind directions (Equation 1).

Site Number	Site Name	Wind Exposure	Site Number	Site Name	Wind Exposure
1	South Clair	481	25	Mid Cadham	1473
2	Second Lead	226	26	S. Cadham	2213
3	Riley	270	27	Pitblado	46
4	Small Bluebill	222	28	Bell Bay	216
5	N Clandeboye	192	29	Naegele Island	534
6	Mid Clandeboye	818	30	Portage Creek	312
7	Souix Pass	361	31	Division Bay	513
8	Mid Waterhen	926	32	22 Bay	538
9	S Waterhen	580	33	Twin Lakes	271
10	SE Bluebill	297	34	Canvasback Bay	171
11	St. Marks	243	35	S. Big Lake	268
12	ME Bluebill	1163	36	Mid Big Lake	384
13	Johnson Lake	182	37	N. Big Lake	339
14	SW Bluebill	1264	38	Weedy Bay	406
15	Gadwell	185	39	W Blind Channel	37
16	NW Gadwell	265	41	Portage Diversion	32
17	E Wilson	489	42	E Blind Channel	20
18	N Wilson	125	43	NW Eaglenest Bay	528
19	High Point Lake	303	45	Portage Bay	302
20	S. Black Fox	1413	46	Mid Simpson	1808
21	Lyttle	1877	76	Thompson Bay	148
22	Home	400	79	Crossroads	67
23	Tin Town	2240	80	Wyes Pond	140
24	Delta Channel	89			