Historical Loading and Current Sorption Capacity

of

Phosphorus in the Sediments of Delta Marsh

By

Christopher Kevin Hope

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Department of Biological Sciences
University of Manitoba
Winnipeg, Manitoba

Abstract

Delta Marsh is a freshwater inland coastal wetland located along the southern basin of Lake Manitoba, Canada. Spanning over 19,000 hectares, Delta Marsh is a eutrophic wetland complex that was formerly a world-renowned destination for waterfowl hunting. Over the past half century, the marsh has been in a state of decline due to invasive plant and animal species, stabilized water levels, and eutrophication due to increased nutrient loading from surrounding development. This prompted a coalition of organizations, including Ducks Unlimited Canada, to begin the "Delta Marsh: Restoring the Tradition" project in 2008 aimed at revitalizing the ecological processes and human traditions of the marsh. The objectives of this study were to investigate the marsh sediments to determine their historic rates of phosphorus accumulation and the current ability to bind phosphorus. I determined how nutrient levels have changed in the marsh over time as well as the ability for surface sediments to act as a sink or a source of phosphorus to the marsh. This was accomplished by collecting two sets of core samples; ten 50-cm cores dated using ^{210}Pb and ^{137}Cs and twenty-two 5-cm cores utilized for phosphorus sorption experiments. All core samples were analyzed for sediment physiochemical parameters including phosphorus concentrations, metal cations, and organic matter. For the most part, surficial sediments acted as a phosphorus sink with some inter-season variability. Furthermore, I found that phosphorus in the sediments, and by proxy the marsh waters, have been both increasing and becoming more labile over the past century (particularly since 1960), potentially indicating increasing eutrophic conditions in the marsh. These results varied across the marsh, with the largest nutrient increases occurring in the west side of the marsh, and higher phosphorus concentrations in the emergent and wet meadow vegetation zones relative to the

open-water. Overall, these results indicate that although the marsh sediments are acting as a sink for phosphorus, the marsh is an increasingly eutrophic system, with a particular risk for algal blooms on the west side.

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Dedication

To my wife Nicole, you would think that after writing so many words here that I would be better at finding the right ones to properly thank you for all of your love and support during this process but I honestly don't think I will ever be able to. You stood with me through this crazy journey and provided support beyond what I could have ever expected. Without you this project would have never made its way onto the page and for that I will forever be thankful. Thank you for being the co-author of my life.

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1: Introduction

Wetlands are an essential component of the natural world whose existence provides many ecological goods and services (Håkanson and Bryhn 2011). Wetlands naturally remove pollutants, excess nutrients, and suspended particulate matter from waterways and are often referred to as "nature's kidneys". These systems also provide habitat to a multitude of species of fish, waterfowl, insects, mammals, and many others. Unfortunately, wetland loss and degradation due to anthropogenic influences, such as increased nutrient loading, sedimentation, and altered hydrology (Main et al. 2014), is increasingly a problem worldwide that threatens the health, function, and existence of these essential systems (UNEP 1995).

Delta Marsh is a freshwater inland coastal wetland located along the southern basin of Lake Manitoba, Canada (Wrubleski et al. 2016). The marsh covers between 150 to 250 km^2 depending on the water level, extends along approximately 32 km of the south shore of Lake Manitoba, and away from the lake for up to 4 km (Wrubleski et al. 2016). Delta Marsh is a eutrophic wetland complex that was formerly a world-renowned destination for waterfowl hunting (Goldsborough and Suggett 2015). Over the past half century, the marsh has been in a state of decline due to eutrophication caused by agricultural and residential development of the surrounding landscape, invasive plant and animal species such as Common Carp and hybrid cattail, and stabilized water levels due to regulation of Lake Manitoba to protect lakefront residential properties (Goldsborough and Suggett 2015). This prompted Ducks Unlimited Canada, along with the Delta Waterfowl Foundation, University of Manitoba, Manitoba Provincial Conservation & Water Stewardship, Department of Fisheries and Oceans,

Environment Canada, and the La Salle Redboine Conservation District, to come together and create a plan for the restoration of the marsh . This initiative resulted in the “Delta Marsh: Restoring the Tradition” project in 2008 aimed at revitalizing the ecological processes and human traditions of the marsh. As a component of that initiative, the objective of this study was to investigate the marsh sediments to determine their historical accumulation of phosphorus, current capacity to bind phosphorus from the water column, if these factors vary spatially across the marsh, and why.

1.1: Phosphorus

Phosphorus (P) is a macronutrient that is essential for all forms of life as it plays a critical role in cellular energetics, as well as a building block for cell growth in plants and animals (Zhang et al. 2009; Zhou et al. 2005). In nature, P is never found in its pure elemental state and exclusively exists as compounds with orthophosphate (PO_4^{2-}) being the only form that is environmentally reactive, or available for uptake by plants (Ironside 2001). The largest reservoir of P is contained within rocks such as apatite whose weathering is the only natural source of P additions to the environment, where it is taken up by plant roots and cycles through the food chain within plant and animal cells (Ironside 2001). P returns to the soil through the decomposition of organic matter or via urine and feces as P salts. When P returns to the soil it is either directly taken up by other organisms, converted from organic P back to orthophosphate by phosphatizing bacteria, or bound into an insoluble and nonreactive form (Miller and Ricklefs 1999). Phosphate is highly reactive and readily binds with many elements such as calcium, aluminum, or iron to form insoluble compounds (Reddy et al. 1999; Tercero et al. 2017). P does not naturally exist in gaseous forms; however, it can be present in the atmosphere within

airborne particulate matter or precipitation as water-soluble phosphate. Atmospheric P is generated through erosion of rock or soil and settles back to aquatic and terrestrial systems through deposition, though a study of Canadian lakes determined that atmospheric sources only account for 1-6% of P inputs (Ahl 1988) and as such reasonably can be ignored in most studies.

The development and increased agricultural use of P-rich fertilizers has altered the natural cycles of P in the environment by allowing P to be applied beyond its natural rate or distribution. In addition to intentional agricultural applications, P is also being added to the environment through anthropogenic activity such as municipal and industrial wastewater discharge, emissions from industrial sources, and runoff from cities and farms. These inputs result in increased levels of biologically active P in terrestrial and aquatic ecosystems that in turn allow for increased rates of plant growth and alter the composition of affected ecosystems (Ironside 2001). The significance of these nutrient increases is particularly important for freshwater ecosystems as plant growth is often limited and determined by the availability of P (Correll 1998; Zhang et al. 2009; Ironside 2001).

Although it is an essential nutrient, at levels beyond what is required for normal plant growth, P can be considered a pollutant as it leads to adverse effects in the environment such as cultural eutrophication (Carpenter et al. 1998; Sharpley 2000). It has been long established that P is often the limiting nutrient for plant growth in freshwater systems (Zhou et al. 2005) and that P enrichment can lead to an overproduction of algae to the detriment of other organisms. These algal blooms are caused because algae are able to utilize additional sources of P more efficiently compared to larger macrophytes, altering the species distribution in an

ecosystem (Ironside 2001). Algal blooms can be considerable in scale and can result in the reduction or elimination of other aquatic plant species (Faassen et al. 2012), decreased light infiltration to the water column, creating anoxic conditions, fish kills (Boyer et al. 2009), disruption of food webs and food availability (Li et al. 2010), and increased resuspension of sediments and turbidity (Ironside 2001; Carpenter et al. 1998). Cyanobacteria, or blue-green algae, that thrive in algal blooms, can produce multiple forms of toxins that are harmful to the health of other organisms and in humans can result in respiratory issues, liver failure, and multiple forms of neurological degradation (Li et al. 2010; Faassen et al. 2012). These blooms are also aesthetically unappealing, produce noxious odors, can impede navigation in waterways, and disrupt the recreational potential of a body of water (Ironside 2001).

To combat cultural eutrophication of aquatic systems, the most effective solution would be a reduction of the external inputs of P to the system (Cavalcante et al. 2018; Ironside 2001). Like all pollutants, however, elimination or reduction of P pollution to the environment is not an easy problem to solve; much of the P loading to the environment is a result of otherwise beneficial activities such as agriculture. Another potential solution to eutrophication is through the sequestration or removal of P from the system, a role for which wetlands such as Delta Marsh are naturally adept (Verhoeven et al. 2006; Zedler 2003). The ability of wetlands to reduce a wide array of pollutants, including excess P, from their waters as well as those in connected aquatic systems through retention is well established (Reddy et al. 1999). P retention is the ability of a wetland to remove P from its waters through physical, chemical, and biological processes and store it in forms not readily released under normal conditions, or transform it into forms that are not biologically available (Reddy et al. 1999). In some studies, it

has been shown that wetlands have the capacity to remove as much as 20-93% of incoming P loads, as well as other pollutants, earning them the nicknames of “green filters” or “natures kidneys” (Tercero et al. 2017).

The retention and removal of P in wetlands takes place through many different biotic and abiotic processes including assimilation by vegetation, periphyton and other microorganisms, chemical precipitation, and sorption by wetland soils and sediments (Haque et al. 2018). Although there are many aspects of wetlands that contribute to P retention and removal, the largest potential P sink is the wetland sediment (Attygalla et al. 2016). Depending on the system, between 80-90% of P in wetlands is stored in the sediments with organic matter, Ca and Mg compounds, metal oxides, and hydroxides representing the main pools of P (Tercero et al. 2017; Jenkins et al. 1971). While the role of wetland vegetation in the retention and removal of P can be significant, many studies have demonstrated that the impact of vegetation on P is considerably less than that of the soil (Tercero et al. 2017). It is of particular interest that, while emergent macrophytes effectively store P, very little of it is drawn directly from the water column; such plants draw their P and other nutrients from the porewater of the wetland sediments (Tercero et al. 2017).

Wetland soils and sediments act as P sinks, reducing the concentration from the overlying water column. However, it is also possible under the right conditions for sediments to act as a P source (Tercero et al. 2017; Kowalczewska-Madura et al. 2018). This internal P source can in some situations contribute P to the water body at levels comparable to or exceeding other external sources (H. Zhang et al. 2016) with some studies finding internal loading to represent as much as 99% of the P in shallow lake or wetland systems (Kowalczewska-Madura

et al. 2018). In some instances, internal loading can continue to release P for decades after external sources have been eliminated, disrupting the recovery of the system (Cavalcante et al. 2018). In this way, a wetland's greatest strength can also be its weakness as the sediment's ability to buffer P can work in both directions (Li et al. 2016). The mechanisms that dictate the sorption or desorption of P in wetland sediments are complex, depending on many physiochemical characteristics such as soil mineralogy, organic matter, particle size and distribution, calcium compounds, redox potential, pH, and the P varieties and concentrations present (Tercero et al. 2017). Obtaining an understanding of these factors, and what drives the P sorption/desorption reactions in the system, is essential to being able to understand, predict, and manage the wetland and its connected aquatic systems (Cavalcante et al. 2018).

1.1.1: Phosphorus Sorption

When P comes into contact with soils or sediments, a multitude of reactions may occur, including those referred to as P sorption where dissolved P binds with the soil and becomes unavailable. The term P sorption includes the fast process of P adsorption surface reactions as well as the slower reactions involving P diffusing into the solid phase of the soil structure (Li et al. 2016; Reddy et al. 1999). The initial reaction occurs quickly between the introduced orthophosphate and the soil minerals and organic compounds along soil reactive surface groups. In this exothermic ligand exchange reaction, a hydroxyl (OH^-) or H_2O molecule is released from the soil surface and replaced with a phosphate complex (Frossard et al. 1995). The speed and maximum capacity of this surface adsorption is governed by the amount and variety of available reactive surface sites and varies between soils depending on composition. Following the initial reaction are slower processes often attributed to solid (Barrow 1985) and

liquid (Radulovich and Sollins 1991) state diffusion within the soil. These reactions are also exothermic and take place via cation exchange within a crystal lattice or, more commonly in soils, by a vacancy mechanism where atoms redistribute to allow room for P atoms to be incorporated into the structure (Frossard et al. 1995) as well as through diffusion into existing micropores (Sui and Thompson 2000). In some cases, the slow P reaction also includes surface adsorption reactions at sites with lower accessibility located within the aggregates of poorly crystallized oxides (Frossard et al. 1995).

The total capacity for P sorption in a soil or sediment is finite and varies considerably between systems due to the content and morphology of clay and organic matter, soil PH, and abundance and availability of exchangeable cations such as aluminum (Al), iron (Fe), and calcium (Ca) (Casson et al. 2006). As P is added or removed from a water body in contact with sediments, sorption or desorption reactions take place. Which sorption reaction occurs is determined by the concentration of P in the solution, the P sorption potential of the sediments, and the amount of P the sediments already contain (Casson et al. 2006). As the P concentration in solution changes, these bidirectional reactions take place until a dynamic equilibrium is attained between the P in the soil and solution (Zhou et al. 2005). These reactions can greatly influence the quantity, bioavailability, and dynamics of P in aquatic and terrestrial systems.

The P sorption capacity of soils and sediments have been studied extensively for agronomic (Ozanne and Shaw 1967; Fox and Kamprath 1970) and environmental purposes (Hongthanat 2010; Zhang et al. 2005; Badiou et al. 2018; Sui and Thompson 2000). Many methods have been developed for the quantitative determination of P sorption parameters. This must be taken into consideration as parameters similarly named and defined may vary in

calculation, and therefore interpretation, between studies. These discrepancies arise due to differences in variables such as time, resources and facilities, personnel expertise, the different environmental systems under consideration that may need to influence procedures, and the individual motivations behind the research in question (Bai et al. 2017; L. Zhang et al. 2016; Wang et al. 2016; Hongthanat et al. 2016; Reddy et al. 1995; Ige, Akinremi, and Flaten 2005).

1.1.2: Phosphorus Parameters: S_{maxT}, S_{max1}, S_{max2}, K_1, K_2, S_n

For any soil or sediment, there is a limited capacity for the potential sorption of P (Frossard et al. 1995). This maximum P sorption capacity (S_{maxT}) represents the point where the soil or sediment is fully saturated and is the limit beyond which no additional P could be sorbed. The S_{maxT} can be compared to a measured concentration of P to determine if there is capacity for the soils or sediments to sorb additional P or not as well as estimate the total amount. The S_{maxT} can be used to assess the likelihood of P loss from soils into aquatic systems as well as the P availability for uptake by plants (Casson et al. 2006).

Corresponding to the sorption capacity is K, a dimensionless constant that represents the P binding energy (Zhang et al. 2009). A higher K value represents a sorption/desorption reaction that is slower and less likely to occur relative to a lower K value. The K values for various soils can be used to compare the relative risk of P loss to waterways based upon the comparative difficulty of a sorption/desorption reaction taking place. The potential for sorption/desorption reactions to occur however depend on a multitude of factors, such as the soil P concentration and maximum capacity, and as such K values are used in conjunction with other parameters.

While the S_{maxT} indicates the total capacity for P sorption in a soil, it does not provide details about the specifics of the sorption reactions taking place, specifically in soils with more than one type of binding reaction or sorption energy (Sui and Thompson 2000). For this purpose, methods (such as that used in this study) have been developed to provide insight as to what reactions were occurring at the high and low energy sorption sites denoted by S_{max1} and S_{max2}, paired with two binding energy constants, K_1 and K_2, respectively (Sui and Thompson 2000). This expanded model enables a greater understanding of the sorption/desorption reactions taking place and allows for better comparisons between samples sites and systems. For example, a soil or sediment with the majority of its sorption capacity associated with the high-energy binding sites would be slower and less likely to sorb P compared to one with the majority of its sorption capacity associated with the low-energy binding sites, even if both samples had the same S_{maxT}. Once the sorption reaction has occurred, high-energy sorption reactions will bind P more strongly and be more resistant to subsequent desorption.

The amount of P present in a soil or sediment upon sampling is called the native P, represented by S_n (Zhou et al. 2005). This parameter can also be referred to by several other names including initial P, or legacy P. The S_n is a useful parameter as it represents the actual amount of P presently sorbed by a soil or sediment sample, giving insight to the current state of the system under consideration. The S_n is valuable for assessing the potential risk of P loss from soils or sediments to waterways and as such is of considerable interest for environmental applications (Zhou et al. 2005). Generally speaking, a higher S_n sample will be more likely to desorb P under the correct conditions than a sample with a lower S_n; however, this reaction

depends on multiple other factors such as the maximum sorption capacity and as such is not often used in isolation.

The S_n is important to consider during the determination of other P sorption parameters as it will readily take part in sorption/desorption reactions and failure to take it into account will yield incorrect and misleading results with all subsequent sorption parameters (Zhang et al. 2009). In soils enriched in P, such as those nearby agricultural lands, the S_n can be particularly important (Zhang et al. 2009).

1.1.3: Equilibrium Phosphorus Concentration: EPC

The S_{maxT}, S_n, and K parameters discussed above provide insight about the P in a system; however, as sorption/desorption reactions are complex in nature these parameters are of limited use in isolation. To make a more thorough determination on the expected behavior of P in soils or sediments in an aquatic system, scientists developed a parameter called the equilibrium P concentration (EPC). The EPC represents the concentration of P in a solution (specifically dissolved reactive P or DRP) in contact with sediment where no sorption/desorption reactions will take place (Zhou et al. 2005). The EPC is used to determine if sediments of an aquatic system will act as a sink or a source of P to the water column, providing useful insight to the potential P cycling in the system (Zhang et al. 2009). If the EPC is greater than the P concentration in the water, then the sediments will act as a P source and release P into the water column until equilibrium is reached (Hongthanat 2010). Conversely, if the EPC is lower than the P concentration in the overlying water, the sediments will act as a P sink and sorb P from the water column, decreasing the P concentration until an equilibrium is reached (Badiou et al. 2018).

Determination of the EPC provides information about the internal movement and interactions of P within a system and can be useful for informing scientific and management decisions for an aquatic system. In sediments that are saturated with P and possess high EPC values, the internal P load of the sediments can accelerate eutrophication of the overlying waterways (Kang et al. 2009). Similarly, sediments with low EPC values will be better able to mitigate eutrophication due to additional P entering the waterway. These sorption/desorption interactions can occur quickly and in localized areas as conditions fluctuate in a water body; inflows of freshwater during precipitation events can strip P from sediments with high enough EPC values and similarly P-rich runoff can deposit P into sediments with a low enough EPC value (Hongthanat 2010).

The EPC has been used to easily categorize or rank which sites within a system are at the highest risk for P loss from the sediment to the water column; the higher the EPC value, the more likely it is that the sediments will act as a source of P (Zhou et al. 2005). In a large system such as Delta Marsh, this could be applicable as the various bays may act differently in response the same P concentration in the water. Determination of a site's risk of P loss based on EPC is superior to that of using S_{maxT} as concentrations as high as S_{maxT} are often not actually found in nature and water quality would generally be impaired at much lower concentrations than the maximum (Brand-Klibanski et al. 2007).

There are several factors that can influence the EPC value of sediments. Inputs of P to the system through soil or water runoff can contribute to the P concentration sorbed in the sediments, incrementally raising the EPC over time (Zhou et al. 2005). Sui and Thompson (2000) found that, following applications of biosolids high in P, the EPC of soil increased significantly.

EPC values are often higher in sediments that are anoxic (Reddy et al. 1998) and can fluctuate appropriately as conditions change. Under anoxic conditions, crystalline forms of Fe can partially dissolve and change to more amorphous forms with increased surface area for P sorption. Although this increases potential P sorption, it can also lower the P bonding energy and raise the overall EPC; these reactions can be complex and difficult to accurately predict (Hongthanat et al. 2016). EPC can also be influenced by sediment particle size distribution, ionic strength of the overlying water, pH, organic matter content, and sediment chemistry (Ige et al. 2005).

1.1.4: Phosphorus Equilibrium Buffering Capacity (PEBC) & Degree of P Saturation (DPS)

Another parameter used to predict the environmental fate of P is the P equilibrium buffering capacity (PEBC). The PEBC represents the capacity of soils to attenuate P additions in the related water column (Schroeder and Kovar 2008). A higher PEBC indicates a greater ability for the soils to sorb additional P and has been used for assessments of plant nutrition and environmental pollution (Sui and Thompson 2000). The PEBC is not as commonly used as EPC or DPS for environmental purposes but can still provide useful information when comparing different sample sites. PEBC is often used to rank various sites within a system as to their relative ability to sorb additional P, and resist a change in P water concentration (Hongthanat 2010).

The degree of P saturation (DPS) is another parameter developed to assess the risk of P loss from soils or sediments to waterways. The DPS is defined as the ratio of occupied P sorption sites (P sorbed in the soil) relative to the total available sorption capacity (Casson et al. 2006). The potential for P loss from soils or sediments depends on both of these variables in

conjunction and, as such, DPS is more useful for predicting the mobility of P than S_{maxT} or S_n alone (Ige et al. 2005). A study by Hooda et al. (2000) determined that the amount of P lost from soils into waterways had little to no correlation with maximum sorption capacity or soil P concentration yet a strong correlation with DPS. The DPS can also be called the P sorption index (PSI) or P saturation index (P_{sat}) depending on the literature.

As the DPS increases, the risk of P loss from soils or sediments to waterways increases (Casson et al. 2006). DPS has therefore been used in agricultural settings to determine what fields are at the highest risk for causing eutrophication to nearby waterways and informing management decisions accordingly. The DPS for agricultural fields has been found to be strongly related to P concentrations in runoff (Casson et al. 2006). One study found that there was little to no risk of P leaching to waterways where DPS was less than 10%, with P leaching increasing linearly with DPS above that threshold (Hooda et al. 2000). Bai et al. (2017) found similar results with negligible desorption at low DPS levels but an exponential increase in P loss as DPS increased.

In many places, critical limit DPS values have been established to guide management of agricultural lands and waterways. In The Netherlands, a critical value of 25% was established where, beyond this value, the risk of P loss due to runoff is unacceptable and additional applications of fertilizer may be prohibited (Zhang et al. 2005). Similarly, DPS thresholds in Florida and Delaware (U.S.) were set at 20% and 15%, respectively, while Quebec (Canada) with the strictest surface water guidelines set their threshold at 9% (Casson et al. 2006). Comparisons between DPS thresholds is difficult, however, as each region utilizes a different calculation to determine DPS, with oxalate-extractable Fe+Al being used in The Netherlands and

Mehlich-3 Fe+Al in the United States. Neither of these methods would be applicable to Delta Marsh as they are not suitable for calcareous soils, and at this time there is no established DPS guideline for determining risk of P loss for Manitoba soils (Ige et al. 2005). Although DPS guidelines from other regions cannot be strictly applied to Manitoba soils, they can still be used informally as a reference to assess general P conditions.

1.1.5: Forms of Phosphorus in Sediments

In soils and sediments, P exists in a number of forms, also referred to as fractions or pools, that exhibit markedly different characteristics and therefore present different implications (Cavalcante et al. 2018). The relative proportions of each form of P depends upon many factors including soil texture, mineralogy, organic matter content, pH, microorganisms, and the P sorption capacity (Reddy et al. 1999; Obikoya 2016; Min Li et al. 2013). Knowledge of the specific P fractions in a soil or sediment, as opposed to just the total P (TP), is essential to understanding and predicting P dynamics within the system and soil P availability (Saleque et al. 2004; Obikoya 2016). Identification of the P fractions is important for the purposes of our study to determine P availability and for the ability to predict if sediments will act as a sink or a source for P (Cavalcante et al. 2018).

The modified Hedley (1982) procedure for P fractionation used in this study identifies H_2O-P, $NaHCO_3$-P, NaOH-P, HCL-P and residual-P fractions. The terminology used for P fractionations is operational in nature, per convention, based upon the extraction method used for each constituent (Reddy et al. 1999). The various fractions of soil P can be divided in many ways with perhaps the most important being labile (H_2O-P and $NaHCO_3$-P) and non-labile (NaOH-P, HCL-P and residual-P) (H. Zhang et al. 2016; Cavalcante et al. 2018). Labile P is

considered to be available for uptake by plants or otherwise released from sediments while non-labile P is non-bioavailable and unlikely to be released from sediments (H. Zhang et al. 2016). In soils and sediments, P can also be found in organic or inorganic forms with the distribution dependent on the nature of their origin (Reddy et al. 1999). Organic forms of P are usually associated with living organisms and therefore, with a few exceptions, generally not available or reactive. Conversely, inorganic P compounds are generally more available and reactive and are associated with cations such as Fe, Al, and Ca (Reddy et al. 1999).

H_2O-P is considered to be the most "loosely sorbed" and bioavailable form of P as it is extractable simply via water (Huang et al. 2015). It corresponds to the non-occluded P in interstitial waters of the sediment and can be rapidly dissolved and desorbed (Cavalcante et al. 2018), or taken up by plant roots (Saleque et al. 2004). The H_2O-P fraction is labile, inorganic P. In this fraction, P may be loosely bound with Al or Fe (Loh et al. 2013).

$NaHCO_3$-P is the most available form of P following H_2O-P, representing the remainder of the non-occluded labile pool of sediment P. Also called reductant-soluble P, $NaHCO_3$-P is associated and bound to redox sensitive oxides such as iron and manganese that are reduced to allow for extraction (Loh et al. 2013). This fraction is labile because its sorption/desorption reactions are greatly influenced by redox conditions and can be released from soil under anoxic conditions (Reddy et al. 1999). This is primarily due to iron hydroxide ($FeOH_3$) which has a strong capacity to sorb inorganic phosphate from the water column under aerobic conditions but under anoxic conditions is reduced from iron III to iron II and readily releases said P, making it bioavailable (Cavalcante et al. 2018). The majority of $NaHCO_3$-P is inorganic P; however, there

can be a portion of $NaHCO_3$ extractable organic P representing species that can undergo rapid microbial breakdown and become available for uptake by plants (Reddy et al. 1999).

NaOH-P, sometimes referred to as metal oxide-bound P or alkali extractable P, is generally considered to be a part of the non-labile P pool although some sources consider it to be an intermediary between the labile and non-labile forms of P as it can become bioavailable under certain conditions. Primarily associated with metal oxides, NaOH-P is primarily non-occluded P bound to clays, aluminum and iron (Reddy et al. 1999; Loh et al. 2013). Similar to $NaHCO_3$-P, NaOH-P can be released from sediments under anoxic conditions although to a lesser extent as NaOH-P is bound more strongly to sediments (Cavalcante et al. 2018). NaOH-P is considered to be unavailable for uptake by plants due to its low solubility in mild extractants. Although primarily represented by inorganic P, NaOH-P can also contain a sizable organic component represented by P bound to humic acids associated with plant roots (Huang et al. 2015; Saleque et al. 2004; Loh et al. 2013).

HCL-P, sometimes called acid-P or calcium-bound P, represents the P fraction that is sensitive to low pH conditions (Reddy et al. 1999) and is a component of the non-labile P pool. HCL-P is occluded in the sediments, associated with calcium, apatite, and other carbonates and can only be released from sediments when the pH of the environment becomes sufficiently acidic (Reddy et al. 1999). HCL-P is non-bioavailable and difficult to release from sediments due to its high stability in the soil matrix resulting in HCL-P being a chief contributor to permanent sequestration of P in sediments (Cavalcante et al. 2018).

Residual-P is defined operationally as the P fraction that is not extracted from the soil or sediment through any step of the extraction sequence (Reddy et al. 1999). Being the most stable P fraction, residual-P is non-labile and non-bioavailable and would be expected to be a contributor to permanent sequestration of P in sediment (Cavalcante et al. 2018). In fact, residual-P is sometimes considered to be separate from the non-labile P pool as it represents the fraction that is even less available (Reddy et al. 1999). Residual-P is usually calculated as the difference between measured TP levels and all P fractions determined through the extraction sequence (Yan et al. 2017). The residual-P fraction, also called the recalcitrant fraction, may contain several different P compounds as it represents anything left over from the extraction sequence but is primarily represented by calcium-bound forms of P.

1.2: Sediment Chronology

The determination of the age of an object or sample is a useful tool, particularly in the environmental sciences. For this reason, there have been several techniques developed to determine the age, or date, of different types of samples.

1.2.1: Radionucleotide Dating

Lead-210 (^{210}Pb) is a naturally occurring radionucleotide that is ubiquitous in soils and sediments around the world (Gundersen 2017) and is one of the most commonly utilized types of dating methods for recently accumulated (100-150-year-old) soils and sediments (Arias-Ortiz et al. 2018). This timeframe often allows for detection of changes due to natural or anthropogenic influences and is well-suited for use in conjunction with management decisions. The integration period of ^{210}Pb into sediments being decades-to-a-century means that ^{210}Pb

profiles are not impacted by inter-annual variability and can be utilized to detect and monitor changes in a system over a period of time (Arias-Ortiz et al. 2018).

Initially developed in the early 1960s (Appleby and Oldfield 1978), the method has been used extensively since that time in all manner of systems including wetlands, fields, forests, and all manners of agricultural activity (Craft 2007; Kuzyk et al. 2004; Clarke 2018; Zuo et al. 1991) and is still widely utilized today. ^{210}Pb is a product of the Uranium-238 (^{238}U) decay series and as such its use in geochronology is based upon the principles of radioactive decay with older samples presenting lower levels of ^{210}Pb (Bowen and Johnson 2019). A common application, which is used in this study, is the extraction of core samples of soils or sediments which are then segmented and analyzed for ^{210}Pb to determine the age of deposition of each layer. The units used to report ^{210}Pb activity are decays per minute per gram (dpm/g) or becquerel (Bq) with 1dpm=60Bq (Clarke 2018).

Sediments in aquatic systems acquire ^{210}Pb from three sources which form the basis for their chronologic determination (Kamula 2015). First, ^{210}Pb is continually produced *in situ* by the decay of naturally occurring radon-226 (^{226}Ra) which is present universally as a constituent of the soil (Bowen and Johnson 2019). As it is continually being produced, this ^{210}Pb is present at a consistent level throughout the sediment matrix and is referred to as "supported ^{210}Pb" or "background ^{210}Pb". The supported ^{210}Pb is deemed to exist in a radioactive equilibrium with the ^{226}Ra in the soil (accumulation rate = decay rate) and as such the supported ^{210}Pb can indirectly be measured by determination of the ^{226}Ra (Clarke 2018). The decay of ^{226}Ra in the soil also results in the production and emission of the inert gas ^{222}Ra to the atmosphere. Following a short lived (3-4 day) decay series, ^{226}Ra decays into ^{210}Pb which returns to the

surface through wet or dry fallout and attaches to soil or sediment particles through ionic bonding (Bowen and Johnson 2019). This atmospheric ^{210}Pb is the second source to the sediments and is referred to as "unsupported ^{210}Pb" or "excess ^{210}Pb". The third source of ^{210}Pb occurs in the surface waters of aquatic systems, once again through the direct decay of ^{226}Ra, and is also classified as unsupported (Krishnaswami and Cochran 2008). When dissolved ^{210}Pb comes into contact with suspended particulate matter, it binds together and consequently settles to the sediment surface through a process called particle scavenging (Kamula 2015). This process is generally considered to be a much smaller source of unsupported ^{210}Pb relative to that from the atmosphere and is often overlooked (Kuzyk et al. 2004). The occurrence of this process has been found to be correlated with salinity in the system and as such is more prevalent in seawater than freshwater systems.

The application of the ^{210}Pb dating method relies upon the satisfaction of several assumptions to be accurate: ^{210}Pb in the atmosphere quickly and efficiently settles and binds to the soils or sediments, ^{210}Pb is immobile once deposited, supported ^{210}Pb levels are uniform across sediments and throughout sediment depth, excess ^{210}Pb does not migrate down to deeper sediments, and that ^{210}Pb is in radioactive equilibrium with ^{226}Ra (Clarke 2018). In practice, these assumptions often are not all fully satisfied which must be kept in mind during analysis and interpretation of the results.

Due to the potential for errors with regards to ^{210}Pb dating, results need to be validated, usually using a second radionucleotide such as cesium-137 (^{137}Cs) (Arias-Ortiz et al. 2018). The ^{137}Cs, a byproduct of nuclear fission, only exists in the environment through anthropogenic inputs and was not present prior to 1945 (Wright et al. 1999). Introduced via atmospheric

nuclear weapons testing ^{137}Cs settled to the surface and bound to soils and sediments similar to ^{210}Pb. ^{137}Cs was primarily produced between 1952 and 1962 with as much as 57% concentrated in a 16-month period between September 1961 and December 1962 (Wright et al. 1999). Following an international treaty banning nuclear testing in 1963, ^{137}Cs inputs to the environment have drastically declined with only negligible additions since that time (Wright et al. 1999). Due to the intensive testing taking place between 1961-1962, there exists a significant and clear (in undisturbed sediments) ^{137}Cs peak representing 1963 when the greatest amount of fallout took place (Clarke 2018). This clear identification of 1963 in the sediment record is commonly used to validate dating profiles created through the analysis of ^{210}Pb. In some areas of northern Europe, a second smaller peak representing 1986 can be detected representing the incident at Chernobyl (Kamula 2015).

1.2.2: Difficulties with Sediment Chronology Profiles

The use of ^{210}Pb and ^{137}Cs dating profiles can be complicated in aquatic systems due to disturbances that mix and redistribute sediments following initial deposition (Kuzyk et al. 2004). The primary source of sediment mixing is due to bioturbation by benthic organisms with physical mixing due to wind and tidal action also playing a factor in systems that are sufficiently shallow (Zuo et al. 1991). Of particular interest to this study is the Common Carp which have been identified to be a significant source of bioturbation and sediment mixing (Huser et al. 2016). In systems with sufficient sediment mixing, it is often impossible to definitively assign dates to sediment layers (Johannessen and Macdonald 2012) although general age ranges and important information can often still be determined. In sediments with sufficient biomixing, it is likely that any dates that are assigned to sample segments will be an overestimate (older) than

the true value, as biomixing will drive objects and sediments deeper than it would be via burial (Johannessen and Macdonald 2012). By accounting for potential mixing and sediment compaction and validating results through the use of multiple radionucleotides, sediment accumulation rates may be able to be determined (Kamula 2015).

Previous studies have demonstrated that vegetated coastal environments, such as that found at Delta Marsh, usually present a greater challenge for dating compared to sediments from lake or marine settings (Saderne et al. 2018). Such open systems receive sediment and fluxes of ^{210}Pb and ^{137}Cs from both internal sources as well as externally from the connected aquatic system or the landscape within its contributing watershed (Arias-Ortiz et al. 2018). These external inputs can vary greatly over time and are especially impacted by storm events (Turner et al. 2007) or in response to changes in land use (Saderne et al. 2018). Once within the system, sediments can also be disturbed and redistributed through wind, currents, bioturbation, or anthropogenic activity such as fishing or hunting (Arias-Ortiz et al. 2018). In these situations, the complex sediment accumulation and redistribution can violate the assumptions of the radionucleotide dating models resulting in inconsistent profiles of ^{210}Pb and ^{137}Cs that are difficult or impossible to interpret.

Both ^{210}Pb and ^{137}Cs have a powerful affinity for binding to fine sediments (clay/silt) and organic matter and as such any process that alters these parameters in the sediment are likely to result in non-ideal radionucleotide profiles (Arias-Ortiz et al. 2018). Delta Marsh sediments contain high amounts of these constituents (Geard 2015), as well as a likely high amount of sediment disturbance due to the prevalence of Common Carp (Badiou and Goldsborough 2010) and, as such, difficulties in determining sediment chronology is expected. A review of the

literature by Arias-Ortiz et al. (2018) determined that the most common process that result in anomalous ^{210}Pb activity and sediment profiles are: sediment mixing, changes in sedimentation rate (primarily increasing), erosion, changes in sediment grain size, and the decay of sediment organic matter. Unfortunately, due to anthropogenic impacts at Delta Marsh most of these factors, to varying degrees, can be expected.

1.2.3: Sediment Chronology Models, Profiles, and Inventories

Due to the usefulness of sediment dating as well as the complexity and variability of aquatic systems many different models have been developed since ^{210}Pb dating was first utilized in 1972. Of these models, three are commonly used: constant flux-constant sedimentation (CF-CS), constant initial concentration (CIC), and constant rate of supply (CRS)(Arias-Ortiz et al. 2018). Each of these models operate under a different set of assumptions; however, they all assume a steady state of ^{210}Pb deposition, no mobility of ^{210}Pb following deposition, and that new radionucleotides are sequentially deposited above previous ones (Arias-Ortiz et al. 2018).

The CF-CS operates under the assumption of a consistent ^{210}Pb input and sediment accumulation rate which makes it unsuitable for many systems where sediment inputs are expected to vary over time (Arias-Ortiz et al. 2018). The CIC model assumes that the initial level of unsupported ^{210}Pb at the sediment-water interface will always be constant regardless of the rate of sediment accumulation such that the ^{210}Pb and mass accumulation rate covary and requires a monotonic decline in ^{210}Pb activity with decreasing depth. The benefit of this model is that the age of any sample in the soil profile can be determined through analysis of the ^{210}Pb

activity if the initial unsupported ^{210}Pb is known (Arias-Ortiz et al. 2018). While useful in many settings these assumptions are rarely satisfied in coastal wetlands such as Delta Marsh.

Of the three main ^{210}Pb dating models the CRS is the most commonly used in lakes, estuaries, and vegetated coastal ecosystems. This is likely due to the model not exhibiting problems with variable sediment accumulation rates or non-monotonic ^{210}Pb decline, which are common factors in many systems. The CRS model can, however, encounter issues in systems with prolific sediment mixing. The CRS model (or constant flux model) assumes a sustained constant rate of ^{210}Pb addition to the sediments. As the unsupported ^{210}Pb is constant but the rate of sediment mass accumulation can vary, the initial specific activity will also be variable and inversely related to the rate of mass accumulation. Dating with the CRS is determined by comparing the levels of ^{210}Pb activity moving down the sediment core and in some cases can be aided by calculating the overall ^{210}Pb inventory (Arias-Ortiz et al. 2018).

Once the segments of a core sample have had associated dates determined, profiles can be constructed to assess how various parameters have changed over time. If the sample segments are analyzed for nutrients, metals, or other physiochemical parameters such as organic matter then the value obtained for each segment can be assumed to be representative of the state of the surface sediments as the corresponding date. Sedimentation rates (cm year^{-1}) or mass accumulation rates (g cm^{-2} year^{-1}) can determine the rates of deposition through history to assess how the system may have changed over time. Caution must be taken in interpreting profiles, however, as many physiochemical parameters change over time within the sediments and cannot be taken to be an exact representation of past historic levels. As an example, some soil bacteria are able to uptake and transform organic P into inorganic P

although it takes time and as a result organic P can decrease as it is buried longer and deeper in the sediments (Min Li et al. 2013).

Nutrients and other physiochemical parameters can be investigated using the raw measured values to see how they have changed over time as well as calculating more sophisticated parameters such as accumulation rates accounting for nutrient mass and area (mg $m^{-2}yr^{-1}$) for a more in-depth assessment of the changes over time. The P in sediments actively interacts with P in the overlying water column, through sorption/desorption reactions seeking a dynamic equilibrium, until they are buried by subsequent sediment deposition and cut off from continued sediment-water interactions. When this happens the P concentrations in each subsequent layer of sediments can act as a preserved "snapshot" of the P concentrations that were present in the water column at the time those sediments were deposited. The nutrient profile can therefore be used as a proxy to provide an estimate of the P status of the aquatic system as a whole and investigate potential historical changes.

By summing the total amounts of $^{210}Pb_{ex}$ and ^{137}Cs in a sediment core inventories can be calculated. The $^{210}Pb_{ex}$ and ^{137}Cs inventories can be used to quickly and easily assess entire cores without further analysis and can be helpful in situations where the dating of cores is not possible (Kuzyk et al. 2004). Inventories can also eliminate the need to account for sediment mixing and bioturbation as vertical disturbance should not alter the total amount of $^{210}Pb_{ex}$ or ^{137}Cs. Inventories can be compared to the regional atmospheric flux/fallout (known or estimated), or if the study site is relatively small atmospheric deposition can be assumed to be consistent across sites (Kamula 2015) allowing for direct comparisons of inventories between sites. Similar inventories indicate consistent sedimentation rates and low rates of sediment

disturbance. Higher or lower inventories indicates differential conditions between sites which could be due to sediment focusing (higher $^{210}Pb_{ex}$; Kuzyk et al. 2013), additional particle scavenging due to resuspension (higher $^{210}Pb_{ex}$; Kamula 2015; Zaborska *et al.*, 2008), considerably changes in sedimentation rates, or horizontal movements of sediments due to disturbance which usually results in lower inventories however may also result in inventories that are higher (Edgington et al. 1991).

1.3: Geologic History of Delta Marsh

Before Delta Marsh or Lake Manitoba, there was glacial Lake Agassiz. At the time of the last ice age much of North America was covered by glaciers. As the glaciers that covered much of North America retreated for the final time approximately 12 000 years ago, their meltwaters pooled in an isostatic depression to create Lake Agassiz (Ojakangas and Matsch 1982). Lake Agassiz retreated from the landscape approximately 10 000 years ago leading to the formation of many lakes in its former basin including Lake Manitoba (Teller and Last 1980; Last and Teller 2002).

Some 6000-7000 years ago marked the formation of the Portage La Prairie alluvial fan (Rannie et al. 1989) which caused the repositioning of what is now known as the Assiniboine River to flow north into Lake Manitoba approximately 6 400 years ago (Teller and Last 1980). As the Assiniboine River began to discharge into Lake Manitoba, it brought large quantities of sediments which were deposited far into the south and west sections of the southern basin and formed a large delta south of the lake (Teller and Last 1980; Batt 2000). The course of the Assiniboine River continued to change over time and, between approximately 3000-4000 years

ago, it became disconnected from Lake Manitoba to move towards its present-day connection with the Red River and Lake Winnipeg (Rannie et al. 1989; Last and Teller 2002). The discharge of the Assiniboine River into Lake Manitoba caused significant changes to the regions geomorphology that continued after the river changed course once again.

Due to its large size and shallow nature, wave action on Lake Manitoba is mainly controlled by wind (Burrows 1969) and between approximately 3500 to 2500 years ago there was a period dominated by northwestern wind and waves (Rannie et al. 1989; Batt 2000). This wave action transported the sediment of the south basin previously imported by the Assiniboine River southeastward and redistributed it along a former shoreline to create a thin beach ridge isolating the southern section of the lake and creating what would become modern-day Delta Marsh (Teller and Last 1980; Rannie et al. 1989).

The establishment of the beach ridge began the formation of Delta Marsh. Physically, the marsh was isolated from Lake Manitoba except for several connecting channels. This resulted in the marsh being shielded from the more extreme physical and hydrological conditions of the lake while also experiencing its fluctuating water levels (Batt 2000). These conditions gradually allowed for the development of a thriving wetland complex that, although changing over time, still exists today.

1.4: Lake Manitoba

With an average surface area of 4 624 km^2, Lake Manitoba is the third-largest lake in Manitoba, 14th-largest in Canada, and 33rd-largest freshwater lake in the world. Located in south central Manitoba, the lake is approximately 225 km in length (north-south) and 52 km

wide (east-west) with 915 km of shoreline (Page 2011). The lake is generally shallow for its size with a mean depth of 4.5 m and a maximum depth of approximately 8 m. It is divided into two basins, north and south, with a connection that is appropriately referred to as "The Narrows". The south basin is considerably larger and comprises approximately 70% of the lakes surface area (Love and Love 1954; Teller and Last 1980; LMRRAC 2003).

The contributing watershed for Lake Manitoba is approximately 79 000 km^2 and includes most of west and central Manitoba and some of east and central Saskatchewan (LMRRAC 2003). The main natural hydrological inputs to Lake Manitoba are to the north basin via the Waterhen River, fed by Lake Winnipegosis, and to the south basin via the Whitemud River fed by southwestern Manitoba. In addition, the lake receives water via overland and small channel flow from its surrounding watershed, direct precipitation, and intermittently from the Assiniboine River via the Portage Diversion which when flowing increases the effective contributing watershed by approximately 142 000 km^2 (Infrastructure 2018). The topography of the Lake Manitoba drainage basin is generally a gentle slope from the western side at 317 masl (meters above sea level) to the south east at 238 masl and as such most of the inflowing water enters from the west with only a few small creeks contributing from the east. The westernmost edge of the drainage basin is approximately 60 km west of Lake Manitoba and generally follows the Manitoba Escarpment while the eastern edge is approximately halfway between Lake Manitoba and Lake Winnipeg. To the north and south the basin is bounded by the Cedar Lake/Summer berry and Assiniboine River drainage basins, respectively (Teller and Last 1980; Page 2011).

The sole outlet for Lake Manitoba is the Fairford River in the northeast corner draining into Lake St. Martin, the Dauphin River, and eventually Lake Winnipeg. Lake Manitoba experiences naturally fluctuating water levels ranging between approximately 246.88 and 248.72 masl (LMRRAC 2003). Waters in the north basin have a relatively short residence time of usually less than two years, while waters in the south basin are much more stagnant and can reside for as much as 20 years with more volume being lost to evaporation than the Fairford River (Goldsborough and Suggett 2015).

1.4.1: The Portage River Diversion

The Portage River Diversion, also known as the Assiniboine River Diversion, was completed in 1969 and reconnects the Assiniboine River to Lake Manitoba. Its purpose is to direct water from the Assiniboine River to Lake Manitoba to prevent flooding downstream in Portage La Prairie, Winnipeg, and the farmland along the river. The diversion of water from the lower Assiniboine River channel also helps to protect downstream dikes and riverbanks from damage caused from the potentially high flows of floodwater. Additionally, water that is always present flowing in the pilot channel of the Portage Diversion can be used by nearby farmers for irrigation (Goldsborough and Suggett 2015). The channel connects to the Assiniboine just west of Portage La Prairie and runs directly north to where it meets Lake Manitoba. The Portage Diversion is not directly connected to Delta Marsh but does bisect the marsh as it approaches the lake. This route through the marsh was chosen as it was deemed to have the least impact on the marsh among the available options (Goldsborough and Suggett 2015). As of 2018, the Portage Diversion has been used for 36 of the 48 years since its construction and has operated for a total of 1546 days (Infrastructure 2018). In general, each year that the Portage Diversion is

used, the increased flow into Lake Manitoba is able to be balanced by increasing flow out from the FRWCS with the exception of 2011 when the outlet was unable to keep up with the volume of water diverted into the lake, resulting in extensive flooding around the lake (LMRRAC 2003; Page 2011). In 2011, the Portage Diversion was in operation for its longest period to date at 126 days and carried approximately 1.72 million cubic decameters of water (Infrastructure 2018).

The Portage Diversion is approximately 29 km in length and has a capacity to carry up to 25 000 cfs (cubic feet per second) along with all of its associated sediment and nutrients (Last 1984). There is a failsafe built into the west dike near Delta Marsh that is the lowest point along its length. When the maximum flow capacity is reached and exceeded, waters overtop and wash out the dike at the failsafe and flood into the west area of Delta Marsh and the surrounding farmland. This failsafe was included to minimize overall damage to the structure. Although predictable, these situations still flood and damage large areas of marsh and agricultural land. Along with the floodwaters, these events result in deposition of large amounts of sediment and influxes of nutrients to these areas (LMRRAC 2003; Page 2011). Due to these flooding events, agricultural lands around the Portage Diversion have seen an increase erosion and experienced an increase in crop loss and decrease in variety of suitable crop options. While breaches in the Portage Diversion primarily flood land to the west, farmland on both sides including the east have also been subjected to increased flooding on an ongoing basis as the construction of the Portage Diversion blocked natural drainage channels that previously emptied into the marsh (Goldsborough and Suggett 2015).

With the volume of water that the Portage Diversion deposits to Lake Manitoba there can be considerable impacts on the lake. Although the Portage Diversion does not flow every

year, on average it contributes 11% of the total annual volume of inflows to the lake. Despite this level of contribution, the Portage Diversion contributes up to 87% of the TP load to Lake Manitoba and 87% of the Total Suspended Solids (TSS) (Nicholson 2012). These levels can be significant but are not consistent; for example, Portage Diversion flows in 2009 accounted for 82% of TP, 38% in 2010, and 87% in 2011 (Nicholson 2012). Due to the close proximity of the Portage Diversion outlet into Lake Manitoba and the nearby channels that connect the lake to Delta Marsh, a large volume of the water discharged by the Portage Diversion is believed to enter into the marsh and can have a considerable impact on the hydrodynamic and nutrient behavior of the marsh (Aminian 2015).

1.5: Delta Marsh

Delta Marsh (50°11'N, 98°19'W) is a freshwater inland coastal wetland located at, and connected to, the southern basin of Lake Manitoba, Canada (Watchorn et al. 2012). The marsh covers between 150 to 250 km^2 depending on the water level, extends along approximately 32 km of the south shore of Lake Manitoba, and away from the lake for up to 4 km (Wrubleski et al. 2016). Of the approximately 564 km^2 of coastal wetlands surrounding the shoreline of Lake Manitoba, Delta Marsh is the single largest (Watchorn et al. 2012). The marsh consists of a wide variety of features including large and small open-water bays, isolated ponds, channels of various size, former rivers sections, flooded emergent vegetation and wet meadows, islands, low prairies and uplands with varying levels of interconnectivity or isolation. The marsh is considered to demonstrate hydrodynamic characteristics comparable to that of a series of interconnected lakes (Aminian 2015).

Delta Marsh is commonly divided into three sections. The Portage Diversion meets Lake Manitoba within Delta Marsh and is a major landmark and influence on the marsh. Although the Portage Diversion bisects the marsh, there is usually no connectivity or direct flow between the Portage Diversion and the marsh and as such it is not considered as part of the marsh. The area of the marsh located to the west of the Portage Diversion is called west marsh and while connected to Lake Manitoba by two channels it is isolated from the rest of the marsh by the Portage Diversion. The area east of the Portage Diversion but west of highway 240 (which also bisects the marsh) is called center marsh and mainly consists of Eaglenest Bay, with a few surrounding ponds. Center marsh is not directly connected to Lake Manitoba or the west marsh but connected to east marsh through a culvert that runs under highway 240. To the east of highway 240 is the area called east marsh which is the largest area of the marsh, and comprises a greater area than both west and center marsh combined. East marsh is highly interconnected and connected to the lake by two channels, Delta Channel and Clandeboye Channel, at the east and west ends, respectively. The east marsh section contains many large open water bays while the west and center marsh sections are more sheltered with considerable amounts of emergent vegetation.

1.5.1: Biophysical Characteristics

Delta Marsh is classified as a barrier-protected wetland as it is separated and protected from Lake Manitoba by a beach ridge but is connected to it by four channels: Cram and Deep Creek in the west marsh and Delta and Clandeboye Channel in the east marsh (Watchorn et al. 2012). As water is able to flow freely through these channels, the water levels in the marsh are directly correlated to those of the lake (De Geus 1987). Through history the number of channels

connecting the lake and marsh has varied with there being a greater number in the past. As channels have been created or obstructed over time, the hydrology and morphology of the different sections of the marsh have changed. For example, center marsh was previously linked to Lake Manitoba through Eaglenest Creek until the mid-1930s when it was filled in by siltation and isolated from the lake (Bortoluzzi 2013).

The marsh is generally shallow with an average depth of approximately one meter and most areas within the range of 10 cm to 1.8 m (Walker 1965). Most of the bays and other areas within the marsh are between 1-1.5 m in depth and exhibit a gently sloping gradient while the deepest channels that were formerly rivers are as deep as 3-4 m and can have relatively steep banks. The gradual change in elevation within most of the marsh carries on to the watershed to the south and as a result the flooded area of the marsh can vary dramatically with a change in water level. This was demonstrated by the flood of 2011 where, according to satellite imagery comparisons, the open water area of the marsh was 56% greater than the same areas in 2009 (Geard 2015). Due to the generally shallow nature of the marsh, the water column does not experience significant thermal stratification and is relatively homogenous.

Due to the connectivity with, and hydraulic control by, Lake Manitoba, the water levels in the marsh vary considerably and are strongly correlated with changes in the lake over both long (annual) and short time periods (hours or days) (De Geus 1987). This trend has been demonstrated to hold true during some of the highest and lowest water level periods on record (Bortoluzzi 2013). Within an open-water season, marsh water levels can fluctuate between a few centimeters to as much as half a meter which is significant in a system like Delta Marsh where average water depth is approximately one meter or less (Bortoluzzi 2013). Long-term

variations in water level are generally due to climactic conditions. Annually, there is typically a rise in water level in the early spring as the freshet brings a rapid inflow of water to the marsh and lake followed by gradual drawdown through the summer due to evapotranspiration. Since the construction of the Fairford River control structure in 1961, long-term variation of the water level of the lake and correspondingly the marsh has decreased to a narrower range although short-term fluctuations are still more severe.

Short-term fluctuations in water level are often a result of seiche activity on Lake Manitoba. Given the large wind fetch along the lakes north-south axis (225 km) and how prevailing winds are primarily from the north/northwest or south/southwest, there can frequently be seiche activity of up to two meters on the lake (Malenchak 2004; Bortoluzzi 2013). During seiche events, wind pushes large amounts of water into or out of the marsh and can have impacts on the water level of the marsh similar to those observed on the lake. The beach ridge between Lake Manitoba and Delta Marsh protects the marsh from much of the destructive weather from the lake but does not prevent the significant changes in water level that occur. During large wind events the beach ridge can be damaged or altered by the high wind and water levels causing changes in the marsh morphology and in some cases the opening of new channels and connections between the lake and the marsh (personal observation).

Delta Marsh receives water via overland and channel flow from its watershed to the south, precipitation, Lake Manitoba, and groundwater recharge. Inputs from its watershed are variable throughout the open-water season with the majority of inflow happening during the spring freshet as snow accumulated over the winter melts and flows (Goldsborough and Suggett 2015). Throughout the remainder of the season, tributaries are typically dry except

after large rainfall events (personal observation). As there are no permanent tributaries to the marsh after the spring freshet (the large influx of water from melting snow), the only sizable water inputs come from Lake Manitoba and overland storm runoff. The largest tributary to Delta Marsh is Portage Creek, flowing northward into Cadham Bay. After the spring freshet, water in Portage Creek often stagnates and begins to flow bidirectionally as water levels in the marsh fluctuate and some consider it to be more so an extension of the marsh as opposed to a flowing stream (Aminian 2015). The marsh has a negative water budget (Geard 2015), meaning that the amount of water gained from precipitation in the average year is less than that which is lost through evapotranspiration. Without other sources of water input, it is estimated that the water level would drop by an average of 18 cm per year. Considering this negative water budget and local precipitation levels, the greatest source of water to the marsh comes from Lake Manitoba (Geard 2015). A study by Aminian (2015) on the east marsh determined that flow into the marsh is not divided equally between channels and that approximately 85% of water flowing into the marsh comes through Clandeboye Channel on the east side with the remainder passing through Delta Channel on the west.

The landscape surrounding Delta Marsh spans a range of different land uses. The northern beach ridge features the Delta Beach development which contains over 200 residential and recreational properties, a public beach, and campsite (Brown 2003). To the south is the Delta Marsh watershed which covers approximately 207 km^2. Dotted across the watershed are several small residential communities, numerous individual homesteads and farmhouses, and several recreational and commercial hunting lodges. The watershed is primarily agricultural land with 75-96% cropland cover (Stanley 2017). In many areas of the

marsh, agricultural land is developed up to the marsh shoreline while in others there is a buffer of natural grasslands and shrub land. Agricultural development across the marsh watershed forms a gradient from somewhat natural grassland and shrub land in the east to increasingly intensive agricultural cropland towards to center and west portions of the marsh (Bortoluzzi 2013). There are a wide variety of crops cultivated in the watershed including canola, wheat, barley, flax, hay, alfalfa, beans, corn, carrots, onions, and potatoes (Bortoluzzi 2013). Areas of land not suitable for intensive agricultural development are often used for cattle grazing with cultivated forage crops or utilized for production of hay.

1.5.2: Marsh Sediments and Substrata

Delta Marsh has an underlying layer of bedrock covered by up to 100 m of lacustrine sediment deposited by glacial Lake Agassiz and more recently river, stream and lake water (Last 1984). These sediments are relatively soft, high in organic matter, and have a generally fine particle size (Geard 2015). The sediments, or soils, are described as sandy loam or silty clay and have been categorized as a mixture of gleysols and regosols (Walker 1965; Batt 2000). According to the Canadian System of Soil Classification, gleysols have properties that indicate prolonged periods of saturation with water, reducing conditions, and anaerobic conditions for at least part of the year. Regosols are classified as mineral soils that have been poorly developed and demonstrate poor drainage (Soil Classification Working Group 1998). The marsh sediments are typically overlaid by 5-10 cm of organic matter due to the slow rate of decomposition on the generally anaerobic marsh floor (Geard 2015). In areas with greater amounts of vegetation, this organic layer is typically much thicker due to a buildup of litter from the emergent vegetation. Like most wetlands, the sediment soils are saturated for most or all

the open water season and as a result are hydric in nature and generally exhibit anaerobic conditions.

A study by Geard (2015) identified that areas with thick plant matter substrate are generally found along the vegetated shorelines of the marsh and transition to a clay/silt substrate towards the center of the open water areas. The different sediment zones appear to be separated according to depth which is expected as sediments with a small particle size (such as clays) generally settle into the deeper areas of a large body of water while larger particles remain in the more shallow regions (Sly 1978). This pattern occurs because most sediments enter a water body from the periphery and settle to the bottom as they drift inwards. Resuspension of sediments occurs due to wind, waves, or the activity of benthic organisms which allows the sediment particles to drift farther into the water body before settling out again. As larger sediment particles are generally heavier, and resuspension and disturbance effects are generally weaker as depth increases, large particles tend to only reach a certain depth. Smaller particles are resuspended much more easily and as such have the opportunity to reach greater depths before settling for the final time (Sly 1978). Fine sediments like the ones found at Delta Marsh can be resuspended by wave action at depths up to four meters so considering that the majority of the marsh is less than two meters deep it would be expected that ongoing resuspension of marsh sediments is quite common and ongoing through the open water season (Cyr 1998).

1.5.3: Cultural and Scientific Significance

Delta Marsh has a long and rich history of nature, culture, and science. Host to an extensive custom of waterfowl hunting since the area was first settled, as early as 1925 sections

within the marsh were officially designated as Game Bird Refuges to protect and maintain this tradition. In 1982, it was given the designation of an Internationally Significant Wetland by the Ramsar Convention (United Nations Convention on Wetlands of International Importance 1971) due to its importance to the migratory waterfowl of North America (Gillespie and Boyd 1991). In 1988, it was recognized as a Manitoba Heritage Marsh and, in 1999, it was designated as an Important Bird Area. In another effort to protect the marsh from further development, it was designated by Manitoba Conservation as a Wildlife Management Area in 2006, resulting in over 110 km^2 within the marsh being set aside to be protected in order to maintain vital habitat for waterfowl and other species using the marsh.

Delta Marsh has had a long history of being used as a site for scientific and conservation efforts. In 1931, James Ford Bell established a duck hatchery at Delta which evolved into the Delta Waterfowl Research Station in 1938 and conducted pioneering research as the first wildlife research station of its kind in North America (Goldsborough and Suggett 2015). The University of Manitoba open the Delta Marsh Field Station in 1966 to conduct research of various other components of the marsh ecosystem including fungi, algae, plants, non-game birds, insects, amphibians, fish, mammals, meteorology and forest biodiversity (Page 2011; Wrubleski et al. 2016). The Delta Marsh Bird Observatory was established in 1995 to take advantage of the greatest songbird nesting density in North America and was the only station of its kind in Manitoba. Delta Marsh was also host to the Marsh Ecology Research Program (MERP) which was one of the largest and longest studies of its kind designed to span 1980-1989 and study the prairie wet-dry cycle and its effect on wetlands (Batt 2000).

1.5.4: The Decline of Delta Marsh

Beginning in the early 1960s, people started to notice negative changes in and around the marsh (Wrubleski et al. 2016). The natural functions and ecological integrity of the marsh were changing and with this was a decline of its corresponding benefits. Reports indicated a reduction in clarity of the water column, increases in algal blooms, reduced amounts and varieties of submersed aquatic vegetation, reduced and changing variety of emergent plant species, increasingly higher populations of Common Carp, infilling and loss of small or shallow ponds, and the loss of islands in the larger bays. Along with these changes, there has been a reduction in the populations of desirable wildlife such as waterfowl and muskrats as well as an overall less attractive marsh in terms of recreation (Bossenmaier et al. 1968; Hertam 2010; Watchorn et al. 2012; Goldsborough and Suggett 2015; Wrubleski et al. 2016). This significant decrease in recreation potential, specifically waterfowl hunting, has been the main factor behind recognizing problems occurring at the marsh and the corresponding efforts for its rehabilitation.

While it is difficult to pinpoint the specific reasons for the deterioration of an ecological complex the size of Delta Marsh, there are several factors that have generally been accepted as being primarily responsible. The exact contribution to the marsh's decline from each of these factors is impossible to quantify precisely but there is little doubt about their influence on the health of the marsh. The main causes for the changes in the marsh include water level stabilization of Lake Manitoba and Delta Marsh, invasive species such as Common Carp and hybrid cattail, and eutrophication due to anthropogenic changes in and around the marsh watershed (Watchorn et al. 2012; Goldsborough and Suggett 2015; Wrubleski et al. 2016).

Cyprinus carpio, or the Common Carp, is an invasive species of considerable concern in many ecosystems across the continent. The species originated in eastern Asia and was intentionally introduced to North America in the 1800s with the intention of providing a self-sustaining food source for the "common man's table" as it was commonly cultivated for this purpose throughout Europe and parts of Asia (McCrimmon 1968; Badiou 2005). The species was successful for itself spreading across the continent and establishing a dominant position in many ecosystems. The first record of introduction was in the United States in 1842 and the fish were first definitively documented in Canada in 1880. Common Carp were not recorded in Manitoba until 1938 and were first documented in the Assiniboine river in 1945 (McCrimmon 1968; Badiou and Goldsborough 2006). By 1947, the fish had made their way to the south basin of Lake Manitoba (Atton 1959) and were confirmed within Delta Marsh in 1952 (Aminian 2015); however, they may have been present prior. Today, Common Carp is the most abundant and widely distributed non-native fish in North America (Parks 2006).

The natural behavior of the Common Carp is often detrimental to the environments where they are present. In many cases, the impacts caused by the species are significant enough that they are referred to as ecosystem engineers. These changes are not only significant but also rapid; as early as 1850 in the US it was being reported that waterways containing Common Carp were being drastically altered "with almost unbelievable rapidity" and, by 1897, there was already widespread condemnation of the presence of Common Carp in Canadian waterways (McCrimmon 1968). In Delta Marsh, the introduction and proliferation of Common Carp is considered one of the significant causes of decline.

The feeding and spawning behavior of the Common Carp both result in significant bioturbation, mixing, and resuspension of the marsh sediment (McCrimmon 1968). This disturbance can result in the release of considerable amounts of nutrients, such as P, from the sediments which contributes to eutrophication (Stewart and Watkinson 2004; Badiou and Goldsborough 2010; Hertam 2010). The vigorous benthic activity also results in the uprooting and destruction of submersed and emergent aquatic vegetation increasing susceptibility to further sediment disturbance via wind (McCrimmon 1968).

The deterioration of Delta Marsh had been taking place prior to when Common Carp entered the ecosystem; however, the most significant periods of decline correspond to times where Common Carp populations would have been increasing. Common Carp were first documented in Delta Marsh in 1952 (Aminian 2015) and the marsh decline is considered to have truly accelerated in the 1950s and 1960s (Wrubleski et al. 2016). Hunters logbooks referring to waterfowl populations explicitly mention that the most rapid and severe decline of the marsh occurred after 1982, between 1982-1990 (Goldsborough and Suggett 2015). Not only did waterfowl populations crash at this time but the water became noticeably more turbid, the submersed vegetation that provided food for waterfowl disappeared, bulrush across the marsh was replaced with cattail and small islands within the marsh eroded away. These dates align with the dredging of Delta Channel and removal of the Clandeboye dam in 1982 which is considered to be the date where Common Carp truly obtained unrestricted access to the marsh (Goldsborough and Suggett 2015).

Studies conducted at Delta Marsh by Badiou (2005), Hnatiuk (2006), Parks (2006), and Hertam (2010), investigated the effects introduction and removal of Common Carp would have

on the marsh. Introduction of Common Carp was demonstrated to rapidly increase turbidity, suspended sediments, rates of eutrophication, rates of sedimentation, phytoplankton and algae while also decreasing submersed vegetation, benthic invertebrates, other fish species, water clarity, and light penetration through the water column. The removal of Common Carp from the system predictably resulted in the reversal of all these changes and in either case the rate of response in the system occurred rapidly. Most notable for this study Badiou (2005) determined that the approximate density of Common Carp expected within Delta Marsh of 400 kg/ha could increase internal loading of P levels in the water column an equivalent of 66%.

The Fairford River Water Control Structure (FRWCS) was completed and began its operation in 1961, beginning the period of water level stabilization on Lake Manitoba. This structure was constructed in an effort to control and regulate the water level of Lake Manitoba following an unusually long period of higher than normal waters in the mid-1950s. Water levels on the lake experienced natural fluctuations which were problematic for residents around the lake resulting in flooding that damaged property and infrastructure. Pressure from residents motivated the government to take action to control the water levels on several occasions ultimately resulting in the construction of the FRWCS which finally proved successful. The FRWCS is constructed on the Fairford River, the sole outlet for water from the lake. The structure operates using a series of stop logs that can be added or removed to raise and lower the level of the lake, respectively to mitigate draughts or floods (LMRRAC 2003).

The goal of the FRWCS was to maintain the lake level at the previous mean value while reducing the variation of water levels previously seen, resulting in an elevation target of 247.55 masl with a desired fluctuation range of ± 0.3 m. Actual water level fluctuations post-

construction have averaged approximately 0.4 m which is a considerable reduction opposed to the pre-construction variation of 1.93 m. Although the mean level of the lake has been preserved pre and post construction the variability of the lake and its wet-dry cycle has not been (De Geus 1987; LMRRAC 2003; Geard 2015).

Water levels in Delta Marsh are strongly correlated with those of Lake Manitoba and therefore the stabilization of the lake has also resulted in the stabilization of the marsh (De Geus 1987; Batt 2000). Unfortunately, the potential impacts of this regulation on the marsh were either not considered or deemed not to be of significant importance. Variation in water levels is essential to the health of a wetland ecosystem such as Delta Marsh. Wetlands such as Delta Marsh naturally go through a "wet-dry" cycle as water levels fluctuate; high water levels flood out emergent vegetation clearing the way for new growth and low water levels expose the seedbed and allow for regeneration and reestablishment of wetland vegetation and diversity (Grosshans and Kenkel 1997; Van Der Valk 2005). This cycle is also essential to replenishment of the wetland seed bank and maintaining the dynamic and productive nature of wetland systems. The ever-changing state of wetland vegetation is a major component of the high diversity of related animal species utilizing the ecosystem. Long-term stabilization of a wetland system has been demonstrated to result in significant degradation (Water and Shay 1992; Kenkel 1995; Van Der Valk 2005).

The stabilization of water levels at Delta Marsh has resulted in the system becoming trapped in an extended "lake marsh phase" and is believed to be responsible for several detrimental effects to the marsh. An extended lake marsh phase results in decreased productivity and species diversity (van der Valk and Davis 1978; Kenkel 1995; Grosshans and

Kenkel 1997) as the formerly dynamic system stagnates (Van Der Valk 2005). Since regulation began Delta Marsh has seen a significant shift in emergent vegetation to a dominance of hybrid cattail as discussed in Section 1.5.5.3. This dominance of cattail has resulted in deteriorated overall health of the marsh as well as decreased habitat for species that previously utilized it. This change is similar to that observed taking place at Netley-Libau Marsh which is also experiencing declined health and function due to a spread of hybrid cattail following water level stabilization (Geard 2015). It has also been recorded that extended periods of high water levels have resulted in increased shoreline erosion and sediment resuspension due to an increase in susceptibility to wind action (Burrows 1969; Malenchak 2004; Bortoluzzi 2013).

In a wetland system, the hydraulic regime is often the main influence on the composition and distribution of emergent and submersed vegetation. A fluctuating hydroperiod results in a diverse and productive community of plants whereas stable water levels often yield systems dominated by the most competitive species which results in lower species diversity (Kantrud and Millar 1989). The contribution from fluctuating water levels to the dynamic nature of a wetland ecosystem is due to the narrow range of tolerance for water levels of most species; as water levels fluctuate species unsuited to their new conditions are replaced by ones that are (Keddy 1983; Keddy and Ellis 1985; Pearsall 1920; Gleason 1926; Geard 2015). Extended periods of high water flood out emergent vegetation that is not suited to these levels and extended periods of lower water levels allow colonization by new species from the seed bank (Grosshans and Kenkel 1997; Shay 1999; Batt 2000; Van Der Valk 2005). In stable water condition, this reduced diversity is often dominated by a small number of highly competitive and often invasive species which has been demonstrated in Delta Marsh with the dominance of

hybrid cattail, *(Typha x glauca)* (Keddy and Ellis 1985; Shay and Shay 1986; Boers and Zedler 2008).

An analysis of aerial photographs by De Geus (1987) indicates that, in 1948, during a period of low water levels, cattail was only present in significant quantities in Clandeboye Bay. From this point, water levels generally increased until regulation in 1961 and along with it the highly competitive invasive cattail gradually expanded throughout the marsh. High water levels in the late 1950s and early 1960s resulted in the overall abundance of emergent vegetation in the marsh declining except for the cattail. After the construction of the FRWCS and the implementation of water regulation, the cattail population began to dominate. Water levels decreased and exposed empty mudflats that were rapidly colonized by cattail. Dominance was slow to shift entirely from phragmites to cattail though by 1972 cattail was dominant in all areas of the marsh except for center marsh. Between 1964 and 1997, cattail cover in Center Marsh increased by 819% while bulrush and open water areas declined 97% and 19%, respectively (Geard 2015). By 1980, cattail was fully dominant in all sections of the marsh and continues to be today.

Eutrophication is the enrichment of an aquatic system by increased levels of nutrients or minerals, usually in reference to nitrogen and P. Although these nutrients are essential to life, when they are present in excess they are considered to be a form of pollution. It is a naturally occurring phenomenon that occurs in all parts of the world however eutrophication can be greatly accelerated by anthropogenic influences (Khan et al. 2007; Hunter et al. 2012). Although not inherently harmful, eutrophication is often responsible for changes to ecosystems that are considered negative by the human populations that utilize them. Beginning in the early 1980s,

the United Nations Environment Programme recognized eutrophication as a result of human activities to be a significant problem on a global scale that requires our attention (UNEP 1995). Cultural eutrophication is described as the primary water quality issue facing the world (Khan et al. 2007).

Worldwide, eutrophication has been identified as deteriorating the health and function of aquatic systems and highlighted as an area of concern for the future (Khan et al. 2007; Li et al. 2010; Faassen et al. 2012). It is an ongoing and cumulative problem with increasing duration and frequency of eutrophic conditions leading to increasing severity of degradation to the impacted aquatic systems (Hunter et al. 2012). Natural eutrophication is caused by geological processes such as erosion, weathering, oceanic upwelling and as a consequence of natural nutrient cycling (Bricker et al. 2008). Cultural eutrophication, eutrophication caused by anthropogenic influences, is caused by a multitude of activities including direct effluent of wastewater from urban centers, industrial and agricultural runoff, consumption of fossil fuels (atmospheric deposition), or accidental discharge such as spills (Arheimer et al. 2004; Khan et al. 2007; Bricker et al. 2008). For both N and P non-point sources, such as agricultural runoff, have been identified as the main source of eutrophication in aquatic systems (Carpenter et al. 1998). Point-sources are, however, still cause for concern, particularly for P, and can contribute large concentrations of nutrients to the environment (Carpenter et al. 1998).

Wetlands such as Delta Marsh are one of the many systems impacted by cultural eutrophication and have been recognized to be deteriorating across the world as a result of anthropogenic activities (Khan et al. 2007). Many of the negative changes taking place at Delta Marsh are symptomatic of eutrophication. Wetland eutrophication results in the loss or

degradation of habitat, reduced biodiversity and shifting species dominance, changes in nutrient dynamics, decreased dissolved oxygen, fish kills, and increasing algal abundance leading to toxic algal blooms (Carpenter et al. 1998; Ironside 2001; Verhoeven et al. 2006).

Delta Marsh is suspected to have always been a eutrophic system; however, this condition is considered to have accelerated over the past century as a result of anthropogenic influences. The Delta Marsh watershed has experienced significant changes due to settlement and development of the region (Goldsborough and Suggett 2015) resulting in numerous new sources of nutrient inputs. The development of the beach ridge for residential and recreational purposes is likely to have resulted in new nutrient inputs considering that urban settlements are an identified and accepted source (Khan et al. 2007; Bricker et al. 2008). Due to the proximity to the marsh, effluent from the settlement would be easily introduced to the aquatic system either directly via runoff or indirectly due to sources such as septic tanks leaching into the groundwater.

Agricultural intensification within the Delta Marsh watershed has resulted in significant changes to the landscape and is considered to be one of the leading causes of eutrophication in the marsh (Nicholson 2012; Bortoluzzi 2013; Goldsborough and Suggett 2015; Wrubleski et al. 2016; Manitoba et al.; Stanley 2017). Prior to settlement, the region was dominated by natural tall grass prairie and forest that has subsequently and increasingly been converted to farm and pasture lands (Sveinson 2003). These tall grass prairies were hydraulically rough meaning that they retained precipitation for longer reducing the runoff rate and volume by allowing for increased infiltration and evaporation (Aminian 2015). Agricultural land comparatively exhibits the opposite conditions and, as such, fosters a much larger and rapid rate of runoff. The

development of agricultural lands in the Delta Marsh watershed also included an ever-increasing efficiency and extent of intentional land drainage moving water downstream to the marsh through systems of ditches, culverts, dikes, and more recently tile drainage (personal observation). These drainage systems function to move as much water as possible off the landscape as quickly as possible leaving no time for infiltration, uptake, or evaporation.

In the early period of settlement, the landscape was fertile and did not require any nutrient inputs; however, in the 1960s, soil nutrients were depleted leading to the application of large quantities of fertilizer rich in N and P (Goldsborough and Suggett 2015). This intensification in agricultural practices corresponded to the same time period that saw the decline of Delta Marsh to accelerate (Goldsborough and Suggett 2015). Not all of the nutrients applied to agricultural land are utilized and the excess that accumulates in surface soils can often be exported downstream via erosion or runoff, primarily in the dissolved phase, during precipitation events (Sui and Thompson 2000; Sharpley et al. 2011). Agricultural development similar to that taking place in the Delta Marsh watershed has been documented across the prairie pothole region to result in eutrophication and deterioration of related water bodies (Main et al. 2014) similar to what is being observed in the marsh. A study by Stanley (2017) determined that watersheds in the western and central sections of the marsh export more N and P to the marsh than areas in the east which matches the degree of agricultural development across the watershed.

The largest hydrologic input to Delta Marsh, like most prairie systems, is during the spring freshet when snow accumulated over the winter melts and flows (Van Der Kamp et al. 1999). During this time there is sufficient overland flow that numerous streams and minor

channels that are dry for the remainder of the year fill and flow into the marsh (Aminian 2015). During this period, the primarily agricultural land of the watershed is mainly uncultivated bare soil resulting in high rates of runoff and nutrient uptake with nothing present on the landscape to prevent or slow it. The combination of nutrient-enriched farmlands, effective and intentional drainage, seasonally bare landscapes and high freshet loading has resulted in increasing nutrient exports to the marsh and eutrophication.

Large portions of the Delta Marsh watershed are also occupied by grazing land for livestock which can be a significant source of nutrients. Livestock only incorporate between 20-40% of the P present in their feed into biomass, with the remainder passed directly as urine and manure (Ironside 2001). In addition to adding nutrient-rich manure onto the landscape, livestock grazing in or around streams or water bodies can deposit directly into the aquatic system and greatly contribute to eutrophication. The N/P ratio of manure is 1-4:1 while uptake by crops and pasture lands is 8:1, resulting in a buildup of excess P in the soils (Zhang et al. 2005). Delta Marsh has seen cattle along its shores since settlement began (Goldsborough and Suggett 2015). Livestock activity and hoof action can also lead to increased erosion along shorelines adding nutrient-rich particulate matter to the marsh as well as increasing turbidity (Hongthanat 2010). Overgrazing activity of livestock also reduces vegetative cover in grazing lands compared to native conditions resulting in decreased retention and uptake of precipitation, as well as increased runoff, erosion, and downstream eutrophication (Hongthanat 2010).

Another potentially significant contributing factor to eutrophication of the marsh is the Portage Diversion (see Section 1.4.2) which has long been a concern for the health of Lake

Manitoba and Delta Marsh. Constructed in 1969 and used more frequently as time progressed (Infrastructure 2018), the timelines of operation of the Portage Diversion and the decline of the marsh appear to be aligned. The Portage Diversion redirects a considerable volume of water from the Assiniboine River to Lake Manitoba that otherwise would not, and previously did not, make its way to the lake/marsh, and could be expected to result in changes to the lake. Water from the Portage Diversion has been recorded to carry high amounts of suspended solids, nutrients, and dissolved ions which increase turbidity and eutrophication in the receiving waters (LMRRAC 2003; Page 2011). During years of use the Portage Diversion accounts for an average of 11% of the annual inflow to the lake but as much as 87% of the total load of P and suspended solids (Page 2011; Nicholson 2012) which indicates that nutrient concentrations in the Portage Diversion waters are much higher than that found in the lake and other natural inflows to the lake.

A study by Aminian (2015) determined that, on years that the Portage Diversion is used, a large quantity of water flows through Lake Manitoba into east Delta Marsh. This water primarily enters the marsh through Delta Channel with much less entering through Clandeboye and as such bays closer to Delta Channel should be impacted to a greater extent than bays towards the far east. Interestingly, it was found that despite Delta Chanel being the main source of Portage Diversion water, it is responsible for a comparatively minimal amount of the total inflow of water to the marsh. Aminian (2015) therefore suggested that, if Delta Channel were to be closed, the eutrophic impact of the Portage Diversion could be minimized without noticeably impacting the hydrologic behavior of the marsh. After entering Lake Manitoba, water is moved into Delta Marsh by wind action where it disperses. Due to this dispersion,

when flow subsequently reverses, the water leaving the marsh is typically older water from the marsh and not the new water from the Portage Diversion. This effect means that the marsh can experience a net gain of Portage Diversion water even in a season where there is overall a net outflow from the marsh into the lake. In years of high water where the Lake St. Martin Emergency Channel is used to lower water levels on Lake Manitoba, the slow drawdown would cause large quantities of water to be drawn from the marsh resulting in concentration of any Portage Diversion water present in the marsh. This effect magnifies the potential impact that water from the Portage Diversion can have on the marsh.

When the Portage Diversion is overloaded and the failsafe breaches, and the waters rich in nutrients and suspended solids from the Portage Diversion discharge directly into the west marsh, it is expected that large quantities of nutrients and sediments are introduced to the system. In these situations, the marsh and its surrounding landscape can be completely flooded and remain so for quite some time. The surrounding landscape that is flooded contains agricultural crop and grazing land which, when inundated with floodwaters, may export nutrients from fertilizers and manure into the marsh system to which it is now hydraulically connected. In addition, during breach events the flow from the Portage Diversion is high enough to potentially transport large quantities of nutrient-rich soil and sediment and deposit them in the marsh. It is reasonable and suspected that due to these factors the west marsh area is considerably impacted by the operation of the Portage Diversion, more so than the remainder of the marsh. This is potentially supported by observations that algal blooms across the marsh are considerably worse in the west marsh area compared to the rest of the marsh.

Of the many negative impacts of eutrophication, the most commonly discussed is an increased abundance of algal blooms, an issue that has been increasingly observed at Delta Marsh over the last few decades. Increases of nutrients to waterways result in large increases of algal populations as their growth is limited by the availability of nitrogen or, in the case of many freshwater systems, P (Correll 1998; Ironside 2001; Zhou et al. 2005). Algae utilize additional nutrients quickly and efficiently to increase in population and as such exhibit a much greater response compared to other organisms such as larger macrophytes. These algal blooms present numerous detriments to aquatic systems such as Delta Marsh and the occurrence of algal blooms is often used as an indicator of wetland health (Boyer et al. 2009).

Cyanobacteria, or blue-green algae, flourish under nutrient rich conditions and can produce multiple forms of toxins that are harmful to the health of other organisms. In humans these toxins can cause respiratory issues, liver failure, dermatological problems, and multiple forms of neurodegeneration (Li et al. 2010; Faassen et al. 2012). These health risks are particularly an issue to public health when present in systems utilized for recreation such as Delta Marsh (Ironside 2001). Toxic cyanobacteria blooms have also been deemed to be responsible for the deaths of animals such as dogs or livestock in multiple cases.

Algal blooms can be considerable in size sometimes blanketing the surface and upper regions of the water column, reducing light penetration and availability for organisms that require it. Extended or frequent algal blooms can therefore result in reductions in the quantity of submersed vegetation, benthic flora and fauna, macrophytes, and any other organisms that utilize these for food or habitat such as waterfowl (Ironside 2001; Li et al. 2010; Sharpley et al. 2011; Faassen et al. 2012). After these large algal blooms die off the associated decomposition

can often consume a sufficient quantity of O_2 from the water body to result in wide scale fish-kills. With the decrease in submersed and benthic vegetation there can also be an increase in the resuspension of sediments that are no longer stabilized, further increasing turbidity as well as releasing additional nutrients which can once again drive more algal growth (Ironside 2001).

1.5.5: Restoring the Tradition (RTT)

The continued decline of Delta Marsh, once a world-renowned landmark, has drawn considerable attention and concern as it steadily deteriorated over the past century. The acceleration of this decline over the past few decades has galvanized stakeholders to more deeply investigate the causes of this decline and potential solutions. In 2009, Ducks Unlimited Canada, Delta Waterfowl Foundation, University of Manitoba, Manitoba Provincial Conservation & Water Stewardship, Department of Fisheries and Oceans, Environment Canada, and the La Salle Redboine Conservation District came together to combine resources and plan the latest attempt to rejuvenate the marsh. The goal of this new coalition was to determine the natural or anthropogenic mechanisms behind the deterioration occurring at the marsh and take steps to address these stressors and slow or ultimately reverse the its decline.

This project ultimately came to fruition in 2013 and was entitled “Restoring the Tradition” (RTT) by Ducks Unlimited Canada in recognition of the longstanding waterfowl hunting tradition that extends over a century and is the most significant contributor to the recognition of Delta Marsh on the world stage. The goal of the project was to restore the ecological and human values and traditions of the marsh. Designed to be a ten-year project, it would be divided in to two equal length phases. Phase one would consist of researching the marsh and gathering background information on topics including possible Common Carp

exclusion, nutrient dynamics, hydrology and morphology of the marsh and surrounding areas. These studies would build upon the considerable previous research that has already occurred at the marsh and would be used to inform the second phase of the project. Phase two would consist of implementing management plans based on the results obtained in phase one and monitoring of the response within the marsh. In addition to the work actively taking place at the marsh, the project would emphasize ongoing collection of information for use in future planning and management of the marsh.

This study is a component of the background research being conducted on Delta Marsh to expand our knowledge and inform future marsh management plans. Other studies conducted include hydrodynamic modelling of Delta Marsh and Lake Manitoba (Aminian 2015), a study on the spatial variation of water quality, algal production, and land use of the marsh (Stanley 2017), and the hydrology of the Delta Marsh watershed (Schellenberg 2017). As mentioned previously there have also been numerous studies conducted prior to the official beginning of the project to obtain background information and help guide scientific efforts.

Previous studies taking place at Delta Marsh, as well as other similar systems around the world, have identified the presence of Common Carp in the marsh to likely be a significant contributor to the deterioration of the marsh. Having already been suspected for decades studies by Badiou (2005), Hnatiuk (2006), Hertam (2010), and preliminary studies by Drs Gordon Goldsborough and Dale Wrubleski established a strong case for the negative impacts of Common Carp in the marsh. Based on the knowledge available about the interactions between Delta Marsh and the Common Carp, a project of Common Carp exclusion was proposed and later implemented.

Following multiple studies on the fish communities of Delta Marsh, the Common Carp exclusion project began in 2013 with the construction of sheet pile weirs across the channels connecting the marsh to Lake Manitoba. These structures were fitted with removable metal screens custom designed to keep 93% of the adult migratory Common Carp out of the marsh while minimizing impact to other native species. Since implementation, the Common Carp exclusion has experienced varying degrees of success each season with preliminary observations suggesting there have already been improvements in aquatic vegetation and water clarity.

The Common Carp exclusion project is expected to continue in perpetuity with responsibility for maintaining the structures being transitioned to the provincial government and, for West Marsh, private entities. There is hope that this project will be joined by others in the future such as efforts to control populations of hybrid cattail, and nutrient management.

1.6: Lake Francis

Farther to the east is another water body named Lake Francis. While not a part of Delta Marsh, Lake Francis has been included in this study as it extends the east-west gradient under investigation, specifically distance from the Portage Diversion, and will allow for additional comparisons to be made with the rest of the study sites in Delta Marsh proper. Being in close proximity to Delta Marsh, and also connected to Lake Manitoba, Lake Francis would be subject to relatively the same climate conditions, soil types, species of plants and animals, hydraulic influence from Lake Manitoba, and geologic history.

Lake Francis consists of an open body of water larger than any of the other sites in this study. The lake is sheltered by marshland around its entire circumference, although to varying degrees. To the south-southwest is a large area of marshland consisting of intermixed ponds and vegetated areas similar in size to the lake itself. To the east of the lake following a strip of marshland are developed agricultural fields. To the northwest of Lake Francis is a strip of marshland, a beach ridge containing residential developments, and Lake Manitoba, sequentially. The development of Lake Francis contains numerous residential properties spread across the length of the Lake Manitoba shoreline and is generally more intensively settled than Delta Marsh.

Lake Francis is utilized for recreation such as hunting by the residents of the surrounding development. An artificial channel was constructed to provide easy access and runs from the road at the entrance to the development into the northern point of the lake. Lake Francis has also been in a state of partial Common Carp exclusion with removable metal screens in the channels connecting it to Lake Manitoba.

1.7: Research Objectives

The overall objective of this project is to gain a greater understanding of the retention capacity, P distribution (concentrations, forms, and fractions), and historical accumulation of P within the sediments of Delta Marsh.

The first objective is to determine the current P sorption capacity of the surficial sediments as well as their ability and likelihood to act as a sink or a source of P to the water column. Surficial sediments will be analyzed to determine their potential sorption capacity and

calculate the equilibrium P concentration. I hypothesize that the marsh sediments are acting as a P sink within the marsh, because wetland systems throughout the literature reviewed were more frequently sinks than sources.

The second objective is to determine how the P forms and concentrations vary across the marsh and attempt to explain why. Sediment samples utilized in the other two objectives will be taken from across Delta Marsh as well as from three landscape positions: the open-water, the emergent vegetation zone, and the wet meadow zone. The sediment samples will also be analyzed for various P species and fractions as well as physiochemical parameters. Correlation and mixed-model analysis will be conducted to determine significant differences between landscape positions and marsh experimental units and evaluate relationships between variables. I hypothesize that the P concentrations will be greatest in the west marsh and decrease towards the east, because of the decreasing gradient of nutrient inputs from west to east and the influence of the Portage Diversion on the west side of the marsh. I also hypothesize that the P concentrations will be highest in the while being highest in the wet meadow and lower in the open-water, because of closer proximity to agricultural landscapes and a suspected higher amount of organic matter in the sediments.

The third objective is to determine if and how the historic rates of sedimentation and nutrient accumulation have changed over time. Sediment core samples will be extracted and analyzed to construct a nutrient profile spanning approximately the past century. Investigation into the accumulation rates of sediment, P, and other parameters will illustrate how conditions within the marsh may have changed. I hypothesize that overall nutrient and sedimentation levels have increased post-settlement of the region, because of increased levels of residential

and agricultural development across the watershed, with the largest increases found on the western side of the marsh where the most intensive development has taken place. I also hypothesize that the nutrient accumulations have been the most significant after the opening of the Portage Diversion, because of the high amount of nutrients it exports to Lake Manitoba and Delta Marsh.

2: Methods

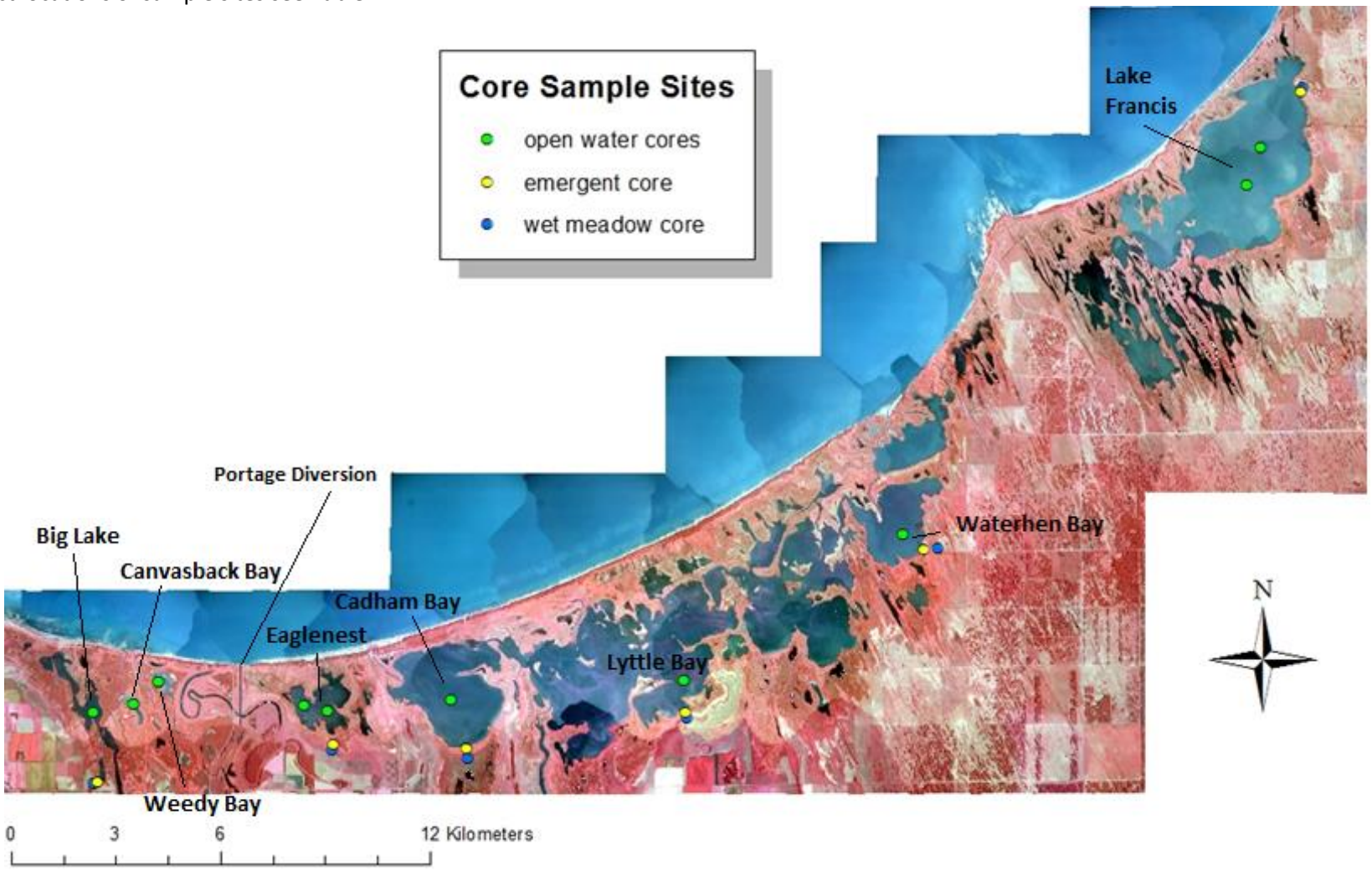
2.1: Site Description

The following is a general description of the bays selected to be the focus of this study. For a detailed description of Delta Marsh, the study area, see Section 1.5.

Big Lake was the westernmost site in this study located at the western edge of Delta Marsh (Figure 1.1). It is a long and relatively thin lake with points stretching from the northwest to the southeast. The west and south sides of Big Lake are encroached upon by developed agricultural land along its entire length although there is a riparian buffer zone present. To the immediate north is Lake Manitoba and to the east of Big Lake is the remainder of the west marsh portion of Delta Marsh. The depth in Big Lake is variable although like all sites in the west marsh is generally shallower than the rest of the sites in this study. The open-water site sampled is in the middle of Big Lake within the largest section of open water. The emergent and wet meadow zone sites associated with Big Lake are located to the south portion of the lake clearly within the associated vegetation zones and were selected based on accessibility as the land surrounding Big Lake is primarily privately owned. The open-water site was not located as close to the emergent and wet meadow zone sites as in the other bays in the study; however,

they are expected to be representative of their respective zones. All the sites within the west marsh are interconnected via channels. The sites within the west marsh are suspected to be greatly influenced by the presence of the Portage Diversion as west marsh is the discharge point when the failsafe is breached on the Portage Diversion. Big Lake is the farthest west marsh site from the Portage Diversion and as such it would be expected to be the least impacted of the west marsh sites.

Figure 2.1: False color infrared map of Delta Marsh and Lake Francis located in Manitoba, Canada, indicating the open-water, emergent and wet meadow zone sample sites, as well as the Portage Diversion. Map produced by Ducks Unlimited Canada and the University of Manitoba in 1997. For exact locations of sample sites see Table A2.



Canvasback Bay is located in the middle of the west marsh section of Delta Marsh (Figure 1.1). It is smaller and more sheltered than any of the other sites. Canvasback Bay is connected to Lake Manitoba via Deep Creek which contains one of the Common Carp exclusion structures from the RTT project. The bay is surrounded by the west marsh with no development in its immediate area save for a few access roads and simple wooden docks for use by hunters.

Weedy Bay is the easternmost site sampled within the west marsh (Figure 1.1). It is smaller and more sheltered than any of the sites outside of the west marsh, but larger than Canvasback Bay. Weedy Bay is the shallowest of any site in this study to the point that the small boats and canoes used in the marsh frequently bottom-out while crossing it. Weedy Bay is connected to Lake Manitoba via Cram Creek which also contains a Common Carp exclusion structure. This is the closest site to the Portage Diversion, specifically the closest to the failsafe, and as such would be expected to be the site most impacted by its overflow events. To the north is Lake Manitoba and on all other sides is the west marsh. The area surrounding Weedy Bay is undeveloped except for some access roads and wooden docks used by hunters who frequently use the bay as an access point into the marsh while hunting.

Eaglenest Bay comprises the majority of what is considered to be center marsh (Figure 1.1). It is larger and more open than any of the sites within the west marsh but smaller than the sites within the east marsh. To the immediate north of the bay is a small area populated by emergent vegetation followed by the residential development of Delta Beach consisting of a paved road and numerous cottages, a residential beach and campsite. To the immediate north of the development is Lake Manitoba. To the east and west is the remainder of Center Marsh consisting of some small ponds and bays within a vegetated area. Center marsh ends to the

east at highway 240 and to the west at the Portage Diversion. To the south are developed agricultural fields following a buffer consisting of wetland vegetation of varying thicknesses. Center marsh is not directly connected to Lake Manitoba but is connected to Cadham Bay in the east marsh via a culvert underneath highway 240, which in turn is connected to Lake Manitoba. The emergent and wet meadow zone sites associated with Eaglenest Bay are located to the south within the vegetated area.

Cadham Bay is the largest open-water area within Delta Marsh and is the westernmost site within the east marsh section (Figure 1.1). Immediately along the northern shore of Cadham Bay contains the majority of the residential development of Delta Beach, spanning nearly the entire length of the shoreline. This beach ridge contains the oldest developments at Delta Marsh, the research stations, and was the site of the original duck hatcheries. The west edge of the bay is bound by highway 240 and to the east is the rest of the east marsh. The south of Cadham Bay is a mixture of natural, undeveloped, vegetated land as well as developed agricultural fields. Cadham Bay is connected to Lake Manitoba via Delta Channel at the northwest of the bay and the rest of east marsh at the northeast of the bay through a channel called “the gap”. The majority of the water from the Portage Diversion that makes its way into the east marsh does so through Delta Channel, and as such Cadham Bay would be expected to be the site (except for West Marsh sites) most influenced by waters from the Portage Diversion (Aminian 2015). All east marsh is interconnected through either channels or across open water.

Lyttle Bay is located in the middle of the east marsh section of Delta Marsh along the southern shore (Figure 1.1). It is partially sheltered from the remainder of the otherwise open area of mid-east marsh named Simpson Bay by a small peninsula of land emerging from the

south shore and stretching along the bay's west and north side. To the south and southeast of the bay is developed agricultural lands following a buffer strip of vegetated natural land. On all other sides, Lyttle Bay is surrounded by the primarily open-water of east marsh. On the south shore of Lyttle Bay is a small "settlement" named Tin Town. The seasonal hunters that live at Tin Town have erected docks on the shore of Lyttle Bay and use it as an access point for entry into the marsh. The emergent and wet meadow zone sites associated with Lyttle Bay are located on the southern shore within the natural land between Tin Town, and the surrounding agriculture, and the open water of the bay.

Waterhen Bay is the easternmost site in the east marsh, as well as in all of Delta Marsh (Figure 1.1). To the east are developed agricultural lands. The buffer of naturalized landscape between the bay and the developed landscape is larger than in many other areas of the marsh. The landscape surrounding Waterhen Bay is less intensively developed than other locations sampled, due to the general gradient of agricultural intensification that declines from west to east across the marsh (Stanley 2017). The rest of the bay is surrounded by more of the marsh. To the south is an area of intermixed small bays and vegetated areas, to the west is the generally open water areas of the east marsh, and to the north is a beach ridge followed by Clandeboye Bay. Waterhen Bay is connected to Clandeboye Bay by Fish Creek and Waterhen Creek, both of which contain structures associated with the Common Carp exclusion project, and Clandeboye Bay is connected via a namesake channel to Lake Manitoba. Eighty-five percent of the water that enters into Delta Marsh from Lake Manitoba does so through Clandeboye Bay and as such Waterhen Bay would be the site most affected by Lake Manitoba waters in our

study (Aminian 2015). The emergent and wet meadow zone sites associated with Waterhen Bay are located on the southeast bank of the bay.

Lake Francis is located to the east of Delta Marsh (Figure 1.1), and for a description of Lake Francis see Section 1.6. The emergent and wet meadow zone sites associated with Lake Francis were on the east side of the lake within the surrounding wetland areas. After the initial sampling it was determined that the emergent zone sample indicated significant anthropogenic influence on the site, consisting primarily of rocky fill material, suggesting that the banks of the lake have been modified at some time in the past possibly for the purpose of facilitating access. The emergent zone was resampled some distance away and although appears to represent a natural soil the potential anthropogenic influence needs to be considered.

2.2: Phosphorus Sorption and Physiochemical Parameters

2.2.1: Experimental Design and Sample Collection

Eight bays within the Delta Marsh complex were chosen to be sampled (Figure 1.1, exact locations Table A2). These sites were chosen to represent the east-west range of the study area with samples taken from each of the marsh units to allow for comparisons across the marsh. Samples were obtained from sites located in the open water as well as in the emergent vegetation and wet meadow zones as delineated by the vegetation present. The emergent vegetation zone was defined as the area hosting cattail and other emergent wetland species, and the wet meadow zone was identified by the presence of species such as foxtail, other grasses and sedges, and saturated soil without any standing water. As water levels fluctuate, different regions of soils will become saturated or submerged and interact with the associated water body and, as such, it is necessary for our understanding to investigate the emergent and wet meadow zone soils as well as those usually expected to be submerged in the open-water.

Sediment core samples from the open-water sites were taken from the approximate center and deepest location of each bay to account for the effects of sediment focusing. Three sites were chosen from the west marsh including Big Lake, Canvasback Bay, and Weedy Bay. Two samples were taken from Eaglenest Bay in the center marsh. Three sites in the east marsh included Cadham Bay, Lyttle Bay, and Waterhen Bay. Two samples were also taken from Lake Francis. Replicate samples were taken from Eaglenest Bay and Lake Francis in order to facilitate statistical comparisons for these smaller marsh units. Emergent and wet meadow zone sample pairs were chosen to be taken at six locations (Big Lake, Eaglenest Bay, Cadham Bay, Lyttle Bay, Waterhen Bay, and Lake Francis) to represent the east-west gradient of the marsh. The

emergent and wet meadow zone samples were taken from the approximate center of their respective zones, relative to the edge of the open water. In total twenty-two samples were taken divided between ten open-water samples, and six each of emergent and wet meadow samples.

Sediment core samples at least 5 cm in depth were collected at ten open-water sites within Delta Marsh and Lake Francis between August and October 2015 (Figure 1.1). In addition, sediment samples were collected from the emergent vegetation zone, and wet meadow zones at six sites. However, due to accessibility issues and problems with some of the initial cores collected, some sites were re-sampled in July 2016. All sediment samples were collected using a handheld Watermark Universal Sediment Corer. Surface sediments (top 5 cm in depth) were sampled in triplicate at each site and composited on site into a single sample for analysis. Samples were collected on calm weather days and care was taken to ensure sediments were not disturbed on approach to the site, the boat was stationary during sample collection, cores were taken from separate footprints, and agitation or disturbance to the cores was minimized while being transported to shore for processing. Samples were processed on shore, placed into new Ziplock freezer bags and transported in a cooler on ice to the laboratory at the University of Manitoba Biological Sciences department. Sediment samples were stored in a refrigerator at 4˚C until analyzed.

2.2.2: Laboratory Methods

Samples were inspected and any non-soil debris such as rocks, shells, or twigs were removed using forceps. Emergent and wet meadow zone cores required additional effort to separate the higher amount of grasses and roots from the soil to be tested as these do not

participate in the P sorption reaction. Once clear of debris, all samples were homogenized using a hand mixer and stainless-steel mixing bowl. The 0-5 cm core samples were divided with approximately one-third of the sample set aside for P sorption experiments and the remainder sent to the University of Manitoba Soil Science department for analysis of metal cations, organic matter content and P fractionation.

P sorption experiments were carried out following the methods described in (Badiou et al. 2018). One gram of cleaned and homogenized wet sediment was added to each of eight 50 mL Pyrex vials along with 25 mL of a 0.01M $CaCl_2$ solution containing P concentrations of 0, 0.2, 0.5, 1, 5, 10 and 20 mg L^{-1}. Wet soil was used to avoid the effects of drying which have been shown to promote errors (Achat et al. 2012). The initial P concentrations were prepared using high purity monobasic potassium phosphate (KH_2PO_4). A randomly selected P concentration was used as a duplicate in the eighth vial. This range of P concentrations is greater than what would typically be found in nature allowing for the adsorption sites to be completely saturated and the calculation of the maximum sorption capacity (Zhou et al. 2005). One drop of chloroform was added to each vial to eliminate any bacteria and inhibit microbial activity (Zhou et al. 2005; Huang et al. 2012). The sediment suspension was shaken on a Labquake Thermolyne shaker (eight rotations per minute) at a constant temperature of 20˚C for 24 hours to reach equilibrium (Bolster and Homberger 2007; Badiou et al. 2018; Cui et al. 2018). After equilibrium, samples were briefly allowed to settle before being decanted through pre-dried and weighed Whatman GF/C filter papers in a vacuum filter. Once most of the filtrate had been removed, vials were rinsed several times through a second filter paper to recover the entire mass of the sample. Analysis of the filtrate for Soluble Reactive P (SRP) was conducted

immediately using the molybdenum ascorbic acid method (Stainton et al. 1977; Mallarino and Sawyer 1999). Filter papers were initially dried for 24 hours in a desiccator, before being further dried for 24 hours in a drying oven at 100˚C and weighed to determine the dry mass of the sediment sample.

When considering the amount of P sorbed by the sediments both the P sorbed from the solution and the native P present in the sediment needed to be considered. The total amount of added P sorbed by the sediment was determined from the difference between the initial and equilibrium P concentrations and was determined using the following equation from (Kang et al. 2009):

(Equation 1.1): $S_a = ((C_o – C_e)*V)/M$

Where,
S_a = Amount of P sorbed by the sediment from the solution (mg kg^{-1})
C_o = Initial concentration of P in the solution (mg L^{-1})
C_e= Equilibrium P concentration in the solution after 24 hours (mg L^{-1})
V = Volume of the solution (L)
M = Dry mass of the sediment sample (kg)

To estimate the native P, or the P originally sorbed naturally by the sediment prior to collection, the modified Freundlich equation with a 1/3 exponent from (Tolner and Fuleky 1995) was utilized:

(Equation 1.2): $S_a = K*C_e^{1/3}-S_n$

Where,
S_a = Amount of P sorbed by the sediment from the solution (mg kg^{-1})
K = Dimensionless constant that represents the steepness of the slope for the Freundlich sorption curve
C_e = Equilibrium P concentration in the solution after 24 hours (mg L^{-1})
S_n = Native P, originally in the sediment (mg kg^{-1})

The S_n value estimates were obtained by plotting the equilibrium concentration (C_e) and P sorbed from the solution (S_a) (X and Y axis, respectively) sorption data and then fitting the modified Freundlich equation using nonlinear regression in SigmaPlot version 11.0 for Windows. This equation was chosen to estimate the native P as previous studies have concluded that it is the most suitable compared with other methods such as the Langmuir and two-surface Langmuir (Badiou et al. 2018). The total amount of P sorbed by the sediment (S_t) was then calculated by combining the native P and the P sorbed from the solution:

(Equation 1.3): $S_t = S_n + S_a$

The equilibrium P concentration (EPC) was defined as the aqueous-phase P concentration at which there is no net exchange of P between the water and sediments. When the dissolved P concentration of the water column was higher than the EPC sediments will act as a P sink and sorb P from the water column, while when dissolved P concentrations in the water column were lower than the EPC sediments will release P and act as a source. The EPC was obtained using a one-step Freundlich equation method as described by (Zhang et al. 2009):

(Equation 1.4): $EPC = [C_{eo}^{n} + C_{eo}V/MK]^{1/n}$

Where,
EPC = Equilibrium P concentration for the sample site (mg L^{-1})
C_{eo} = Equilibrium P concentration of the 0 mg L^{-1} vial after 24 hours (mg L^{-1})
V = Volume of solution in the vial (L)
M = Dry weight of the sediment added to the vial (kg)
K = Dimensionless constant that represents the steepness of the slope for the Freundlich sorption curve
n = Dimensionless constant of given value of 0.333

The K value used in this equation was obtained previously from the Freundlich equation while determining the native P (Tolner and Fuleky 1995; Badiou et al. 2018).

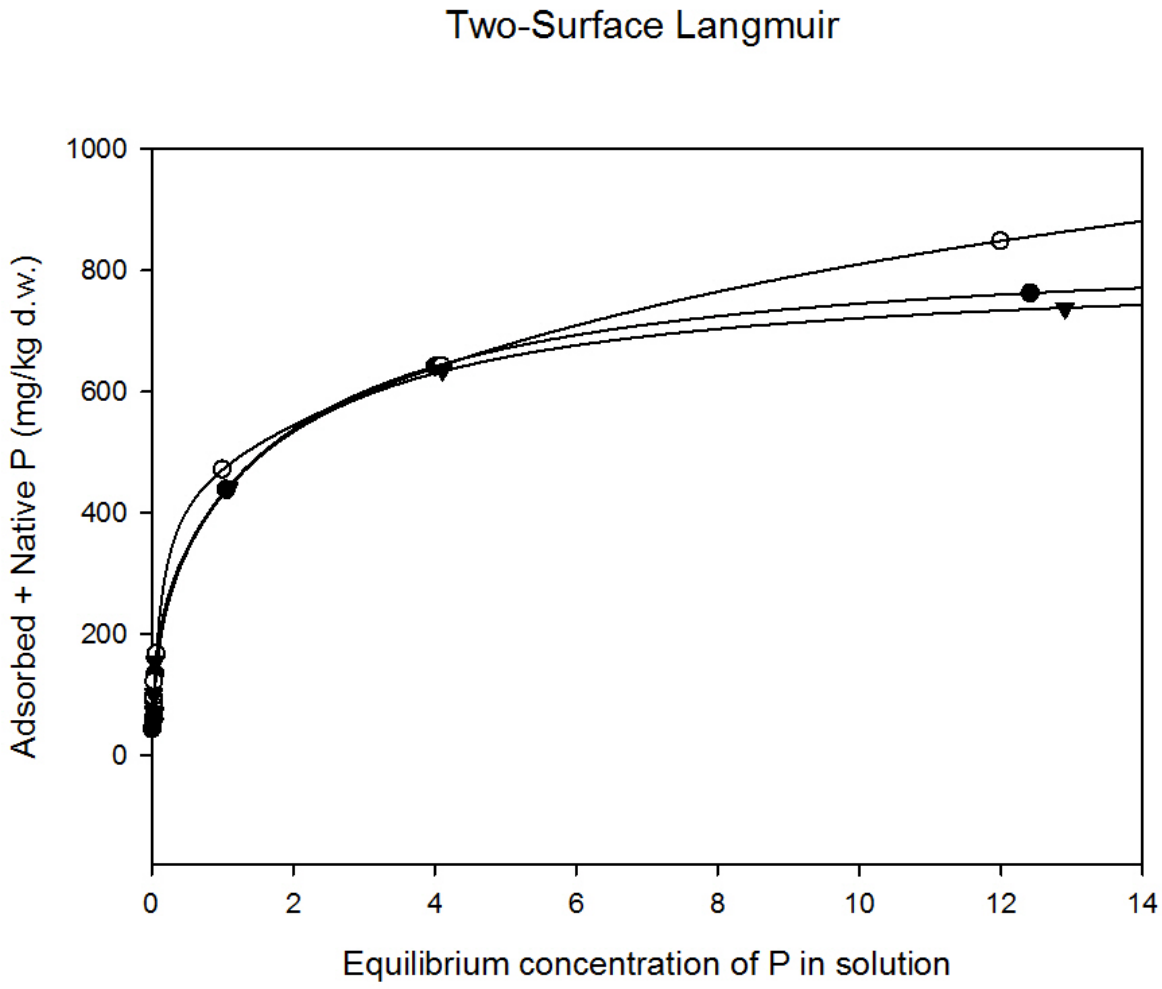
The equilibrium concentrations (C_e) and total P sorbed (S_t) were plotted (X and Y axis, respectively) and P sorption isotherms were modeled using the Two-Surface Langmuir equation (Sui and Thompson 2000; Bolster and Homberger 2007; Badiou et al. 2018). For an example of the P sorption isotherm see Figure 2.1:

(Equation 1.5): $S_t = ((S_{max1}*K_1*C_e)/(1+(K_1*C_e)))+((S_{max2}*K_2*C_e)/(1+(K_2*C_e)))$

Where,
S_t = Total P sorbed by the sediment (mg kg^{-1})
S_{max1} = Low-affinity P sorption maxima (mg kg^{-1})
K_1 = Low-affinity bonding energy for P sorption
C_e = Equilibrium P concentration in the solution after 24 hours (mg L^{-1})
S_{max2} = High-affinity P sorption maxima (mg kg^{-1})
K_2 = High -affinity bonding energy for P sorption

Estimates for the S_{max1}, K_1, S_{max2}, and K_2 were obtained by fitting the Two-Surface Langmuir equation using nonlinear regression in SigmaPlot version 11.0 for Windows. As a method of validation for the estimates obtained using SigmaPlot the same analysis was also conducted using the Solver Add-In function with an excel spreadsheet developed by (Bolster and Homberger 2007).

Figure 1.2: Sample of the Two-Surface Langmuir P sorption isotherm plotting the equilibrium concentration of P in solution and total P sorbed for triplicate samples from Northeast Eaglenest Bay at Delta Marsh, Canada.



The total maximum sorption capacity (S_{maxT}) for the soil was then determined by summing the high and low affinity P sorption maxima:

(Equation 1.6): $S_{maxT} = S_{max1}+S_{max2}$

The PEBC, representing the capacity of the soil to attenuate P additions to the overlying water column (Schroeder and Kovar 2008), was calculated by revisiting Equation 2 except only plotting the low concentration values where the sorption isotherm becomes approximately linear (Hongthanat et al. 2016). In this case the PEBC was represented by the slope (K) of the linear isotherm (Haque et al. 2018).

The degree of P saturation (DPS) is a function of the portion of soil exchange sites that are currently bound with sorbed P relative to the number of sites that are available (the maximum sorption capacity). The equation used to calculate DPS was from (Zhou and Li 2001) and (Casson et al. 2006):

(Equation 1.7): $DPS(\%) = (S_n/S_{maxT})*100\%$

A portion of each homogenized sample was transported to the Soil Science department of the University of Manitoba for chemical analysis following the laboratories standard methods as described in (Ige, et al. 2005). Total P and N were measured by digesting samples with a H_2SO_4–H_2O_2 mixture using the wet oxidation method described in (Akinremi et al. 2003). Organic P (mg kg^{-1}) was determined using the NaOH-EDTA (ethylenediaminetetraacetic acid) extraction procedure described in (O'Halloran and Cade-Menun 2007). Bioavailable P (mg kg^{-1}) was measured by extraction using iron oxide-impregnated filter paper strips as described in

(Menon et al. 1989). Organic matter (%) was determined though a wet combustion titration method involving oxidation by chromic acid and back titration with ferrous sulphate (Carter and Gregorich 2007).

Mehlich-3 extractable P, Ca, Mg, Fe and Al (mg kg^{-1}) were determined by equilibrating 2.5 g of air-dried soil sample with 25 mL of Mehlich-3 extracting solution for five minutes and filtering through Whatman No. 40 filter paper (Mehlich 1984). The various soil extracts were analyzed colorimetrically for P using the molybdate-blue method (Murphy and Riley 1962). Extractable Ca, Mg, Fe, and Al (mg kg^{-1}) were determined by inductively coupled plasma atomic emission spectrometry (Thermo iCAP 6500 Duo).

P fractionation was conducted following a modified version of the methods described by Hedley and Stewart (1982). Sediment samples were sequentially exposed to different extracting solutions of increasing strength and effectiveness to determine the concentrations of different P species, which were subsequently operationally defined by their corresponding extracting solutions. H_2O was used to remove the solution or loosely bound P, $NaHCO_3$ removed the adsorbed P, NaOH extracted P mainly associated with Fe and Al, and HCL contained some organic P but mainly Ca-bound P. The remaining P in the sample was then digested and termed Residual-P. The different soil extracts obtained were then analyzed for P colorimetrically using the molybdate-blue method as described in (Murphy and Riley 1962).

2.2.3: Statistical Analysis

Sorption and physiochemical parameters were tested for normality using the Shapiro-Wilk test as well as evaluated visually using Q-Q plots and histograms. Data found to be non-normal were natural-log-transformed and retested for normality.

Mixed model analysis (split-plot) were conducted using R v3.3.2 to investigate for significant differences in each parameter between marsh units and landscape positions. The marsh unit (west marsh, center marsh, east marsh, and Lake Francis), landscape position (open-water, emergent zone, and wet meadow zone) and their interaction (marsh unit X landscape position) were all treated as fixed effects while the individual sample transect number was included as a random effect. The Satterthwaite approximation was used to calculate degrees of freedom due to an imbalance in the sample sizes between marsh units and landscape positions.

Correlations between P sorption parameters and soil physiochemical properties were investigated by assessing Pearson correlation coefficients between all variables (Table 1.4). All sample sites were analyzed together as well as divided by landscape position (open-water, emergent zone, and wet meadow zone) to investigate the potential effect of the different soil conditions of each landscape position. Analysis was conducted both including and excluding Lake Francis. Delta Marsh proper is the main area under consideration and, as such, needed to be assessed in isolation without the influence of data collected from Lake Francis. At the same time, including Lake Francis allowed for expansion of the small sample size with a system that is geographically close to Delta Marsh and was expected to share many of the same geological and environmental conditions, which may result in the emergence of additional relationships not apparent by investigating the main marsh in isolation. The investigation of Lake Francis may

also help determine the potential impacts of the Portage Diversion and Common Carp on Delta Marsh as Lake Francis is not expected to be significantly influenced by either factor. As the sample size was relatively low in this study, assessing the data with multiple "slices" may allow for the identification of relationships that would otherwise be obscured.

To determine if the marsh sediments act as a sink or a source for P relative to the water column the EPC values were compared to total dissolved phosphorus (TDP) measured near each site during summer 2015 and 2016. Phosphorus flux between wetland sediments and the water column is dynamic and may change over time as the TDP concentration fluctuates and for this reason it is important to compare EPC and TDP values at multiple points over time. If the EPC were to be compared to a single datapoint or the average TDP value it would be implied that the sediment acts as either a sink or source at all times, whereas when compared to a series of measurements over time it may become apparent that the sediments can alternate between being a sink or a source as conditions in the overlying water change. Comparisons were made separately between TDP and the open-water, emergent, and wet meadow zone for each site as data availability allowed, to understand how these different soil zones will react when exposed to the water column separately. The assessment of the emergent and wet meadow zone sediments is important because these sites are likely to be inundated due to the relatively small differences in elevation seen around the marsh. Soils, and their nutrient loads, from these zones are also likely to make their way into the open-water through erosion (Wolfe 2007), and there can be inter-site P cycling within the system (Zhou et al. 2011).

2.3: Sediment Chronology and Nutrient Accumulation

2.3.1: Experimental Design and Sample Collection

Eight bays within the Delta Marsh complex were chosen to be sampled (Figure 1.1, for exact locations see Table A2). These sites were chosen to represent the east-west range of the study area with samples taken from each of the marsh units to allow for comparisons across the marsh. Samples were taken from the approximate center and deepest location of each bay to account for the effects of sediment focusing and minimize any influence from shorelines. The sample sites selected were the same as the open-water sites discussed in Section 2.2. Three sites were chosen from the west marsh including Big Lake, Canvasback Bay, and Weedy Bay. Two samples were taken from Eaglenest Bay in center marsh. Three sites in the east marsh included Cadham Bay, Lyttle Bay, and Waterhen Bay. Two samples were also taken from Lake Francis. Replicate samples were taken from Eaglenest Bay and Lake Francis in order to facilitate statistical comparisons for these smaller marsh units.

Sediment core samples of at least 50-cm were collected at the ten sites in Delta Marsh and Lake Francis between August and October 2015. All sediment samples were collected using a handheld Watermark Universal Sediment Corer. For each site, six cores of at least 50-cm were collected within a close proximity with care taken to ensure each sample was from a new and undisturbed location. Samples were collected on calm weather days and care was taken to ensure sediments were not disturbed on approach to the site. The boat was stationary during sample collection, cores were taken from separate footprints, cores remained vertical until processed, and agitation or disturbance to the cores was minimized while being transported to shore for processing. The cores samples were processed *in situ* on the shoreline and divided

into 1-cm segments with each successive layer composited from the six cores into one sample. Samples were placed into Ziplock freezer bags and transported in a cooler on ice to the laboratory at the University of Manitoba Biological Sciences department. Sediment samples were stored in a refrigerator at 4˚C until analyzed.

2.3.2: Laboratory Methods

Sediment samples were submitted to the University of Manitoba Soil Sciences department for analysis of ^{210}Pb and ^{137}Cs. Of the fifty samples (pooled) from each site, a representative subset of twenty-one were selected to be analyzed with the remainder stored should they be required. Only a subset of samples was submitted due to budgetary and time restrictions and because determination of an accurate dating profile did not require every one-cm segment to be analyzed. For the more recent sediments near the surface of the cores, more sample segments were submitted with an increasingly lower frequency of samples being submitted from lower in the cores. The fewest samples were submitted from the bottom of the core as while the background ^{210}Pb levels and full ^{137}Cs profile needed to be located and identified it was not expected that this would reach the full 50-cm depth. The sample segments that were analyzed were 0-1, 1-2, 2-3, 3-4, 4-5, 6-7, 8-9, 10-11, 12-13, 14-15, 16-17, 18-19, 20-21, 23-24, 26-27, 29-30, 33-34, 37-38, 41-42, 45-46, and 49-50 cm in depth.

All selected one-cm core samples were submitted to the Radio Isotope laboratory at the University of Manitoba Soil Science department for the non-destructive analysis of ^{137}Cs and ^{210}Pb. Sediment samples were first massed and dried to determine percent moisture and prepare samples for analysis. Samples were ground and sieved through a two-mm sieve. As much mass as possible was placed in a plastic container on a gamma spectroscopy system with

a lithium drifted germanium detector (Ge(Li)) and multichannel analyzer and left to count for 24 hours. The activities of ^{137}Cs and ^{210}Pb were determined from the gamma spectrum obtained at 662 KeV and 47 KeV, respectively, and compared to an IAEA certified sample of the same geometry. Sample analysis decay was back-corrected to the day of sampling. The activity for ^{137}Cs and ^{210}Pb was determined by the equation:

(Equation 2.1) For ^{137}Cs: Cs activity = (ΣCs / 0.85) * ^{137}Eff * (1/e-λt)

Where:
Cs activity = Activity of ^{137}Cs or ^{210}Pb at time of sampling (dps)
ΣCs = Counts per second (cps) of 662 KeV or 47 KeV photo peak
^{137}Eff = Counting efficiency for ^{137}Cs at 662 KeV for the sample geometry used
λ = Decay constant for ^{137}Cs or ^{210}Pb (years)
t = Elapsed time (years) between sampling and counting

The activities for ^{137}Cs and ^{210}Pb were both found using the same equation; however, ^{210}Pb uses a different detector counting efficiency and decay correction values.

Following the ^{137}Cs and ^{210}Pb analysis, samples were resubmitted for analysis of total, organic, and biologically available P (mg kg^{-1}), total nitrogen (mg kg^{-1}), and organic matter (%) at the Soil Science department of the University of Manitoba. Samples were analyzed alongside the 0-5 cm core samples following standard laboratory methods based on (Ige *et al*. 2005) as described in Section 2.2.2.

2.3.3: Statistical Analysis

The following analysis utilizes many similar variables and as such there was a consistent notation used throughout:

X(i)= Value at the surface of sample segment i, counting downwards from the surface of the core

X_i= Value at the geometric center of the sample segment i, counting downwards from the surface of the core

Where,
X = The variable under consideration
Note: Using this notation the sample depth z(0)= 0cm, z_1= 0.5cm, z(1)= 1cm, etc.

Following determination of ^{210}Pb and ^{137}Cs in each successive soil/sediment layer, dating profiles were created for each sediment core. Due to the potential effects of sediment compaction dating must be determined relative to mass depth m_i (kg m^{-2}) and not directly from the sample segment depth z(i) (m)(Sanchez-Cabeza and Ruiz-Fernández 2012). Mass depth (m_i) for each sample segment was determined by:

(Equation 2.2): $m_i = (m(i-1)+m(i))/2$

Where,
m_i = Mass depth of sample segment i (middle of sample segment) (kg m^{-2})
m(i) = Mass depth of sample segment i (top of sample segment) (kg m^{-2})

and,

(Equation 2.3): $m(i) = m(i-1)+(\Delta m_i/S)$

Where,
Δm_i = Dry mass of the sample segment i, measured in laboratory (kg)
S = Surface area of sediment core sample (m^2)

Following the above formula, the mass depth m_i was the average of the mass depth m(i) above and below m_i, while m(i) was the total cumulative mass depth from the surface of the sediment core to the depth i. As the cumulative mass depth was required to be calculated to construct profiles, requiring values for every sample segment, missing values were interpolated as described by (Sanchez-Cabeza and Ruiz-Fernández 2012). The dating profiles were then created by plotting the mass depth m_i against the measured ^{210}Pb and ^{137}Cs activities (Y and X, respectively) for each sample segment using SigmaPlot version 11.0 for Windows.

All dating models based on ^{210}Pb activity were based on the unsupported or excess (atmospheric) $^{210}Pb_{ex}$ (Sanchez-Cabeza and Ruiz-Fernández 2012; Mabit et al. 2014). The $^{210}Pb_{ex}$ was determined by:

(Equation 2.4): $^{210}Pb_{ex} = {}^{210}Pb - {}^{226}Ra$

Where,
$^{210}Pb_{ex}$ = Excess or unsupported ^{210}Pb (Bq kg^{-1})
^{210}Pb = Total ^{210}Pb measured in the laboratory (Bq kg^{-1})
^{226}Ra = Supported or background ^{210}Pb (Bq kg^{-1})

The ^{226}Ra can be determined through direct measurement via gamma spectrometry or, in the case of this study, calculated as the mean ^{210}Pb of at least three consecutive sample segments from the bottom of the core sample that demonstrate approximately constant concentrations (Sanchez-Cabeza and Ruiz-Fernández 2012). If the sediment core sample was sufficiently deep, the ^{210}Pb activity will eventually become constant at a concentration equal to the supported or background ^{210}Pb (or the ^{226}Ra) allowing for its determination without the need of additional analytical measurements. Once the $^{210}Pb_{ex}$ was determined for each sample segment the $^{210}Pb_{ex}$ deposit ΔA, (the activity per unit surface area) was determined for each segment by:

(Equation 2.5): $\Delta A_i = (C_i \Delta m_i)/S$

Where,
ΔA_i = $^{210}Pb_{ex}$ in segment i (Bq m^{-2})
C_i = $^{210}Pb_{ex}$ concentration in segment i, determined using Equation 2.3 (Bq kg^{-1})
Δm_i = Dry mass of the sample segment i, measured in laboratory (kg)
S = Surface area of sediment core sample (m^2)

From there the accumulated $^{210}Pb_{ex}$ deposit for each segment i was calculated sequentially from the bottom of the core (where $^{210}Pb_{ex} = 0$) by:

(Equation 2.6): $A(i) = A(i+1)+\Delta A_i$

Where,
A(i) = cumulative $^{210}Pb_{ex}$ deposit from core bottom (Bq m^{-2})
ΔA_i = $^{210}Pb_{ex}$ in segment i (Bq m^{-2})

The time elapsed since the formation of each sample segment was then able to be determined by:

(Equation 2.7): $t(i) = (LN(A(i)/A(0))) / \lambda$

Where,
t(i) = Time elapsed since formation of sediment segment (year)
LN() = Natural logarithm function
A(i) = cumulative $^{210}Pb_{ex}$ deposit from core bottom (Bq m^{-2})
A(0) = cumulative $^{210}Pb_{ex}$ deposit from core bottom to the surface of the core (Bq m^{-2})
λ = Disintegration constant ($years^{-1}$)

And finally, to determine the year each sample segment represents in the historical record:

(Equation 2.8): $T(i) = T(0) - t(i)$

Where,
T(i) = Date represented by sample segment (year)
T(0) = Date represented by the top of core (date of sample collection)(year)
t(i) = Time elapsed since formation of sediment segment (year)

In determining the rate of accumulation in an aquatic system there are two commonly used criteria; the mass accumulation rate (MAR) measuring the rate of mass of sediment accumulated per area per year (kg $m^{-2}yr^{-1}$), and sediment accumulation rate (SAR) measuring

the vertical accumulation of sediment per year (m yr^{-1})(Sanchez-Cabeza and Ruiz-Fernández 2012; Eleftheriou et al. 2018). The MAR was calculated by:

(Equation 2.9): $MAR = (\lambda A(i))/ C_i$

Where,
MAR = Mass accumulation rate (kg $m^{-2}yr^{-1}$)
λ = Disintegration constant ($years^{-1}$)
A(i) = cumulative $^{210}Pb_{ex}$ deposit from core bottom (Bq m^{-2})
C_i = $^{210}Pb_{ex}$ concentration in segment i, determined using Equation 2.3 (Bq kg^{-1})

The SAR is proportionally related to the MAR and was calculated by:

Equation (2.10): $SAR = MAR ((S \Delta z_i)/ \Delta m_i)$

Where,
SAR = Sediment accumulation rate (m yr^{-1})
MAR = Mass accumulation rate (kg $m^{-2}yr^{-1}$)
S = Surface area of sediment core sample (m^2)
Δz_i = Width of sediment sample segment (m^2)
Δm_i = Dry mass of the sample segment i, measured in laboratory (kg)

Following the determination of the MAR and SAR for each sediment sample segment, these parameters were plotted against the dates determined by Equation 2.8 (X and Y axis, respectively) using SigmaPlot version 11.0 for Windows.

To investigate the change over time in the physiochemical parameters analyzed, their rates of accumulation were determined. This was conducted for total P, organic P, biologically available P, total N, total C, and organic matter, all following the same method. Using total P as an example, this was done by:

(Equation 2.11): TPAR = MAR * p

Where,
TPAR = Total P accumulation rate (mg $m^{-2}yr^{-1}$)
MAR = Mass accumulation rate (kg $m^{-2}yr^{-1}$)
P = Concentration of P in sample segment measured in laboratory (mg kg^{-1})

Following the determination of the accumulation rates for each sediment sample segment these parameters were plotted against the dates determined by Equation 2.8 (X and Y axis, respectively) using SigmaPlot version 11.0 for Windows.

Due to the expected presence of mixing between layers and other source of sediment disturbance core inventories were calculated for ^{210}Pb and ^{137}Cs in order to facilitate comparisons between sites on a whole-core basis and investigate the potential sediment disturbance and mobility (Craft 2007b; Kuzyk et al. 2004; Kamula 2015). Inventories were calculated by determining the $^{210}Pb_{ex}$ or ^{137}Cs activity in each sample segment and summing the results to the depth where no further activity was detected:

(Equation 2.12): $^{210}Pb_{ex}inv_i = (P\ \Delta m_i)/ S$

Where,
$^{210}Pb_{ex}$ = $^{210}Pb_{ex}$ activity contained in sediment segment i (Bq m^{-2})
P = $^{210}Pb_{ex}$ activity in sample segment measured in laboratory (Bq kg^{-1})
Δm_i = Dry mass of the sample segment i, measured in laboratory (kg)
S = Surface area of sediment core sample (m^2)

and,

(Equation 2.13): $^{210}Pb_{ex}inv$ = $^{210}Pb_{ex}$ core inventory = sum of $^{210}Pb_{ex}inv_i$ for the entire core.

The ^{137}Cs inventory was also calculated using Equations 2.12 and 2.13 except using ^{137}Cs instead of $^{210}Pb_{ex}$.

For all sediment profiles the uncertainty, or error, (as standard deviation) was calculated following quadratic uncertainty propagation through the core to determine the uncertainty for each sample segment (Sanchez-Cabeza and Ruiz-Fernández 2012).

To assess potential changes in the forms of P within the marsh over time OP and BAP as a percentage of TP was calculated for each core sample segment. To investigate the possibility of nutrient limitation within the marsh and how it may have changed over time molar "Redfield" ratios of TN/TP were calculated (Khan et al. 2007; Kinsman-Costello et al. 2014).

3: Results

3.1: P Sorption and Physiochemical Parameters

3.1.1: Sediment Physiochemical Parameters

For all physiochemical parameters there were significant differences due to at least one location parameter in the marsh. Analysis was conducted both including and excluding Lake Francis to investigate if its inclusion in the analysis changed the significance of the results for Delta Marsh proper, the primary area of interest. Except for S_n including LF resulted in additional information without affecting the results of the marsh proper and as such it was included. For the S_n parameter including LF erased the significance of the results otherwise obtained without contributing new information and as such LF was excluded from this analysis.

The total P (TP) concentrations ranged from 750 mg kg^{-1} (Weedy Bay – open water) to 2,031 mg kg^{-1} (Lake Francis – wet meadow) and averaged 1,141 mg kg^{-1} across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position ($p < 0.0001$)(open-water, emergent, or wet meadow

zone), marsh unit (p = 0.0143)(west marsh, center marsh, east marsh, or Lake Francis), and for the interaction between marsh unit and landscape position (p = 0.0174; Table 1.3). Across landscape positions, mean TP concentrations were 858 mg kg^{-1}, 1,247 mg kg^{-1}, and 1,507 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean TP concentrations were 891 mg kg^{-1}, 1,150 mg kg^{-1}, 1,185 mg kg^{-1}, and 1,345 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.2). In the WM unit TP concentrations were not significantly different between landscape positions. In the CM unit TP concentrations were significantly different between the open-water landscape position (mean 941 mg kg^{-1}) and wet meadow landscape position (1,425 mg kg^{-1}; Table 1.2; p = 0.0365; Table 1.4). In the EM unit TP concentrations were significantly different between the open-water landscape position (mean 821 mg kg^{-1}) and both the emergent landscape position (mean 1,210 mg kg^{-1}; p = 0.0067), and the wet meadow landscape position (mean 1,523 mg kg^{-1}; Table 1.2; p = 0.0001; Table 1.4). In the LF unit TP concentrations were significantly different between the open-water landscape position (mean 809 mg kg^{-1}) and both the emergent landscape position (1,731 mg kg^{-1}; p = 0.0003), and the wet meadow landscape position (2,031 mg kg^{-1}; Table 1.2; p < 0.0001; Table 1.4). For the open-water landscape positions across the marsh TP concentrations were not significantly different between marsh units. For the emergent landscape positions across the marsh TP concentrations were significantly different between the WM unit (825 mg kg^{-1}) and the LF unit (1,731 mg kg^{-1}; Table 1.2; p = 0.0022; Table 1.4). For the wet meadow landscape positions across the marsh TP concentrations were significantly different between the WM unit (1,019 mg kg^{-1}) and both the EM unit (mean 1,523 mg kg^{-1}; p = 0.0734), and the LF unit (2,031 mg kg^{-1}; Table 1.2; p = 0.0044; Table 1.4).

Table 1.1. Physiochemical properties of surficial (0-5cm) sediments and soils collected from twenty-two locations in Delta Marsh (Figure 1.1), Manitoba, Canada, between August and October 2015.

Site Name	Total P	Mehlich P	Organic P	BAP*	Total N	Mehlich Al	Mehlich Fe	Mehlich Mg	Mehlich Ca	Organic Matter
	----------mg kg^{-1}----------									%
Big Lake Open Water	838	16.6	128	29	6,823	4.5	511	2,350	8,640	13.3
Big Lake Emergent	825	9.1	81	39	2,875	6.4	523	1,660	7,870	4.7
Big Lake Wet Meadow	1,019	25.9	220	52	8,688	23.1	333	4,080	6,120	13.9
Canvasback Bay Open Water	1,025	17.2	98	43	5,638	3.5	507	1,670	8,240	7.3
Weedy Bay Open Water	750	4.0	37	31	2,960	3.1	539	1,300	7,220	10.5
Eaglenest 1 Open Water	794	24.2	114	23	7,240	34.3	428	2,060	10,100	11.1
Eaglenest 2 Open Water	1,088	25.1	37	49	9,230	37.3	376	2,290	10,200	13.3
Eaglenest Emergent	1,294	33.2	415	42	19,778	16.7	482	4,480	5,790	37.4
Eaglenest Wet Meadow	1,425	60.0	230	40	8,203	54.7	176	5,310	9,190	11.9
Cadham Bay Open Water	813	11.1	91	29	4,680	0.05	382	2,260	15,700	8.1
Cadham Bay Emergent	925	7.9	68	56	3,513	0.05	401	2,000	15,400	8.4
Cadham Bay Wet Meadow	1,481	26.4	354	13	3,492	15.6	245	6,780	7,440	18.3
Lyttle Bay Open Water	875	12.6	105	25	8,048	0.9	397	2,220	13,400	13.9
Lyttle Bay Emergent	1,175	16.6	167	26	11,048	0.05	463	2,920	10,000	22.1
Lyttle Bay Wet Meadow	1,175	26.8	500	42	10,583	44.2	353	4,360	4,820	19.4
Waterhen Bay Open Water	775	16.5	121	22	5,480	4.4	352	1,540	7,080	9.2
Waterhen Bay Emergent	1,531	19.2	222	40	11,638	4.8	471	2,330	7,170	22.8
Waterhen Bay Wet Meadow	1,913	24.9	349	53	15,523	35.8	262	5,670	5,300	24.6
Lake Francis 1 Open Water	813	21.0	123	34	7,900	1.7	372	2,360	8,360	15.2
Lake Francis 2 Open Water	806	13.6	136	27	8,998	0.05	391	2,300	8,390	19.4
Lake Francis Emergent	1,731	19.7	313	64	19,412	0.1	292	4,170	11,400	27.0
Lake Francis Wet Meadow	2,031	73.0	671	87	21,808	39.0	202	7,890	8,220	38.4
Overall Mean	1,141	23.0	208	39	9,252	15.0	385	3,272	8,911	16.8

*BAP, biologically available P

Table 1.2 Mean values by location, landscape position, and marsh unit for physiochemical properties of surficial (0-5cm) sediments and soils collected from twenty-two locations in Delta Marsh (Figure 1.1), Manitoba, Canada, between August and October 2015.

Location	Total P	Mehlich P	Organic P	BAP	Total N	Mehlich Al	Mehlich Fe	Mehlich Mg	Mehlich Ca	Organic Matter
West Marsh	---mg kg^{-1}---									%
Open-water	871	12.6	88	34	5140	3.7	519	1773	8033	10.4
Emergent	825	9.1	81	39	2875	6.4	523	1660	7870	4.7
Wet meadow	1019	25.9	220	52	8688	23.1	333	4080	6120	13.9
Center Marsh										
Open-water	941	24.7	76	36	8235	35.8	402	2175	10150	12.2
Emergent	1294	33.2	415	42	19778	16.7	482	4480	5790	37.4
Wet meadow	1425	60.0	230	40	8203	54.7	176	5310	9190	11.9
East Marsh										
Open-water	821	13.4	106	25	6069	1.8	377	2007	12060	10.4
Emergent	1210	14.6	152	41	8733	1.6	445	2417	10857	17.8
Wet meadow	1523	26.0	401	36	9866	31.9	287	5603	5853	20.7
Lake Francis										
Open-water	809	17.3	129	31	8449	0.9	382	2330	8375	17.3
Emergent	1731	19.7	313	64	19412	0.1	292	4170	11400	27.0
Wet meadow	2031	73.0	671	87	21808	39.0	202	7890	8220	38.4
Landscape Position	---mg kg^{-1}---									%
Open-water (n=10)	858	16.2	99	31	6700	9.0	426	2035	9733	12.1
Emergent (n=6)	1247	17.6	211	44	11377	4.7	439	2927	9605	20.4
Wet meadow (n=6)	1507	39.5	387	48	11382	35.4	262	5682	6848	21.1
Marsh Unit										
West Marsh (n=5)	891	14.6	113	39	5397	8.1	483	2212	7618	10.0
Center Marsh (n=4)	1150	35.6	199	39	11113	35.8	366	3535	8820	18.4
East Marsh (n=9)	1185	18.0	219	34	8222	11.7	370	3342	9590	16.3
Lake Francis (n=4)	1345	31.8	311	53	14529	10.2	314	4180	9093	25.0
Overall Mean	1141	23.0	208	39	9252	15.0	385	3272	8911	16.8

Table 1.3a. Results from mixed-model analysis looking at statistically significant differences of surficial (0-5cm) sediment sample physiochemical properties across marsh units and landscape positions at Delta Marsh (Figure 1.1), Canada, collected between August and October 2015. Analysis includes Lake Francis.

Parameter	Variable	NumDF	DenDF	F-value	P-value	
Total P	Marsh Unit	3	7.5	7.005	**0.0143**	*
	Landscape Position	2	14.3	33.792	**3.75E-06**	***
	Unit x Position	6	14.2	3.847	**0.0174**	*
Mehlich P	Marsh Unit	3	9.5	5.631	**0.0173**	*
	Landscape Position	2	11.3	56.969	**1.29E-06**	***
	Unit x Position	6	11.2	4.208	**0.0185**	*
Organic P	Marsh Unit	3	8.8	4.618	**0.0332**	*
	Landscape Position	2	10.7	57.245	**1.89E-06**	***
	Unit x Position	6	10.6	9.998	**0.0007**	***
BAP	Marsh Unit	3	22.0	7.262	**0.0015**	**
	Landscape Position	2	22.0	10.533	**0.0006**	***
	Unit x Position	6	22.0	2.618	**0.0454**	*
Total N	Marsh Unit	3	11.4	5.670	**0.0128**	*
	Landscape Position	2	13.7	5.085	**0.0224**	*
	Unit x Position	6	13.6	5.711	**0.0038**	**
Mehlich Al	Marsh Unit	3	12.2	17.910	**9.10E-05**	***
	Landscape Position	2	15.3	119.592	**4.45E-10**	***
	Unit x Position	6	15.2	15.811	**9.33E-06**	***
Mehlich Fe	Marsh Unit	3	10.4	22.624	**7.28E-05**	***
	Landscape Position	2	15.1	136.312	**2.14E-10**	***
	Unit x Position	6	14.9	9.996	**0.0002**	***
Mehlich Mg	Marsh Unit	3	22.0	14.534	**1.94E-05**	***
	Landscape Position	2	22.0	78.602	**9.55E-11**	***
	Unit x Position	6	22.0	2.888	**0.0313**	*
Mehlich Ca	Marsh Unit	3	12.5	1.154	0.3659	
	Landscape Position	2	14.9	8.914	**0.0028**	**
	Unit x Position	6	14.8	8.421	**0.0004**	***
Total Calcium	Marsh Unit	3	10.1	10.376	**0.0020**	**
	Landscape Position	2	11.9	148.079	**4.05E-09**	***
	Unit x Position	6	11.8	19.116	**1.92E-05**	***
Total Carbon	Marsh Unit	3	10.5	21.516	**8.66E-05**	***
	Landscape Position	2	15.7	17.126	**0.0001**	***
	Unit x Position	6	15.5	8.550	**0.0003**	***
Organic Matter	Marsh Unit	3	10.0	11.554	**0.0014**	**
	Landscape Position	2	14.3	8.383	**0.0039**	**
	Unit x Position	6	14.1	10.198	**0.0002**	***

*,**,*** Significant at the 0.05, 0.01, and 0.001 probability levels, respectively

Table 1.3b. Results from mixed-model analysis looking at statistically significant differences of surficial (0-5cm) sediment sample physiochemical properties across marsh units and landscape positions at Delta Marsh (Figure 1.1), Canada, collected between August and October 2015. Analysis does not include Lake Francis

Parameter	Variable	NumDF	DenDF	F-value	P-value	
Total P	Marsh Unit	2	6.1	3.800	0.0852	
	Landscape Position	2	11.6	11.666	**0.0017**	**
	Unit x Position	4	11.6	1.919	0.1743	
Mehlich P	Marsh Unit	2	7.6	5.283	**0.0365**	*
	Landscape Position	2	9.4	24.021	**0.0002**	***
	Unit x Position	4	9.3	1.680	0.2352	
Organic P	Marsh Unit	2	7.1	3.726	0.0782	
	Landscape Position	2	9.0	33.164	**7.12E-05**	***
	Unit x Position	4	8.9	12.253	**0.0011**	**
BAP	Marsh Unit	2	18.0	0.946	0.4066	
	Landscape Position	2	18.0	1.858	0.1847	
	Unit x Position	4	18.0	0.378	0.8213	
Total N	Marsh Unit	2	9.2	3.326	0.0820	
	Landscape Position	2	11.3	0.902	0.4333	
	Unit x Position	4	11.2	5.797	**0.0089**	**
Mehlich Al	Marsh Unit	2	10.2	15.058	**0.0009**	***
	Landscape Position	2	12.9	40.191	**2.95E-06**	***
	Unit x Position	4	12.8	8.043	**0.0018**	**
Mehlich Fe	Marsh Unit	2	8.5	13.293	**0.0024**	**
	Landscape Position	2	12.5	108.067	**1.34E-08**	***
	Unit x Position	4	12.4	6.676	**0.0043**	**
Mehlich Mg	Marsh Unit	2	18.0	9.512	**0.0015**	**
	Landscape Position	2	18.0	44.486	**1.08E-07**	***
	Unit x Position	4	18.0	2.775	0.0588	
Mehlich Ca	Marsh Unit	2	10.1	1.018	0.3958	
	Landscape Position	2	12.2	10.670	**0.0021**	**
	Unit x Position	4	12.2	7.875	**0.0023**	**
Total Calcium	Marsh Unit	2	8.1	1.610	0.2580	
	Landscape Position	2	9.9	77.114	**9.51E-07**	***
	Unit x Position	4	9.8	19.695	**0.0001**	***
Total Carbon	Marsh Unit	2	8.6	8.278	**0.0099**	**
	Landscape Position	2	12.9	4.862	**0.0267**	*
	Unit x Position	4	12.8	7.567	**0.0023**	**
Organic Matter	Marsh Unit	2	8.3	7.537	**0.0136**	*
	Landscape Position	2	11.9	3.114	0.0817	
	Unit x Position	4	11.8	12.403	**0.0003**	***

*,**,*** Significant at the 0.05, 0.01, and 0.001 probability levels, respectively

Table 1.4a. Detailed results from mixed-model analysis looking at statistically significant differences for surficial (0-5cm) sediment sample physiochemical properties between landscape positions within marsh units of Delta Marsh (Figure 1.1), collected between August and October 2015.

	West Marsh	Center Marsh	East Marsh	Lake Francis
Total P	-	OM-WM[a]	OW-E[b], OW-WM[c]	OW-E[c], OW-WM[c]
Mehlich3 P	E-WM[b]	OW-WM[b], E-WM	OW-WM[c], E-WM[c]	OW-WM[c], E-WM[c]
Organic P	E-WM[a]	OW-E[c], OW-WM[c]	OW-WM[c], E-WM[c]	OW-E[a], OW-WM[c]
Bioavailable P	-	-	-	OW-E[a], OW-WM[c]
Total N	OW-E[a], E-WM[b]	OW-E[a], E-WM[a]	-	OW-E[a], OW-WM[a]
Mehlich3 Al	OW-WM[c], E-WM[a]	E-WM[a]	OW-WM[c], E-WM[c]	OW-E[c], OW-WM[c], E-WM[c]
Mehlich3 Fe	OW-WM[c], E-WM[c]	OW-E[b], OW-WM[c], E-WM[c]	OW-E[b], OW-WM[c], E-WM[c]	OW-E[b], OW-WM[c], E-WM[a]
Mehlich3 Mg	OW-WM[c], E-WM[c]	OW-E[b], OW-WM[c]	OW-WM[c], E-WM[c]	OW-E[b], OW-WM[c], E-WM[a]
Mehlich3 Ca	-	OW-E[b], E-WM[a]	OW-WM[c], E-WM[c]	-
Organic Matter	OW-E[b], E-WM[b]	OW-E[c], E-WM[b]	OW-E[a], OW-WM[c]	OW-WM[a]

Landscape Position: OW = Open-water; E = Emergent; WM = Wet meadow
Significance: [a] $P \leq 0.05$, [b] $P \leq 0.01$, [c] $P \leq 0.001$.

Table 1.4b. Detailed results from mixed-model analysis looking at statistically significant differences for surficial (0-5cm) sediment sample physiochemical properties between landscape positions within landscape positions of Delta Marsh (Figure 1.1), collected between August and October 2015.

	Open Water	Emergent	Wet Meadow
Total P	-	W-LF[b]	W-LF[b]
Mehlich3 P	-	W-C[b], W-LF[a]	W-C[a], W-LF[b], E-LF[a]
Organic P	-	W-C[c,] W-LF[b], C-E[b]	W-LF[a]
Bioavailable P	-	-	C-LF[b], E-LF[c]
Total N	-	W-C[c], W-E[a], W-LF[c]	-
Mehlich3 Al	W-C[c], C-E[c], C-LF[c],	W-LF[c], C-E[c], C-LF[c], E-LF[c]	-
Mehlich3 Fe	W-C[c], W-E[c], W-LF[c]	W-E[a], W-LF[c], C-LF[c], E-LF[c]	W-C[c], W-LF[b], C-E[b], E-LF[a]
Mehlich3 Mg	-	W-C[c], W-LF[c], C-E[b], E-LF[a]	W-LF[a]
Mehlich3 Ca	-	C-E[a], C-LF[a]	-
Organic Matter	-	W-C[c], W-E[c], W-LF[c], C-E[a]	W-LF[b], C-LF[b]

Marsh Units: W = West; C = Centre; E = East; LF = Lake Francis
Significance: [a] $P \leq 0.05$, [b] $P \leq 0.01$, [c] $P \leq 0.001$.

The Mehlich-3 P concentrations ranged from 4 mg kg^{-1} (Weedy Bay – open water) to 72 mg kg^{-1} (Lake Francis – wet meadow) and averaged 23 mg kg^{-1} across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position ($p < 0.0001$), marsh unit ($p = 0.0173$), and for the interaction between marsh unit and landscape position ($p = 0.0185$; Table 1.3). Across landscape positions, mean Mehlich-3 P concentrations were 16.2 mg kg^{-1}, 17.6 mg kg^{-1}, and 39.5 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean Mehlich-3 P concentrations were 14.6 mg kg^{-1}, 35.6 mg kg^{-1}, 18.0 mg kg^{-1}, and 31.8 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.2). In the WM unit Mehlich-3 P concentrations were significantly different between the emergent landscape position (9.1 mg kg^{-1}) and wet meadow landscape position (25.9 mg kg^{-1}; Table 1.2; $p = 0.0017$; Table 1.4). In the CM unit Mehlich-3 P concentrations were significantly different between the wet meadow landscape position (60 mg kg^{-1}) and both the open water (24.7 mg kg^{-1}; $p = 0.0042$) and emergent landscape positions (33.2 mg kg^{-1}; Table 1.2; $p = 0.0507$; Table 1.4). In the EM unit Mehlich-3 P concentrations were significantly different between the wet meadow landscape position (mean 26 mg kg^{-1}) and both the open water (mean 13.4 mg kg^{-1}; $p = 0.0007$) and emergent landscape positions (mean 14.6 mg kg^{-1}; Table 1.2; $p = 0.0010$; Table 1.4). In the LF unit Mehlich-3 P concentrations were significantly different between the wet meadow landscape position (73 mg kg^{-1}) and both the open water (mean 17.3 mg kg^{-1}; $p < 0.0001$) and emergent landscape positions (19.7 mg kg^{-1}; Table 1.2; $p = 0.0003$; Table 1.4). For the emergent landscape positions across the marsh Mehlich-3 P concentrations were significantly different between the WM unit (9.1 mg kg^{-1}) and both the CM unit (33.2 mg kg^{-1}; $p = 0.0045$), and LF unit

(19.7 mg kg^{-1}; Table 1.2; p = 0.0234; Table 1.4). For the wet meadow landscape positions across the marsh Mehlich-3 P concentrations were significantly different between the LF unit (73 mg kg^{-1}) and both the WM unit (25.9 mg kg^{-1}; p = 0.0060), and EM unit (mean 26 mg kg^{-1}; p = 0.0303), as well as between the WM unit (25.9 mg kg-1) and CM unit (60 mg kg^{-1}; Table 1.2; p = 0.0424; Table 1.4). For the open-water landscape positions across the marsh Mehlich-3 P concentrations were not significantly different between marsh units.

The organic P (OP) concentrations ranged from 37 mg kg^{-1} (Eaglenest Bay 2 – open water) to 671 mg kg^{-1} (Lake Francis – wet meadow) and averaged 208 mg kg^{-1} across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position (p < 0.0001), marsh unit (p = 0.0332), and for the interaction between marsh unit and landscape position (p = 0.0007; Table 1.3). Across landscape positions, mean OP concentrations were 99 mg kg^{-1}, 211 mg kg^{-1}, and 387 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean OP concentrations were 113 mg kg^{-1}, 199 mg kg^{-1}, 219 mg kg^{-1}, and 311 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.2). In the WM unit OP concentrations were significantly different between the emergent landscape position (81 mg kg^{-1}) and the wet meadow landscape position (220 mg kg^{-1}; Table 1.2; p = 0.0134; Table 1.4). In the CM unit OP concentrations were significantly different between the open-water landscape position (mean 76 mg kg^{-1}) and both the emergent landscape position (415 mg kg^{-1}; p < 0.0001), and the wet meadow landscape position (230 mg kg^{-1}; Table 1.2; p = 0.0002; Table 1.4). In the EM unit OP concentrations were significantly different between the wet meadow landscape position (mean 106 mg kg^{-1}) and both the open-water landscape position (mean 152 mg kg^{-1}; p

< 0.0001), and the emergent landscape position (mean 401 mg kg^{-1}; Table 1.2; p = 0.0002; Table 1.4). In the LF unit OP concentrations were significantly different between the open-water landscape position (mean 129 mg kg^{-1}) and both the emergent landscape position (313 mg kg^{-1}; p = 0.0249), and the wet meadow landscape position (671 mg kg^{-1}; Table 1.2; p = 0.0002; Table 1.4). For the open-water landscape positions across the marsh OP concentrations were not significantly different between marsh units. For the emergent landscape positions across the marsh OP concentrations were significantly different between the WM unit (81 mg kg^{-1}) and both the CM unit (415 mg kg^{-1}; p = 0.0002), and the LF unit (313 mg kg^{-1}; Table 1.2; p = 0.0071; Table 1.4), as well as between the CM unit (415 mg kg^{-1}) and the EM unit (mean 152 mg kg^{-1}; p = 0.0002). For the wet meadow landscape positions across the marsh OP concentrations were significantly different between the WM unit (220 mg kg^{-1}) and LF unit (671 mg kg^{-1}; Table 1.2; p = 0.0227; Table 1.4). The organic P was lower than that found in intact and drained wetlands in the prairie pothole region (322-450 mg kg^{-1}; Badiou et al. 2018) with the exception of the Lake Francis and Lyttle Bay wet meadow samples (671 and 500 mg kg^{-1}, respectively).

Biologically available P (BAP) concentrations ranged from 13 mg kg^{-1} (Cadham Bay – wet meadow) to 87 mg kg^{-1} (Lake Francis – wet meadow) and averaged 39 mg kg^{-1} across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position (p = 0.0006), marsh unit (p = 0.0015), and for the interaction between marsh unit and landscape position (p = 0.0454; Table 1.3). Across landscape positions, mean BAP concentrations were 31 mg kg^{-1}, 44 mg kg^{-1}, and 48 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean BAP concentrations were 39 mg kg^{-1}, 39 mg kg^{-1}, 34 mg kg^{-1}, and 53

mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.2). In the WM, CM, and EM units BAP concentrations were not significantly different between landscape positions. In the LF unit BAP concentrations were significantly different between the open-water landscape position (31 mg kg^{-1}) and both the emergent landscape position (64 mg kg^{-1}; p = 0.0179), and the wet meadow landscape position (87 mg kg^{-1}; Table 1.2; p = 0.0002; Table 1.4). For the open-water and emergent landscape positions across the marsh BAP concentrations were not significantly different between marsh units. For the wet meadow landscape positions across the marsh BAP concentrations were significantly different between the LF unit (87 mg kg^{-1}) and both the CM unit (40 mg kg^{-1}; p = 0.0085), and the EM unit (mean 36 mg kg^{-1}; Table 1.2; p = 0.0005; Table 1.4).

The total nitrogen (TN) concentrations ranged from 2,875 mg kg^{-1} (Big Lake - emergent) to 21,808 mg kg^{-1} (Lake Francis – wet meadow) and averaged 9,252 mg kg^{-1} across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position (p = 0.0224), marsh unit (p = 0.0128), and for the interaction between marsh unit and landscape position (p = 0.0038; Table 1.3). Across landscape positions, mean TN concentrations were 6,700 mg kg^{-1}, 11,377 mg kg^{-1}, and 11,382 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean TN concentrations were 5,397 mg kg^{-1}, 11,113 mg kg^{-1}, 8,222 mg kg^{-1}, and 14,529 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.2). In the WM unit TN concentrations were significantly different between the emergent landscape position (2,875 mg kg^{-1}) and both the open-water landscape position (mean 5,140 mg kg^{-1}; p = 0.0351), and the wet meadow landscape position (8,688 mg kg^{-1}; Table 1.2; p = 0.0057; Table

1.4). In the CM unit TN concentrations were significantly different between the emergent landscape position (19,778 mg kg^{-1}) and both the open-water landscape position (mean 8,235 mg kg^{-1}; p = 0.0281), and the wet meadow landscape position (8,203 mg kg^{-1}; Table 1.2; p = 0.0242; Table 1.4). In the EM unit TN concentrations were not significantly different between landscape positions. In the LF unit TN concentrations were significantly different between the open-water landscape position (mean 8,449 mg kg^{-1}) and both the emergent landscape position (19,412 mg kg^{-1}; p = 0.0299), and the wet meadow landscape position (21,808 mg kg^{-1}; Table 1.2; p = 0.012; Table 1.4). For the open-water and wet meadow landscape positions across the marsh TN concentrations were not significantly different between marsh units. For the emergent landscape positions across the marsh TN concentrations were significantly different between the WM unit (2,875 mg kg^{-1}) and all of the CM unit (19,778 mg kg^{-1}; p = 0.0004), the EM unit (mean 8,733 mg kg^{-1}; p = 0.0135), and LF unit (19,412 mg kg^{-1}; Table 1.2; p = 0.0004; Table 1.4).

The Mehlich-3 Al concentrations ranged from 0.05 mg kg^{-1} (Cadham Bay – open water and emergent; Lake Francis 2 – open water) to 54.7 mg kg^{-1} (Eaglenest – wet meadow) and averaged 15 mg kg^{-1} across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position, marsh unit, and for the interaction between marsh unit and landscape position (all p < 0.0001; Table 1.3). Across landscape positions, mean Mehlich-3 Al concentrations were 9 mg kg^{-1}, 5 mg kg^{-1}, and 35 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean Mehlich-3 Al concentrations were 8 mg kg^{-1}, 36 mg kg^{-1}, 12 mg kg^{-1}, and 10 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.2). In the

WM unit Mehlich-3 Al concentrations were significantly different between the wet meadow landscape position (23 mg kg^{-1}) and both the open-water landscape position (mean 4 mg kg^{-1}; p = 0.0006), and the emergent landscape position (6 mg kg^{-1}; Table 1.2; p = 0.0198; Table 1.4). In the CM unit Mehlich-3 Al concentrations were significantly different between the emergent landscape position (17 mg kg^{-1}) and wet meadow landscape position (55 mg kg^{-1}; Table 1.2; p = 0.0314; Table 1.4). In the EM unit Mehlich-3 Al concentrations were significantly different between the wet meadow landscape position (mean 32 mg kg^{-1}) and both the open-water landscape position (mean 2 mg kg^{-1}; p < 0.0001), and the emergent landscape position (mean 2 mg kg^{-1}; Table 1.2; p < 0.0001; Table 1.4). In the LF unit Mehlich-3 Al concentrations were significantly different between the open-water (1 mg kg^{-1}) and emergent landscape position (0 mg kg^{-1}; p = 0.0002), the emergent (0 mg kg^{-1}) and wet meadow landscape position (39 mg kg^{-1} p <0.0001), and the open-water (1 mg kg^{-1}) and wet meadow landscape position (39 mg kg^{-1}; Table 1.2; p <0.0001; Table 1.4). For the open-water landscape positions across the marsh Mehlich-3 Al concentrations were significantly different between the CM unit (36 mg kg^{-1}) and all of the WM unit (4 mg kg^{-1}; p = 0.0001), the EM unit (2 mg kg^{-1}; p < 0.0001), and the LF unit (1 mg kg^{-1}; Table 1.2; p < 0.0001; Table 1.4). For the emergent landscape positions across the marsh Mehlich-3 Al concentrations were significantly different between the LF unit (0 mg kg^{-1}) and all of the WM unit (6 mg kg^{-1}; p < 0.0001), the CM unit (17 mg kg^{-1}; p < 0.0001), and the EM unit (2 mg kg^{-1}; Table 1.2; p = 0.0003; Table 1.4), as well as between the CM unit and the EM unit (p = 0.0003). For the wet meadow landscape positions across the marsh Mehlich-3 Al concentrations were not significantly different between marsh units.

The Mehlich-3 Fe concentrations ranged from 176 mg kg^{-1} (Eaglenest Bay – wet meadow) to 539 mg kg^{-1} (Weedy Bay – open water) and averaged 384 mg kg^{-1} across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position ($p < 0.0001$), marsh unit ($p < 0.0001$), and for the interaction between marsh unit and landscape position ($p = 0.0002$; Table 1.3). Across landscape positions, mean Mehlich-3 Fe concentrations were 426 mg kg^{-1}, 439 mg kg^{-1}, and 262 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean Mehlich-3 Fe concentrations were 483 mg kg^{-1}, 366 mg kg^{-1}, 370 mg kg^{-1}, and 314 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.2). Across all marsh units, Mehlich-3 Fe was significantly lower at the wet meadow landscape position relative to both the open water and emergent landscape positions. For the open-water landscape positions across the marsh Mehlich-3 Fe concentrations were significantly different between the WM unit (mean 519 mg kg^{-1}) and the CM unit (mean 402 mg kg^{-1}; $p = 0.0002$), the EM unit (mean 377 mg kg^{-1}; $p < 0.0001$), and LF unit (mean 382 mg kg^{-1}; Table 1.2; $p < 0.0001$; Table 1.4). For the emergent landscape positions across the marsh Mehlich-3 Fe concentrations were significantly different between the LF unit (292 mg kg^{-1}) and all of the WM unit (523 mg kg^{-1}; $p < 0.0001$), the CM unit (482 mg kg^{-1}; $p < 0.0001$), and EM unit (mean 445 mg kg^{-1}; Table 1.2; $p = 0.0001$; Table 1.4). For the wet meadow landscape positions across the marsh Mehlich-3 Fe concentrations were significantly different between the CM unit (176 mg kg^{-1}) and both the WM unit (333 mg kg^{-1}; $p = 0.0008$), and the EM unit (mean 287 mg kg^{-1}; $p = 0.0061$), as well as between the LF unit (202 mg kg^{-1}) and both the WM unit (333 mg kg^{-1}; $p = 0.0023$), and the EM unit (mean 287 mg kg^{-1}; Table 1.2; $p = 0.0199$; Table 1.4).

The Mehlich-3 Mg concentrations ranged from 1,300 mg kg^{-1} (Weedy Bay – open water) to 7,890 mg kg^{-1} (Lake Francis – wet meadow) and averaged 3,273 mg kg^{-1} across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position ($p < 0.0001$), marsh unit ($p < 0.0001$), and for the interaction between marsh unit and landscape position ($p = 0.0313$; Table 1.3). Across landscape positions, mean Mehlich-3 Mg concentrations were 2,035 mg kg^{-1}, 2,927 mg kg^{-1}, and 5,682 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean Mehlich-3 Mg concentrations were 2,212 mg kg^{-1}, 3,535 mg kg^{-1}, 3,342 mg kg^{-1}, and 4,180 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.2). Across all marsh units Mehlich-3 Mg was significantly higher at the wet meadow landscape position relative to the open-water landscape position with values at the emergent landscape position generally falling between those of the open-water and wetland meadow landscape positions. For the open-water landscape positions across the marsh Mehlich-3 Mg concentrations were not significantly different between marsh units. For the emergent landscape positions across the marsh Mehlich-3 Mg concentrations were significantly different between the CM unit (4,480 mg kg^{-1}) and both the WM unit (1,660 mg kg^{-1}; $p = 0.0003$), and the EM unit (mean 2,417 mg kg^{-1}; $p = 0.0045$), as well as between the LF unit (4,170 mg kg^{-1}) and both the WM unit (1,660 mg kg^{-1}; $p = 0.0007$), and the EM unit (mean 2,417 mg kg^{-1}; Table 1.2; $p = 0.0121$; Table 1.4). For the wet meadow landscape positions across the marsh Mehlich-3 Mg concentrations were significantly different between the WM unit (4,080 mg kg^{-1}) and the LF unit (7,890 mg kg^{-1}; Table 1.2; $p = 0.0169$; Table 1.4).

The Mehlich-3 Ca concentrations ranged from 4,820 mg kg^{-1} (Lyttle Bay – wet meadow) to 15,700 mg kg^{-1} (Cadham Bay – open water) and averaged 8,911 mg kg^{-1} across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position ($p = 0.0028$), and for the interaction between marsh unit and landscape position ($p = 0.0004$; Table 1.3). There was not a significant difference due to the marsh unit. Across landscape positions, mean Mehlich -3 Ca concentrations were 9,733 mg kg^{-1}, 9,605 mg kg^{-1}, and 6,848 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean Mehlich-3 Ca concentrations were 7,618 mg kg^{-1}, 8,820 mg kg^{-1}, 9,590 mg kg^{-1}, and 9,093 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.2). In the CM unit Mehlich -3 Ca concentrations were significantly different between the emergent landscape position (5,790 mg kg^{-1}) and both the open-water landscape positions (mean 10,150 mg kg^{-1}; $p = 0.0035$) and wet meadow landscape position (9,190 mg kg^{-1}; Table 1.2; $p = 0.0217$; Table 1.4). In the EM unit Mehlich -3 Ca concentrations were significantly different between the wet meadow landscape positions (mean 5,853 mg kg^{-1}) and both the open water landscape positions (mean 12,060 mg kg^{-1}; $p < 0.0001$), and emergent landscape positions (mean 10,857 mg kg^{-1}; Table 1.2; $p < 0.0001$; Table 1.4). In the WM and LF units Mehlich -3 Ca concentrations were not significantly different between landscape positions. For the emergent landscape positions across the marsh Mehlich -3 Ca concentrations were significantly different between the CM unit (5,790 mg kg^{-1}) and both the EM unit (mean 10,857 mg kg^{-1}; $p = 0.0255$) and the LF unit (11,400 mg kg^{-1}; Table 1.2; $p = 0.0280$; Table 1.4). For the open-water and wet meadow landscape positions across the marsh Mehlich -3 Ca concentrations were not significantly different between marsh units.

The organic matter (OM) percentage ranged from 4.7% (Big Lake - emergent) to 38.4% (Lake Francis – wet meadow) and averaged 16.8% across all sites and landscape positions (Table 1.1). Statistical contrasts identified significant differences between locations due to the landscape position ($p = 0.0039$), marsh unit ($p = 0.0014$), and for the interaction between marsh unit and landscape position ($p = 0.0002$; Table 1.3). Across landscape positions, mean OM concentrations were 12%, 20%, and 21% for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.2). Across marsh units, mean OM concentrations were 10%, 18%, 16%, and 25%, for the WM, CM, EM, and LF units, respectively (Table 1.2). In the WM unit OM concentrations were significantly different between the emergent landscape position (5%) and both the open-water landscape position (mean 10%; $p = 0.0033$), and the wet meadow landscape position (14%; Table 1.2; $p = 0.0024$; Table 1.4). In the CM unit OM concentrations were significantly different between the emergent landscape position (37%) and both the open-water landscape position (mean 12%; $p = 0.0006$), and the wet meadow landscape position (12%; Table 1.2; $p = 0.0016$; Table 1.4). In the EM unit OM concentrations were significantly different between the open-water landscape position (mean 10%) and both the emergent landscape position (mean 18%; $p = 0.0165$), and the wet meadow landscape position (mean 21%; Table 1.2; $p = 0.0009$; Table 1.4). In the LF unit OM concentrations were significantly different between the open-water landscape position (mean 17%) and wet meadow landscape position (38%; Table 1.2; $p = 0.0112$; Table 1.4). For the open-water landscape positions across the marsh OM concentrations were not significantly different between marsh units. For the emergent landscape positions across the marsh OM concentrations were significantly different between the WM unit (5%) and all of the CM unit

(37%; $p < 0.0001$), the EM unit (mean 18%; $p = 0.0001$), and the LF unit (27%; $p < 0.0001$), as well as between the CM unit (37%), and the EM unit (mean 18%; Table 1.2; $p = 0.0220$; Table 1.4). For the wet meadow landscape positions across the marsh OM concentrations were significantly different between the LF unit (38%) and both the WM unit (14%; $p = 0.0092$), and the CM unit (12%; Table 1.2; $p = 0.0052$; Table 1.4).

3.1.2: Phosphorus Fractionation:

P fractionation was conducted for the open-water sites to determine the proportion of each species and the results can be found in Table 1.5. The same mixed model analysis was used to assess the P fractionation data as for the physiochemical properties and the results can be found in Table 1.6. Only samples from the open-water landscape position were assessed using P fractionation.

For the residual-P, statistical contrasts identified significant differences between marsh units ($p = 0.0308$; Table 1.6) with higher levels in Lake Francis and to a lesser but still significant extent center marsh. There were no significant differences detected between marsh units for the other P fractions. Due to the small sample size for the P fractionation caution should be taken with regards to any statistical interpretation of the results; however, they are still of use for guiding my interpretation of the data.

Table 1.5: Results from sequential P fractionation for surficial (0-5cm) sediment samples from ten locations at Delta Marsh (Figure 1.1), Manitoba, Canada, collected between August and October 2015.

Site Name	H2O-P	NaHCO3-P	NaOH-P	HCL-P	Residual-P
	----------mg kg-1----------				
Big Lake open-water	4.08	72.60	141.0	331.8	146.4
Canvasback Bay open-water	9.66	89.40	151.8	403.8	118.2
Weedy Bay open-water	5.41	52.98	82.8	412.2	93.6
Eaglenest 1 open-water	6.12	50.28	123.0	433.2	140.4
Eaglenest 2 open-water	8.22	91.80	213.0	389.4	171.0
Cadham Bay open-water	5.71	54.42	111.6	449.4	127.8
Lyttle Bay open-water	4.79	49.56	111.0	454.2	136.2
Waterhen Bay open-water	10.20	38.04	107.4	354.6	101.4
Lake Francis 1 open-water	6.06	55.26	147.0	306.0	151.2
Lake Francis 2 open-water	5.59	56.22	191.4	292.2	179.4
Overall Mean	6.58	61.10	138.0	382.7	136.6

Table 1.6: Results from mixed-model analysis looking at statistically significant differences of P fractionation from surficial (0-5cm) sediment samples across marsh units at Delta Marsh (Figure 1.1), Canada, collected between August and October 2015.

Parameter	Variable	NumDF	DenDF	F-value	P-value	
H2O-P	Marsh Unit	3	8.9	0.136	0.9359	
NaHCO3-P	Marsh Unit	3	8.9	2.025	0.1809	
NaOH-P	Marsh Unit	3	10.0	2.816	0.0935	
HCL-P	Marsh Unit	3	8.7	3.304	0.0731	
Residual-P	Marsh Unit	3	10.0	4.474	0.0307	*

*,**,*** Significant at the 0.05, 0.01, and 0.001 probability levels, respectively

The relative concentration of each P fraction followed the order of HCL-P > NaOH-P or Residual-P > $NaHCO_3$-P > The H_2O-P (Table 1.5). The concentrations of NaOH-P and Residual-P were similar, and the larger fraction varies between samples; however, otherwise all samples contain fraction concentrations in the same relative order. Averaging across sample sites to facilitate comparisons across the marsh, the relative proportions of each fraction are 53% HCL-P, 19% both NaOH-P and Residual-P, 8% $NaHCO_3$-P, and 1% H_2O-P (Figure 1.3). When combining the labile and non-labile fractions together, the labile P represented 9% of the total P while non-labile P represented 91% (Figure 1.4).

Figure 1.3. Proportions of each P fraction (mg kg^{-1}) contained within surficial (0-5cm) sediment samples collected between August and October 2015, averaged across open-water sites in Delta Marsh, Canada (Figure 1.1).

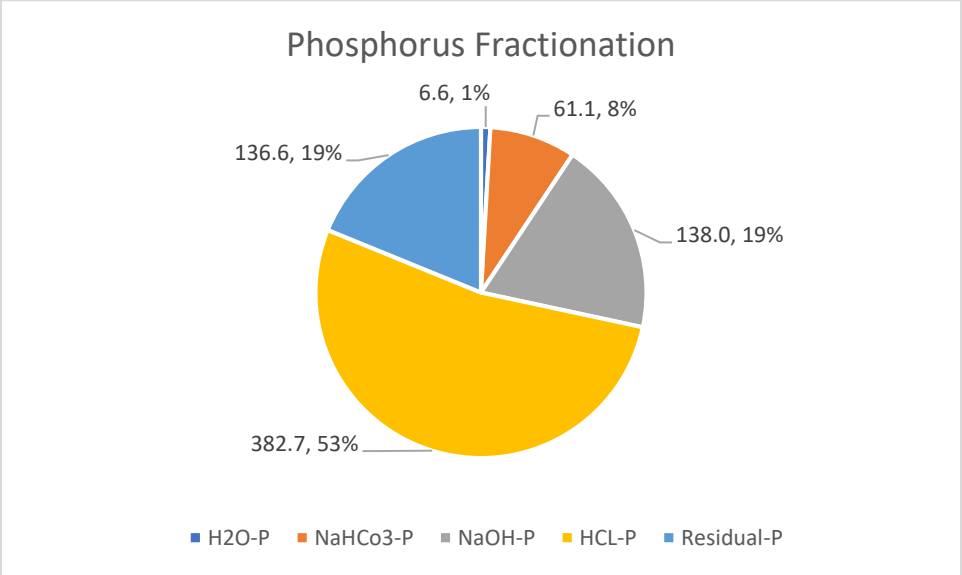
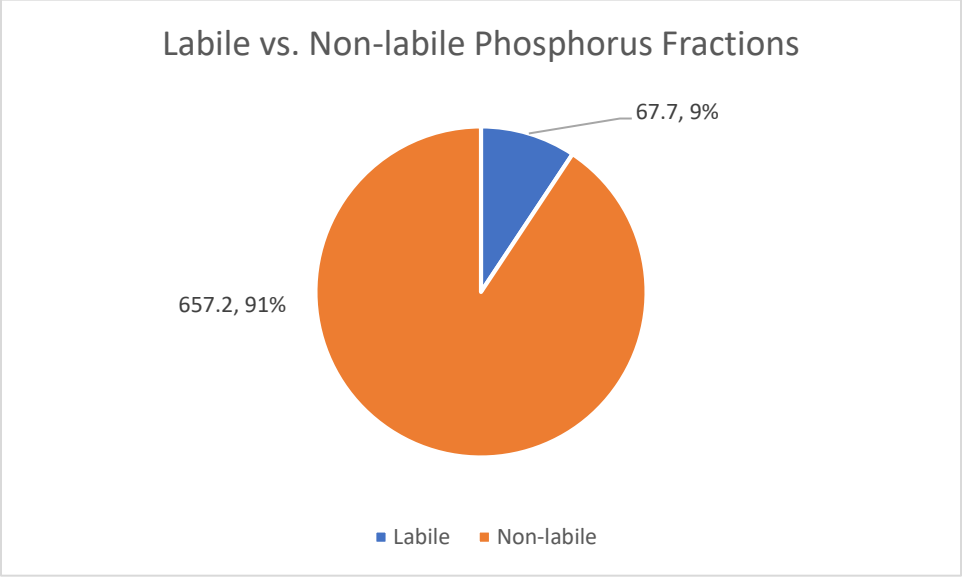


Figure 1.4. Comparison of Labile vs. Non-labile P fractions (mg kg^{-1}) contained within surficial (0-5cm) sediment samples collected between August and October 2015, averaged across open-water sites in Delta Marsh, Canada (Figure 1.1).



3.1.3: Phosphorus Sorption Parameters & Correlations with Physiochemical Parameters

The Freundlich and Two-Surface Langmuir equations demonstrated a good agreement for the isotherm fit and resulting sorption parameters, for all sites, using both SigmaPlot and the Bolster & Homberger (2007) excel spreadsheet. Coefficients of determination can be found in Table A1 (appendix) from isotherms fit using SigmaPlot as it was deemed to be the more powerful and accurate program for use in the final analysis.

The P originally sorbed by the soil (S_n) concentrations ranged from 29 mg kg^{-1} (Lyttle Bay – wet meadow) to 128 mg kg^{-1} (Lake Francis - emergent) and averaged 74 mg kg^{-1} across all sites and landscape positions (Table 1.7). Excluding LF from the analysis the highest remaining value becomes 120 mg kg^{-1} at Weedy Bay - open water. Statistical contrasts indicated that there were no significant differences within the marsh however when conducting the analysis excluding Lake Francis there were significant differences in S_n due to the marsh unit ($p = 0.0063$; Table 1.10). For this reason, analysis was conducted excluding Lake Francis for this parameter. There was not a significant difference due to the landscape position or the interaction between the landscape position and marsh unit. Across landscape positions, mean S_n concentrations were 67 mg kg^{-1}, 80 mg kg^{-1}, and 79 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.8). Across marsh units, mean S_n concentrations were 83 mg kg^{-1}, 80 mg kg^{-1}, and 55 mg kg^{-1}, for the WM, CM, and EM, units, respectively (Table 1.8). In the WM unit and EM unit S_n concentrations were not significantly different between landscape positions. In the CM unit S_n concentrations were significantly different between the open-water landscape position (mean 55 mg kg^{-1}) and wet meadow landscape position (108 mg kg^{-1}; Table 1.22; $p = 0.0323$; Table 1.11). For the open-water

landscape positions across the marsh S_n concentrations were significantly different between the WM unit (mean 90 mg kg^{-1}) and EM unit (mean 47 mg kg^{-1}; Table 1.22; p = 0.0085; Table 1.11). For the emergent landscape positions across the marsh S_n concentrations were not significantly different between marsh units. For the wet meadow landscape positions across the marsh S_n concentrations were significantly different between the CM unit (108 mg kg^{-1}) and EM unit (mean 57 mg kg^{-1}; Table 1.22; p = 0.0268; Table 1.11).

When considering all landscape positions together there were no significant correlations between S_n and any of the physiochemical properties. However, when only considering certain landscape positions significant correlations were detected (Table 1.9). For the emergent landscape position significant correlations were detected between S_n and BAP (r= 0.82; p = 0.0036), TN (r= 0.64; p = 0.0468), Mehlich-3 Mg (r= 0.68; p = 0.0305), and with Mehlich-3 Fe (r= -0.68; p = 0.0296). For the wet meadow landscape position significant correlations were detected between S_n and Mehlich-3 P (r= 0.79; p = 0.0062), Mehlich-3 Mg (r= 0.64; p = 0.0450), Mehlich-3 Ca (r= 0.94; p < 0.0001), and Mehlich-3 Fe (r= -0.83; p = 0.0030).

Table 1.7. P sorption parameters of surficial (0-5cm) sediments and soils collected from twenty-two locations in Delta Marsh (Figure 1.1), Manitoba, Canada, between August and October 2015.

Site Name	S_n	S_{max1}	S_{max2}	S_{maxT}	K_1	K_2	EPC_0	PEBC	DPS
	----------mg kg^{-1}----------				-------L mg^{-1}-------		mg L^{-1}	L kg^{-1}	%
Big Lake Open Water	63	1037	332	1369	0.26	39.4	0.033	527	4.6
Big Lake Emergent	69	647	294	941	0.16	13.6	0.113	333	7.4
Big Lake Wet Meadow	76	637	142	779	0.13	31.6	0.090	228	9.8
Canvasback Bay Open Water	87	1248	545	1793	0.20	20.9	0.062	667	4.9
Weedy Bay Open Water	120	968	449	1417	0.67	42.8	0.022	789	8.6
Eaglenest 1 Open Water	49	643	314	958	0.35	17.6	0.031	373	5.2
Eaglenest 2 Open Water	62	2386	565	2951	0.05	23.3	0.112	611	2.1
Eaglenest Emergent	99	1640	760	2400	0.08	15.7	0.038	733	4.1
Eaglenest Wet Meadow	108	459	174	632	1.38	16.5	0.151	221	17.1
Cadham Bay Open Water	42	862	355	1217	0.15	19.4	0.022	410	3.4
Cadham Bay Emergent	74	1318	280	1598	0.06	20.2	0.065	347	4.7
Cadham Bay Wet Meadow	89	691	145	836	0.17	44.9	0.285	269	10.6
Lyttle Bay Open Water	62	1617	410	2026	0.06	22.4	0.030	474	3.1
Lyttle Bay Emergent	43	975	242	1217	0.03	8.2	0.070	220	3.6
Lyttle Bay Wet Meadow	29	835	214	1049	0.06	19.7	0.060	235	2.9
Waterhen Bay Open Water	38	744	273	1017	0.09	11.6	0.166	276	3.8
Waterhen Bay Emergent	66	934	238	1172	0.10	27.7	0.112	309	5.7
Waterhen Bay Wet Meadow	52	492	119	611	0.08	19.6	0.169	151	8.6
Lake Francis 1 Open Water	35	742	222	965	0.21	23.9	0.013	316	3.7
Lake Francis 2 Open Water	113	1382	497	1879	0.21	18.2	0.003	666	6.0
Lake Francis Emergent	128	1547	338	1885	0.16	44.2	0.333	588	6.8
Lake Francis Wet Meadow	118	1170	182	1353	0.08	45.9	0.104	308	8.7
Overall Mean	74	1044	322	1366	0.22	24.88	0.095	411	6.15

* S_o, originally sorbed P; S_{max1}, low-affinity maximum P sorption capacity; S_{max2}, high-affinity maximum P sorption capacity; S_{max}, total maximum P sorption capacity; K_1, low-affinity binding energy of P sorption, K_2, high-affinity binding energy of P sorption; EPC_0 equilibrium P concentration; PEBC, P equilibrium buffering capacity; DPS, degree of P saturation.

Table 1.8. Mean values by location, landscape position, and marsh unit for P sorption parameters of surficial (0-5cm) sediments and soils collected from twenty-two locations in Delta Marsh (Figure 1.1), Manitoba, Canada, between August and October 2015.

Location	S_n	S_{max1}	S_{max2}	S_{maxT}	K_1	K_2	EPC_0	PEBC	DPS
West Marsh	----------mg kg^{-1}----------				-------L mg^{-1}-------		mg L^{-1}	L kg^{-1}	%
Open-water	90	1084	442	1526	0.38	34.4	0.039	661	6.0
Emergent	69	647	294	941	0.16	13.6	0.113	333	7.4
Wet meadow	76	637	142	779	0.13	31.6	0.090	228	9.8
Center Marsh									
Open-water	55	1515	439	1954	0.20	20.4	0.072	492	3.7
Emergent	99	1640	760	2400	0.08	15.7	0.038	733	4.1
Wet meadow	108	459	174	632	1.38	16.5	0.151	221	17.1
East Marsh									
Open-water	47	1074	346	1420	0.10	17.8	0.073	387	3.4
Emergent	61	1076	253	1329	0.06	18.7	0.082	292	4.7
Wet meadow	57	673	159	832	0.11	28.1	0.172	218	7.4
Lake Francis									
Open-water	74	1062	360	1422	0.21	21.1	0.008	491	4.8
Emergent	128	1547	338	1885	0.16	44.2	0.333	588	6.8
Wet meadow	118	1170	182	1353	0.08	45.9	0.104	308	8.7
Landscape Position									
Open-water (n=10)	67	1163	396	1559	0.23	24.0	0.049	511	4.5
Emergent (n=6)	80	1177	359	1535	0.10	21.6	0.122	422	5.4
Wet meadow (n=6)	79	714	163	877	0.32	29.7	0.143	235	9.6
Marsh Unit									
West Marsh (n=5)	83	907	352	1260	0.29	29.7	0.064	509	7.0
Center Marsh (n=4)	80	1282	453	1735	0.47	18.3	0.083	485	7.1
East Marsh (n=9)	55	941	253	1194	0.09	21.5	0.109	299	5.1
Lake Francis (n=4)	98	1210	310	1520	0.16	33.1	0.113	469	6.3
Overall Mean	74	1044	322	1366	0.22	24.88	0.095	411	6.15

Table 1.9a. Correlation coefficients between soil physiochemical properties and P sorption parameters of surficial (0-5cm) sediments and soils collected from twenty-two locations in Delta Marsh (Figure 1.1), Manitoba, Canada, between August and October 2015. This analysis includes all landscape positions.

	S_n	S_{max1}	S_{max2}	S_{maxT}	k_1	k_2	EPC_0	PEBC	DPS	Mehlich Ca	Mehlich P	Mehlich Al	Mehlich Fe	Mehlich Mg	Total P	Total N	BAP	Organic P
S_n	1																	
S_{max1}	0.26	1																
S_{max2}	0.26	**0.72**[a]	1															
S_{maxT}	0.27	**0.98**[c]	**0.84**[b]	1														
k_1	0.38	-0.35	-0.11	-0.31	1													
k_2	0.54	0.09	-0.17	0.03	0.03	1												
EPC_0	0.28	-0.08	-0.37	-0.16	0.02	0.36	1											
PEBC	0.51	**0.66**[a]	**0.89**[c]	**0.76**[a]	0.02	0.21	-0.27	1										
DPS	0.55	-0.5	-0.45	-0.51	**0.73**[a]	0.29	0.42	-0.28	1									
Mehlich Ca	-0.04	0.32	0.11	0.28	-0.03	-0.07	-0.11	0.11	-0.25	1								
Mehlich P	0.32	-0.09	-0.26	-0.14	0.33	0.21	0.22	-0.3	0.52	-0.26	1							
Mehlich Al	0.01	-0.17	-0.27	-0.2	0.37	-0.01	0.15	-0.37	0.44	-0.37	**0.74**[a]	1						
Mehlich Fe	-0.17	0.19	0.55	0.3	-0.22	-0.23	-0.55	0.52	-0.54	0.02	**-0.69**[a]	-0.58	1					
Mehlich Mg	0.33	-0.17	-0.41	-0.25	0.07	0.41	0.49	-0.39	0.54	-0.32	**0.79**[b]	0.59	**-0.77**[b]	1				
Total P	0.36	-0.02	-0.33	-0.1	-0.04	0.37	0.61	-0.3	0.42	-0.27	**0.66**[a]	0.45	-0.62	**0.83**[b]	1			
Total N	0.31	0.28	0.07	0.24	-0.22	0.19	0.24	0.02	0.02	-0.25	0.58	0.31	-0.36	0.6	**0.75**[a]	1		
BAP	0.39	0.24	-0.1	0.16	-0.13	0.32	0.21	-0.04	0.17	-0.03	0.52	0.34	-0.36	0.45	0.63	**0.65**[a]	1	
Organic P	0.24	-0.11	-0.28	-0.16	-0.14	0.33	0.33	-0.29	0.26	-0.44	**0.72**[a]	0.51	-0.56	**0.88**[c]	**0.78**[b]	**0.74**[a]	0.52	1
O.M %	0.35	0.23	0.06	0.19	-0.26	0.29	0.21	0.02	0.05	-0.34	0.56	0.23	-0.32	**0.68**[a]	**0.74**[a]	**0.92**[c]	0.49	**0.81**[b]

a,b,c Significant at the 0.05, 0.01, and 0.001 probability levels, respectively

Table 1.9b. Correlation coefficients between soil physiochemical properties, P sorption parameters, and P fractionation of surficial (0-5cm) sediments and soils collected at Delta Marsh (Figure 1.1), Manitoba, Canada, between August and October 2015, for the open-water sites only.

	S_n	S_{max1}	S_{max2}	S_{maxT}	k_1	k_2	EPC_0	PEBC	DPS	Mehlich Ca	Mehlich P	Mehlich Al	Mehlich Fe
S_n	1												
S_{max1}	0.25	1											
S_{max2}	**0.68[a]**	**0.77[b]**	1										
S_{maxT}	0.34	**0.99[c]**	**0.84[b]**	1									
k_1	0.56	-0.43	-0.03	-0.37	1								
k_2	0.46	0.03	0.11	0.04	**0.67[a]**	1							
EPC_0	-0.34	0.21	0.03	0.18	-0.42	-0.39	1						
PEBC	**0.92[c]**	0.47	**0.83[b]**	0.55	0.46	0.56	-0.27	1					
DPS	**0.76[a]**	-0.39	0.12	-0.31	**0.91[c]**	0.52	-0.4	0.57	1				
Mehlich Ca	-0.37	0.16	0	0.14	-0.39	-0.22	-0.26	-0.24	-0.5	1			
Mehlich P	-0.55	0.21	-0.1	0.16	-0.49	-0.44	0.32	-0.43	-0.6	-0.07	1		
Mehlich Al	-0.2	0.36	0.2	0.35	-0.07	-0.14	0.31	-0.04	-0.26	0	0.71a	1	
Mehlich Fe	0.6	-0.13	0.29	-0.06	**0.71[a]**	**0.77[b]**	-0.32	**0.66[a]**	**0.68[a]**	-0.34	-0.4	-0.14	1
Mehlich Mg	-0.38	0.28	-0.12	0.22	-0.54	-0.15	-0.37	-0.29	-0.59	0.49	0.46	0.14	-0.44
Total P	0.01	**0.79[b]**	**0.68[a]**	**0.80[b]**	-0.49	-0.11	0.32	0.31	-0.51	0.1	0.48	0.43	0
Total N	-0.17	0.54	0.14	0.49	-0.6	-0.35	-0.03	-0.15	-0.54	0.07	**0.67[a]**	0.38	-0.52
BAP	0.15	**0.64[a]**	0.63	**0.66[a]**	-0.17	0.16	0.16	0.45	-0.25	-0.09	0.33	0.32	0.15
Organic P	-0.3	-0.48	-0.53	-0.51	-0.31	-0.4	-0.17	-0.53	-0.12	-0.08	0.19	-0.38	-0.26
O.M %	0.25	0.3	0.04	0.26	-0.11	0.06	-0.39	0.11	0.02	-0.15	0.12	-0.03	-0.28
H2O-P	-0.17	0.1	0.2	0.12	-0.34	-0.56	**0.81[b]**	-0.13	-0.25	-0.34	0.33	0.2	-0.23
NaHCO3-P	0.21	**0.63[a]**	**0.68[a]**	**0.67[a]**	-0.18	0.22	0.08	0.51	-0.22	-0.07	0.37	0.34	0.33
NaOH-P	0.1	**0.67[a]**	0.5	**0.66[a]**	-0.47	-0.22	0.08	0.23	-0.37	-0.11	0.6	0.4	-0.28
HCL-P	-0.11	0.08	0.18	0.1	0.06	-0.03	0.01	0.03	-0.1	0.63	-0.19	0.23	0.14
Residual-P	-0.02	0.5	0.22	0.47	-0.43	-0.17	-0.27	0.03	-0.38	0.15	0.53	0.32	-0.39

	H2O-P	NaHCO3-P	NaOH-P	HCL-P
H2O-P	1			
NaHCO3-P	0.23	1		
NaOH-P	0.18	**0.66[a]**	1	
HCL-P	-0.01	-0.03	-0.48	1
Residual-P	-0.28	0.35	**0.85[b]**	-0.44

	Mehlich Mg	Total P	Total N	BAP	Organic P	O.M %
Mehlich Mg	1					
Total P	0.19	1				
Total N	**0.75[a]**	0.41	1			
BAP	0.08	**0.86[b]**	0.19	1		
Organic P	0.37	-0.39	0.31	-0.56	1	
O.M %	0.58	-0.11	**0.73[a]**	-0.1	0.32	1
H2O-P	-0.47	0.43	-0.09	0.36	-0.13	-0.48
NaHCO3-P	0.19	**0.88[c]**	0.26	**0.91[c]**	-0.37	-0.08
NaOH-P	0.55	**0.66[a]**	**0.78[b]**	0.63	0.01	0.52
HCL-P	-0.24	0.14	-0.38	-0.04	-0.48	-0.62
Residual-P	**0.85[b]**	0.34	**0.90[c]**	0.27	0.24	**0.77[b]**

a,b,c Significant at the 0.05, 0.01, and 0.001 probability levels, respectively

Table 1.9c. Correlation coefficients between soil physiochemical properties and P sorption parameters of surficial (0-5cm) sediments and soils collected at Delta Marsh (Figure 1.1), Manitoba, Canada, between August and October 2015, for emergent zone sites only.

	S_n	S_{max1}	S_{max2}	S_{maxT}	k_1	k_2	EPC_0	PEBC	DPS	Mehlich Ca	Mehlich P	Mehlich Al	Mehlich Fe	Mehlich Mg	Total P	Total N	BAP	Organic P
S_n	1																	
S_{max1}	**0.74**[a]	1																
S_{max2}	0.47	**0.65**[a]	1															
S_{maxT}	**0.70**[a]	**0.96**[c]	**0.84**[b]	1														
k_1	0.57	-0.13	-0.03	-0.1	1													
k_2	**0.78**[b]	0.44	-0.11	0.27	0.54	1												
EPC_0	**0.69**[a]	0.23	-0.24	0.08	**0.67**[a]	**0.87**[c]	1											
PEBC	**0.84**[b]	**0.82**[b]	**0.87**[b]	**0.91**[c]	0.29	0.38	0.24	1										
DPS	0.39	-0.32	-0.29	-0.34	**0.96**[c]	0.52	**0.64**[a]	0.03	1									
Mehlich Ca	0.04	0.17	-0.48	-0.06	-0.21	0.22	0.21	-0.29	-0.07	1								
Mehlich P	0.42	0.62	**0.82**[b]	**0.74**[a]	-0.11	0.1	-0.04	**0.75**[a]	-0.35	**-0.64**[a]	1							
Mehlich Al	0.2	0.26	**0.88**[c]	0.51	0.08	-0.28	-0.42	**0.65**[a]	-0.13	**-0.7**[6a]	**0.72**[a]	1						
Mehlich Fe	**-0.68**[a]	-0.61	0.14	-0.39	-0.18	**-0.82**[b]	**-0.79**[b]	-0.3	-0.16	-0.61	0.01	0.52	1					
Mehlich Mg	**0.68**[a]	**0.83**[b]	**0.71**[a]	**0.86**[b]	0	0.36	0.32	**0.83**[b]	-0.26	-0.27	**0.86**[b]	0.4	-0.43	1				
Total P	0.57	0.48	0.1	0.38	0.18	**0.77**[b]	**0.65**[a]	0.41	0.1	-0.19	0.56	-0.04	-0.6	**0.66**[a]	1			
Total N	**0.64**[a]	**0.72**[a]	0.6	**0.74**[a]	0.05	0.46	0.39	**0.74**[a]	-0.18	-0.39	**0.89**[c]	0.37	-0.41	**0.96**[c]	**0.82**[b]	1		
BAP	**0.82**[b]	0.59	0.07	0.45	0.45	**0.81**[b]	**0.65**[a]	0.49	0.42	0.52	-0.07	-0.2	**-0.80**[b]	0.25	0.35	0.21	1	
Organic P	0.63	**0.71**[a]	**0.76**[b]	**0.80**[b]	0.07	0.34	0.22	**0.83**[b]	-0.18	-0.53	**0.96**[c]	0.6	-0.22	**0.93**[c]	**0.71**[a]	**0.96**[c]	0.16	1
O.M %	0.44	**0.68**[a]	**0.67**[a]	**0.74**[a]	-0.18	0.24	0.1	**0.68**[a]	-0.4	-0.48	**0.96**[c]	0.5	-0.2	**0.91**[c]	**0.72**[a]	**0.95**[c]	0	**0.96**[c]

a,b,c Significant at the 0.05, 0.01, and 0.001 probability levels, respectively

Table 1.9d. Correlation coefficients between soil physiochemical properties and P sorption parameters of surficial (0-5cm) sediments and soils collected at Delta Marsh (Figure 1.1), Manitoba, Canada, between August and October 2015, for wet meadow zone sites only.

	S_n	S_{max1}	S_{max2}	S_{maxT}	k_1	k_2	EPC_0	PEBC	DPS	Mehlich Ca	Mehlich P	Mehlich Al	Mehlich Fe	Mehlich Mg	Total P	Total N	BAP	Organic P
S_n	1																	
S_{max1}	0.23	1																
S_{max2}	-0.07	0.54	1															
S_{maxT}	0.21	**0.99**[c]	0.62	1														
k_1	0.46	-0.5	0.11	-0.45	1													
k_2	0.52	**0.65**[a]	-0.09	0.6	-0.45	1												
EPC_0	0.27	-0.33	-0.55	-0.38	0.11	0.35	1											
PEBC	0.57	**0.80**[b]	0.5	**0.81**[b]	-0.12	**0.79**[b]	0	1										
DPS	**0.72**[a]	-0.49	-0.33	-0.5	**0.84**[b]	-0.07	0.41	-0.04	1									
Mehlich Ca	**0.94**[c]	0.08	0.07	0.08	**0.69**[a]	0.31	0.31	0.49	**0.82**[b]	1								
Mehlich P	**0.79**[b]	0.47	0.38	0.48	0.44	0.23	-0.19	0.54	0.43	**0.79**[b]	1							
Mehlich Al	0.04	-0.05	0.55	0.02	0.6	**-0.65**[a]	-0.5	-0.17	0.19	0.24	0.55	1						
Mehlich Fe	-0.83	0	0.12	0.02	-0.61	-0.19	-0.43	-0.23	**-0.75**[a]	**-0.88**[c]	**-0.76**[a]	-0.32	1					
Mehlich Mg	0.64	0.55	-0.06	0.5	-0.12	**0.71**[a]	0.45	0.57	0.15	0.54	0.59	-0.13	**-0.69**[a]	1				
Total P	0.37	0.32	-0.22	0.27	-0.13	0.29	0.24	0.08	0.07	0.27	0.5	0.14	**-0.64**[a]	**0.82**[b]	1			
Total N	0.16	0.59	0.13	0.56	-0.3	0.12	-0.48	0.12	-0.28	-0.01	0.55	0.35	-0.21	0.45	**0.72**[a]	1		
BAP	0.27	0.58	0.16	0.56	-0.2	0.13	**-0.64**[a]	0.2	-0.17	0.08	0.61	0.35	-0.16	0.3	0.51	**0.94**[c]	1	
Organic P	0.07	**0.91**[c]	0.53	**0.91**[c]	-0.49	0.45	-0.27	0.58	-0.56	-0.03	0.44	0.14	-0.08	0.61	0.56	**0.71**[a]	0.57	1
O.M %	0.24	**0.79**[b]	0.12	**0.75**[a]	-0.5	0.52	-0.14	0.42	-0.37	0.05	0.49	0.02	-0.26	**0.75**[a]	**0.81**[b]	**0.87**[b]	**0.73**[a]	**0.89**[c]

a,b,c Significant at the 0.05, 0.01, and 0.001 probability levels, respectively

Table 1.10a. Results from mixed-model analysis looking at statistically significant differences of sorption parameters of surficial (0-5cm) sediments across locations at Delta Marsh (Figure 1.1), collected between August and October 2015. Analysis includes Lake Francis.

Parameter[i]	Variable	NumDF	DenDF	F-value	P-value	
S_n	Marsh Unit	3	6.2	2.254	0.1801	
	Landscape Position	2	8.6	4.184	0.0538	
	Unit x Position	6	8.5	1.050	0.4578	
S_{max1}	Marsh Unit	3	9.4	0.590	0.6362	
	Landscape Position	2	11.2	25.420	**6.94E-05**	***
	Unit x Position	6	11.1	6.201	**0.0046**	**
S_{max2}	Marsh Unit	3	10.0	1.715	0.2270	
	Landscape Position	2	11.8	215.499	**5.24E-10**	***
	Unit x Position	6	11.8	33.695	**9.54E-07**	***
S_{maxT}	Marsh Unit	3	9.6	0.055	0.9821	
	Landscape Position	2	11.4	35.192	**1.36E-05**	***
	Unit x Position	6	11.3	6.347	**0.0040**	**
K_1	Marsh Unit	3	9.9	4.215	**0.0366**	*
	Landscape Position	2	11.6	11.309	**0.0019**	**
	Unit x Position	6	11.6	28.341	**2.88E-06**	***
K_2	Marsh Unit	3	22.0	4.435	**0.0139**	*
	Landscape Position	2	22.0	1.802	0.1885	
	Unit x Position	6	22.0	2.284	0.0726	
EPC_0	Marsh Unit	3	6.3	0.183	0.9046	
	Landscape Position	2	11.0	19.908	**0.0002**	***
	Unit x Position	6	10.8	7.130	**0.0028**	**
PEBC	Marsh Unit	3	5.8	2.020	0.2151	
	Landscape Position	2	9.0	33.528	**6.67E-05**	***
	Unit x Position	6	8.9	3.037	0.0668	
DPS	Marsh Unit	3	8.6	2.141	0.1679	
	Landscape Position	2	11.5	27.724	**3.99E-05**	***
	Unit x Position	6	11.4	3.551	**0.0317**	*

S_o, originally sorbed P; S_{max1}, low-affinity maximum P sorption capacity; S_{max2}, high-affinity maximum P sorption capacity; S_{max}, total maximum P sorption capacity; K_1, low-affinity binding energy of P sorption, K_2, high-affinity binding energy of P sorption; EPC_0 equilibrium P concentration; PEBC, P equilibrium buffering capacity; DPS, degree of P saturation.
*,**,*** Significant at the 0.05, 0.01, and 0.001 probability levels, respectively

Table 1.10b. Results from mixed-model analysis looking at statistically significant differences of sorption parameters of surficial (0-5cm) sediments across locations at Delta Marsh (Figure 1.1), collected between August and October 2015. Analysis excludes Lake Francis.

Parameter[i]	Variable	NumDF	DenDF	F-value	P-value	
S_n	Marsh Unit	2	18.0	6.810	**0.0063**	**
	Landscape Position	2	18.0	1.646	0.2205	
	Unit x Position	4	18.0	2.552	0.0747	
S_{max1}	Marsh Unit	2	7.4	0.348	0.7169	
	Landscape Position	2	9.2	26.282	**0.0002**	***
	Unit x Position	4	9.2	6.226	**0.0106**	*
S_{max2}	Marsh Unit	2	7.9	1.898	0.2122	
	Landscape Position	2	9.7	152.549	**4.61E-08**	***
	Unit x Position	4	9.7	38.828	**6.01E-06**	***
S_{maxT}	Marsh Unit	2	7.5	0.080	0.9239	
	Landscape Position	2	9.3	38.043	**3.37E-05**	***
	Unit x Position	4	9.2	6.114	**0.0111**	*
K_1	Marsh Unit	2	7.9	4.942	**0.0405**	*
	Landscape Position	2	9.7	25.309	**0.0001**	***
	Unit x Position	4	9.6	28.436	**2.50E-05**	***
K_2	Marsh Unit	2	18.0	1.028	0.3777	
	Landscape Position	2	18.0	2.393	0.1198	
	Unit x Position	4	18.0	1.215	0.3392	
EPC_0	Marsh Unit	2	6.7	0.227	0.8026	
	Landscape Position	2	11.5	4.754	**0.0313**	*
	Unit x Position	4	11.4	1.254	0.3429	
PEBC	Marsh Unit	2	5.9	4.511	0.0645	
	Landscape Position	2	10.7	27.556	**5.94E-05**	***
	Unit x Position	4	10.6	4.257	**0.0265**	*
DPS	Marsh Unit	2	7.1	2.853	0.1231	
	Landscape Position	2	9.8	25.014	**0.0001**	***
	Unit x Position	4	9.7	3.975	**0.0361**	*

S_o, originally sorbed P; S_{max1}, low-affinity maximum P sorption capacity; S_{max2}, high-affinity maximum P sorption capacity; S_{max}, total maximum P sorption capacity; K_1, low-affinity binding energy of P sorption, K_2, high-affinity binding energy of P sorption; EPC_0 equilibrium P concentration; PEBC, P equilibrium buffering capacity; DPS, degree of P saturation.
*,**,*** Significant at the 0.05, 0.01, and 0.001 probability levels, respectively

Table 1.11a. Detailed results from mixed-model analysis looking at statistically significant differences for P sorption parameters of surficial (0-5cm) sediments, between landscape positions within marsh units of Delta Marsh (Figure 1.1). Samples were collected between August and October 2015

	West Marsh	Center Marsh	East Marsh	Lake Francis
Sn	-	OW-WM[a]	-	-
Smax1	-	OW-WM[c], E-WM[c]	OW-WM[a], E-WM[b]	-
Smax2	OW-WM[c], E-WM[b]	OW-E[c], OW-WM[c], E-WM[c]	OW-E[c], OW-WM[c], E-WM[c]	OW-E[b], OW-WM[c], E-WM[b]
SmaxT	-	OW-WM[c], E-WM[c]	OW-WM[b], E-WM[a]	-
K1	-	OW-WM[c], E-WM[c]	OW-E[a]	OW-WM[b]
K2	OW-E[a]	-	-	-
EPC0	-	-	OW-WM[a]	OW-E[c], OW-WM[a]
PEBC	OW-E[a], OW-WM[b]	OW-WM[b], E-WM[b]	OW-WM[b]	OW-WM[a]
DPS	-	OW-WM[c]	OW-WM[b]	-

Landscape Position: OW = Open-water; E = Emergent; WM = Wet meadow
Significance: [a] $P \leq 0.05$, [b] $P \leq 0.01$, [c] $P \leq 0.001$.

Table 1.11b. Detailed results from mixed-model analysis looking at statistically significant differences for P sorption parameters of surficial (0-5cm) sediments, between marsh units within landscape positions of Delta Marsh (Figure 1.1). Samples were collected between August and October 2015

	Open Water	Emergent	Wet Meadow
Sn	W-E[b]	-	C-E[a]
Smax1	-	-	C-LF[a]
Smax2	-	C-E[b], C-LF[b]	-
SmaxT	-	-	-
K1	-	-	W-C[c], C-E[c], C-LF[c]
K2	-	W-LF[a], E-LF[a]	-
EPC0	W-LF[a], C-LF[b], E-LF[b]	C-LF[a]	-
PEBC	W-E[a]	-	-
DPS	-	-	C-E[b]

Marsh Units: W = West; C = Centre; E = East; LF = Lake Francis
Significance: [a] $P \leq 0.05$, [b] $P \leq 0.01$, [c] $P \leq 0.001$.

The S_{max1} ranged from 459 mg kg^{-1} (Eaglenest – wet meadow) to 2386 mg kg^{-1} (Eaglenest – open-water) and averaged 1044 mg kg^{-1} across all sites and landscape positions (Table 1.7). Statistical contrasts indicated that there were no significant differences between locations for S_{max1} due to the marsh unit however there was due to landscape position ($p < 0.0001$) as well as the interaction between marsh unit and landscape position ($p = 0.0046$; Table 1.10). Across landscape positions, mean S_{max1} concentrations were 1,163 mg kg^{-1}, 1,177 mg kg^{-1}, and 714 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.8). Across marsh units, mean S_{max1} concentrations were 907 mg kg^{-1}, 1,282 mg kg^{-1}, 941 mg kg^{-1}, and 1,210 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.8). In the WM unit and LF unit S_{max1} concentrations were not significantly different between landscape positions. In the CM unit S_{max1} concentrations were significantly different between the wet meadow landscape position (459 mg kg^{-1}) and both the open-water landscape position (mean 1,515 mg kg^{-1}; $p < 0.0001$), and the emergent landscape position (1,640 mg kg^{-1}; Table 1.8; $p = 0.0003$; Table 1.11). In the EM unit S_{max1} concentrations were significantly different between the wet meadow landscape position (mean 673 mg kg^{-1}) and both the open-water landscape position (mean 1,074 mg kg^{-1}; $p = 0.0125$), and the emergent landscape position (mean 1,076 mg kg^{-1}; Table 1.8; $p = 0.0066$; Table 1.11). For the open-water and emergent landscape positions across the marsh S_{max1} concentrations were not significantly different between marsh units. For the wet meadow landscape positions across the marsh S_{max1} concentrations were significantly different between the CM unit (459 mg kg^{-1}) and LF unit (1,170 mg kg^{-1}; Table 1.8; $p = 0.0292$; Table 1.11).

When considering all landscape positions together there were no significant correlations between S_{max1} and any of the physiochemical properties however when only considering certain landscape positions significant correlations were detected (Table 1.9). For the open-water landscape position sites significant correlations were detected between S_{max1} and Total P (r= 0.79; p = 0.0069), BAP (r= 0.64; p = 0.0454), $NaHCO_3$-P (r= 0.63; p = 0.0490), and NaOH-P (r= 0.67; p = 0.0344). For the emergent landscape position sites significant correlations were detected between S_{max1} and Organic P (r= 0.71; p = 0.0203), TN (r= 0.72; p = 0.0187), Mehlich-3 Mg (r= 0.83; p = 0.0032), and OM (r= 0.68; p = 0.0299). For the wet meadow landscape position sites significant correlations were detected between S_{max1} and Organic P (r= 0.91; p = 0.0002), and OM (r= 0.79; p = 0.0068).

The S_{max2} ranged from 119 mg kg^{-1} (Waterhen Bay – wet meadow) to 760 mg kg^{-1} (Eaglenest – emergent) and averaged 322 mg kg^{-1} across all sites and landscape positions (Table 1.7). Statistical contrasts indicated that there were no significant differences between locations for S_{max2} due to the marsh unit however there was due to landscape position ($p < 0.0001$) as well as the interaction between marsh unit and landscape position ($p < 0.0001$; Table 1.10). Across landscape positions, mean S_{max2} concentrations were 396 mg kg^{-1}, 359 mg kg^{-1}, and 163 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.8). Across marsh units, mean S_{max2} concentrations were 352 mg kg^{-1}, 453 mg kg^{-1}, 253 mg kg^{-1}, and 310 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.8). In the WM unit S_{max2} concentrations were significantly different between the wet meadow landscape position (142 mg kg^{-1}) and both the open-water landscape position (mean 442 mg kg^{-1}; p = 0.0001), and the emergent landscape position (294 mg kg^{-1}; Table 1.8; p = 0.0012; Table 1.11).

In the CM unit S_{max2} concentrations were significantly different between the open-water landscape position (mean 439 mg kg^{-1}) and both the emergent landscape position (760 mg kg^{-1}; $p = 0.0001$), and the wet meadow landscape position (174 mg kg^{-1}; $p < 0.0001$), as well as between the emergent landscape position (760 mg kg^{-1}) and the wet meadow landscape position (174 mg kg^{-1}; Table 1.8; $p < 0.0001$; Table 1.11). In the EM unit S_{max2} concentrations were significantly different between the open-water landscape position (mean 346 mg kg^{-1}) and both the emergent landscape position (mean 253 mg kg^{-1}; $p = 0.0008$), and the wet meadow landscape position (mean 159 mg kg^{-1}; $p < 0.0001$), as well as between the emergent landscape position (mean 253 mg kg^{-1}) and the wet meadow landscape position (mean 159 mg kg^{-1}; Table 1.8; $p = 0.0007$; Table 1.11). In the LF unit S_{max2} concentrations were significantly different between the open-water landscape position (mean 360 mg kg^{-1}) and both the emergent landscape position (338 mg kg^{-1}; $p = 0.0011$), and the wet meadow landscape position (182 mg kg^{-1}; $p < 0.0001$), as well as between the emergent landscape position (338 mg kg^{-1}) and the wet meadow landscape position (182 mg kg^{-1}; Table 1.8; $p = 0.0011$; Table 1.11). For the open-water and wet meadow landscape positions across the marsh S_{max2} concentrations were not significantly different between marsh units. For the emergent landscape positions across the marsh S_{max2} concentrations were significantly different between the CM unit (760 mg kg^{-1}) and both the EM unit (mean 253 mg kg^{-1}; $p = 0.0059$), and the LF unit (338 mg kg^{-1}; Table 1.8; $p = 0.0051$; Table 1.11).

When considering all landscape positions together there were no significant correlations between S_{max2} and any of the physiochemical properties; however, significant correlations were detected when only considering certain landscape positions (Table 1.9). For the open-water

landscape position sites significant correlations were detected between S_{max2} and Total P (r= 0.68; p = 0.0292), $NaHCO_3$-P (r= 0.68; p = 0.0310). For the emergent landscape position sites significant correlations were detected between S_{max2} and Mehlich-3 P (r= 0.82; p = 0.0040), Organic P (r= 0.76; p = 0.0099), Mehlich-3 Al (r= 0.88; p = 0.0009), Mehlich-3 Mg (r= 0.71; p = 0.0218), and OM (r= 0.67; p = 0.0325). For the wet meadow landscape position sites there were no significant correlations detected between S_{max2} and any of the physiochemical properties.

The S_{maxT} ranged from 611 mg kg^{-1} (Waterhen Bay – wet meadow) to 2951 mg kg^{-1} (Eaglenest 2 – open-water) and averaged 1367 mg kg^{-1} across all sites and landscape positions (Table 1.7). Statistical contrasts indicated that there were no significant differences between locations for S_{maxT} due to the marsh unit however there was due to landscape position ($p < 0.0001$) as well as the interaction between marsh unit and landscape position (p = 0.0040; Table 1.10). Across landscape positions, mean S_{maxT} concentrations were 1,559 mg kg^{-1}, 1,535 mg kg^{-1}, and 877 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.8). Across marsh units, mean S_{maxT} concentrations were 1,260 mg kg^{-1}, 1,735 mg kg^{-1}, 1,194 mg kg^{-1}, and 1,520 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.8). In the WM unit and LF unit S_{maxT} concentrations were not significantly different between landscape positions. In the CM unit S_{maxT} concentrations were significantly different between the wet meadow landscape position (632 mg kg^{-1}) and both the open-water landscape position (mean 1,954 mg kg^{-1}; $p < 0.0001$), and the emergent landscape position (2,400 mg kg^{-1}; Table 1.8; p = 0.0001; Table 1.11). In the EM unit S_{maxT} concentrations were significantly different between the wet meadow landscape position (mean 832 mg kg^{-1}) and both the open-water landscape position (mean 1,420 mg kg^{-1}; p = 0.0074), and the emergent landscape

position (mean 1,329 mg kg^{-1}; Table 1.8; p = 0.0202; Table 1.11). For the all landscape positions across the marsh S_{maxT} concentrations were not significantly different between marsh units

When considering all landscape positions together there were no significant correlations between S_{maxT} and any of the physiochemical properties however when only considering certain landscape positions significant correlations were detected (Table 1.9). For the open-water landscape position sites significant correlations were detected between S_{maxT} and Total P (r= 0.80; p = 0.0058), BAP (r= 066; p = 0.0365), $NaHCO_3$-P (r= 0.67; p = 0.0357), and NaOH-P (r= 0.66; p = 0.0367). For the emergent landscape position sites significant correlations were detected between S_{maxT} and Mehlich-3 P (r= 0.74; p = 0.0134), Organic P (r= 0.80; p = 0.0059), TN (r= 0.74; p = 0.0147), Mehlich-3 Mg (r= 0.80; p = 0.0016), and OM (r= 0.74; p = 0.0146). For the wet meadow landscape position sites significant correlations were detected between S_{maxT} and Organic P (r= 0.91; p = 0.0002), and OM (r= 0.75; p = 0.0130).

The P binding energy value K_1 ranged from 0.03 L mg^{-1} (Lyttle Bay – emergent) to 1.38 L mg^{-1} (Eaglenest – wet meadow) and averaged 0.22 L mg^{-1} across all sites and landscape positions (Table 1.7). Statistical contrasts identified significant differences between locations for K_1 due to the landscape position (p = 0.0019), marsh unit (p = 0.0366), and for the interaction between marsh unit and landscape position (p < 0.0001; Table 1.10). Across landscape positions, mean K_1 concentrations were 0.23 mg kg^{-1}, 0.10 mg kg^{-1}, and 0.32 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.8). Across marsh units, mean K_1 concentrations were 0.29 mg kg^{-1}, 0.47 mg kg^{-1}, 0.09 mg kg^{-1}, and 0.16 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.8). In the WM unit K_1 concentrations were not significantly different between landscape positions. In the CM unit K_1

concentrations were significantly different between the wet meadow landscape position (1.38 mg kg^{-1}) and both the open-water landscape position (mean 0.20 mg kg^{-1}; $P < 0.0001$), and the emergent landscape position (0.08 mg kg^{-1}; Table 1.8; $P < 0.0001$; Table 1.11). In the EM unit K_1 concentrations were significantly different between the open-water landscape position (mean 0.10 mg kg^{-1}) and emergent landscape position (mean 0.06 mg kg^{-1}; Table 1.8; $p = 0.0204$; Table 1.11). In the LF unit K_1 concentrations were significantly different between the open-water landscape position (mean 0.21 mg kg^{-1}) and wet meadow landscape position (0.08 mg kg^{-1}; Table 1.8; $p = 0.0079$; Table 1.11). For the open-water and emergent landscape positions across the marsh K_1 concentrations were not significantly different between marsh units. For the wet meadow landscape positions across the marsh K_1 concentrations were significantly different between the CM unit (1.38 mg kg^{-1}) and all of the WM unit (0.13 mg kg^{-1}; $p = 0.0005$), the EM unit (mean 0.11 mg kg^{-1}; $p = 0.0001$), and the LF units (0.08 mg kg^{-1}; Table 1.8; $p = 0.0001$; Table 1.11).

When considering all landscape positions together there were no significant correlations between k_1 and any of the physiochemical properties; however, significant correlations were detected when only considering certain landscape positions (Table 1.9). For the open-water landscape position sites significant correlations were detected between k_1 and Mehlich-3 Fe ($r= 0.71$; $p = 0.0218$). For the emergent landscape position sites there were no significant correlations detected between k_1 and any of the physiochemical properties. For the wet meadow landscape position sites significant correlations were detected between k_1 and Mehlich-3 Ca ($r= 0.69$; $p = 0.0273$).

The P binding energy value K_2 ranged from 8.2 L mg^{-1} (Lyttle – emergent) to 45.9 L mg^{-1} (Lake Francis – wet meadow) and averaged 24.88 L mg^{-1} across all sites and landscape positions (Table 1.7). Statistical contrasts identified significant differences between locations for K_2 due to the marsh unit ($p = 0.0366$; Table 1.10), with no effect from the landscape position or the interaction between marsh unit and landscape position. Across landscape positions, mean K_2 concentrations were 24.0 mg kg^{-1}, 21.6 mg kg^{-1}, and 29.7 mg kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.8). Across marsh units, mean K_2 concentrations were 29.7 mg kg^{-1}, 18.3 mg kg^{-1}, 21.5 mg kg^{-1}, and 33.1 mg kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.8). In the WM unit K_2 concentrations were significantly different between the open-water landscape position (mean 34.4 mg kg^{-1}) and emergent landscape position (13.6 mg kg^{-1}; Table 1.8; $p = 0.0406$; Table 1.11). In the CM, EM, and LF units K_2 concentrations were not significantly different between landscape positions. For the open-water and wet meadow landscape positions across the marsh K_2 concentrations were not significantly different between marsh units. For the emergent landscape positions across the marsh K_2 concentrations were significantly different between the LF unit (44.2 mg kg^{-1}) and both the WM unit (13.6 mg kg^{-1}; $p = 0.0421$), and the EM unit (18.7 mg kg^{-1}; Table 1.8; $p = 0.0371$; Table 1.11).

When considering all landscape positions together there were no significant correlations between k_2 and any of the physiochemical properties however when only considering certain landscape positions significant correlations were detected (Table 1.9). For the open-water landscape position sites significant correlations were detected between k_2 and Mehlich-3 Fe ($r = 0.77$; $p = 0.0091$). For the emergent landscape position sites significant correlations were

detected between k_2 and Total P (r= 0.77; p = 0.0093), BAP (r= 0.81; p = 0.0042), and negatively with Mehlich-3 Fe (r= -0.82; p = 0.0037). For the wet meadow landscape position sites significant correlations were detected between k_2 and Mehlich-3 Mg (r= 0.71; p = 0.0223), and negatively with Mehlich-3 Al (r= -0.65; p = 0.0406).

The equilibrium P concentration (EPC_0) ranged from 0.03 mg L^{-1} (LF – open-water) to 0.333 mg L^{-1} (LF - emergent) and averaged 0.095 mg L^{-1} across all sites and landscape positions (Table 1.7). Statistical contrasts identified significant differences between locations due to the landscape position (p = 0.0002), and for the interaction between marsh unit and landscape position (p = 0.0028; Table 1.10). There was not a significant difference due to the marsh unit. Across landscape positions, mean EPC_0 was 0.049 mg L^{-1}, 0.122 mg L^{-1}, and 0.143 mg L^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.8). Across marsh units, mean EPC_0 was 0.064 mg L^{-1}, 0.083 mg L^{-1}, 0.109 mg L^{-1}, and 0.113 mg L^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.8). In the WM and CM units EPC_0 were not significantly different between landscape positions. In the EM unit EPC_0 were significantly different between the open-water landscape position (mean 0.073 mg L^{-1}) and wet meadow landscape position (mean 0.172 mg L^{-1}; Table 1.8; p = 0.0401; Table 1.11). In the LF unit EPC_0 were significantly different between the open-water landscape position (mean 0.008 mg L^{-1}) and both the emergent landscape position (0.333 mg L^{-1}; p < 0.0001), and the wet meadow landscape position (0.104 mg L^{-1}; Table 1.8; p = 0.0004; Table 1.11). For the open-water landscape positions across the marsh EPC_0 were significantly different between the LF unit (mean 0.008 mg L^{-1}) and all of the WM unit (mean 0.039 mg L^{-1}; p = 0.0114), the CM unit (mean 0.072 mg L^{-1}; p = 0.0033), and the EM units (mean 0.073 mg L^{-1}; Table 1.8; p = 0.0034; Table

1.11). For the emergent landscape positions across the marsh EPC_0 were significantly different between the EM unit (mean 0.082 mg L^{-1}) and LF unit (0.333 mg L^{-1}; Table 1.8; p = 0.0108; Table 1.11). For the wet meadow landscape positions across the marsh EPC_0 were not significantly different between marsh units.

When considering all landscape positions together there were no significant correlations between EPC_0 and any of the physiochemical properties however when only considering certain landscape positions significant correlations were detected (Table 1.9). For the open-water landscape position sites significant correlations were detected between EPC_0 and H_2O-P (r= 0.81; p = 0.0048). For the emergent landscape position sites significant correlations were detected between EPC_0 and Total P (r= 0.65; p = 0.0427), BAP (r= 0.65; p = 0.0425), and negatively with Mehlich-3 Fe (r= -0.79; p = 0.0066). For the wet meadow landscape position sites significant negative correlations were detected between EPC_0 and BAP (r= -0.64; p = 0.0474).

The comparisons between EPC and TDP indicated that for the open-water sites, Big Lake, Weedy Bay, Canvasback Bay, Northwest Eaglenest Bay, Northeast Eaglenest Bay, Cadham Bay, and Lyttle Bay all generally demonstrated EPC levels that were lower than the TDP indicating that they would primarily act as a sink for water column P (Figure 1.5). If the average TDP values were to be used, then these sites would all be identified as P sinks. There were, however, some sample dates where EPC was higher than the TDP and at these times the sediments would switch and act as a source of P to the water column. Sediments in Big Lake became a P source in May 2015 (TDP=0.025 mg L^{-1}, EPC=0.033 mg L^{-1}). Weedy Bay was a P source in July 2016 (TDP=0.02 mg L^{-1}, EPC=0.022 mg L^{-1}). Canvasback Bay was a P source for

three of seven sampling dates in May 2015, May 2016, and June 2016 (TDP=0.038 mg L^{-1}, 0.016 mg L^{-1}, 0.03 mg L^{-1}, respectively, EPC=0.062 mg L^{-1}). Northwest Eaglenest Bay sediments released P in May 2016 (TDP=0.077 mg L^{-1}, EPC=0.112 mg L^{-1}), while Northeast Eaglenest Bay sediments were never a P source. Cadham Bay acted as a P sink for all sampling dates while Lyttle Bay was a P source once in July 2015 (TDP=0.024 mg L^{-1}, EPC=0.030 mg L^{-1}).

Figure 1.5a. Total dissolved P measured between 2015-2016 at selected sites at Delta Marsh (Figure 1.1) compared to calculated equilibrium P concentrations in the open-water, emergent vegetation, and wet-meadow zone sediments. Only open-water data was obtained for Canvasback and Weedy Bay's

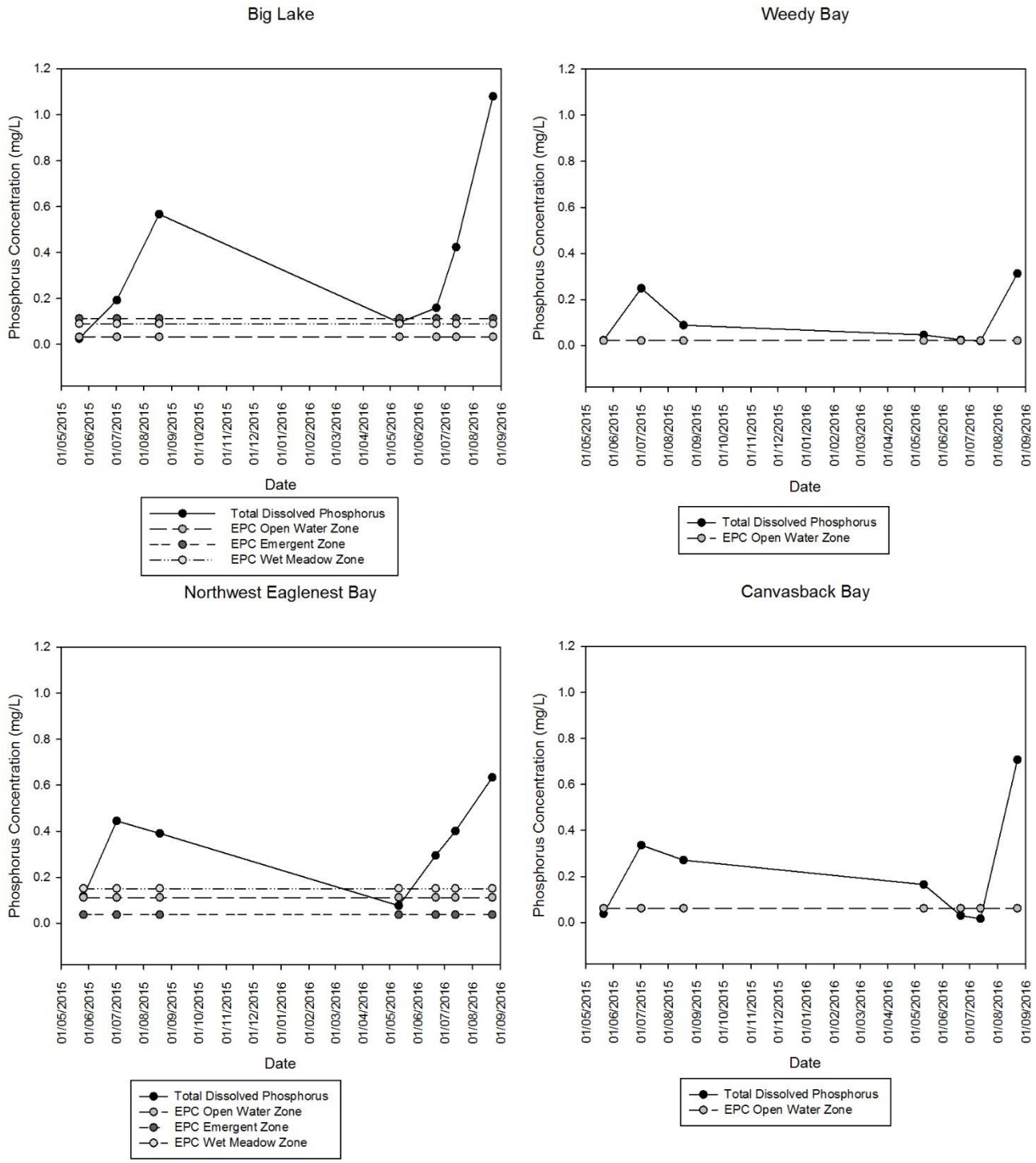
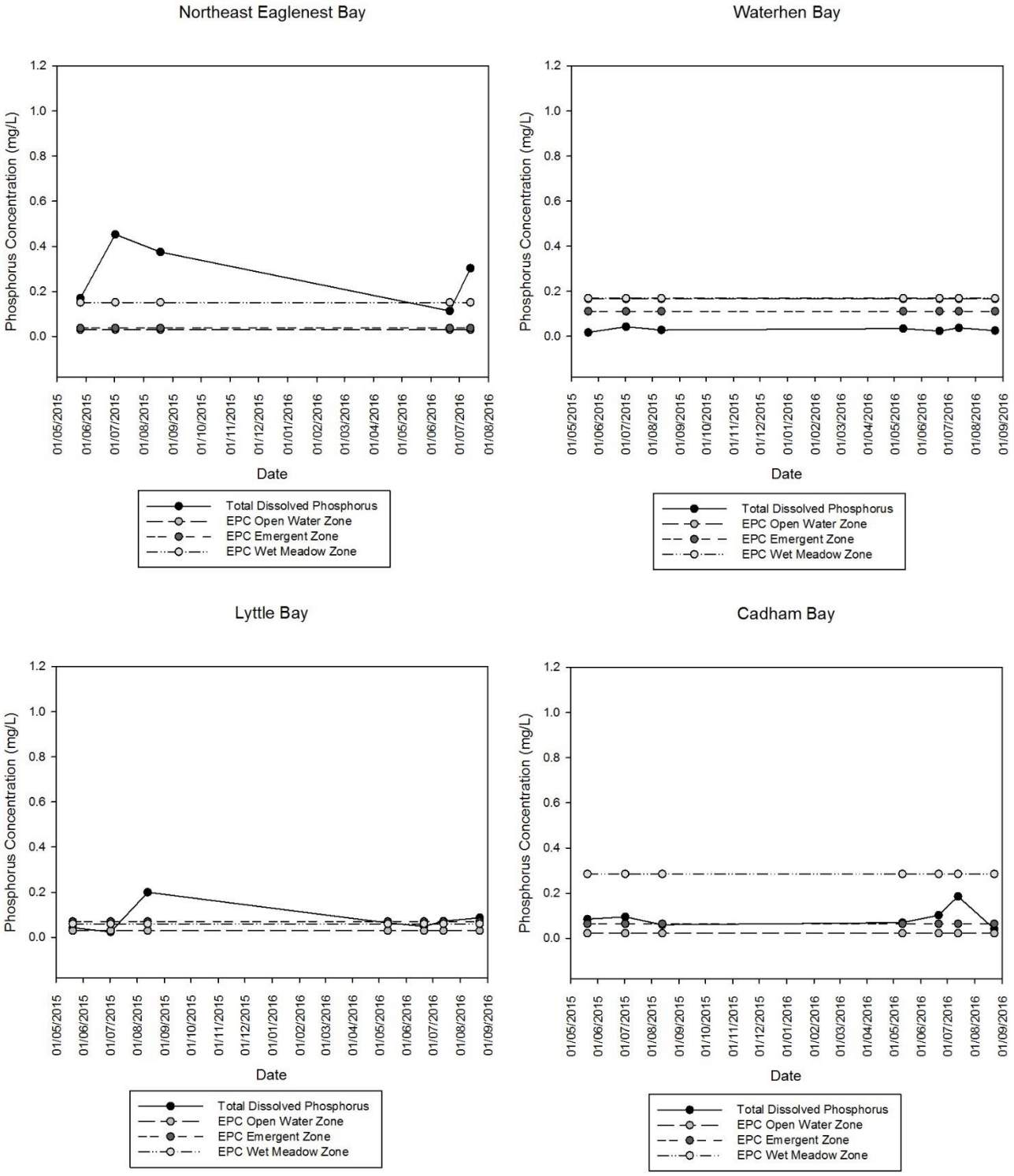


Figure 1.5b. Total dissolved P measured between 2015-2016 at selected sites at Delta Marsh (Figure 1.1) compared to calculated equilibrium P concentrations in the open-water, emergent vegetation, and wet-meadow zone sediments.



Comparing the EPC and TDP values indicated that sample sites in the emergent vegetation and wet meadow zones would also potentially act as a P sink for the majority of the sample period; however, these sites switched to possible P sources more frequently than the open-water sites. This again suggests that overall, the marsh sediments and soils will act to reduce P levels in the corresponding waters; however, this effect will be reduced in situations where there are higher water levels. For Big Lake the emergent vegetation zone was a potential source in May 2015 and 2016 (TDP=0.0255 mg L^{-1}, 0.095 mg L^{-1}, respectively, EPC= 0.113 mg L^{-1}) and the wet meadow zone in May 2015 (TDP=0.033 mg L^{-1}, EPC=0.090 mg L^{-1}; Figure 1.4). In Eaglenest the emergent zone was never a potential source for P; however, the wet meadow zone was relative to the Northwest site in May 2015 and 2016 (TDP=0.117 mg L^{-1}, 0.077 mg L^{-1}, respectively, EPC=0.151 mg L^{-1}) and the Northeast site in June 2016 (TDP=0.114 mg L^{-1}, EPC=0.151 mg L^{-1}). For Cadham Bay the emergent vegetation zone was a potential P source in August 2015 and 2016 (TDP=0.06 mg L^{-1}, 0.042 mg L^{-1}, respectively, EPC=0.065 mg L^{-1}). The Cadham Bay wet meadow zone was always a potential source for P (TDP=0.06-0.186 mg L^{-1}, EPC=0.285 mg L^{-1}) and had the second highest EPC of any site in the study. For Lyttle Bay the emergent vegetation zone was a potential P source in May 2015, May 2016, and June 2016 (TDP=0.043 mg L^{-1}, 0.064 mg L^{-1}, 0.049 mg L^{-1}, respectively, EPC=0.070 mg L^{-1}), and the wet meadow zone in May 2015 and June 2016 (TDP=0.043 mg L^{-1}, 0.049 mg L^{-1}, EPC=0.060 mg L^{-1}). Weedy Bay and Canvasback bay emergent and wet meadow zone sites were not sampled.

The one open-water sample site that behaved differently in comparison to the rest was Waterhen Bay. For all sample dates the TDP concentration was lower than the EPC indicating that Waterhen Bay would always act as a P source to the overlying water column (Figure 1.5).

Waterhen Bay had both the highest EPC (0.166 mg L^{-1}) and generally lowest TDP concentrations (TDP=0.017-0.043 mg L^{-1}) compared to other open-water sites. Similar to the open-water sites the emergent and wet meadow zone sites at Waterhen Bay were a potential P source for the duration of the sample period. The emergent vegetation and wet meadow zones had EPC's of 0.112 mg L^{-1} and 0.169 mg L^{-1}, respectively, which were higher than the TDP range of 0.017-0.043 mg L^{-1} in the bay.

The P equilibrium buffering capacity (PEBC) ranged from 151 L kg^{-1} (Waterhen Bay – wet meadow) to 789 L kg^{-1} (Weedy Bay – open-water) and averaged 411 L kg^{-1} across all sites and landscape positions (Table 1.7). Statistical contrasts identified significant differences between locations due to the landscape position ($p < 0.0001$; Table 1.10). There was not a significant difference due to the marsh unit or the interaction between the landscape position and marsh unit. Across landscape positions, mean PEBC were 511 L kg^{-1}, 422 L kg^{-1}, and 235 L kg^{-1} for the open-water, emergent, and wet meadow landscape positions, respectively (Table 1.8). Across marsh units, mean PEBC were 509 L kg^{-1}, 485 L kg^{-1}, 299 L kg^{-1}, and 469 L kg^{-1}, for the WM, CM, EM, and LF units, respectively (Table 1.8). In the WM unit PEBC were significantly different between the open-water landscape position (mean 661 L kg^{-1}) and both the emergent landscape position (333 L kg^{-1}; $p = 0.0405$), and the wet meadow landscape position (228 L kg^{-1}; Table 1.8; $p = 0.0015$; Table 1.11). In the CM unit PEBC were significantly different between the wet meadow landscape position (221 L kg^{-1}) and both the open-water landscape position (mean 492 L kg^{-1}; $p = 0.0041$), and the emergent landscape position (733 L kg^{-1}; Table 1.8; $p = 0.0024$; Table 1.11). In the EM unit PEBC were significantly different between the open water landscape position (mean 387 L kg^{-1}) and wet meadow landscape position (mean 218 L kg^{-1}; Table 1.8; p =

0.0068; Table 1.11). In the LF unit PEBC were significantly different between the open water landscape position (mean 491 L kg^{-1}) and wet meadow landscape position (308 L kg^{-1}; Table 1.8; p = 0.0475; Table 1.11). For the open-water landscape positions across the marsh PEBC were significantly different between the WM unit (mean 661 L kg^{-1}) and EM unit (mean 387 L kg^{-1}; Table 1.8; p = 0.0451; Table 1.11). For the emergent and wet meadow landscape positions across the marsh PEBC were not significantly different between marsh units.

When considering all landscape positions together there were no significant correlations between PEBC and any of the physiochemical properties however when only considering certain landscape positions significant correlations were detected (Table 1.9). For the open-water landscape position sites significant correlations were detected between PEBC and Mehlich-3 Fe (r= 0.66; p = 0.0392). For the emergent landscape position sites significant correlations were detected between PEBC and Mehlich-3 P (r= 0.75; p = 0.0124), Organic P (r= 0.83; p = 0.0026), TN (r= 0.74; p = 0.0140), Mehlich-3 Al (r= 0.65; p = 0.0430), Mehlich-3 Mg (r= 0.83; p = 0.0032), and OM (r= 0.68; p = 0.0300). For the wet meadow landscape position sites there were no significant correlations detected between PEBC and any of the physiochemical properties.

The DPS ranged from 2.1% (Eaglenest 2 – open water) to 17.1% (Eaglenest – wet meadow) and averaged 6.1% across all sites and landscape positions (Table 1.7). Statistical contrasts identified significant differences between locations due to the landscape position ($p < 0.0001$), and for the interaction between marsh unit and landscape position (p = 0.0317; Table 1.10). There was not a significant difference due to the marsh unit. Across landscape positions, mean DPS were 4.5%, 5.4%, and 9.6% for the open-water, emergent, and wet meadow

landscape positions, respectively (Table 1.8). Across marsh units, mean DPS were 7.0%, 7.1%, 5.1%, and 6.3%, for the WM, CM, EM, and LF units, respectively (Table 1.8). In the WM and LF units DPS were not significantly different between landscape positions. In the CM unit DPS were significantly different between the open-water landscape position (mean 3.7%) and wet meadow landscape position (17.1%; Table 1.8; $p < 0.0001$; Table 1.11). In the EM unit DPS were significantly different between the open-water landscape position (mean 3.4%) and wet meadow landscape position (mean 7.4%; Table 1.8; $p = 0.0083$; Table 1.11). For the open-water and emergent landscape positions across the marsh DPS were not significantly different between marsh units. For the wet meadow landscape positions across the marsh DPS were significantly different between the CM unit (17.1%) and EM unit (mean 3.4%; Table 1.8; $p = 0.0079$; Table 1.11).

When considering all landscape positions together there were no significant correlations between DPS and any of the physiochemical properties; however, when only considering certain landscape positions significant correlations were detected (Table 1.9). For the open-water landscape position sites significant correlations were detected between DPS and Mehlich-3 Fe ($r= 0.68$; $p = 0.0291$). For the emergent landscape position sites there were no significant correlations detected between DPS and any of the physiochemical properties. For the wet meadow landscape position sites, significant correlations were detected between DPS and Mehlich-3 Ca ($r= 0.82$; $p = 0.0039$), and negatively with Mehlich-3 Fe ($r= -0.75$; $p = 0.0119$).

3.2: Sediment Chronology and Nutrient Accumulation

3.2.1: Sediment Chronologic Profiles

Overall, the core samples indicated a variable but distinguishable amount of disturbance due to irregular ^{210}Pb and ^{137}Cs activity profiles and disagreement between dates determined via ^{210}Pb and ^{137}Cs for some sites. For most cores the 1963 and 1954 dates, determined by the peak ^{137}Cs activity and zero ^{137}Cs activity depths, respectively, are not in agreement with the dates determined by ^{210}Pb (Table 2.1) indicating that the ^{210}Pb chronology cannot be perfectly verified using ^{137}Cs for those cores. For Eaglenest 1, Eaglenest 2, Cadham Bay and Waterhen Bay, the dates determined by ^{210}Pb and ^{137}Cs are within a few cm of the core profile, giving confidence to the assigned chronology. The disagreement in sample layer dates for some cores was expected as the sediment cores indicate a high amount of disturbance and mixing between layers which can make determining dates difficult. The agreement between the ^{210}Pb and ^{137}Cs 1954 dates was worse than that for the 1963 dates for all sites potentially indicating that sediment chronology of these samples became less reliable with increasing depth.

Figure 2.1: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Big Lake, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

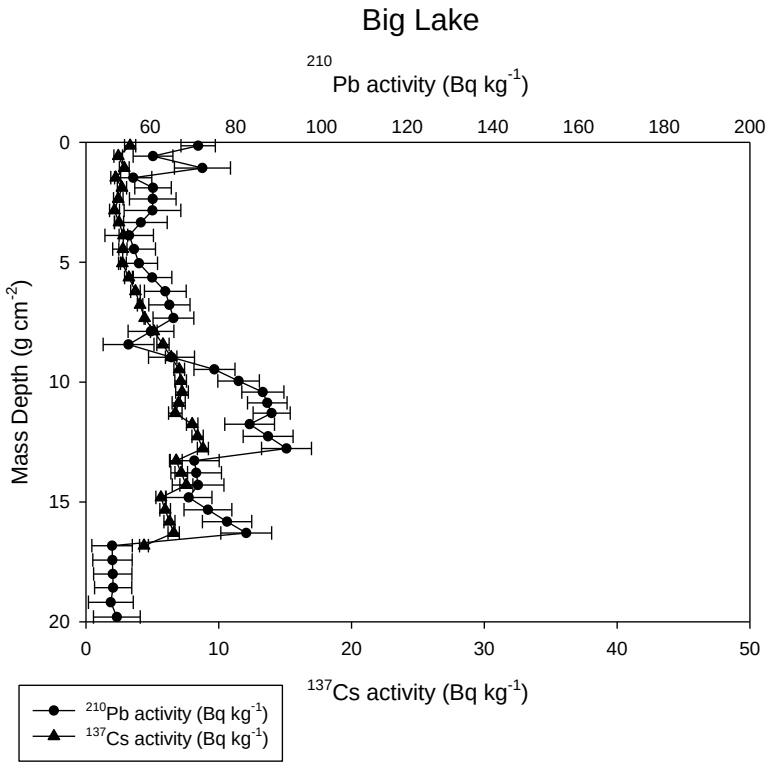


Figure 2.2: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Canvasback Bay, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

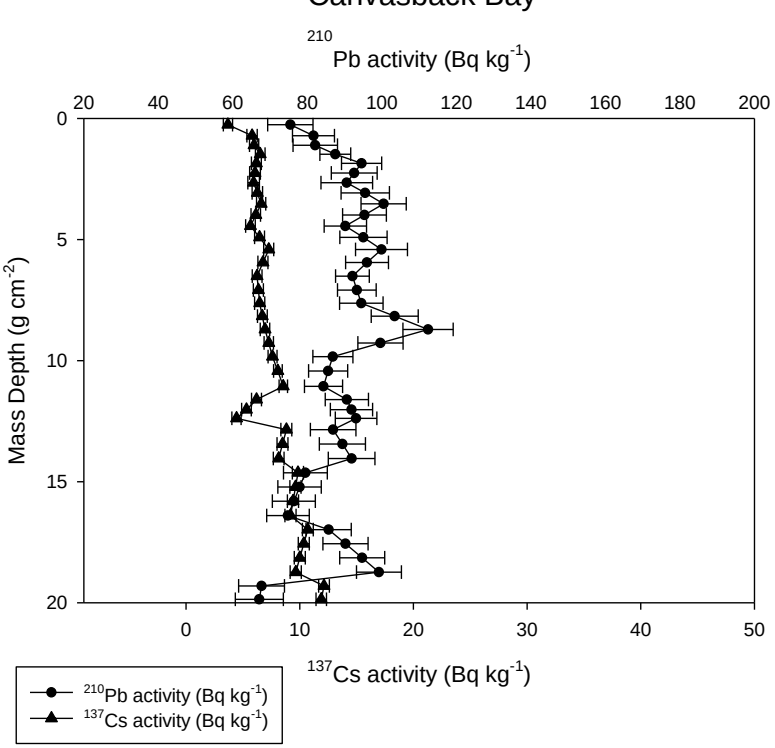


Figure 2.3: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Weedy Bay, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

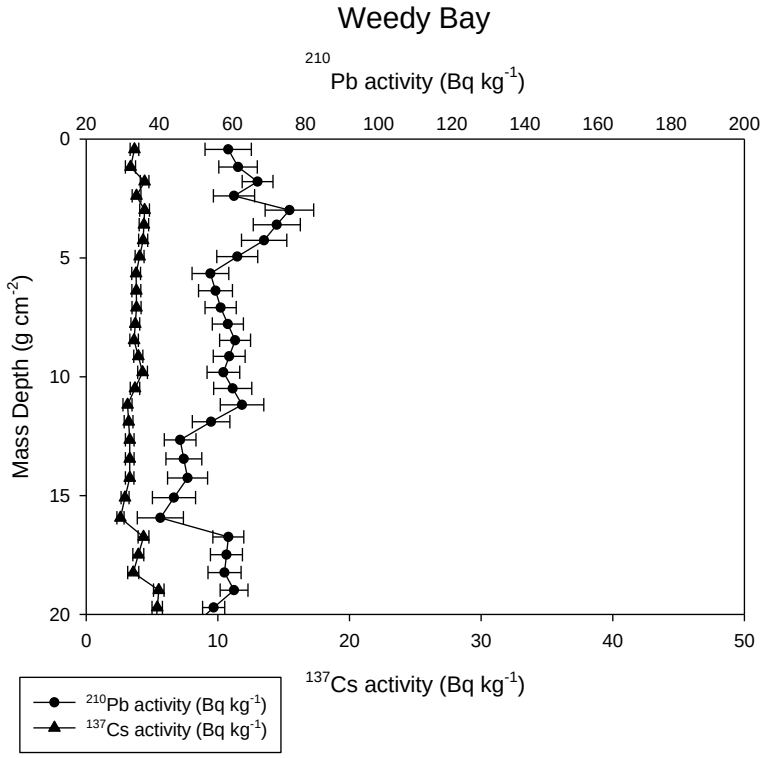


Figure 2.4: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Northeast Eaglenest, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

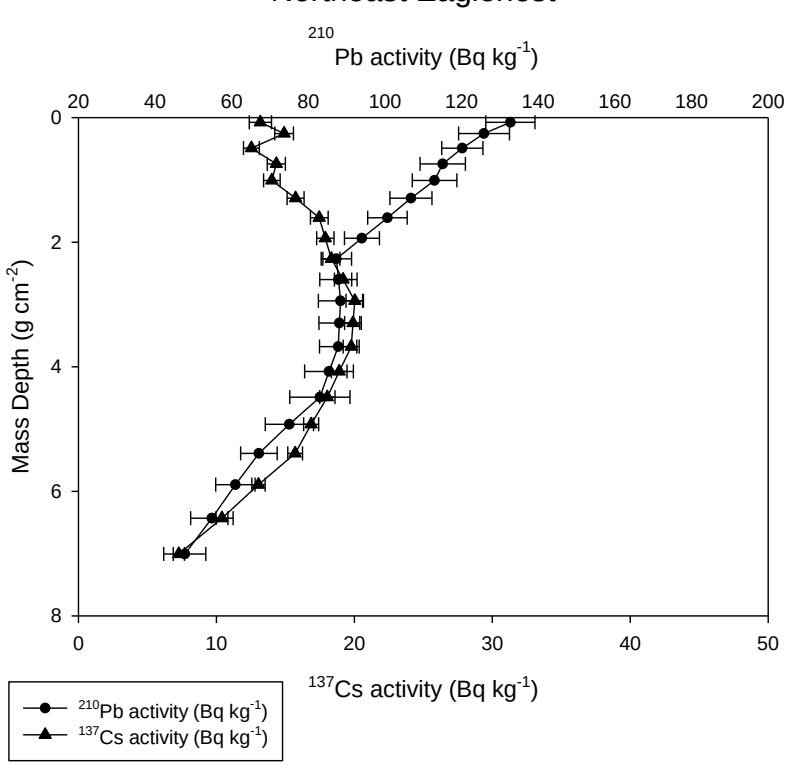


Figure 2.5: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Northwest Eaglenest, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

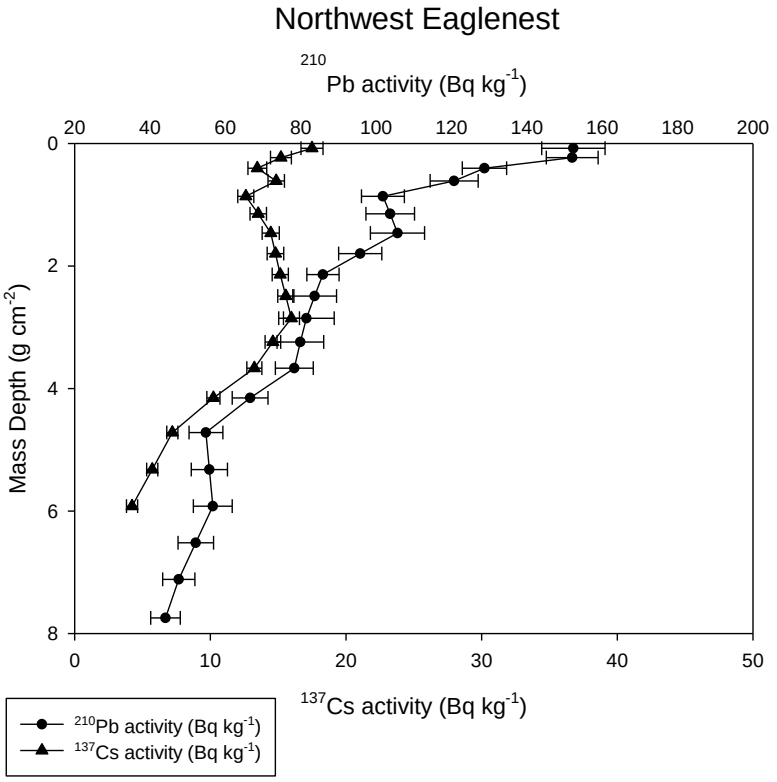


Figure 2.6: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Cadham Bay, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

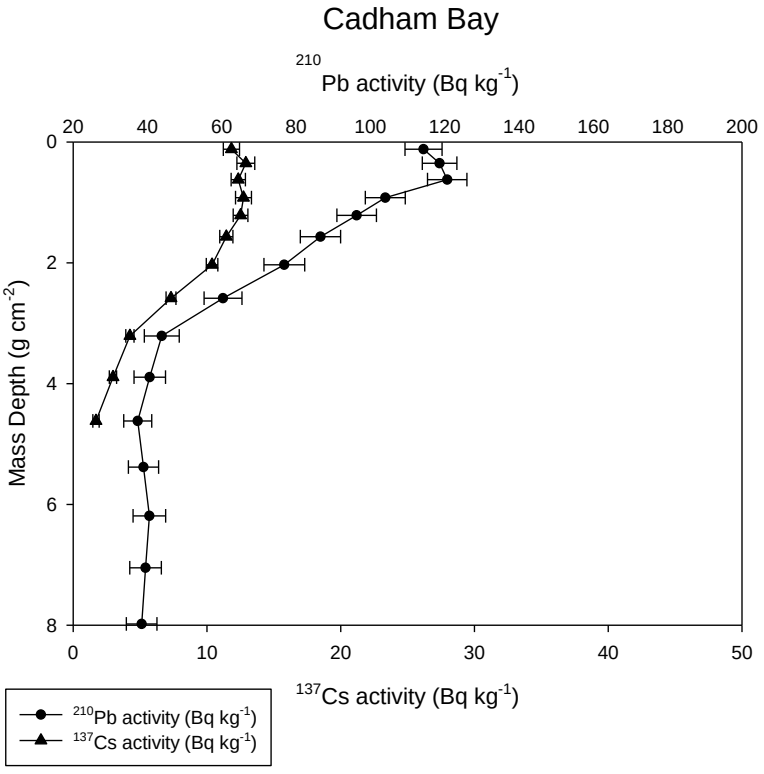


Figure 2.7: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Lyttle Bay, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

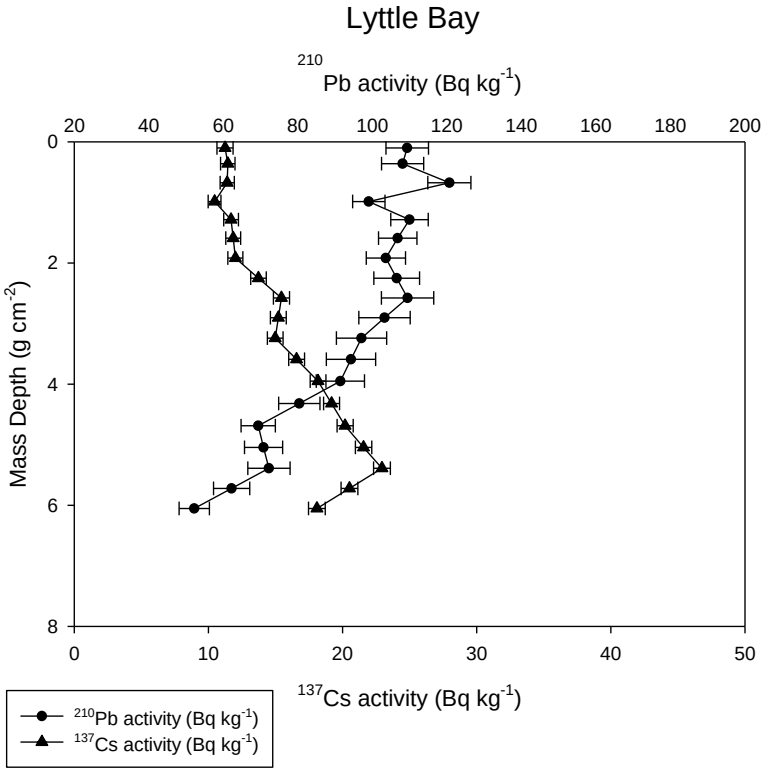


Figure 2.8: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Waterhen Bay, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

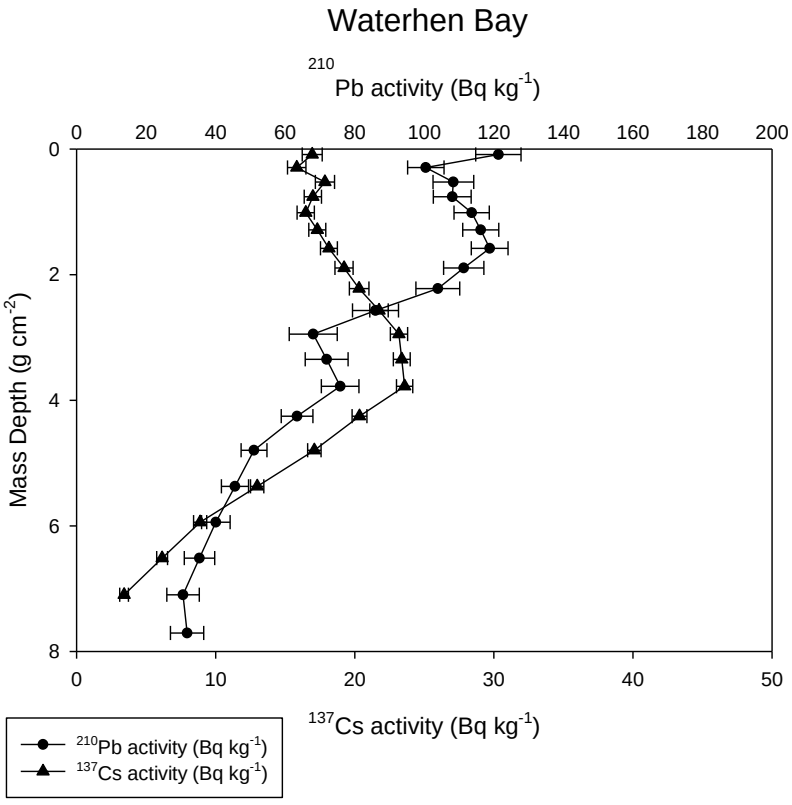


Figure 2.9: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Lake Francis 1, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

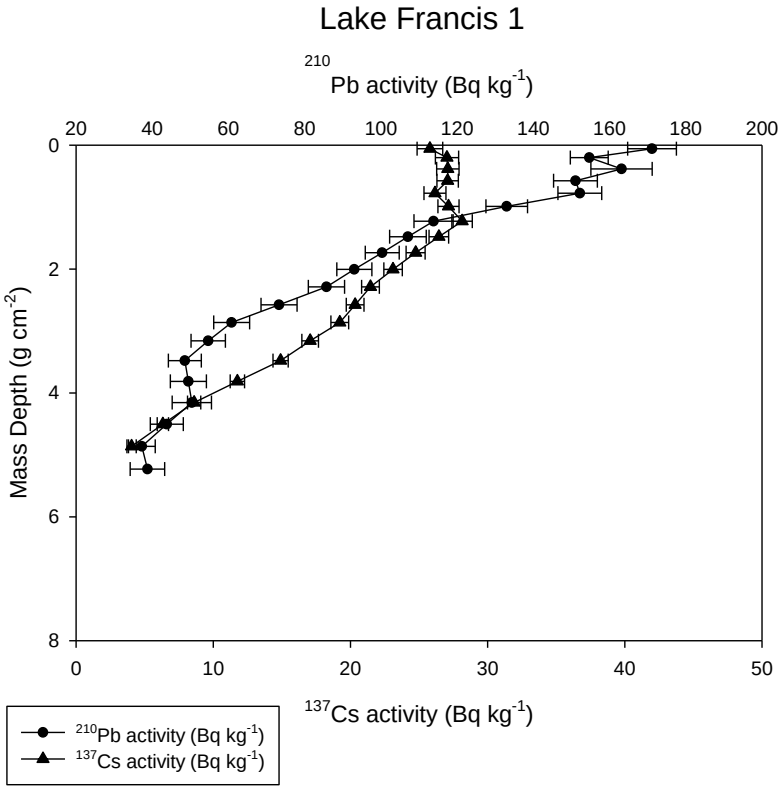


Figure 2.10: ^{210}Pb and ^{137}Cs activity vs mass depth a for 50-cm sediment core sample obtained from Lake Francis 2, Delta Marsh (Figure 1.1), Canada, between August and October 2015. (±S.D.)

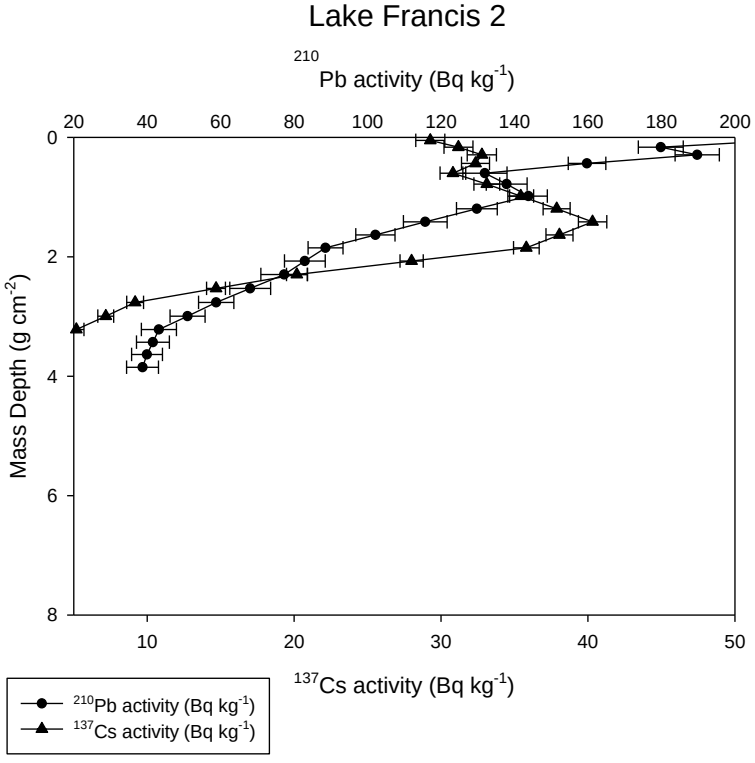


Table 2.1: Dates determined for sample segments from sediment cores collected at Delta Marsh and Lake Francis (Figure 1.1) between August and October 2015, determined based on ^{210}Pb and ^{137}Cs activity.

Depth (cm)	Big Lake	Eaglenest 1	Eaglenest 2	Cadham	Lyttle	Waterhen	Lake Francis 1	Lake Francis 2
0-1	2016	2015	2015	2016	2015	2015	2015	2015
1-2	2014	2013	2013	2013	2014	2013	2013	2013
2-3	2011	2011	2010	2010	2011	2011	2010	2010
3-4	2008	2008	2008	2004	2008	2009	2007	2007
4-5	2006	2005	2004	2000	2005	2007	2003	2003
5-6	2004	2002	2001	1994	2002	2003	1999	2000
6-7	2001	1998	1997	1987	1998	1999	1994	1995
7-8	1997	1994	1991	1978	1994	1994	1988	1989
8-9	1994	1989	1986	1968	1989	1988	1982	1982
9-10	1991	1985	1980	1962	1983	1981	1975	1974
10-11	1987	1979	1974	1956	1976	1974	1967	1966
11-12	1982	1973	1965	1951	1967	1968	1956	1957
12-13	1974	1964	1954	1943	1956	1959	1943	1945
13-14	1961	1951	1935	1929	1939	1942	1928	1928
14-15	1940	1928	1902	1906	1912	1914	1904	1902
15-16	1930	1923	1904	1834	1912	1905	1906	1892
16-17	1916	1917	1906		1912	1893	1907	1878
17-18		1910	1892		1902	1877	1900	1873
18-19		1900	1867		1887	1842	1890	1868
19-20		1878	1781		1874	1857	1894	1869

*Dates could not be estimated for Weedy or Canvasback Bays
*Black box indicates ^{137}Cs peak date of 1963
*Highlighted box represents bottom of ^{137}Cs activity, approximately 1954
*^{137}Cs dates could not be determined for Big Lake

The cores collected from Canvasback Bay and Weedy Bay were the poorest in quality and these cores were not suitable for dating due to the high degree of disturbance suggested by the profiles. The ^{210}Pb and ^{137}Cs activities for these sites oscillated considerably but did not demonstrate any increasing or decreasing trends with depth and did not approximate that of an

expected profile. For all other sites the highest levels of ^{210}Pb were detected at the surface, as would be expected; however, for Canvasback and Weedy the highest values recorded were at 19-cm and 30-cm, respectively. Additionally, the ^{137}Cs activity was present to the bottom of the cores, with no clear peak.

The Big Lake site demonstrated the highest degree of disturbance of the sites where chronology was able to be estimated; however, due to the quality of this profile these dates have a very low degree of confidence. The ^{210}Pb activity in the Big Lake profile increased with depth in some sections instead of decreasing steadily for the length of the core and the determination of background unsupported ^{210}Pb carried the least confidence of the cores dated. The ^{137}Cs activity for Big Lake posed two problems; the peak activity that should represent 1963 did not agree with the ^{210}Pb chronology determination (^{210}Pb 1963: 13cm, ^{137}Cs 1963: 24cm) and was instead below the oldest date determined by ^{210}Pb (1916: 17cm) while the ^{137}Cs activity was also present to the bottom of the core samples analyzed. Overall, the sites in the west marsh appeared to be the most disturbed of the sites sampled in Delta Marsh and Lake Francis.

The ^{210}Pb and ^{137}Cs activities for Eaglenest 1, Eaglenest 2, Cadham Bay, Lyttle Bay, Waterhen Bay, Lake Francis 1 and Lake Francis 2 all displayed appropriate-but-disturbed profiles. The ^{210}Pb activity was the highest in the surface sediments and decreased with depth while the ^{137}Cs activity demonstrated an identifiable peak. The decrease of ^{210}Pb activity was not consistent with ideal core profiles indicating possible disturbance or changing sedimentation rates. For all cores with identifiable ^{137}Cs peaks they were represented across multiple sample segments as a large rounded peak instead of the ideal sharp peak indicating

vertical mixing between layers (Kuzyk et al. 2004). This mixing reduces the confidence of sediment chronology but cores can still be utilized so long as this limitation is understood (Johannessen and Macdonald 2012).

The best ^{210}Pb and ^{137}Cs profiles were obtained from sites within center marsh (Eaglenest 1 and 2) and Lake Francis (Lake Francis 1 and 2) and suggested the least amount of disturbance compared to the other sites. This was indicated by ^{210}Pb profiles that consistently decreased with depth, demonstrating the least mixing in the upper layers, and not exhibiting any considerable jumps in activity levels throughout the profile. The ^{137}Cs profiles similarly demonstrated values increasing to a clear maximum and then decreasing until a zero value as depth increased. Although considerably broader than in an ideal profile these sites still demonstrate a clear subsurface ^{137}Cs peak. The center marsh sites had the best agreement between 1963 dates determined by ^{210}Pb and ^{137}Cs of all sites sampled (Table 2.1) and although the 1954 dates are not in agreement, they are the closest of all sites. For the Lake Francis sites the 1963 dates determined by ^{210}Pb and ^{137}Cs are not in agreement with the ^{210}Pb dates being much deeper than the ^{137}Cs dates (Table 2.1). The 1954 dates determined by ^{210}Pb and ^{137}Cs are similarly not in agreement but with the ^{210}Pb 1954 date deeper than the ^{137}Cs 1954 date. For both center marsh and Lake Francis the ^{210}Pb and ^{137}Cs chronology were very similar between the two sites within each marsh unit which adds confidence to these results.

In the east marsh unit, the site with the best ^{210}Pb and ^{137}Cs profiles was Lyttle Bay, followed by Waterhen Bay and Cadham Bay, although these profiles were all more irregular compared to those from center marsh or Lake Francis. All east marsh sites demonstrate a surface mixed layer with ^{210}Pb and ^{137}Cs activities fluctuating indicating vertical mixing which

may help explain the irregular nature of the lower profiles if the sources of intermixing have been present at these sites for some time (Johannessen and Macdonald 2012). Waterhen Bay had the best agreement between ^{210}Pb and ^{137}Cs 1963 dates of the east marsh sites while Lyttle Bay had the clearest ^{137}Cs peak. Cadham Bay demonstrated ^{210}Pb and ^{137}Cs profiles that were considerably compressed into the immediate surface sediments. The ^{137}Cs peak, if Cadham Bay can be considered to have one, is immediately at the surface of the core which is unique for the sites sampled at Delta Marsh. The bottom of the ^{137}Cs activity (^{137}Cs =0) within the core sample is also the shallowest of any site sampled.

For all sites except Canvasback Bay and Weedy Bay the $^{210}Pb_{ex}$ and ^{137}Cs activities reached zero within the core and as such full inventories were able to be calculated (Table 2.2). For Canvasback Bay and Weedy Bay both the $^{210}Pb_{ex}$ and ^{137}Cs activities extended beyond the 50-cm core depth so inventories were calculated by summing the entire core. These inventories would not represent the true site inventory (likely providing an under-estimate as activity did not reach zero) but would still provide a basis for comparison with the other site. As the ^{210}Pb did not reach a steady-state that would indicate level of supported ^{210}Pb the activity from Big Lake was used to approximate as this was the nearest site and would be expected to approximate similar background conditions.

Table 2.2: $^{210}Pb_{ex}$ and ^{137}Cs inventories for core samples collected at Delta Marsh (Figure 1.1) and depth of the corresponding radionuclei activity. The $^{210}Pb_{ex}$ inventory sums all unsupported $^{210}Pb_{ex}$ while the ^{137}Cs inventory sums all ^{137}Cs present in the core sample. Averages and standard deviation including and excluding Weedy Bay and Canvasback Bay

Site	**$^{210}Pb_{ex}$**		**Total ^{137}Cs**	
	Depth (cm)	Bq m^{-2}	Depth (cm)	Bq m^{-2}
Big Lake	38	3851	34	859
Weedy Bay*	50	5028	50	1373
Canvasback Bay*	50	11790	50	1504
Eaglenest 1	19	3076	24	1140
Eaglenest 2	17	2448	17	725
Cadham Bay	9	1809	11	376
Lyttle Bay	21	3647	21	1043
Waterhen Bay	19	3265	19	1180
Lake Francis 1	21	3445	19	945
Lake Francis 2	21	2760	19	891
Average including Weedy/Canvasback:		4112 ±2833		996 ±675
Average excluding Weedy/Canvasback:		3037 ±330		885 ±262

*Weedy Bay and Canvasback Bay inventories do not represent the full site inventory and were approximated using the background ^{210}Pb activity from Big Lake

The estimated inventories for Weedy Bay and Canvasback Bay were considerably higher than those found in any of the other sites, particularly for the $^{210}Pb_{ex}$. The highest $^{210}Pb_{ex}$ inventory, Canvasback Bay, was nearly an order of magnitude higher than the lowest, Cadham Bay. The average inventories across the marsh for $^{210}Pb_{ex}$ and ^{137}Cs when including all sites were 4112 ±2833 and 996 ±675, respectively. When excluding Weedy Bay and Canvasback Bay, as these represent highly disturbed sites, the average inventories across the marsh for $^{210}Pb_{ex}$ and ^{137}Cs were 3037 ±330 and 885 ±262, respectively. The average values excluding Canvasback Bay and Weedy Bay were better suited to be used for comparisons, as the inventories of those sites do not represent natural conditions.

For both the $^{210}Pb_{ex}$ and ^{137}Cs inventories, the lowest values were for Cadham Bay which was considerably lower than the marsh average. Eaglenest 2 and Lake Francis 2 ($^{210}Pb_{ex}$ only) were also below average but to a much lesser degree. Big Lake ($^{210}Pb_{ex}$), Eaglenest 1 (^{137}Cs), Lyttle Bay ($^{210}Pb_{ex}$ and ^{137}Cs), Waterhen Bay ($^{210}Pb_{ex}$ and ^{137}Cs) and Lake Francis 1 ($^{210}Pb_{ex}$ and ^{137}Cs) were all above average although to varying degrees. The sites with inventory values the closest to the averages were those in center marsh and Lake Francis. The depths of each inventory varied between sites with Cadham Bay being the most shallow and Big Lake the deepest (with a bottom) by a considerable amount.

3.2.2: Physiochemical Accumulation Rates

Overall for all sample sites, the sediment accumulation rate (SAR), mass accumulation rate (MAR) and accumulation rates for TP (TPAR), OP (OPAR), BAP (BAPAR), TN (TNAR), and OM (OMAR) increase in more recent sediments for all sites sampled (Figures 2.11-2.26). The accumulation rates were not determined for Canvasback Bay and Weedy Bay due to the inability to determine chronology for these sites and the highly irregular nature of those cores. Generally, the accumulation rates increased exponentially from the oldest and deepest segments up to the present day and demonstrated the most considerable increases after approximately 1960 and again after approximately 1995.

Figure 2.11: Sediment accumulation rate and mass accumulation rate by year for sediment core sample obtained from Big Lake, Delta Marsh (Figure 1.1), Canada in August 2015. (±S.D.)

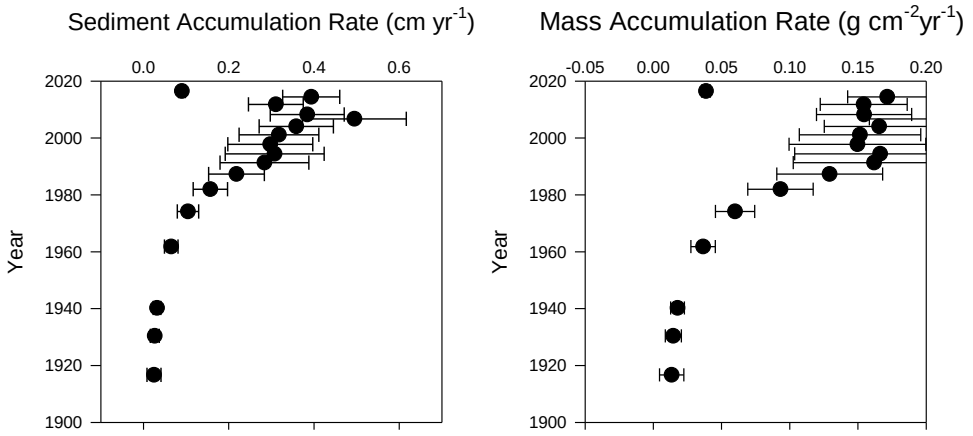


Figure 2.12: Sediment accumulation rate and mass accumulation rate by year for sediment core sample obtained from Northeast Eaglenest, Delta Marsh (Figure 1.1), Canada in August 2015. (±S.D.)

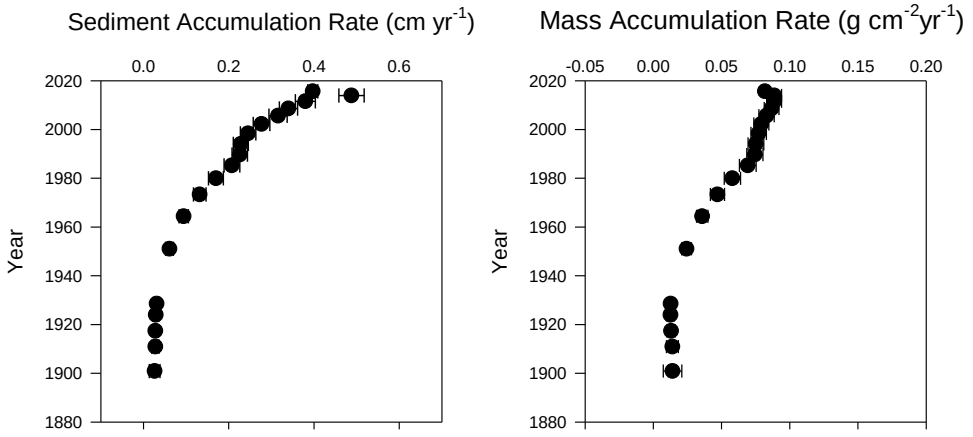


Figure 2.13: Sediment accumulation rate and mass accumulation rate by year for sediment core sample obtained from Northwest Eaglenest, Delta Marsh (Figure 1.1), Canada in September 2015. (±S.D.)

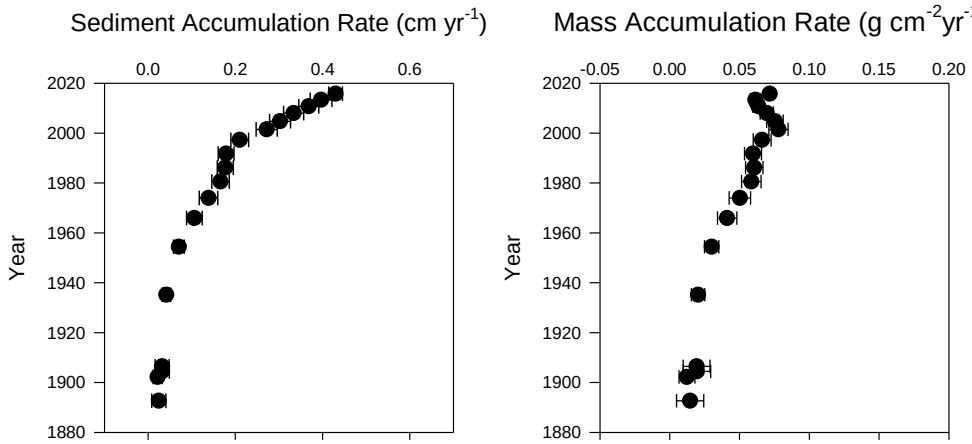


Figure 2.14: Sediment accumulation rate and mass accumulation rate by year for sediment core sample obtained from Cadham Bay, Delta Marsh (Figure 1.1), Canada in August 2015. (±S.D.)

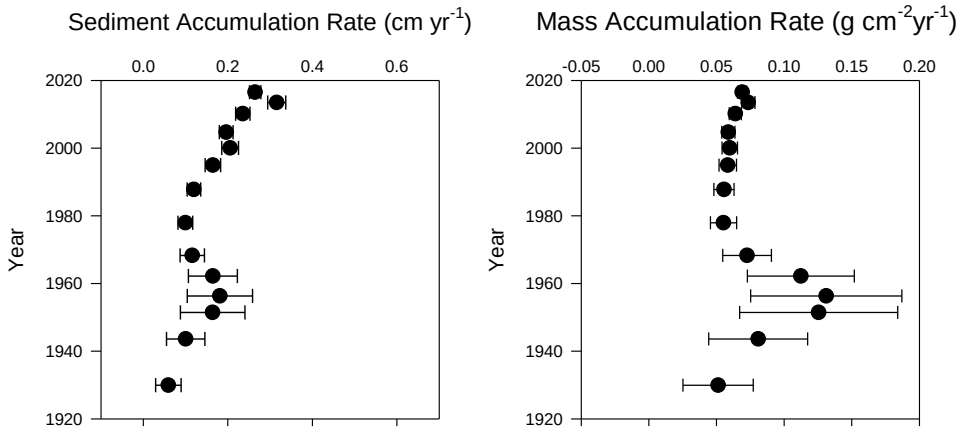


Figure 2.15: Sediment accumulation rate and mass accumulation rate by year for sediment core sample obtained from Lyttle Bay, Delta Marsh (Figure 1.1), Canada in August 2015. (±S.D.)

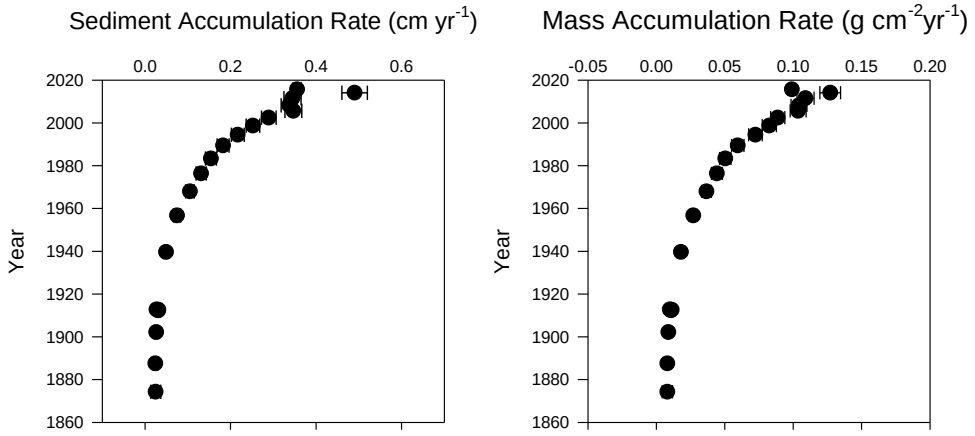


Figure 2.16: Sediment accumulation rate and mass accumulation rate by year for sediment core sample obtained from Waterhen Bay, Delta Marsh (Figure 1.1), Canada in August 2015. (±S.D.)

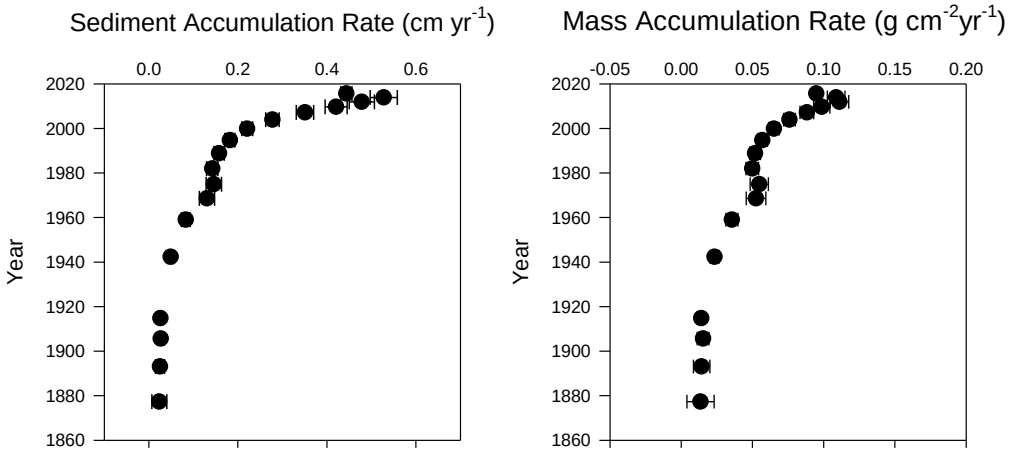


Figure 2.17: Sediment accumulation rate and mass accumulation rate by year for sediment core sample obtained from Lake Francis 1, Delta Marsh (Figure 1.1), Canada in August 2015. (±S.D.)

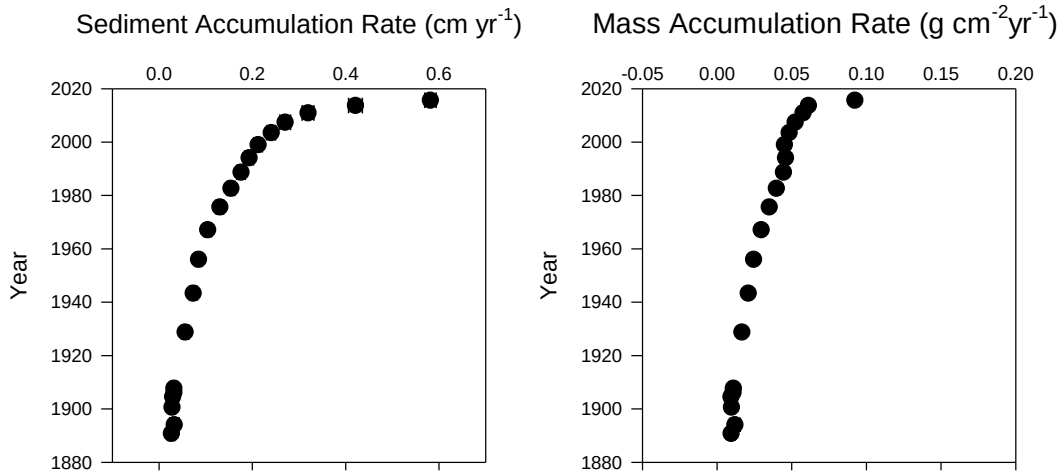


Figure 2.18: Sediment accumulation rate and mass accumulation rate by year for sediment core sample obtained from Lake Francis 2, Delta Marsh (Figure 1.1), Canada in September 2015. (±S.D.)

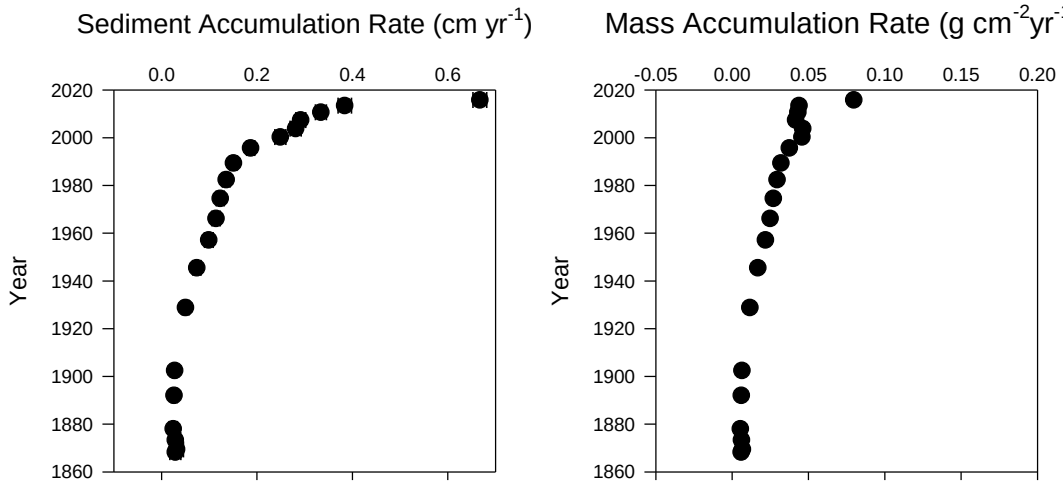
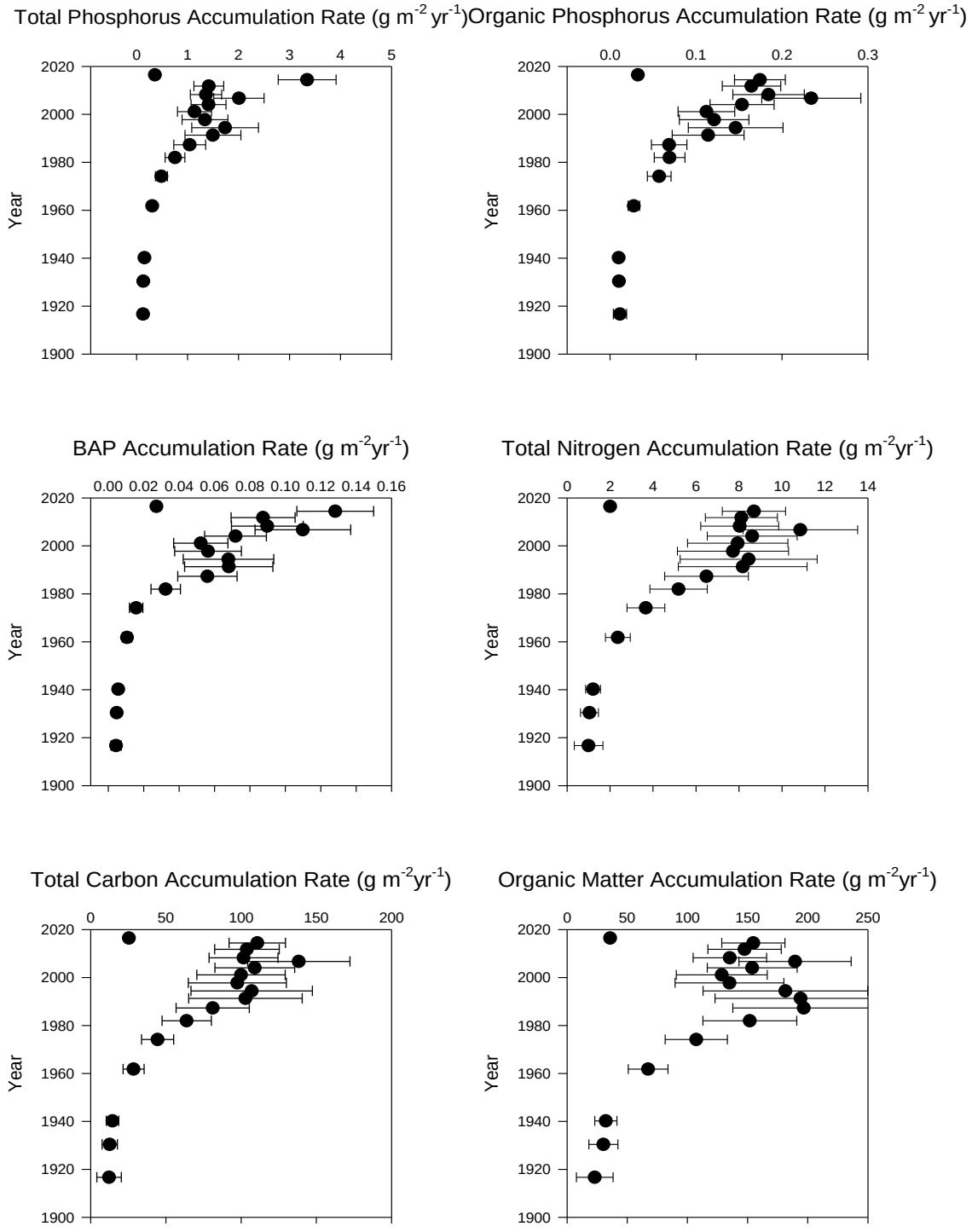


Figure 2.19: Accumulation rates for total P, organic P, biologically available P, total nitrogen, total carbon, and organic matter by year for sediment core sample obtained from Big Lake, Delta Marsh (Figure 1.1), Canada, in August 2015. (±S.D.)



*Axis for accumulation rates in Big Lake are different from the rest of the marsh as levels were considerably higher

Figure 2.20: Accumulation rates for total P, organic P, biologically available P, total nitrogen, total carbon, and organic matter by year for sediment core sample obtained from Northeast Eaglenest, Delta Marsh (Figure 1.1), Canada, in August 2015. (±S.D.)

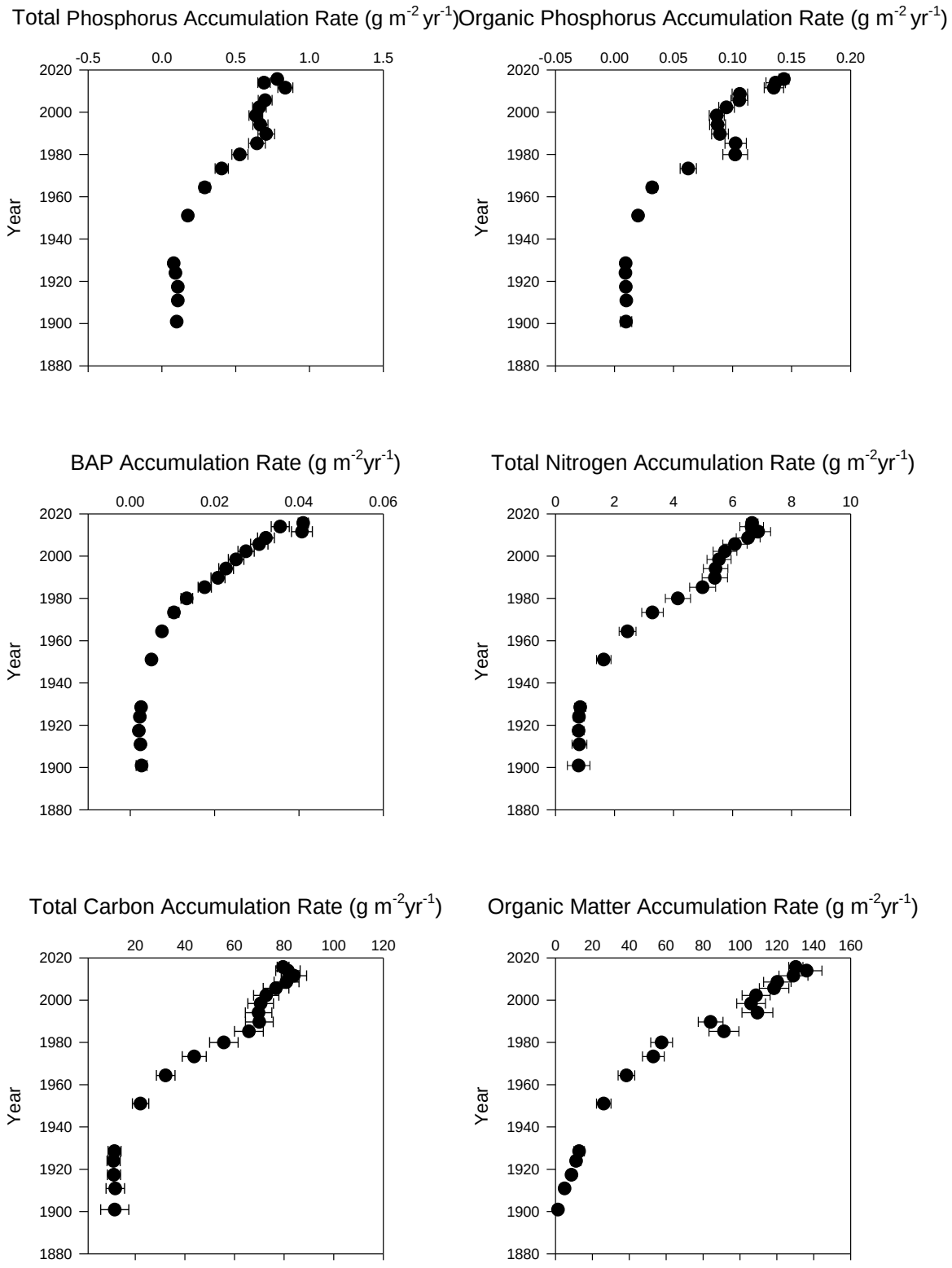


Figure 2.21: Accumulation rates for total P, organic P, biologically available P, total nitrogen, total carbon, and organic matter by year for sediment core sample obtained from Northwest Eaglenest, Delta Marsh (Figure 1.1), Canada, in September 2015. (±S.D.)

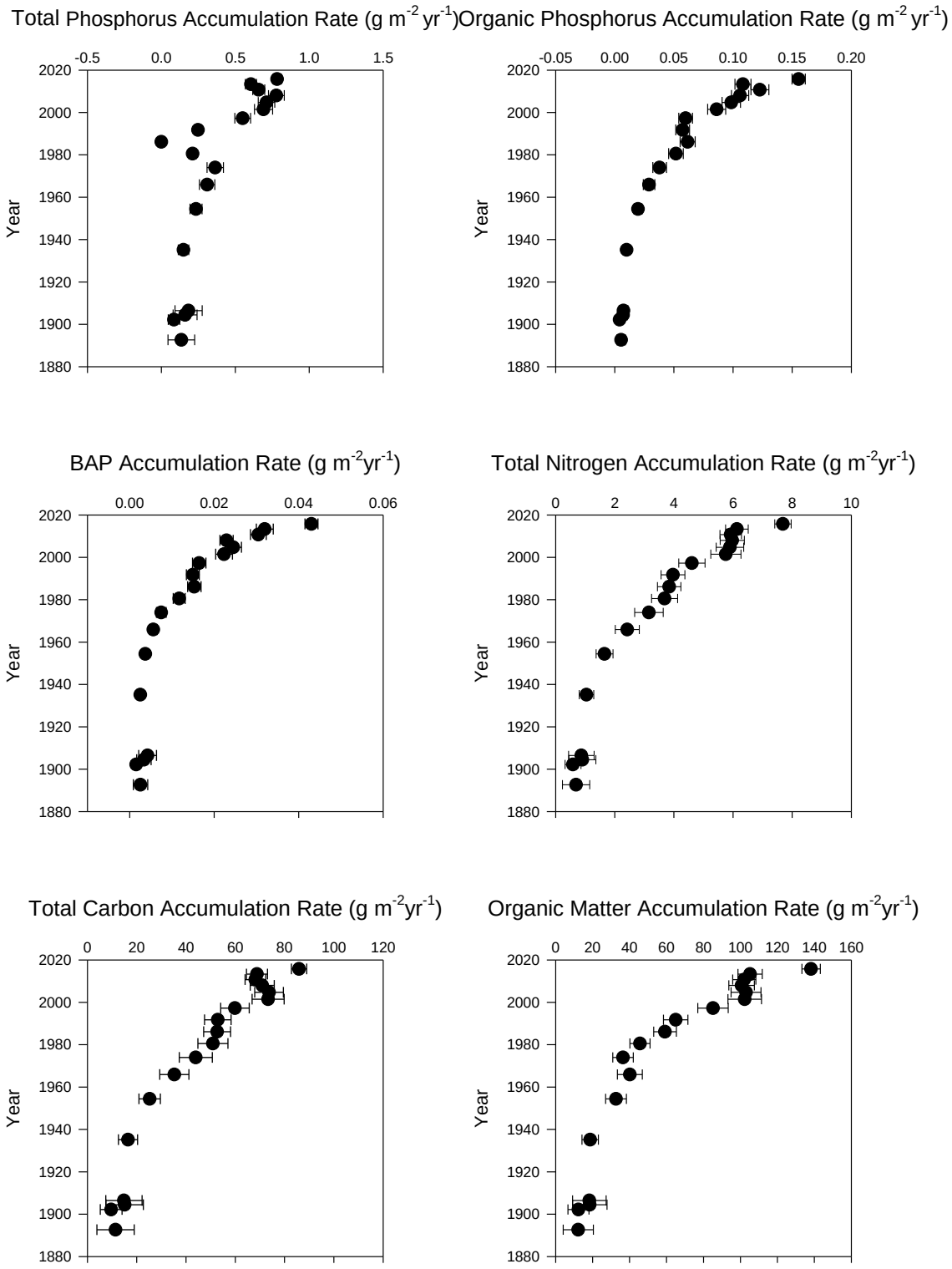


Figure 2.22: Accumulation rates for total P, organic P, biologically available P, total nitrogen, total carbon, and organic matter by year for sediment core sample obtained from Cadham Bay, Delta Marsh (Figure 1.1), Canada, in August 2015. (±S.D.)

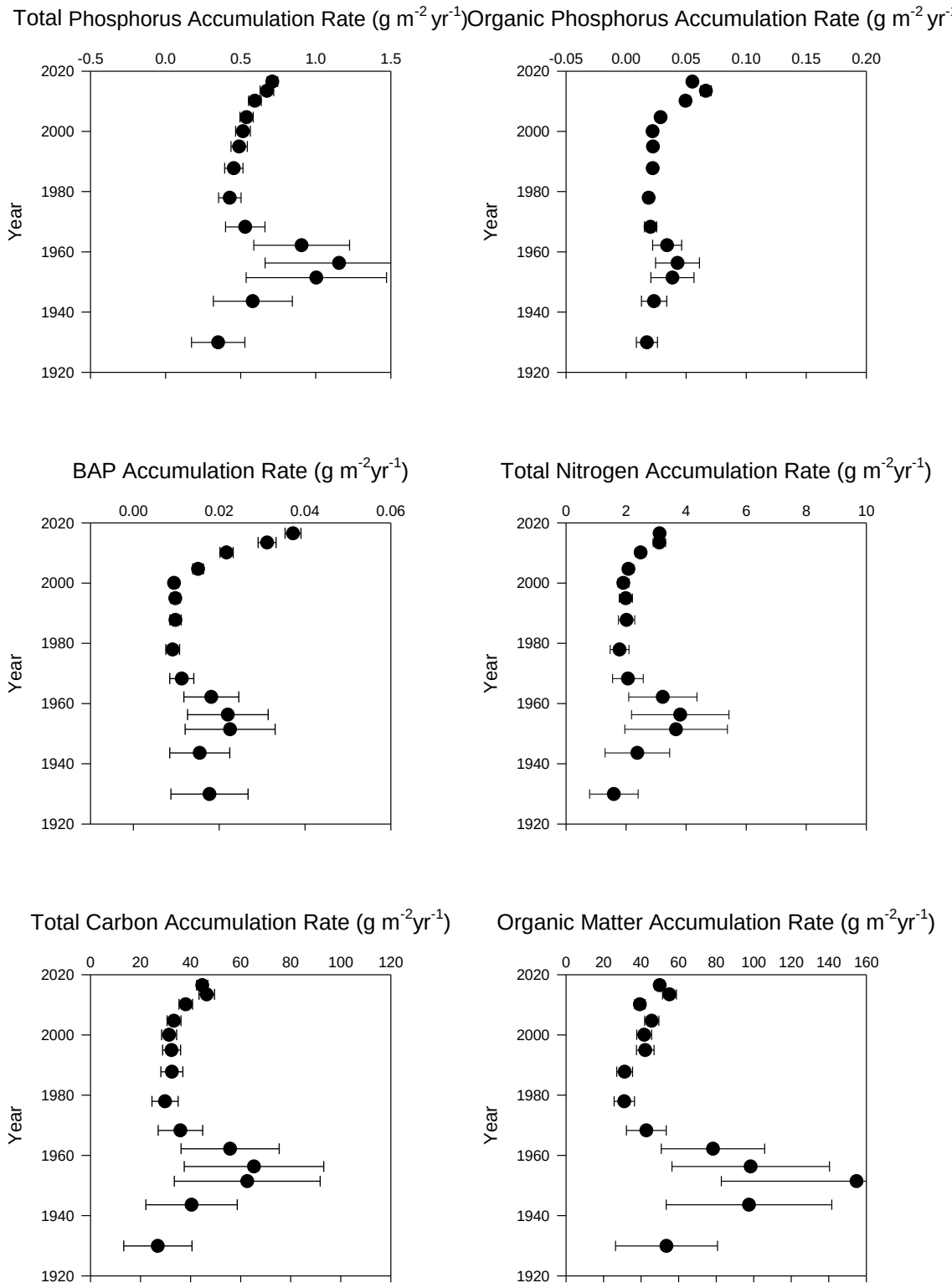


Figure 2.23: Accumulation rates for total P, organic P, biologically available P, total nitrogen, total carbon, and organic matter by year for sediment core sample obtained from Lyttle Bay, Delta Marsh (Figure 1.1), Canada, in August 2015. (±S.D.)

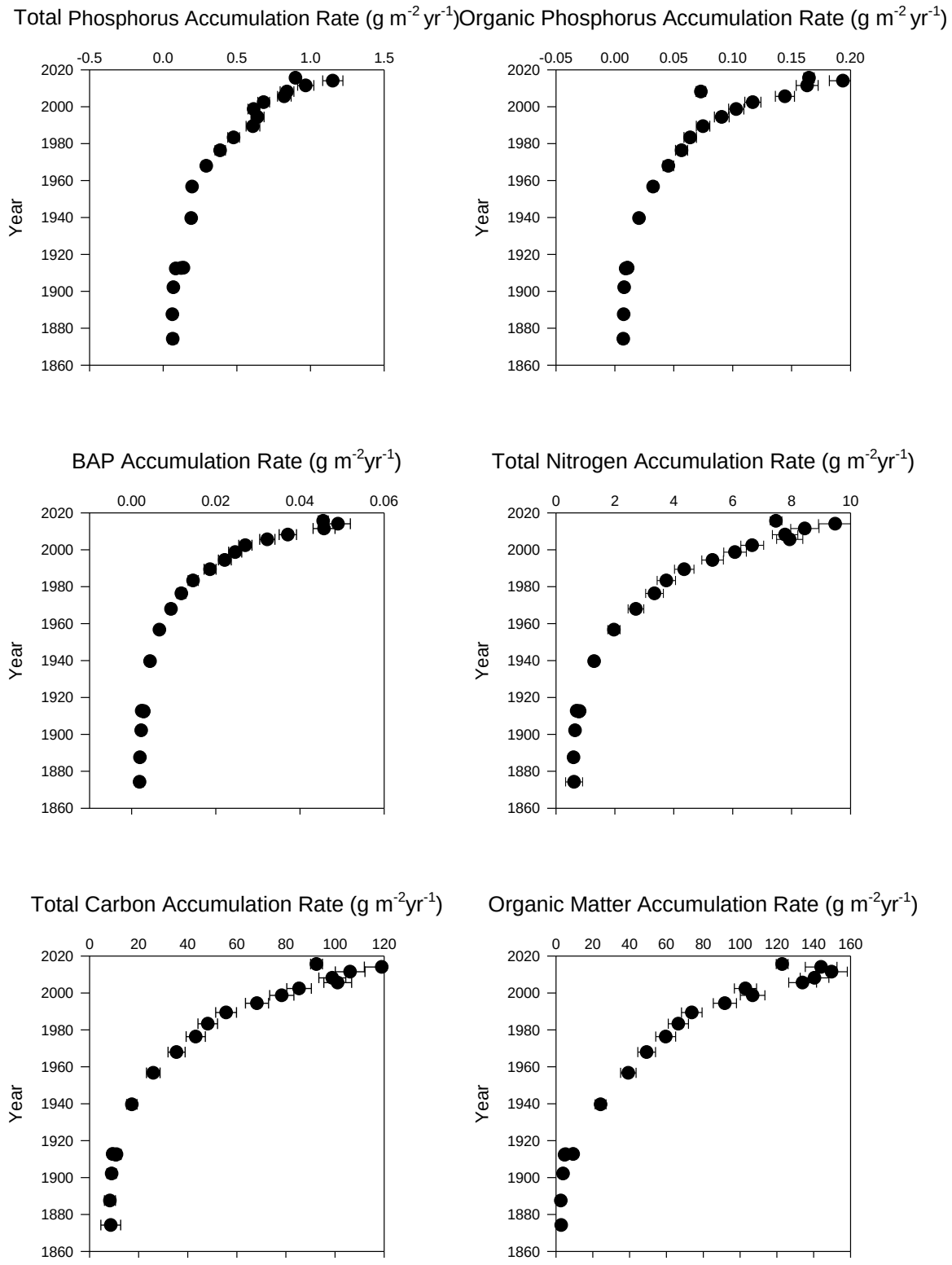


Figure 2.24: Accumulation rates for total P, organic P, biologically available P, total nitrogen, total carbon, and organic matter by year for sediment core sample obtained from Waterhen Bay, Delta Marsh (Figure 1.1), Canada, in August 2015. (±S.D.)

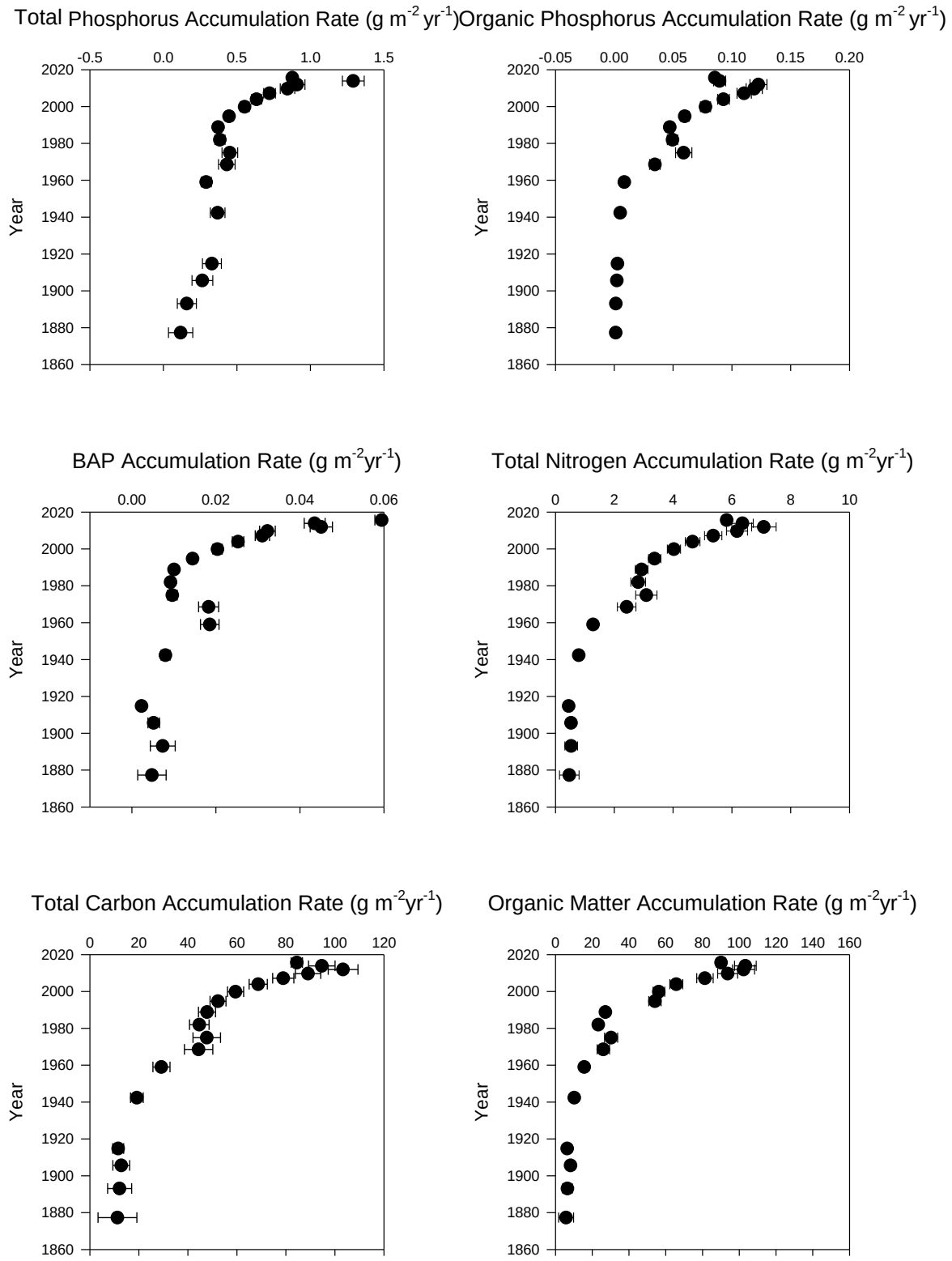


Figure 2.25: Accumulation rates for total P, organic P, biologically available P, total nitrogen, total carbon, and organic matter by year for sediment core sample obtained from Lake Francis 1, Delta Marsh (Figure 1.1), Canada, in August 2015. (±S.D.)

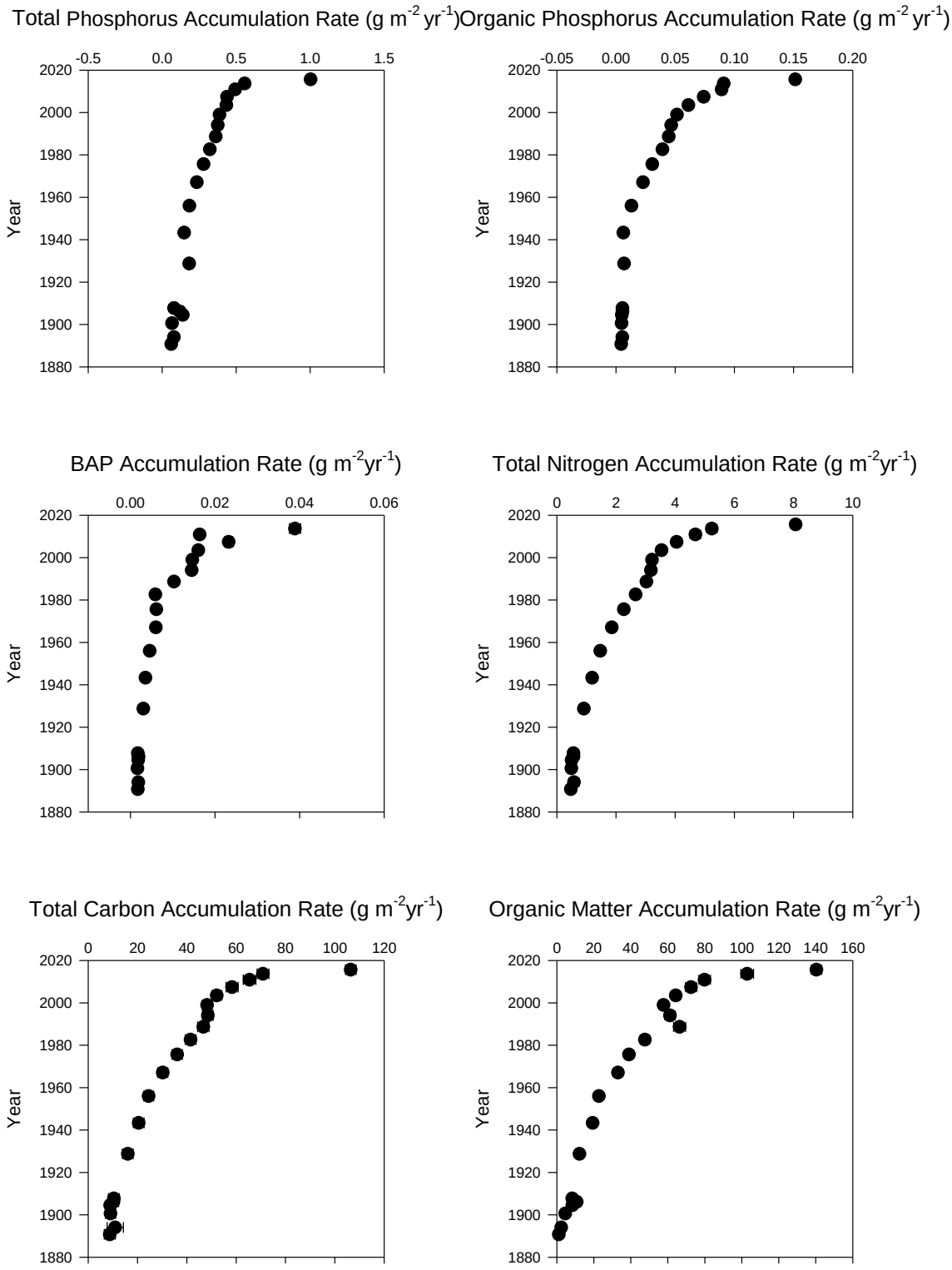
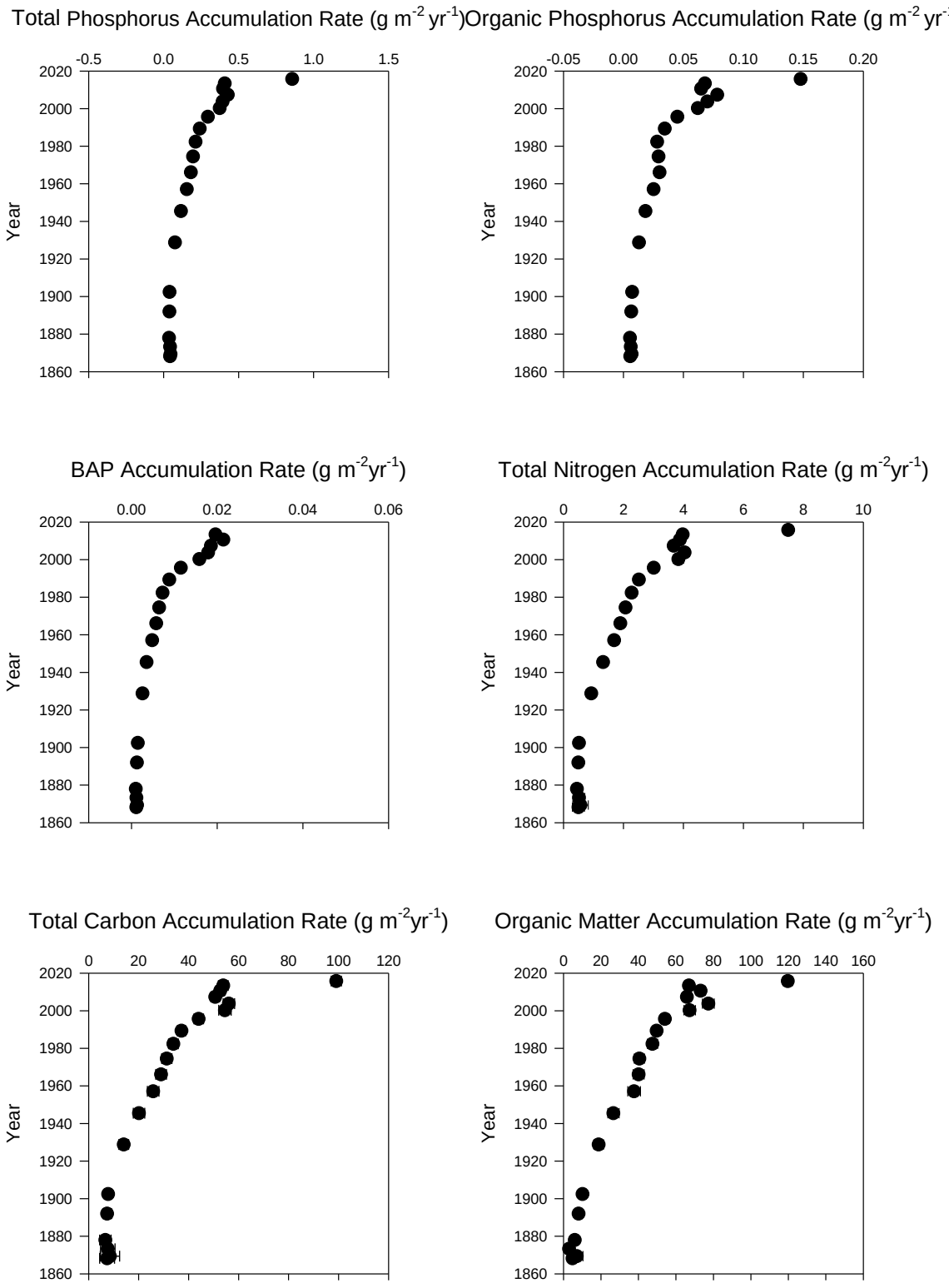


Figure 2.26: Accumulation rates for total P, organic P, biologically available P, total nitrogen, total carbon, and organic matter by year for sediment core sample obtained from Lake Francis 2, Delta Marsh (Figure 1.1), Canada, in September 2015. (±S.D.)



Graphical investigation into the raw nutrient values measured for Canvasback Bay and Weedy Bay, in lieu of being able to determine accumulation rates, indicated that for most parameters the concentrations fluctuated over time without any clear pattern. This is likely due to the high amount of sediment mixing between layers that may have homogenized the sediments as well due to potentially high rates of sediment deposition from the Portage Diversion. The exception was OM which consistently decreased from the deeper sediments to surface sediments for both sites, and the BAP in Canvasback Bay which was consistent in lower sediments and increases considerably in the surface sediments. Considering the OM, this may be explained due to the deposition of less organic sediments from the Portage Diversion as OM levels at the bottom of the Canvasback Bay and Weedy Bay cores were similar (10-15%) to those found in the surface sediments throughout the west marsh.

Unlike the profiles for sediment accumulation and chronology, there is no assumption that the profiles for accumulation rates will be smooth. The shape of the accumulation rate profiles does, however, indicate if the rates of change have been consistent over time. The accumulation rates for Eaglenest 1 and 2, Lyttle Bay, and Lake Francis 1 and 2, have the smoothest profiles indicating the lowest degree of sediment disturbance and consistently increasing rates. Deviations from the smooth profiles were minor but include the PAR for Eaglenest 1 which remained somewhat consistent post-1990, and Eaglenest 2 which contains some values deviating from the smooth trend for the PAR around 1980-1990. The accumulation profiles for Waterhen Bay were similar but not as smooth or consistent. The Big Lake accumulation profiles were similar except with considerable intermixing in the surface layers and a much larger standard deviations compared to the other sites. The accumulation profiles

for Cadham Bay demonstrated increasing values post-1980, although not as smooth and consistent; however, below 1980 the values also increased but with considerably larger standard deviations relative to the rest of the core.

The accumulation rates for all parameters were the highest in Big Lake relative to the other sites with the MAR and TPAR demonstrating the most considerable difference at double-to-triple the rates seen across the rest of the marsh. Conversely the MAR and TPAR in the center marsh and Lake Francis sites behave similarly and are much lower than those from Big Lake. Lake Francis had the lowest TPAR of any marsh unit. Lyttle Bay demonstrated similar accumulation rates to center marsh and Lake Francis. The accumulation rates for Cadham Bay were generally the lowest of all sites measured, likely due to the considerable sediment disturbance and removal taking place at this site, as indicated by the considerably lower inventory values.

3.2.3: Phosphorus Dynamics

For all sites except for Canvasback Bay and Weedy Bay the OP and BAP as a percentage of TP increase over time with the most considerable increases in the top 10-20cm of sediment which represent approximately the last 50-100 years, respectively (Table 2.1). For Canvasback Bay and Weedy Bay there was no consistent pattern, likely due to the high degree of disturbance and mixing between sediment layers. The increasing proportion of OP and BAP in the TP is due to increasing levels of OP and BAP in more recent sediments while the TP either remained consistent or increased less than the OP and BAP. As a result, while the TP levels were increasing in the marsh sediments over time the proportion of TP that was represented by OP and BAP was increasing at a faster rate suggesting a redistribution in the forms of P present

over time. These increases were the most pronounced and consistent in center marsh, Lake Francis, and Lyttle Bay with OP proportions roughly twice those seen in other sites (Table 2.1). In Waterhen Bay the OP as a percentage of TP was higher in the more recent sediments though the increase was more sudden around 1960 relative to the other sites which demonstrated slower and more consistent increases. For Waterhen Bay the BAP as a percentage of TP generally increases in more recent sediments (similar to the OP) however it is not a clear trend and there is considerable fluctuation in levels over time. In Cadham Bay the OP and BAP as a percentage of TP in the deeper and older sediments fluctuated around approximately consistent levels until the most recent sediments where they spiked upwards, post approximately 2000. Cadham Bay had the lowest levels of OP (both measured and as a percentage of TP) while the remainder of the marsh was variable without a consistent pattern. In the west marsh OP and BAP as a percentage of TP was variable but without a clear trend except for BAP as a percentage of TP in Big Lake which increased over time similar to the rest of the marsh and Lake Francis. The BAP (both measured and as a percentage of TP) in the west marsh was the highest of all sites with levels not demonstrating any clear pattern across the other marsh units.

Table 2.3: Organic P and biologically available P as a percentage of total P for sample segments from sediment cores collected at Delta Marsh and Lake Francis (Figure 1.1), Canada, in August and September 2015. Average of top 20-cm represents approximately the past century.

Sample Depth	Big Lake		Canvasback		Weedy Bay		Eaglenest 1		Eaglenest 2		Cadham Bay		Lyttle Bay		Waterhen		Lake Francis 1		Lake Francis 2	
	OP	BAP	OP	BAP	OP	BAP	OP	BAP	OP	BAP	OP	BAP	OP	BAP	OP	BAP	OP	BAP	OP	BAP
(cm)	----------as a % of TP----------																			
0-1	9.0	7.6	11.5	10.1	8.9	4.5	18.3	5.2	19.8	5.5	7.8	5.2	18.4	5.1	9.7	6.8	15.1	7.2	17.2	7.3
1-2	5.2	3.8	14.0	7.0	9.8	5.6	19.7	5.1	17.9	5.3	9.9	4.6	16.8	4.3	6.9	3.4	16.3	7.0	16.7	4.8
2-3	11.6	6.2	12.5	5.1	11.1	5.8	16.1	4.9	18.6	4.6	8.3	3.7	16.9	4.7	13.5	5.0	18.1	3.3	16.4	5.4
3-4	13.5	6.6	15.9	5.2	10.0	5.1	-	-	13.6	2.9	5.3	2.8	8.7	4.4	14.1	3.8	16.9	5.3	18.2	4.3
4-5	11.7	5.5	15.0	4.7	11.8	6.1	15.1	4.4	13.8	3.4	4.3	1.8	17.6	3.9	15.3	4.3	14.1	3.7	17.8	4.5
6-7	9.9	4.6	16.9	4.4	9.7	4.9	13.6	3.9	10.9	3.0	4.9	2.2	16.8	4.0	14.0	3.7	12.4	3.9	15.3	3.9
8-9	8.4	3.9	13.8	5.4	7.0	3.2	12.6	2.9	11.6	2.9	3.9	2.1	12.3	3.1	12.7	2.7	12.2	1.8	13.2	3.4
10-11	6.6	5.4	10.2	3.2	8.1	3.8	19.3	2.5	10.4	2.1	3.7	1.9	14.6	3.1	13.1	2.1	9.7	2.6	16.6	3.2
12-13	11.8	3.2	9.3	2.8	6.9	3.6	11.0	2.6	8.4	1.6	4.0	2.7	16.6	3.4	3.0	6.4	4.2	2.4	16.0	3.1
14-15	6.5	3.6	11.3	3.6	7.8	4.4	11.9	3.2	4.9	1.8	6.0	7.7	-	-	-	-	3.6	1.3	18.2	3.7
16-17	9.1	3.4	11.0	4.4	6.3	3.5	8.9	1.9	4.0	2.3	5.5	1.9	10.9	3.4	-	-	6.8	2.2	15.4	2.8
18-19	13.7	2.9	11.7	5.2	4.8	4.3	9.8	2.7	3.9	1.5	7.8	2.2	12.0	3.2	1.4	3.1	7.3	2.9	13.2	2.6
20-21	9.1	2.3	13.4	5.6	6.6	6.3	8.1	5.8	5.3	1.0	4.5	2.0	10.0	2.6	2.1	2.6	6.3	1.9	17.0	3.1
23-24	12.1	3.2	16.6	6.2	5.1	4.4	7.5	2.1	6.1	1.1	5.0	1.9	9.9	2.5	1.5	1.3	4.5	2.0	12.7	3.1
26-27	14.7	4.1	14.7	8.4	7.2	5.4	6.8	1.8	4.9	1.1	5.3	2.1	5.4	2.6	1.0	1.5	5.3	1.9	12.4	3.0
29-30	7.8	1.4	15.8	5.8	10.4	4.6	6.4	2.1	3.7	0.6	4.0	2.6	4.0	2.2	0.2	2.8	4.5	2.1	12.6	3.2
33-34	-	-	16.3	5.1	9.5	4.7	6.3	2.2	5.1	1.5	4.5	2.7	3.5	2.8	0.8	3.9	3.0	2.5	12.3	2.9
37-38	-	-	17.9	4.9	7.4	6.0	4.7	2.4	4.1	1.1	3.8	3.4	2.0	3.0	-	3.2	2.4	2.4	6.9	3.0
41-42	9.1	3.7	17.5	3.5	11.2	4.4	2.7	4.1	4.5	1.8	3.5	3.1	2.6	2.8	-	3.8	0.2	2.8	6.9	2.9
45-46	10.4	2.6	12.4	3.0	10.4	7.0	-	3.1	3.7	1.7	3.6	3.0	1.8	2.3	-	-	-	2.4	9.4	3.2
49-50	-	-	18.6	3.0	-	-	-	4.2	3.7	1.6	2.3	1.7	2.1	2.6	2.5	4.1	-	-	-	-
Average of top 20-cm	9.7	4.5	12.8	5.1	8.4	4.7	13.7	3.8	11.0	2.9	5.8	3.1	14.3	3.8	9.6	4.0	11.0	3.5	16.2	4.0

For the center marsh sites and Lake Francis 1 the TN/TP has consistently increased in more recent sediments with the highest values found near the surface. Lyttle Bay shows similar increases in more recent sediments except the increase is not as consistent and in the top 20-cm TN/TP remained somewhat consistent. The TN/TP for Lake Francis 2, Waterhen Bay and Cadham Bay fluctuated over time however did not show any consistent pattern. Lake Francis 2 was the site with the overall highest TN/TP ratios with values generally higher for the entire core relative to any sample segment in the other sites so although there was not a clear trend over time (like with Lake Francis 1 increasing) the values have always been elevated. All sites in the west marsh demonstrated consistently decreasing TN/TP ratios in more recent sediments. The west marsh sites and Cadham Bay had the lowest TN/TP ratios of the sites sampled with the lowest values in across the marsh found in Cadham and Weedy Bays. When considering the sediments associated with approximately the past century (roughly the top 20-cm) the sites in center marsh, Lake Francis, and Lyttle Bay all had TN/TP ratios greater than 16 generally indicating that these sites would be P limited (Håkanson and Bryhn 2011). The west marsh sites and Cadham Bay had TN/TP ratios less than 16 generally indicating that these sites would be N limited (Khan et al. 2007; Kinsman-Costello et al. 2014). Waterhen Bay had TN/TP ratios that fluctuated around the 16 mark indicating that it could be either N or P limited although this likely means the site doesn't demonstrate any strong N or P limitations (Håkanson and Bryhn 2011).

Table 2.4: Molar ratios of TN to TP for sample segments from 50-cm sediment cores collected from Delta Marsh and Lake Francis (Figure 1.1), in August and September 2015.

Depth	Big Lake	Canvasback	Weedy Bay	Eaglenest 1	Eaglenest 2	Cadham Bay	Lyttle Bay	Waterhen	Lake Francis 1	Lake Francis 2
(cm)	TN/TP (molar)									
0-1	12.3	8.1	8.1	18.8	21.7	9.7	18.4	14.7	17.8	19.3
1-2	5.7	10.0	9.0	21.2	22.4	10.2	18.2	10.9	20.8	21.6
2-3	12.7	10.3	10.0	18.2	19.9	9.2	19.3	17.2	21.0	21.7
3-4	13.0	11.9	8.8		17.0	8.5	20.5	16.2	20.4	19.0
4-5	12.0	12.9	10.9	19.2	18.3	8.2	21.3	16.4	18.0	22.7
6-7	15.4	14.4	10.0	19.2	18.5	9.8	21.9	16.1	18.7	22.6
8-9	10.8	13.2	9.5	16.9	16.0	8.6	15.8	17.4	18.2	23.6
10-11	13.8	9.0	10.1	17.4	19.2	7.3	19.1	15.1	17.5	23.1
12-13	16.7	9.6	10.5	18.6	15.6	9.0	22.2	9.7	17.8	25.3
14-15	17.2	11.8	12.4	23.2	15.2	11.2			7.9	28.6
16-17	17.3	12.9	10.4	16.1	10.5	12.2	20.2		15.6	27.7
18-19	23.2	13.6	8.8	17.3	12.3	16.5	21.1	11.5	17.0	25.5
20-21	17.8	13.9	9.0	15.5	11.8	16.3	20.5	10.6	15.4	29.2
23-24	23.3	13.5	9.2	15.0	15.1	15.0	19.7	6.2	13.4	24.9
26-27	27.0	15.8	11.7	14.3	14.7	12.2	13.8	6.0	14.7	23.1
29-30	12.9	18.0	14.0	12.7	9.2	10.6	10.5	9.7	14.6	23.4
33-34		18.5	13.4	11.0	13.7	9.6	10.6	10.5	11.2	21.2
37-38		21.8	13.1	6.3	9.5	7.2	11.1	11.2	8.4	14.6
41-42	22.6	20.0	16.4	5.3	14.0	7.7	8.3	11.9	4.1	14.9
45-46	20.0	14.7	14.6	2.3	12.3	10.0	7.5		4.1	17.8
49-50		25.3		2.3	7.1	8.3	7.9	14.3		
Average*	14.5	11.7	9.8	18.5	16.8	10.5	19.9	14.2	17.4	23.8

*Average of top 20-cm of sample, representing approximately the past century

4: Discussion:

4.1: Phosphorus Sorption and Physiochemical Parameters

4.1.1: Are Wetland Sediments a Sink or Source for Phosphorus?

Investigation into the question of whether the marsh sediments will act as a sink or a source of P to the overlying water column suggests that the majority of the time the marsh sediments can be expected to be a sink for P, although at times they may be a source depending on fluctuations in the water column TDP. Wetland sediments acting overall as P sinks, but varying over time due to conditions in the water column, is a common result throughout the literature (Haggard et al. 2007; Hongthanat 2010; Haque et al. 2018). The primary method of determination was comparisons of the sediment EPC values with recorded levels of TDP from the water column, a commonly used approach from the literature. These conclusions were further supported by investigations into the S_{maxT} and DPS parameters, with agreements between the results.

The S_{maxT} values obtained were within the range of those found in wetlands of the Canadian prairie pothole region (266-3527 mg kg^{-1}; Badiou et al. 2018), and palustrine forested and emergent marsh depressional wetlands (51.3–5115.0 mg kg^{-1}; Lane and Autrey 2016). S_{maxT} was much higher than soils in Iowa (357-667 mg kg^{-1}; Hongthanat 2010), croplands (563.7-607.6 mg kg^{-1}) and buffer strips near agricultural fields (412.2-435.2 mg kg^{-1}; Schroeder and Kovar 2008), wetlands impacted by dairy effluent (11-645 mg kg^{-1}; K. R. Reddy et al. 1998), wetlands elsewhere in the prairie pothole region (6.9-333.3 mg kg^{-1}; Haque et al. 2018), and streams with large particle sizes (19-228 mg kg^{-1}; Agudelo et al. 2011) which is reasonable as larger particles

have less mineral cations and surface area thus a lower capacity to adsorb P. The EPC values for this study were similar to those obtained for wetlands in the prairie pothole region (0.001-0.799 mg L^{-1}; Badiou et al. 2018), (0-2.1 mg L^{-1}; Haque et al. 2018), wetlands in Iowa (0.02-0.12 mg L^{-1}; Hongthanat 2010), lakes (0.024-0.119 mg L^{-1}; Zhou et al. 2005) and bays (0.061-0.057 mg L^{-1}; M. Li et al. 2013) in China, shoreline sediments of a freshwater lake (0.03-0.4 mg L^{-1}; Zhou et al. 2011), palustrine forested and emergent marsh depressional wetlands (0.01–27.18 mg L^{-1}; Lane and Autrey 2016), and riparian and floodplain wetlands in the Lake Okeechobee Basin in Florida (0.03–1.28 mg L^{-1}; Reddy et al. 1995). These similarities indicate that the sediment EPC values and sorption dynamics at Delta Marsh would not be considered to be unusual. Throughout the literature wetland sediments have generally been found to agree with my findings and act as P sinks, with the exception of systems where there has been exceptional amounts of past nutrient loading which saturated the sediments with P. The S_{maxT} being higher than many other systems from the literature indicates that the sediments at Delta Marsh have a high potential capacity to sorb and sequester P, and further supports that the sediments may act as a P sink. The high S_{maxT} values also likely contributes to lower EPC and higher PEBC values, encouraging P sorption, as they were negatively and positively correlated, respectively. Schroeder and Kovar (2008) studied riparian buffer strips adjacent to wetlands and found values that are similar to our emergent and wet meadow zone sites (0.10-0.50 mg L^{-1}) which supports the conclusion that these sites have been acting as buffers between the open-water and My finding of marsh sediments consistently acting as a P sink suggests that internal loading is not a major contributor to eutrophication at Delta Marsh that would need to be overcome during restoration efforts. A potential reduction of the external nutrient inputs to the marsh

could therefore be expected to be successful in reducing nutrient levels in the water column and reducing the frequency and severity of algal blooms. The sediments would be expected to sorb and sequester P, aiding in restoration of the marsh as opposed to hindering it (Belmont et al. 2009).

The emergent and wet meadow zone sites presenting a greater likelihood of being a P source compared to the open-water sites could be explained due to the fluctuations in water level resulting in alternating periods of wet and dry for the sediments. Wet/dry cycles can affect the redox conditions of sediments resulting in increased crystallinity of Fe cations, decreasing the sediments affinity for sorbing P (Attygalla et al. 2016). This intermittent drying and rewetting can also shift P sediment chemistry towards more labile forms of P (Kröger et al. 2012) due to the lysis of microbial cells and mineralization of P within OM (Achat et al. 2012), that are more easily released from sediments (Bai et al. 2017). Unfortunately, this cannot be confirmed as fractionation was not conducted for emergent and wet meadow zone sites. The desorption of P from sediments also increases in likelihood with increasing initial P loading (Huang et al. 2012); for all sites the measured TP levels were higher in the emergent and wet meadow zone sediments and for all sites except Lyttle Bay. The S_n and DPS values were also higher, providing another potential explanation for the expected increase in desorption from these sites (Koski-Vähälä and Hartikainen 2001).

The one location that behaved differently from the rest of the marsh was Waterhen Bay with all landscape positions identified as a possible source for P to the water column for all sample dates (Figure 1.4). Located in the easternmost section of Delta Marsh Waterhen Bay is the site most connected to Lake Manitoba. Aminian (2015) identified that the majority of water

inflowing from Lake Manitoba (85%) moves through Clandeboye Bay which is then directly connected to Waterhen Bay and as such of all sites sampled Waterhen Bay should be the site most impacted by the lake chemistry. Waterhen Bay demonstrated water chemistry more similar to Lake Manitoba than other sites within Delta Marsh (Page 2011), which likely explains the low values for TDP.

A study conducted by Stanley (2017) concluded that the east section of Delta Marsh and its surrounding area had the least amount of agricultural intensification and correspondingly the lowest nutrient loading across the marsh. This lower amount of P loading could help further explain the low levels of TDP found in Waterhen Bay relative to the rest of the marsh as the P levels in the water column are directly affected by the P loading from the watershed. Uptake by phytoplankton could also contribute to the low TDP in the sediments as Stanley (2017) also determined that phytoplankton biomass was roughly consistent across the marsh despite unequal nutrient inputs from the landscape and hypothesized that in the east marsh the phytoplankton was drawing nutrients from the sediments instead.

An analysis of sediment particle size conducted by Geard (2015) indicated that Waterhen Bay demonstrated somewhat lower amounts of clay/silt compared to the remainder of the marsh, and therefore would possess a higher proportion of sand. As clay/silt represent the smaller particle sizes in sediments with a higher sorption potential (Mechtensimer and Toor 2016), this may explain the lower sorption potential and higher EPC compared to the other bays (Reddy et al. 1999). It has been demonstrated that a higher sand content results in a lower sorption capacity in Manitoba soils (Ige et al. 2005) and as such this is the most likely explanation for the low sorption capacity found in Waterhen Bay. Geard (2015) also found a

sample consisting almost entirely of gravel in Waterhen Bay which was unique in the analysis of the marsh. Gravel will not contribute in any meaningful way to sediment sorption potential and as such if this sample indicates the possibility of substrate higher in gravel in Waterhen Bay then this could further explain the low P sorption potential and therefore high EPC.

The emergent and wet meadow zone sites at Waterhen Bay also demonstrated generally higher EPC's and lower PEBC and S_{maxT} values, consistent with a lower sorption capacity. The emergent and wet meadow zone sites at Waterhen Bay also contained small amounts of gravel (not found anywhere else in the marsh), which would not contribute to P sorption. Any visible gravel pieces were removed from the sediments prior to conducting the sorption experiments however it is possible that small fragments could have been included which would have impacted the sorption capacity determined. The emergent and wet meadow zone sites for Waterhen Bay initially sampled primarily consisted of gravel and were therefore resampled at another location. The presence of gravel, to varying degrees, in all the above samples indicates that it cannot be ignored as an outlier and that the sediment sorption capacity for Waterhen Bay is likely influenced and potentially reduced by its presence.

Waterhen Bay also demonstrated the lowest PEBC and one of the lowest S_{maxT} levels for the open-water sites. This further supports that this region has a lower sorption potential relative to the remainder of the marsh and agrees with conclusions drawn from the EPC. The EPC was found to be significantly negatively correlated with PEBC and all S values and so Waterhen Bay having the highest and lowest of those values, respectively, is a clear indication of poor sorption potential.

The status of Waterhen Bay as an exclusive P source is dependent upon both its high EPC and low TDP values. If the TDP values from Waterhen Bay (up to 0.043 mg $^{L-1}$) were to be compared to the EPC of many other sites in the marsh the result would be a predicted P sink; Big Lake open water (EPC=0.033 L mg^{-1}), Weedy Bay open-water (EPC=0.022 L mg^{-1}), Eaglenest 1 open-water (EPC=0.031 L mg^{-1}), Eaglenest emergent (EPC=0.038 L mg^{-1}), Cadham Bay open-water (EPC=0.022 L mg^{-1}), Lyttle Bay open-water (EPC=0.030 L mg^{-1}), and Lake Francis 1 and 2 (EPC=0.013, 0.003 L mg^{-1}) all demonstrate EPC values below the high range of Waterhen Bay's TDP.

These EPC vs TDP comparisons can be useful for indicating if soils and sediments will behave as a P sink or source to the associated water body in general terms. However, it is important to note that the extent of what conclusions can be made regarding this information is otherwise limited. The TDP levels within the water body can change considerably within an open-water season, as well as diurnally, so classifying a site as generally a P sink or source is possible but determining how it will at any given time is not possible without obtaining a TDP measurement for comparison. As this study has multiple sample dates spanning the open-water season and the majority identify the marsh as mostly a P sink this is a reasonable conclusion. Between sampling dates the exact sorption/desorption relationship between the sediments and water column are unknown. This comparison also may identify the sink/source relationship present but does not determine the speed of the reaction taking place and the magnitude of P actually adsorbed/desorbed will depend on many factors such as the time of interaction, degree of intermixing, area of contact, species of cations interacting, ionic strength of solution and other factors (Schroeder and Kovar 2008).

The relationship between sediment and water column P can also be assessed by investigating the sorption parameters S_{max1}. S_{max2}, S_{max}, K_1, and K_2 (Table 1.7). The S_{maxT} demonstrates the maximum amount of P that the sediments are predicted to be able to sorb before being completely saturated (Casson et al. 2006). Considering the S_{maxT} compared to the P levels measured in the sediments (Table 1.1) there is still considerable capacity for the sediments to sorb P and it can be concluded that all sites have the potential to act as a P sink under the correct conditions. The S_{maxT} values alone are of limited practical use however as P levels comparable to the S_{maxT} values obtained are unlikely to ever be found in nature and water quality in any system would be impaired long before these concentrations could be reached (Hongthanat 2010). The capacity for sediments to adsorb P also does not mean that it necessarily will, as in application the sorption or desorption of P from sediments also depends on the P concentration in the water column. In addition Brand-Klibanski, Litaor and Shenker, (2007) determined that sediment samples exposed or analyzed in aerobic conditions, which ours were, can result in S_{maxT} values that are inflated in some situations which potentially brings their validity into question. For this reason, it has been recommended that EPC levels are more relevant and reliable than assessments based on S_{maxT}. The S_{maxT} parameter is still useful however as it agrees with and validates conclusions drawn from the EPC levels discussed previously.

The S_{max1}. S_{max2}, K_1, and K_2 can provide additional insight to the sorption potential of the system. The K values represent the binding energy, or the ease of which the sediments can adsorb/desorb P, with a lower K value representing easier and faster reactions. The corresponding S values indicate the potential P capacity associated with each K. In this study

the K_1 values were considerably lower than the K_2 values indicating the fast and slow sorption reactions discussed in Section 1.1, respectively. The S_{max1} values are all considerably larger than the S_{max2} values indicating that the majority of the sorption potential in Delta Marsh is associated with the fast and easy sorption reactions with less associated with the slower sorption reactions. These results are consistent with those found by Badiou et al. (2018) who also studied calcareous soils in the Canadian prairies. This means that sediments which have available sorption capacity, or that were identified as P sinks according to EPC, will overall act rapidly to adsorb surplus P from the water column and act as an effective mechanism for removing excess P.

The majority of the sorption capacity being represented by low-energy sites suggests that potential desorption from the sediments is possible should the sediments become resuspended and act as a potential P source (Cavalcante et al. 2018), despite what is suggested by EPC values (Li et al. 2016). Delta Marsh currently, and historically, has high rates of sediment resuspension due to bioturbation caused by the prevalence of invasive Common Carp within the system (Section 1.5.5) as well as wind action across the more open bays in the marsh. This is further supported by the evidence of considerable mixing within the surface layers of the sediment as discussed in Section 3.2. This potential desorption of P from resuspended sediments could help explain the low levels of sorbed P relative to the potential maximum sorption capacity and the difference observed between EPC and TDP which should be continually moving towards equilibrium. If the presence of Common Carp in the marsh is suppressing the overall P sorption of the sediments then it is possible that with the implementation of the carp exclusion component of the RTT project that the sediment P

sorption, and therefore sequestration, will increase going forward. These benefits would likely have a synergistic effect as decreasing Common Carp activity would also increase coverage of submersed and emergent vegetation (also benefitting from increased sediment P levels) which would further decrease sediment resuspension and result in additional reductions in P desorption from the sediments (H. Zhang et al. 2016). An increase in submersed and emergent vegetation could also result in increases to the soil OM and related cations such as Al and Fe, further increasing the P sorption capacity over time (Bai et al. 2017; Jin et al. 2013).

Considering the results obtained from the 50-cm cores (Section 4.2) for Lake Francis and center marsh it is suggested that sites that sites that are isolated from Common Carp experience increasing levels of OM (Figure 2.19-2.26) and increasing proportions of P present in the form of OP (Table 2.4) compared to sites exposed to Common Carp. As OP is usually one of the less labile forms of P found in sediments (Min Li et al. 2013), these findings further support that lower numbers of Common Carp in Delta Marsh will result in increases in P sorption and retention.

The DPS is another parameter commonly utilized to determine the likelihood of P desorption from sediment (Casson et al. 2006) and if the system would act as a sink or source for P (Ige, Akinremi, and Flaten 2005). Use of the DPS is more accurate and preferred to direct measurements of soil P or assessment of the S_{maxT} as it takes into account the current P load as well as the maximum capacity instead of considering them in isolation (Casson et al. 2006). Once calculated DPS is also appropriate for all soil types (Ige, Akinremi, and Flaten 2005). The DPS has been found to be strongly correlated with measurements of P desorption from soil under experimental conditions (Bai et al. 2017; Hooda et al. 2000; Zhou and Li 2001) while S_{maxT}

was not directly correlated. As DPS increases the number of available sorption sites for additional P sorption decreases and the sites being utilized are of increasingly poor quality resulting in an increase in potentially available or exchangeable P (Yan et al. 2017).

Review of the literature indicates that the DPS can be calculated using a number of different methods which may yield different results from the same data. For each method the DPS is similarly defined as the proportion of available sorption sites that are currently occupied by P and as such the results obtained are often similar. In this study several methods were tested for the calculation of DPS and results obtained between the various methods were similar enough that the resulting conclusions would not be changed. All methods of DPS calculation used in studies referenced below were explored for this study.

Generally speaking the likelihood of P desorption increases with DPS (Casson et al. 2006). Hooda et al. (2000) determined that below 10% there is negligible desorption of P with desorption rates increasing linearly above that level while Bai et al. (2017) found similar results at low levels and an exponential increase in P desorption at higher sediment P concentrations. A study by Casson et al. (2006) identified DPS levels from 25-40% to be generally associated with a high risk for P losses. The DPS values for Delta Marsh sediments were all relatively low (Table 1.7). All the sample sites exhibited DPS levels below the 10% threshold with the exception of one (Center Marsh – Wet Meadow, DPS =17.1%) and as such should not be expected to be at risk for P desorption or to be a source for P based on DPS. The higher DPS value of Center Marsh Wet Meadow is likely due to its close proximity to agricultural fields resulting in elevated S_n concentrations (due to increased loading) and low S_{maxT}. The DPS of 17.1%, although our highest value, is still lower than the threshold suggested by Casson et al.

(2006) and as such may suggest a higher risk for P desorption but does not classify the site as high-risk.

As discussed in section 1.1.4 many governments have established critical values for DPS used to identify sites at a high risk for P desorption (Zhang et al. 2005). These critical values need to be established for the specific system under investigation due to regional differences in analytical techniques and system properties (Casson et al. 2006). There is currently no established critical value for Manitoba soils (Ige, Akinremi, Flaten, et al. 2005) however these guidelines can still be utilized to support our analysis as long as this is understood and not used in isolation (Hongthanat 2010). Critical values of 25%, 20%, 15%, and 9% were established for the Netherlands (Zhang et al. 2005), Florida, Delaware, and Quebec (Casson et al. 2006), respectively. These guidelines vary due to the method of analysis but primarily are different due to the desired strictness of water quality protections in each area. Following the critical values of the Netherlands and Florida none of our sites would be an unacceptable risk for P desorption, for Delaware Center Marsh wet meadow is identified as an unacceptable risk, and for Quebec the West Marsh wet meadow is also identified as a risk site. This is in support of our other analysis that suggests the marsh would generally be a sink for P. The two sites (Eaglenest and Cadham wet meadow) identified that could be considered a risk, depending on criteria, were both sites expected to be anthropogenically impacted and in close proximity to agricultural fields suggesting that disturbance and/or the presence of agriculture is likely the reason for these sediments to be P desorption risks.

As discussed in section 3.1.3 there were significant differences detected between landscape positions for DPS (Table 1.10) with the highest DPS values for each marsh unit found

in the wet meadow zone (Table 1.7). For each marsh unit the DPS values were lowest in the open-water and increased as the sample sites moved to higher elevations. With emergent and wet meadow zone sites increasingly and consistently having higher DPS values this suggests that higher water levels in the marsh would result in increased risk of overall P desorption from the surrounding landscape. As explained in Section 1.5 the elevation at Delta Marsh is relatively flat and small increases in water level can considerably increase the open-water area of the marsh, inundating large areas not normally in contact with the water column. As well with the increased use of the Portage Diversion in recent years these higher water levels can be expected to be more common in the future.

Assessment of the DPS levels indicate that overall there is little risk for P desorption from wetland sediments and as such they should not be expected to act as a P source. Methodologically these conclusions carry less weight compared to those from assessing the EPC as without prior established guidelines assessment of DPS is more qualitative in nature. The DPS is still useful in that the conclusions drawn from the DPS are generally in agreement with, and support the conclusions drawn from the EPC and sorption capacity.

4.1.2: Phosphorus Fractionation

A study by Huang et al. (2015) demonstrated similar results to those found at Delta Marsh with fraction concentrations in the decreasing order HCl-P > Residual-P > NaOH-P > $NaHCO_3$-P, while Linquist et al. (2011) reported fraction orders of HCl-P > NaOH-P > $NaHCO_3$-P, so our results are not unprecedented. There were however also considerably different results found in the literature from those at Delta Marsh. Considering the HCl-P fraction, studies have found it to represent from as little as 9% (Linquist et al. 2011) to as much as 70% (Huang et al.

2015) of the TP. Studies conducted by Yan et al., (2017), Saleque *et al.*, (2004), and Koski-Vähälä and Hartikainen, (2001) found NaOH-P to be the dominant fraction accounting for as much as 60% of TP. As well, Cavalcante *et al.*, (2018) studied the sediments of multiple systems and found the labile fractions to be dominant representing up to 62% of TP with $NaHCO_3$-P being the dominant fraction and the HCL-P the smallest fraction. These examples demonstrate that the composition of P fractions can vary considerably between different systems and we could not have predicted the P fraction speciation without this study.

The P fractionation results support the conclusion from Section 4.1.1 that the marsh sediments are unlikely to be a source of P under normal conditions. Only approximately 9% of sediment P is in labile forms (H_2O-P and $NaHCO_3$-P) that are likely to be released from the sediments, contributing to eutrophication, or otherwise bioavailable to be taken up by plants. Of that 9%, only 1% of the total P in the sediments is H_2O-P that represents the most readily available and extractable form of P in the sediments (Saleque et al. 2004). Approximately 53% of the wetland sediment P is contained in the HCL-P fraction that is occluded, highly stable, and generally associated with permanent sequestration of P in sediments (Cavalcante et al. 2018). The remainder of the non-labile fraction is divided roughly evenly at 19% each between Residual-P, the most stable P constituent considered to represent permanently sequestered P (Reddy et al. 1999), and NaOH-P, which is unlikely to be released from sediments except under very specific anoxic redox conditions (Cavalcante et al. 2018). Considering these fractions, it is unlikely that large amounts of P would be desorbed from the wetland sediments under normal conditions and the sediments at Delta Marsh should be considered to be effective for the long-term storage of P (Koski-Vähälä and Hartikainen, 2001; H. Zhang *et al.*, 2016). It is important to

remember that even small amounts of P desorption can impair water quality resulting in algal blooms and the less labile forms of P can still be desorbed with the difference being the relative level of risk.

With regards to the relatively small pool of labile P in the wetland sediments, one possible explanation is that most of the easily-desorbable P has already been stripped from the sediments through desorption reactions triggered through sediment resuspension (Li *et al.*, 2016; Koski-Vähälä and Hartikainen, 2001) due to the bioturbation of Common Carp (Badiou and Goldsborough 2010) or wind action. Another possible explanation for the labile P found in Delta Marsh could be due to the influence of phytoplankton and other algae. Stanley, (2017) found that nutrient loading across the marsh was considerably higher in the west and center marsh units relative to the east however levels of algae and phytoplankton were consistent across the marsh, contrary to her hypothesis. This algal distribution could be explained with the additional nutrients coming from the sediments in the east marsh which has the lowest proportion of labile P (7-8%) relative to the west and center (9-13%) where nutrient inputs were speculated to be sufficient enough to support the algal population unassisted. Stanley, (2017) also determined that phytoplankton populations across the marsh were more strongly correlated with nutrient levels in the sediment than in the water column. Stanley, (2017) did not include Lake Francis. If this is correct then it can reasonably be concluded that decreasing nutrient inputs to the marsh would likely result in additional uptake of labile nutrients by phytoplankton communities from the sediments, increasing P sorption capacity and resulting in further decreased nutrients in the marsh. P in sediments can also be converted between the various fractions given the right conditions (Saleque et al. 2004); if the labile pool is depleted

then the more stable forms can be converted to replenish the supply while excess labile P can be converted to more stable forms over time if it remains in the sediment. This means that reductions to the labile P pool could also result in decreases in the other fractions.

It has been suggested that high concentrations and proportions of Residual-P and HCL-P could represent primary mineral P (apatite) that has simply not had the opportunity to be weathered (Obikoya 2016) as opposed to representing formerly mobile P that has been sorbed by the sediments (Huang et al. 2015). It is unfortunately impossible to distinguish between P sorbed by sediment or represented by un-weathered apatite within the HCL-P and Residual-P fractions given the data available. In a study by Huang *et al.*, (2015) it was suggested that the HCL-P detected arose from soil minerals as opposed to sorption mechanics or plant residue due to the fact that the concentrations were very consistent across all sample sites. In our results the HCL-P ranged from 292.2-454.2 mg kg^{-1} representing a much larger range and suggesting that although some HCL-P detected could represent poorly weathered rock material it is unlikely to account for all of it. This suggests that some of these non-labile fractions are due to the sorption and sequestration taking place within the marsh which supports the hypothesis that marsh sediments are acting as a sink for P within the system. Regardless of source, the HCL-P and Residual-P fractions are highly stable and unlikely to contribute to P desorption (Koski-Vähälä and Hartikainen 2001).

Across the marsh the only statistically significant difference found for the P fractions was that Lake Francis had the highest levels of Residual-P at 23-25% (Table 1.5) of total compared to the average of 19% across all sites. Apart from the Residual-P the other fractions did not exhibit any noticeable pattern in proportions. Adding the other non-labile P fractions

together the west marsh represented the lowest proportion, with levels relatively consistent across the remainder of the marsh. This suggests that the mechanisms that regulate P sorption and the exchange between the different P fractions are otherwise somewhat consistent across the open-water sites of the marsh. These fractionation results indicate that the sediments of the west marsh are the most likely to desorb P with the remainder of the marsh approximately equal or possibly favoring Lake Francis for P retention.

4.1.3: Correlations and Significant Differences Between Variables Across the Marsh

Sediment resuspension due to wind action may be able to explain the significantly lower OM in the open-water sites as it has been demonstrated to reduce OM by accelerating decomposition, inducing cell lysis, delivering O_2 to previously buried sediments, and through promoting changes in redox conditions (Kamula, 2015; Lenzi and Renzi, 2011). This can be seen in the east marsh as the more open (and exposed to wind action) Cadham and Waterhen Bays demonstrated lower OM levels while the more sheltered Lyttle Bay exhibited higher levels of OM. Higher OM in the emergent and wet meadow zones is also likely influenced by the higher amount of living and decomposing plant matter relative to the open-water.

Resuspended sediments are also at a much higher risk for P desorption due to an increase in surface area contact between the sediment particles and the water column as well as due to the continual flushing of sediment particles with a higher volume of water (Koski-Vähälä and Hartikainen 2001). This desorption may be a contributing factor to the significantly lower S_n values found in the open-water sites, especially considering that the sediments of Delta Marsh are suspected to be susceptible to P desorption under conditions of resuspension (Section 4.1.1). For the sites where samples were within larger bays (Cadham Bay and

Waterhen Bay) that would have a larger wind fetch (Stanley 2017) and correspondingly expect more resuspension the open-water sites demonstrated the lowest S_n, while the more sheltered sites (Big Lake and Lyttle Bay) represented higher levels (Table 1.6). This P loss due to resuspension has been found for both resuspension due to wind action and bioturbation (Reddy et al. 1999), and usually is represented by a decrease in the more labile forms of P (Cavalcante et al. 2018).

The significantly higher OM and S_n found in Lake Francis could be explained due to the absence of bioturbation caused by Common Carp as Lake Francis has been in a state (of at least partial) Common Carp exclusion. The lower magnitude of disturbance would allow for an increase in living submersed and emergent vegetation as well as increased opportunity for detritus to settle into the sediments and remain without interference (Lenzi and Renzi 2011). As bioturbation increases P desorption and lowers S_n, the absence of Common Carp in Lake Francis may be a contributing factor to the higher S_n in these sites.

Disturbance caused by the operation of the Portage Diversion may explain the significantly lower levels of OM in the west marsh as while operating it can import sediment while also resuspending and scouring away large quantities of sediment from the west marsh. This resuspension and removal would be expected to result in lower levels of OM (Lenzi and Renzi 2011; Kamula, 2015). During breach events the west marsh is inundated with waters known to be considerably richer in P (Nicholson 2012) than the waters usually found in Delta Marsh or Lake Manitoba. This may explain the open-water sites with the highest S_n within Delta Marsh proper being Canvasback Bay and Weedy Bay, and to a lesser extent Big Lake (Table 1.6), as the sites most impacted by Portage Diversion overflow (see Section 4.2). Conversely the east

marsh, subject to less direct Portage Diversion influence and watershed nutrient loading, was found to have significantly lower levels of Sn compared to the other marsh units (Goldsborough and Suggett, 2015; Stanley, 2017). The conclusion that elevated P levels in Canvasback Bay and Weedy Bay is due to the Portage Diversion is additionally supported by three more points. First, the nutrient loading to the west and center marsh units (excluding the Portage Diversion) are somewhat consistent (Stanley 2017) however the S_n was considerably lower in Big Lake and Eaglenest Bay sites which are expected to be considerably less impacted by the Portage Diversion. Secondly, comparing Cadham Bay to Waterhen Bay, the two east marsh sites with the closest connection to Lake Manitoba, higher S_n values (for all landscape positions) are found in Cadham Bay which has been identified to receive considerably more water from the Portage Diversion (Aminian 2015). Of course, it must be considered that Cadham Bay is also expected to be influenced by the Delta Beach development which would be expected to result in increased nutrient loading and as such like in most cases the higher sediment nutrients can not be exclusively assigned to just one source given the information available. The third and most compelling point is from Section 4.2 where sediment profiles for Canvasback Bay and Weedy Bay demonstrate considerable amounts of disturbance incomparable to any other site within the marsh that could only be caused by an exceptional source of disturbance such as the breaching of the Portage Diversion. Overall these findings suggest that the operation of the Portage Diversion potentially results in increased levels of sediment P, and likely eutrophication, in areas of Delta Marsh that receive its waters which is important to be understood for the purposes of management of the Diversion, marsh, and Lake Manitoba.

Investigation into sediment disturbance indicates that higher levels of sediment OM could be encouraged throughout Delta Marsh via decreasing potential disturbances to the sediments which would likely result in an increase in P sorption potential and sequestration. This is important as the Common Carp exclusion component of the RTT project already underway should be expected to result in decreases in bioturbation, an increase in submersed and emergent vegetation, decreasing resuspension via wind, and ultimately improved P sorption by the sediments. These results are in agreement with a study by Badiou (2005) that determined the approximate density of Common Carp expected within Delta Marsh of 400 kg ha^{-1} (at the time) could increase internal loading of P levels in the water column an equivalent of 66%. Future monitoring of the marsh sediments for changes in OM content and improvements to sorption capacity is suggested.

Due to the alternating periods of inundation and drying in the emergent and wet meadow zone sites as water levels fluctuate the soils and sediments in these sites are exposed to different redox conditions compared to the open-water. It is well known that Fe is very sensitive to changes in redox conditions (Huang et al. 2015) and alternating periods of drying and rewetting can result in Fe being reduced and oxidized between ferric iron (Fe-III) and ferrous iron (Fe-II) which exhibit very different affinities for sorption of P (Bai et al. 2017). As the sediments alternate between wet and dry, the soil Fe oxidizes and reduces and the sediment P can shift towards more labile forms of P that are subsequently released each time the sediments are inundated with high water levels (Achat *et al.*, 2012; Bai *et al.*, 2017). Through this mechanism these wet/dry cycles result in a continually decreasing sediment

affinity for P, which has been associated with an increase in crystallinity of Fe phases (Attygalla et al. 2016).

It is potentially due to these redox changes that S_n, BAP, EPC, PEBC, and other related parameters demonstrate positive correlations to M3 Fe in the open-water sites and negative correlations in the emergent and wet meadow zone sites. The positive correlations were expected as Fe is well established to be a key positive factor in P sorption (Cui *et al.*, 2018; Fink *et al.*, 2016) and is consistent with the literature (Badiou, Page and Akinremi, 2018; Kowalczewska-Madura *et al.*, 2018), however the negative correlations were not expected. All three S_{max} parameters also had significantly lower values in the wet meadow zone which may also be an influence from the intermittent drying and rewetting of the zones, crystallinity of sediment Fe and lability of sediment P resulting in decreased sorption capacity and a loss of sorbed P (Attygalla *et al.*, 2016; Achat *et al.*, 2012; Bai *et al.*, 2017).

The BAP was positively correlated with TP in all landscape positions although only significantly in the open-water. It was also significantly correlated with the highly labile $NaHCO_3$-P as well as less labile NaOH-P fractions. Together this suggests that the highly labile BAP related to sediment Fe is being retained in the open-water sites but being desorbed from the emergent and wet meadow landscape positions due to the effects of Fe. Similar relationships have been identified for BAP in sediments exposed to drying and rewetting in numerous other studies (Achat *et al.*, 2012; Badiou, Page and Akinremi, 2018). This may also explain why the relationship between BAP and TP was strongest in the open-water as in the emergent and wet meadow zone sites the BAP was comparatively being lost to desorption and was proportionally a lesser constituent of the TP. The BAP concentrations were however higher

in the emergent and wet meadow zone sites relative to the open-water. This suggests that although the P associated with Fe may be lower in the emergent and wet meadow zone sites the overall P concentrations were higher. This could be explained due to the OP which was also significantly lower in the open-water and higher in the emergent and wet meadow zone sites , as well as negatively correlated to BAP in the open-water and positively correlated in the emergent and wet meadow zone sites . The OP can be mineralized into BAP through a process catalyzed by phosphatase enzymes which are commonly found associated with plant roots (Huang et al. 2015) which would be abundant in the emergent and wet meadow zone sites . As OP and BAP were positively correlated in the emergent and wet meadow zone sites , where mineralization would be more likely due to the cyclical wet-dry cycles (Achat et al. 2012) and larger amount of plant roots and OM, it is possible that the BAP in the emergent and wet meadow zone sites is being supported by the OP. Similarly, the transformation of OP to BAP was not prevalent in the open-water sites indicated by the negative relationship between the two P forms as well as the significantly lower levels of BAP in the open-water sites.

These redox conditions suggest that at higher water levels, when the emergent and wet meadow zone sites become saturated, the Fe bound P is more susceptible to desorption which could have important implications for management of the water levels of the marsh and Lake Manitoba. Revisiting Section 4.1.1, the EPC vs TDP comparisons also indicated that the emergent and wet meadow zone sites were more likely to participate in P desorption relative to the open-water sites. Fluctuating water levels are important to the long-term health of the marsh ecosystem and one of the proposed courses of action for marsh restoration however it

would be beneficial to be aware of all the potential implications of this course of action to anticipate otherwise unexpected results.

Sediment OM was found to be one of the most influential variables on the P sorption parameters. The M3 P, M3 Al, M3 Mg, TP, TN, OP, TC, and the residual-P fraction were all found to be significantly positively correlated with OM. This is not surprising as OM is well established to be positively associated with all of the above parameters (Achat *et al.*, 2012; Jin *et al.*, 2013; Min Li *et al.*, 2013; Yan *et al.*, 2017; Badiou, Page and Akinremi, 2018; Cui *et al.*, 2018; Haque *et al.*, 2018). For most variables these significant correlations were found to hold across all landscape positions and both including and excluding Lake Francis which is not the case for many of the correlations found between other variables. This indicates that OM has a very strong influence on the other physiochemical and sorption capacity parameters in the marsh. The OM being higher in Lake Francis and lower in the west marsh was also true of TP, M3 P, OP, TN, M3 Mg, M3 Ca, and TC, which indicates that the OM could be responsible for or contribute to these other physiochemical parameters by creating complexes with the sediment cations and providing additional sites for P sorption (Reddy et al. 1999).

The S_{maxT}, S_{max1}, S_{max2}, and PEBC were all also found to be significantly positively correlated with OM. Sediment OM often results in higher concentrations of cations associated with sorption through organic-inorganic complexes (Cui et al. 2018) as well as promoting forms of cations with greater sorption potential via inhibiting crystallization (Reddy *et al.*, 1995; Yan *et al.*, 2017), ultimately resulting in a higher sorption capacity in soils with higher OM (Kang et al. 2009). The OM in sediments can also result in bacteria colonies that absorb P from the overlying water column, enhancing the sorption potential of the sediments (Jin et al. 2013). This

increased sorption potential results in an increase in sorbed sediment P and potentially long-term P sequestration as well as a corresponding decrease in P levels in the overlying water column (Jin *et al.*, 2013; Min Li *et al.*, 2013).

S_n was significantly correlated M3 Al, M3 Mg, and M3 Ca in the emergent and wet meadow zone sites which, similar to Fe, is not surprising as these parameters are all also well established to be key drivers of P sorption in soils and sediments and consistent with the literature (Li *et al.*, 2016; Casson *et al.*, 2006; Tercero *et al.*, 2017; Bai *et al.*, 2017; Reddy *et al.*, 1999). Interestingly these relationships were not present in the open-water sites where a very weak negative correlation was found instead, which was not expected. These results suggest that the M3 Al, M3 Mg, and M3 Ca are influential on the S_n in the marsh sediments; however, in the open-water sites other variables may be overpowering and obscuring these positive relationships. Another possible explanation is that in the open-water sites the M3 Fe is dominating the P sorption reactions relative to the other cations while in the emergent and wet meadow zone sites where Fe exhibits a reduced role in P sorption the other cations are able to more clearly demonstrate their affinity for P sorption (Cui et al. 2018)

While not statistically significant, there was a clear pattern present for all three parameters with center marsh and Lake Francis demonstrating higher sorption capacities while in the west and east marsh units the sorption capacities were markedly lower. A possible explanation for this trend could be due to the effects of OM as significantly lower levels were found in the west and east marsh units with higher levels in center marsh and Lake Francis. Significant positive correlations were found between all sorption capacity parameters and OM. Higher sediment OM is typically associated with higher sorption potential which would be in

agreement with the trend seen across the marsh. The center marsh and Lake Francis also exhibit higher levels of M3 P, TN, Residual-P, and TC which could all also be attributed to the higher levels of OM.

Another possible explanation for this trend is that the west and east marsh units are directly connected to Lake Manitoba while center marsh and Lake Francis are more isolated and exposed to less agricultural runoff. Additionally, when considering the east marsh, the Lyttle Bay open-water site which is the most removed from Lake Manitoba exhibited the highest levels for all of S_{maxT}, S_{max1}, and S_{max2} compared to Cadham and Waterhen Bays which are located near to the lake-marsh connections. It can be speculated that connectivity with Lake Manitoba is resulting in a decrease in sorption capacity compared to sites that are isolated from the lake. The mechanism of this influence cannot be known for certain; however, a possibility is due to the influence of the Portage Diversion which due to proximity would impart high amounts of nutrients to the marsh sites relative to their connectivity to Lake Manitoba.

If lake-marsh connectivity is resulting in decreased sorption capacity in the marsh it is worthwhile to revisit a possible course of action suggested by Aminian, (2015) for improvement to marsh water quality which was to close off Delta Channel, the direct connection between Cadham Bay and Lake Manitoba. Aminian, (2015) identified that the closing of Delta Channel would have minimal impact on the hydrological conditions of the marsh but would considerably reduce the quantity of high-nutrient inflows of water from the Portage Diversion to the east marsh. This action would likely have a beneficial impact on sediment sorption capacity regardless of if lake-marsh connectivity is a factor as the increased P loading from the Portage Diversion would itself negatively affect sediment sorption capacity, and Delta Marsh overall.

The wet meadow zone exhibited significantly lower levels of M3 Fe, and M3 Ca which are all known to be important to P sorption capacity, particularly in calcareous soils (Cui *et al.*, 2018; Kowalczewska-Madura *et al.*, 2018; Tercero *et al.*, 2017), which could potentially help explain the reduced sorption capacity of these sites. Furthermore there were significantly higher levels of M3 Mg in the wet meadow zone which in some cases can result in preferential formation of more soluble Mg-P compounds over more stable Ca-P compounds (Badiou et al. 2018). All three sorption capacity variables were found to be significantly positively correlated with M3 Mg indicating that Mg is likely responsible for contributing to the sediment sorption capacity. If this is the case then these more labile P compounds which could be in higher amounts in the wet meadow would be at a higher risk for desorption compared to other forms of sorbed P, conversely resulting in a decrease in sorption potential.

Interestingly the wet meadow zone also exhibited significantly higher levels of TP, M3 P, OP, and DPS, which all indicate a potentially higher level of P loading in these sediments. It has been established that a higher background level of P in sediments can result in a lower sediment sorption capacity (Cui et al. 2018), and as such the higher levels of P loading in the wet meadow zone could help explain the depressed sorption capacity at those sites. The wet meadow zone, more so than any other, is subjected to nutrient inputs from the developed agricultural and recreational land surrounding Delta Marsh (Stanley 2017) which likely contributes to the elevated levels of sediment P loading. Many of the wet meadow sites sampled were in very close proximity to active agricultural fields which likely utilize fertilizers and/or graze cattle. If increased P loading to the marsh is resulting in a decreased sorption capacity, as illustrated in the wet meadow zone, then actions to reduce P loading from the

surrounding landscape would likely result in an increase in the ability for sediments to sorb P and reduce eutrophication in the marsh and Lake Manitoba. As explained previously many areas of the marsh have agricultural activity encroaching immediately up to the edge of the open water area; the establishment or increase of buffer zones between agriculture and the marsh could be one possible action to reduce nonpoint P inputs currently entering into the marsh system.

For all S_{maxT}, S_{max1}, and S_{max2} significant positive correlations were found with TP, M3 P, OP, and BAP. These relationships are expected as sediments with a higher sorption capacity typically exhibit higher concentrations of sorbed P. In the case of these parameters it is likely that the higher sorption capacity is the cause of the higher P concentration within the sediments and not the inverse as higher levels of P loading can work to lower the corresponding sorption capacity. From this it can be concluded that marsh sediments are functioning to sorb and potentially sequester P from the marsh waters. Any changes within the marsh that would be expected to aid in increasing the sorption capacity, such as increased OM, decreased sediment resuspension, or decreased external nutrient loading could result in increased removal of water column P and decreased eutrophication. This provides additional support for management and land use changes that may reduce external nutrient loading to the marsh, which in itself would reduce eutrophication, as the resulting increase in sediment sorption capacity would act to amplify the reductions to eutrophication in the marsh.

The S_{maxT}, S_{max1}, and S_{max2} were all positively correlated with the sediment chemistry M3 Al, M3 Fe, and M3 Mg which was expected as these cations are all well established to be positive contributors to P sorption capacity in the literature (Reddy *et al.*, 1999; Bai *et al.*, 2017;

Yan *et al.*, 2017; Cui *et al.*, 2018). The strength of these correlations was however not all equal. The M3 Al was only significantly correlated with S_{max1}, and not S_{max2} while the M3 Fe was only significantly correlated with S_{max2} and not S_{max1}. The S_{max1} is associated with the faster, lower energy, and weaker sorption processes while S_{max2} is more associated with the slower, higher energy, and stronger sorption processes. From this we can speculate that Al in the sediments is contributing more to the fast/weak P sorption while Fe contributes more to the slower/stronger P sorption. The M3 Mg was significantly correlated with both S_{max1} and S_{max2} and as such would significantly contribute to both forms of sorption.

The EPC was found to be significantly lower among open-water sites and higher in the emergent and wet meadow zone sites which could be explained as TP, M3 P, and OP were all found to be significantly higher in the emergent and wet meadow zone sites. This is further supported as the S_n and DPS (also measurements of sediment loading) were also found to be significantly lower in the open-water landscape position which agrees with the EPC. These results suggest that a decrease in the overall P load in the sediments from outside sources would result in a lower EPC (and increased PEBC) and therefore a decreased likelihood of the sediments acting as a source of P. This is encouraging as any mitigation measures taken at the marsh or within the surrounding watershed to decrease external loading would be amplified internally by an increased sorption capacity of the sediments.

There were significant positive correlations found between M3 Mg and the sorption capacity parameters, indicating that Mg in the sediments of Delta Marsh contribute to increased sorption capacity, which is consistent with prior studies of Manitoba soils (Ige, Akinremi and Flaten, 2008; Frossard *et al.*, 1995). The EPC was found to be significantly

positively correlated with M3 Mg in the wet meadow zone but negatively with the emergent zone. In both cases the inclusion of Lake Francis eliminates this correlation almost entirely. When assessing the data graphically for the emergent zone the EPC/M3 Mg datapoint for Lake Francis appears to be an outlier; the M3 Mg value is within reasonable range, but the EPC value is considerably higher than the rest of the sites. With regards to the relationship between EPC and M3 Mg being positive, meaning a higher M3 Mg would result in a lower expectation for the sediment to act as a sink for P, this could be explained as P sorbed by Mg can result in more stable forms such as HCL-P or Residual-P (Reddy et al. 1999). This is supported by the significant positive correlation found between M3 Mg and Residual-P. As these forms are more stable and therefore likely to remain sorbed in the sediments, and these represented the two largest fractions found at Delta Marsh, it is possible that the Mg bound P in the sediments has built up over time decreasing the remaining sorption capacity. This could potentially explain the positive relationship between EPC and M3 Mg, as the more Mg present and bound to P in the soils the less sorption capacity would remain, represented by a higher EPC (Frossard et al. 1995).

The EPC was significantly negatively correlated with M3 Fe in the emergent zone alone as well as when considering all sites together, with the correlation present in open-water and wet meadow sites although not statistically significant. These results are reasonable as the role of Fe in P sorption has been well established (Bai *et al.*, 2017; Yan *et al.*, 2017; Cui *et al.*, 2018) and as such higher levels of M3 Fe should be expected to result in a lower EPC. The consistency of this relationship between the various analysis, and its significance, suggests that M3 Fe is an important influence on the EPC at Delta Marsh which is in agreement with the literature for calcareous soils and coastal wetlands (Zhang *et al.*, 2005; Bai *et al.*, 2017).

Both M3 Fe and M3 Mg are well established to contribute to P sorption capacity and as such could be expected to exhibit similar relationships to the sorption parameters and EPC however this is not what was observed. The relationship between EPC and M3 Fe can be potentially illuminated by looking at the P fractionation. The $NaHCO_3$-P and NaOH-P are the P fractions most commonly associated with Fe and unlike the non-labile fractions associated with Mg these represent the labile and semi-labile P pools in the sediments. The $NaHCO_3$-P in particular is a readily desorbable form of P which, due to resuspension caused by bioturbation of Common Carp, is unlikely to be saturated in its ability to sorb P and therefore retains P sorption capacity as indicated by a lower EPC. The $NaHCO_3$-P fraction only represents 8% of P in the sediments (Figure 1.3) which supports this conclusion. The NaOH-P is less labile and understandably represents a higher fraction of the sediment P (19%, Figure 1.3). If these conclusions are correct then it can be assumed that increasing the sediment stability by reduction of Common Carp activity in the marsh or increased submersed vegetation cover, both expected outcomes of the RTT Common Carp exclusion project, would in turn result in an increase in P sorption via the available sediment Fe contributing to decreased P eutrophication in the marsh (Bai et al. 2017). It is suggested therefore that continued, or follow-up, monitoring of the wetland sediment nutrients and cations would be beneficial to assess the impacts of the ongoing management which may help validate the project.

The EPC and M3 calcium were significantly positively correlated in the emergent zone with the inclusion of Lake Francis, as well as positively correlated non-significantly across all the other landscape position analysis conducted although in most cases quite weakly.

With regards to the significant correlations between EPC and M3 calcium, this relationship is well established; especially in calcareous soils (Frossard *et al.*, 1995; Ige, Akinremi and Flaten, 2005; Zhou *et al.*, 2011). The positive relationship between EPC and sediment Ca was, similar to M3 Mg, unexpected as more Ca should increase the sorption capacity and therefore decrease EPC. This effect of Ca however was consistent across the other sorption parameters as higher Ca correlated to lower S_{max1}, $S_{max2,}$ S_{maxT}, and PEBC while also higher S_n, K_1 and DPS, all significantly. These results would indicate that higher levels of Ca result in lower sorption capacity (S values and PEBC) and higher levels of current P sorbed (S_n and DPS). In addition, the P fractions typically associated with Ca, Residual-P and HCL-P, represent the majority of the P currently sorbed in the sediments. These relationships seem to indicate that Ca does greatly influence the sorption capacity of the sediments as higher levels of Ca correspond to higher rates of P already sorbed, which in turn can result in a lower P sorption capacity to remain due to the existing P load. Sorption relative to sediment Ca has been identified to represent the longer-term P sorption and retention in wetland sediments (Tercero *et al.*, 2017; Kowalczewska-Madura *et al.*, 2018), and as such it could be expected that the large amount of Ca-bound P has the potential to be sequestered long term.

The fact that higher concentrations of Ca do not result in a lower EPC, along with the above observations, may be indicative that the Ca within the marsh sediments is relatively saturated with P. The relationship between Ca and the K values supports this conclusion as K_1 is significantly positively correlated with Ca possibly indicating that more Ca present at a site corresponds to more of the lower-energy binding sites being occupied thus leaving only higher energy sites signified by a higher K_1. When considering K_2 a similar relationship is observed in

the emergent and wet meadow zone sites. In the open-water sites Ca and K_2 are significantly negatively correlated however Ca concentrations are higher in the open-water sites relative to the emergent and wet meadow zone sites and as such in those locations the Ca may not be as saturated resulting in available sites for P sorption and therefore the negative correlation that was observed. With the K_1 values being lower than the K_2 values it is reasonable that K_1 would exclusively be driven upwards by increased Ca saturation while K_2 (already representing higher energy sites) would react relative to the abundance of Ca at the site. If the Ca in the marsh sediments is in fact largely saturated with P this suggests that Ca is very effective at binding P within this system.

The EPC was also positively correlated with H_2O-P across the open-water sites (the only landscape position where fractionation was conducted). This suggests that higher levels of the easily desorbed H_2O-P relate to an increased chance of P desorption which could be as expected. Between the EPC and the $NaHCO_3$-P (the other labile P pool) there is only a weak positive correlation however if the Waterhen Bay site were to be excluded the relationship becomes significantly positive, which would be expected similarly to the H_2O-P. The H_2O-P only represents up to 1% of the P within the wetland sediments however (and 8% for $NaHCO_3$-P) so these relationships are not likely to correspond to a large amount of P desorption taking place. Additionally, the EPC is generally positively correlated to all of the P fractions, though not significantly, as these fractions represent current P loading in the sediments and therefore decreased capacity for additional P sorption.

The PEBC was found to be significantly positively correlated in the emergent and wet meadow zone sites with sediment OM while in the open water there was no apparent

relationship detected. It would be expected that the PEBC and OM would be positively correlated as OM, and the metal cations commonly associated with OM, is one of the main influences of the P sorption taking place in the sediments. The OM was significantly lower in the open-water sites compared to the emergent and wet meadow zone sites which could possibly explain the absence of a clear relationship in the open-water. It is unclear if the lower amount of OM is resulting in the positive relationship not being able to be detected or if OM is having a reduced role in P sorption in the open water where other variables may be having more significant effects. Overall, it appears that higher concentrations of OM in the sediments promote a higher PEBC. As a higher PEBC signifies a greater ability for sediments to mitigate inputs of P and therefore eutrophication, increased levels of OM in the marsh sediments should be encouraged.

Assessing the PEBC across the marsh it was found to be significantly lower in the wet meadow zone relative to the other positions as well as lower in the east marsh relative to the other units. This is similar to the results for the S_{max} parameters which are likely responsible for these differences in the PEBC. The wet meadow zone exhibited significantly lower levels of M3 Fe, and M3 Ca which are all known to be important to P sorption capacity, particularly in calcareous soils (Tercero *et al.*, 2017; Cui *et al.*, 2018; Kowalczewska-Madura *et al.*, 2018), which could potentially help explain the reduced sorption and buffering capacity of these sites. The wet meadow zone also exhibited significantly higher levels of TP, M3 P, OP, and DPS, which all indicate a potentially higher level of P loading in these sediments and are negatively correlated to the PEBC. It has been established that a higher background level of P in sediments can result in a lower sediment sorption capacity (Cui et al. 2018), and as such the higher levels

of P loading in the wet meadow zone could help explain the depressed buffering capacity of that zone.

4.2: Sediment Chronology and Nutrient Accumulation

The core samples used in this study demonstrated considerable amounts of disturbance and sediment mixing, which somewhat limited their use for analysis. This was not unexpected as discussed in Section 1.2.2. Although dates were determined for sample segments on most cores these should be treated as approximate as opposed to definite. This high degree of sediment disturbance is however informative in its own right as to the conditions in certain regions within the marsh. This is particularly true for Canvasback Bay and Weedy Bay (Figure 2.2 and 2.3) where the extensively disturbed sediments, compared to the rest of the marsh, support the idea that the Portage Diversion exerts a considerable influence. Similarly, the relatively lower degree of sediment disturbance in center marsh and Lake Francis demonstrate the impact of Common Carp on the rest of the marsh, and suggest similarities between these two bodies of water due to their absence of Common Carp activity.

There are two sites, Canvasback Bay and Weedy Bay, that demonstrate different historic profiles relative to the rest of the marsh that are erratic, highly disturbed, and generally do not demonstrate any clear pattern for most parameters. This is not unexpected given that these sites are directly situated on of the flow path between the Portage Diversion failsafe and Lake Manitoba. As a result, sediment profiles at these sites are heavily impacted and are omitted as they are not expected to be in agreement with the rest of the sample sites.

4.2.1: Changes in Phosphorus and Sedimentation

The sediments in a wetland such as Delta Marsh create a record that can be used as a proxy to estimate its conditions over approximately the last century and reveal information about how it has changed. Despite difficulties in obtaining chronologic determinations for a system with such high degrees of sediment disturbance (Table 2.1), it was clear that the rates of accumulation for sediment depth and mass, TP, OP, BAP, TN, and OM have all increased considerably throughout the marsh over time (Figures 2.11-2.26). These increasing physiochemical accumulation rates support that Delta Marsh has long been a eutrophic system to some degree. In the lowest depths of the sediment cores the accumulation rates were quite low and relatively flat indicating that in the early 1900s nutrient levels were increasing very slowly, if at all. Around approximately 1960 the accumulation rates began increasing exponentially across the marsh, indicating that major changes were taking place and eutrophication was accelerating. Pre-1960 the levels of nutrients and sedimentation were increasing, but only gradually. The accumulation rates noticeably accelerated a second time around approximately 1995, with increases that appear to be greater than those seen in 1960.

The proportion of TP being represented by BAP (BAP%ofTP) and OP (OP%ofTP) have also been increasing overall indicating that these two P species are increasing faster relative to the TP, changing the P dynamics within the marsh (Table 2.2). The increase of the BAP%ofTP and OP%ofTP has also been accelerating in more recent sediments with considerable increases in the top approximately 10 cm, representing roughly the 1960's-1970's indicating the rate of redistribution of P species is increasing.

The BAP generally represents labile forms of P (significantly correlated to $NaHCO_3$-P and NaOH-P) that are more readily desorbable and available for uptake by plants (Bai et al. 2017), so the increasing amounts of BAP in the marsh sediments may contribute to P desorption, marsh eutrophication, and therefore algal blooms (Reddy et al. 1999; Reddy et al. 2011; Nicholson 2012). The OP may be present in any of several different forms, some labile and some non-labile, and the analysis to differentiate between OP species was not conducted. Depending on form the OP can be an important and readily available P source for wetland plants, (Lenzi and Renzi 2011) especially in the presence of plant roots (abundant at Delta Marsh) that can release organic acid anions and mobilize the more resistant forms of OP (Min Li et al. 2013). It is also possible that what appears to be OP increasing in more recent sediments is instead, in whole or in part, explained instead by OP decreasing in deeper sediments due to being transformed into inorganic P by soil bacteria over time (Min Li et al. 2013). Increases in labile forms of P could help explain the unsaturated sorption capacities determined across the marsh in Section 4.1, as losses of easily-desorbed sediment P may be ongoing due to sediment resuspension.

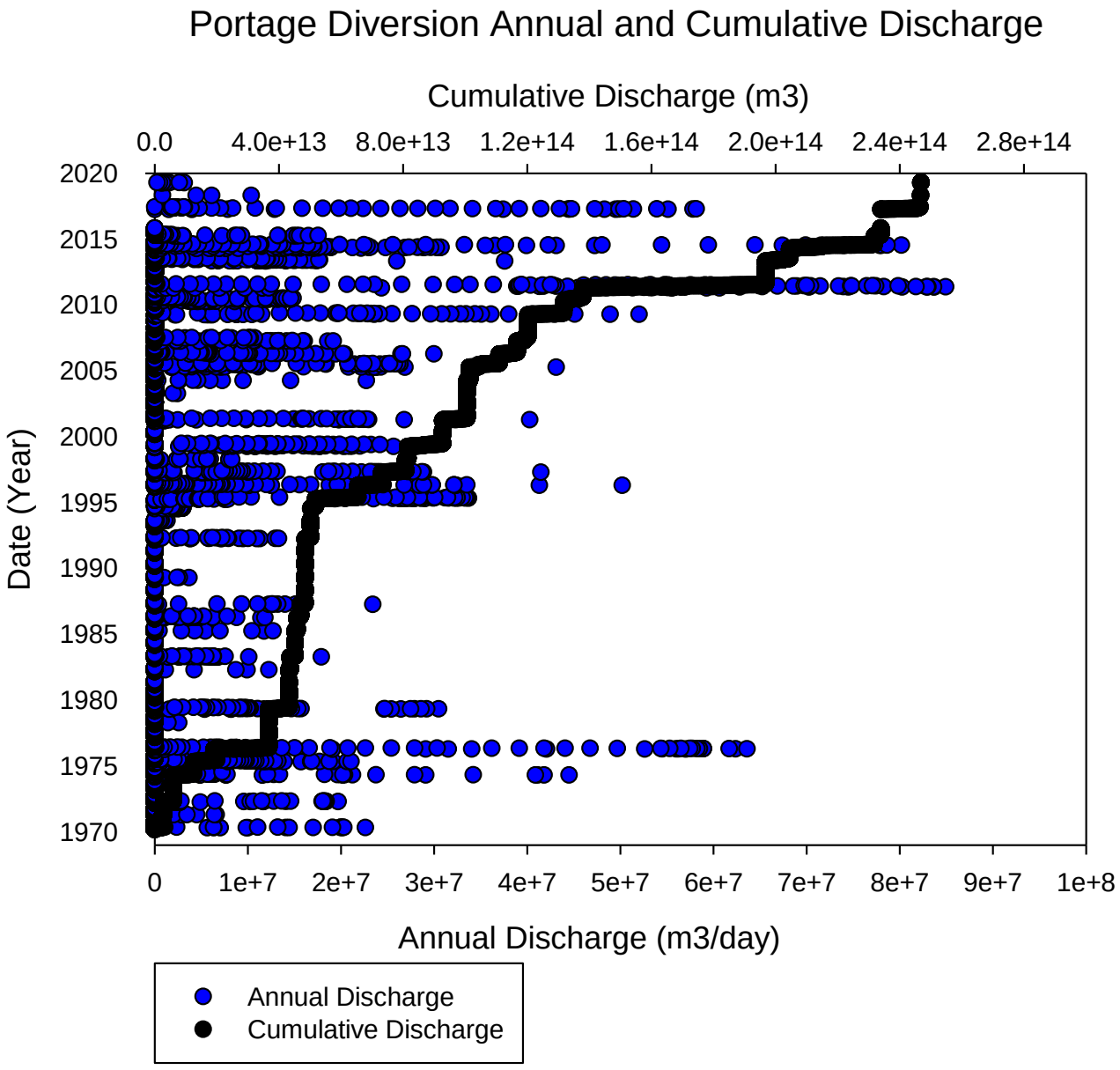
Investigation into the classic molar Redfield ratios of TN/TP (Table 2.3) suggest that center marsh, Lake Francis, and Lyttle Bay are likely to be P limited, the west marsh and Cadham Bay may be N limited, and Waterhen Bay is close enough to the benchmark value of 16 that it is unlikely to be limited by either nutrient specifically (Khan et al. 2007; Kinsman-Costello et al. 2014; Ansari et al. 2011). These ratios suggest that the west marsh and Cadham Bay are a particular risk for algal blooms due to an excess supply of sediment P.

The increasing eutrophication, P species redistribution, and TN/TP ratios support the suggestion that increasing algal blooms in recent years may be driven by eutrophication of the marsh, especially for the west marsh unit and Cadham Bay. It was determined in Section 4.1 that the marsh sediments are generally acting as a P sink (indicating internal loading isn't an important factor for eutrophication) for the marsh with considerable sorption capacity remaining, which suggests that the eutrophication of the marsh may increase in the future if nothing is done to change the course for this historic wetland.

The Portage Diversion is suggested to be a major source of nutrients and sediment to the marsh due to the high levels of each transported to Lake Manitoba on years when it operates (Nicholson 2012). The west marsh receives considerable amounts of water from the Portage Diversion on years that it breaches the failsafe, and Aminian (2015) determined that large quantities of water also enter into the east marsh. The influence of the Portage Diversion is also supported by comparing the timeline of its operation to the accumulation profiles in the sediments. The Portage Diversion commenced operation in 1969 which is similar to the beginning of the exponential increases seen in the physiochemical accumulation profiles, considering that the dates identified are approximate. Similarly, the Portage Diversion saw a considerable increase in both the frequency of use and volume of water discharged around 1995 (Figure 2.27), when the physiochemical profiles also saw an increase. Over the first 25 years of operation (1969-1994) the Portage Diversion discharged approximately 5-billion m^3 of nutrient-rich floodwater to Lake Manitoba and Delta Marsh while in the second 25 years of operation, during the time period of the most considerable eutrophication of the marsh (1994-2019), the discharge was approximately 20-billion m^3 (Figure 2.27). Although not definitive

proof this strongly suggests that the Portage Diversion discharge may be having a considerable influence on Lake Manitoba and Delta Marsh nutrient levels.

Figure 2.27: Annual and cumulative discharge from the Portage Diversion to Lake Manitoba (Figure 1.1) from beginning of operation in 1969 to April 2018



The increasing rates of nutrient accumulation in the marsh are also likely due to the activity of agriculture, as 75-96% of the surrounding area is utilized for agriculture (Stanley 2017). Due to the depletion of soil nutrients, in 1960 large quantities of P-rich fertilizer began to be used across the Delta Marsh watershed (Goldsborough and Suggett 2015). The application

of these fertilizers has increased since that time due to expanding agriculture and continues to the present day. The 1960s is also when residents first began to report deterioration in the quality of the marsh (Wrubleski et al. 2016), and agrees with the beginning of the exponential increases in the physiochemical accumulation profiles. As with most fertilizer applications, not all nutrients applied are utilized and the excess accumulates in surface soils. Similarly livestock, who have been a feature of the landscape since settlement began, only incorporate 20-40% of the P from their food into biomass while the rest is returned directly to the soil via their waste (Ironside 2001). The N/P ratio of manure is 1-4:1 while uptake by crops and pasture lands is 8:1, resulting in a buildup of excess P in the soils (Zhang et al. 2005). These nutrient rich soils can be exported downstream due to erosion or runoff during precipitation events, find their way into the marsh, and contribute to eutrophication (Sui and Thompson 2000; Sharpley et al. 2011). A study by Stanley (2017) determined that watersheds associated with the west marsh and center marsh export more N and P to Delta Marsh than the watershed associated with the east marsh, which matches the degree of agricultural development across the watershed and supports that agriculture is an active and ongoing contributor to eutrophication of the marsh. Stanley (2017) also suggested that grazing lands, such as those more commonly found surrounding the east side of the marsh, export very little P to the marsh. This study, however, primarily focused on nutrient loading via tributaries and due to runoff and did not quantify more direct additions. Livestock congregating around the Delta Marsh shoreline has been a consistent feature since early settlement (Goldsborough and Suggett 2015), where they can contribute to eutrophication through the direct input of waste (Zhang et al. 2005), or through hoof action

that increases shoreline erosion and adds nutrient-rich particulate matter to the marsh (Hongthanat 2010).

While the effects of marsh deterioration were noticed in the 1960s it likely began some time earlier in order to have advanced to the stage where it would be noticed and generate complaints. As the 1960s were when both the Portage Diversion began operation and P fertilizer application began, the eutrophication and subsequent deterioration pre-1960 must be due to another source. Early settlement of the region that saw the establishment of farms and grazing lands that would have exported nutrients to the marsh, construction of homes and vacation properties that required the disposal of waste, commercial and recreational ventures including the first major public beach in Manitoba, and even a railway connecting the marsh to the rest of civilization (Goldsborough and Suggett 2015). The potential influence of the Delta Beach settlement and associated impacts of on-site sewage treatment systems is supported by the significantly increased levels of M3-P found in center marsh. These early activities may account for the slow, but present, increases in sediment P recorded in the deeper sections of the core samples.

Nutrient increases pre-1960 (and post) may also be due to the activity of Common Carp. Common Carp have been found to increase nutrient levels in a system due to the excretion of waste (Hertam 2010; Badiou and Goldsborough 2010), increasing shoreline erosion of nutrient-rich sediments (Parks 2006), or the uprooting and subsequent decomposition of submersed and emergent vegetation. The first confirmed record of Common Carp in Delta Marsh was in 1952 however they may have been present prior and were confirmed in Lake Manitoba in 1947 (Atton 1959). Badiou (2005), Hnatiuk (2006), Parks (2006), and Hertam (2010) investigated the effects of Common Carp introduction and

removal from marsh systems and determined that the negative impacts of the fish start almost immediately once they are present in a system. From this we can surmise that the marsh would have been experiencing the negative influence and eutrophication of Common Carp as early as 1947-1952.

4.2.2: Differences in the Changes Seen Across the Marsh

The observed changes in nutrients and sedimentation were not consistent across the marsh, likely due to different influences acting on the various marsh units. The west marsh unit demonstrated the greatest differences relative to the rest of the marsh which can be primarily attributed to the influence of the Portage Diversion. Considering the sediment inventories, all sites in the west marsh have elevated levels of $^{210}Pb_{ex}$ of 11 790, 5 028, and 3 851 Bq m^{-2} for Canvasback Bay, Weedy Bay, and Big Lake, respectively (Table 2.2). Compared to the marsh average (4 112 ±2 833 or 3 037 ±330, including or excluding Canvasback Bay and Weedy Bay) this indicates considerable additional sedimentation at these sites. This is particularly remarkable as the inventories in Canvasback Bay and Weedy Bay are likely underestimates as, unlike the rest of the marsh, the bottom of the $^{210}Pb_{ex}$ and ^{137}Cs activities were not contained within the cores. This deposition is most likely due to the Portage Diversion, especially considering highest inventories were in Canvasback Bay and Weedy Bay, with lower values in Big Lake. Furthermore, the erratic nature of the profiles, the buried ^{210}Pb maximum (Anderson et al. 1987) and the nature of the ^{137}Cs profile (Radakovitch et al. 1999) indicate a high amount of disturbance for these sediments such as vertical mixing, resuspension, or horizontal sediment mobility (Edgington et al. 1991). Although these factors are all likely influencing the west marsh, the high sediment inventories suggest that overall the excessive sedimentation is the most influential factor affecting west marsh.

The west marsh had the highest accumulation rates for all parameters (represented by Big Lake, as Weedy Bay and Canvasback Bay were not determined), of all sites across the marsh and Lake Francis (Figure 2.19). The TPAC, BAPAR, and MAR were particularly higher in Big Lake with levels double-to-triple those found across the rest of the sites. The BAP (both measured and as a percentage of TP) was also the highest in the west marsh (Table 2.3). Knowing the suspected magnitude of influence the Portage Diversion has on the west marsh, and the high amounts of nutrients and sediment it carries, it is likely to be a major contributor to these high physiochemical accumulation rates. The high BAP in the west marsh may also be due to the bioturbation of Common Carp in that region of the marsh, as sediment disturbance can shift P to more labile forms (Reddy et al. 1999; Cavalcante et al. 2018). The west marsh was the only place across the marsh or Lake Francis where TN/TP decreased (Table 2.4), had the lowest values compared to the other units, and was firmly N limited. Together these results suggest that the west marsh experiences the worst eutrophication and validates observations that it has the highest frequency and severity of algal blooms.

Center marsh and Lake Francis were the two study units that demonstrated results and sediment conditions that were the most similar. This difference from the rest of the marsh was likely due to the isolation of these sites from Lake Manitoba, resulting in the absence of both the influence of water from the Portage Diversion and Common Carp. These two study units demonstrated the best (least disturbed and closest to ideal) chronologic (Figures 2.1-2.10) and physiochemical accumulation profiles (Figures 2.11-2.17 and 2.19-2.26) which can likely be attributed to the absence (or reduced) bioturbation from Common Carp. The absence of Common Carp may also explain the OM at these sites as sediment disturbance such as

bioturbation results in decreases in OM (Lenzi and Renzi 2011; Reddy et al. 1999); Lake Francis had significantly higher levels of OM compared to the rest of the marsh, and center marsh demonstrated the highest accumulation rates for OM. Similarly, these sites had the highest OP%ofTP, particularly in sediments near the surface where levels were nearly double those in other sites (Table 2.4). The higher levels of OP found in these sites can likely be attributed to the higher OM and sediment stability. As OP is usually found in less-labile forms (Min Li et al. 2013), this means these more isolated sites are less likely to desorb P. This may also help explain the significantly higher levels of Residual-P found in Lake Francis (Min Li et al. 2013). The absence of any influence from the nutrient-rich waters of the Portage Diversion may explain why these sites had the overall lowest accumulation rates for sediment and all forms of P (Figures 2.11-2.17 and 2.19-2.26). The TPAR and MAR were particularly low in these isolated sites. Lake Francis demonstrated the lowest TPAR across all sites, including center marsh, which may be due to the suspected influence of sewage in center marsh that results in the elevated levels of some forms of P in that unit.

These more isolated sites also had the highest sorption capacity (Tables 1.7-1.8) which may be attributed to the absence of both Common Carp and the influence of the Portage Diversion. Sorption capacity can be reduced by both the bioturbation of Common Carp through desorption, and by nutrient loading via an increased number of sorption sites being occupied. These two study units also demonstrated the highest TN/TP ratios (Table 2.4), indicating they were the most likely to be P limited as opposed to N limited (Ansari et al. 2011), once again likely due to the absence of both Common Carp and the Portage Diversion. Considering all of the results for these isolated sites together it is likely that these study units will be at the lowest

risk for P desorption, and subsequently algal blooms. This suggests that decreasing the activity, and influence, of Common Carp throughout the marsh will likely promote more favorable conditions in the marsh including decreased sediment P desorption, and conversely greater P sequestration. This is supported by a study by Badiou (2005) that determined the approximate density of Common Carp expected within Delta Marsh of 400 kg ha^{-1} (at the time) could increase internal loading of P levels in the water column by as much as 66%.

Interestingly, Lyttle Bay demonstrated many results that were similar to the more isolated center marsh and Lake Francis, different from the rest of the east marsh. Lyttle Bay had similarly high OP%ofTP (Table 2.3) and TN/TP (Table 2.4), and low sediment profile disturbance. This was unexpected as Lyttle Bay is exposed to the activity of Common Carp, unlike the isolated sites, and potentially waters from the Portage Diversion. Of the sites in the east marsh Lyttle Bay would be the site least influenced by the Portage Diversion; Aminian (2015) determined that counterintuitively any water from the Portage Diversion that would reach Lyttle Bay would do so via entering the east marsh through Clandeboye Channel on the far east end and then working its way back west towards Lyttle Bay. With this considerable and winding distance (Figure 1.1) it is possible that the Portage Diversion has very little influence on Lyttle Bay. The Common Carp similarly enter Delta Marsh each spring after over-wintering in Lake Manitoba and as Lyttle Bay is somewhat removed from any lake-marsh connection it may receive less Common Carp activity, although this is only speculative. A final possibility for the results found at Lyttle Bay may be that it is somewhat more sheltered from wind action relative to Cadham Bay and Waterhen Bay, which would result in less sediment disturbance and possibly similar conditions to the disturbed sites.

Cadham Bay, like the west marsh, demonstrated results that were very different from the rest of the sites, although for different reasons. Cadham Bay is the site outside the west marsh most impacted by the nutrient-rich waters of the Portage Diversion (Aminian 2015), however it would not be impacted physically like the west marsh, or experience any sediment deposition. This is because the majority of the water from the Portage Diversion that enters the east marsh does so though Delta Channel, directly into Cadham Bay, and remains there as little water leaves Cadham into the rest of the east marsh (Aminian 2015). In some instances, the water flowing into Cadham Bay was comprised of as much as 40-50% water from the Portage Diversion which may account for the higher levels of S_n, TP, and BAP relative to Waterhen Bay, which is similarly connected to Lake Manitoba (Figure 1.1) but receives much less water from the Portage Diversion. With the influx of nutrient-rich waters from the Portage Diversion it would be expected that Cadham Bay would demonstrate higher physiochemical accumulation profiles like the west marsh. However, the physiochemical profiles at Cadham Bay are instead the lowest seen anywhere in the marsh, suggesting that there is another factor exerting a greater influence on the site.

The main influence on Cadham Bay, not see elsewhere in the marsh, is likely due to wind action. Cadham Bay has the greatest wind fetch across the marsh (Stanley 2017) and generally experiences the most severe wave action (personal observation). This is supported by the $^{210}Pb_{ex}$ and ^{137}Cs inventories as Cadham Bay has the lowest values of any site across the marsh (1 809, 376 Bq m^{-2}, respectively; Table 2.2), much lower than the averages even with the exclusion of Canvasback Bay and Weedy Bay (3 037 ±330 and 885 ±262, respectively; Table 2.2). These low inventory values suggest that there has been considerable horizontal movement and

removal of the sediments (Edgington et al. 1991). This agrees with other unusual characteristics of the site, such as the shallow radionucleotide profile depths and shallow compressed ^{137}Cs peak. The low physiochemical parameters can be explained due to this influence of wind; the SAR and MAR are lower due to the removal of sediment from the site, while the TPAR, OPAR, and BAPAR may be lower due to increased rates of desorption caused by the sediment resuspension (Koski-Vähälä and Hartikainen, 2001; Li *et al.*, 2016; H. Zhang et al. 2016). The BAP%ofTP at Cadham Bay was also the lowest on average (Table 2.3) which can be similarly explained as BAP is labile and easily desorbed. Cadham Bay also had a decreasing and low TN/TP, similar to that of the west marsh, and would similarly be expected to be N limited. The activity of Common Carp would likely also contribute to the sediment disturbance at Cadham Bay, but is likely less influential compared to the effects of wind. Common Carp are present throughout the east marsh but these levels of sediment removal were only observed at Cadham Bay

Considering the differences found across the marsh it could be suggested that sediment disturbance was an important factor regulating the accumulation rates of the physiochemical parameters. Each site was subjected to different, and often multiple, sources of sediment disturbance and the definitive determination of which are the most influential is not possible with the data obtained in this study. It appears, however, that the influence of the Portage Diversion is significant and results in increasing physiochemical accumulation profiles. Conversely, sediment disturbance (and removal) caused by wind action results in decreasing physiochemical accumulation profiles. The influence of Common Carp on the accumulation profiles is difficult to determine as it was likely overshadowed in the west marsh by the Portage

Diversion, overshadowed in Cadham Bay by wind action, and absent in the center marsh and Lake Francis. Considering the results from the isolated sites compared to the rest of the marsh it is could be suggested that the activity of Common Carp results in lower OM, OP and TN/TP ratios. It is clear though that the presence of Common Carp results in considerable intermixing between sediment layers and disruption of the sediment chronology profiles.

5: Conclusions

5.1: Objective 1

My first objective was to determine the current P sorption capacity of the surficial sediments as well as their ability and likelihood to act as a sink or a source of P to the water column. The primary method of determination was comparisons of the sediment EPC values with recorded levels of TDP from the water column which indicated that seven of the eight sample sites act as a P sink for the majority of the study period. Northeast Eaglenest and Cadham Bay were never a P source, Big Lake, Weedy Bay, Northwest Eaglenest and Lyttle Bay were all a P source one time (12.5-20.0% of samples), and Canvasback Bay was a source for three of the seven sample dates (42.9%). Waterhen Bay was always a source for P, possibly due to sediments with a greater proportion of sand compared to the rest of the marsh, as well as lower TDP values potentially due to a higher influence from Lake Manitoba. Results for the emergent and wet meadow zone sites were similar except presented a greater frequency of acting as a P source which may be attributed to changing redox conditions due to intermittent flooding or increased P loading from the surrounding landscape.

Investigation into the sediment sorption capacity, degree of P saturation, and forms of P fractionation agreed with these conclusions and actually made a stronger argument for the marsh to be a P sink. The sorption capacity and degree of P saturation identified that there was considerable potential capacity for the sediments to sorb additional P compared to the current levels. The P fractionation identified that only 9% of sediment P was in forms that would be considered labile and at risk for desorption. The results for the P fractionation were: 53% HCL-P, 19% both NaOH-P and Residual-P, 8% $NaHCO_3$-P, and 1% H_2O-P

My hypothesis was that the sediments are acting as a P sink within the marsh and was supported.

5.2: Objective 2

My second objective was to determine how the P forms and concentrations vary across the marsh and attempt to explain why. The sorption capacity parameters were determined to be lower in the wet meadow zone and higher in the open-water as well as lower in the west and east marsh units and higher in center marsh and Lake Francis. The S_n was lower in the open-water compared to the emergent and wet meadow zones as well as lower in the east marsh unit overall. The EPC was lower in the west and center marsh units overall while also lower in the open-water sites across the marsh. The DPS was higher in the west and center marsh units and also increased moving from the open-water to the wet meadow zone. The TP, M3 P, OP, and BAP were generally similar and were lower in the open-water sites and higher in the wet meadow zone. They were also higher in Lake Francis. The organic matter generally

increased from west to east across the marsh with larger increases in the emergent and wet meadow zones.

The variables that were found to be the most influential within the marsh were the organic matter, and sources of disturbance that can cause sediment resuspension such as the Portage Diversion, wind action, and bioturbation by Common Carp. The OM was significantly correlated to M3-P, M3-Al, M3-Mg, TP, TN, OP, TC, and the residual-P fraction and was demonstrated to influence the distribution of all these variables. The M3 Fe was found to be an important factor for P sorption across the different landscape positions. The Portage Diversion had a strong influence on the west marsh and Cadham Bay for their nutrient loading, sediment disturbance and transport. Resuspension of sediments due to disturbance was found to result in lowering OM, reducing labile forms of P, causing sediment P desorption (contributing to eutrophication) and increasing the difficulty of interpreting sediment cores.

My hypothesis was that the P concentrations will be greatest in the west marsh and decrease towards the east, while being highest in the wet meadow and lower in the open-water. When considering S_n my hypothesis was supported. Considering the TP, M3 P, OP and BAP they were lower in the open-water and higher in the wet meadow; however, between marsh units did not demonstrate a consistent pattern as suspected.

5.3: Objective 3

My third objective was to determine if and how the historic rates of sedimentation and nutrient accumulation have changed over time. It was determined that sedimentation and nutrient accumulation has increased exponentially across the marsh over approximately the

last century. The increases observed were slower in the older sediments with the majority of the increases taking place post approximately 1960, and increasing again post-1995. Sediment profiles indicated a high degree of disturbance and sediment mixing throughout the profiles. Canvasback Bay and Weedy Bay had been exposed to such a high degree of disturbance, attributed to the Portage Diversion, that these profiles were largely unusable. The potential sources for nutrient accumulation to Delta Marsh were the Portage Diversion, agricultural activity in the watershed, and the residential development of Delta Beach.

It was also determined that the proportion of P in the sediments represented by organic P and biologically available P has been increasing over time across the marsh with much larger increases occurring since roughly the 1960s to 1970s. Investigation of TN/TP ratios suggested that center marsh, Lake Francis, and Lyttle Bay are likely to be P limited, the west marsh and Cadham Bay may be N limited, and Waterhen Bay was neither. Collectively these results demonstrate that Delta Marsh is an increasingly eutrophic system with a particular risk for algal blooms in the west marsh.

The west marsh unit demonstrated considerably higher levels of sedimentation and P accumulation compared to the other marsh units which was attributed to the influence of the Portage Diversion. Center marsh and Lake Francis demonstrated similar results to each other with the clearest sediment profiles, lower accumulation rates, and highest accumulation of OM and OP. The changes in center marsh and Lake Francis were attributed to these areas being isolated from Lake Manitoba, Common Carp and the influence of the Portage Diversion. Cadham Bay demonstrated the lowest accumulation profiles which was attributed to the removal of sediment due to the action of wind.

My hypothesis was that the overall nutrients and sedimentation levels have increased post-settlement of the region with the largest increases found on the western side of the marsh where the most intensive development has taken place. I also hypothesized that the nutrient accumulations have been the most significant after the opening of the Portage Diversion, with the west marsh being the most impacted. My hypothesis was supported on both points.

6: Recommendations

6.1: Future Research

1. My study suggested that the ongoing sediment disturbance by Common Carp may be resulting in the desorption of labile sediment P. As there is currently a Common Carp exclusion project taking place at Delta Marsh it would be useful to asses the impact of reduced carp activity on the marsh sediments. I would suggest that there should be future assessments of the P levels in the sediments including:
    a. Ongoing monitoring of sediment P to asses the potential changes. Higher levels of sediment P post-exclusion would indicate that sediments are sorbing and sequestering greater amounts of P, validating that the Common Carp exclusion is working to improve conditions in the marsh.
    b. A follow-up study reassessing the surficial sediments for sorption capacity and EPC, following several years of successful Common Carp exclusion. In addition to building on the goals in point 1a, this would allow an assessment of the changes in maximum sorption capacity and EPC in the sediments in response to the exclusion project. This may be important as if the sediments sorb and retain greater amounts of P, this

increased sediment loading may increase the EPC and therefore likelihood that the sediments could act as an internal P source to the marsh. The response of EPC and sorption capacity to Common Carp exclusion may be useful in informing management decisions for other systems where exclusion projects (or other sediment stabilization) may be considered.

2. Continuation of my study via the collection and analysis of additional sediment cores from across the marsh to expand the dataset may be beneficial in a number of ways. Future sediment cores would be able to be used to build on this existing dataset as beyond the surficial sediments there should not be significant changes.

    a. Expansion of the dataset would allow for a more robust analysis of the differences between sections of the marsh, as well as over time. The limited sample size of this study limits the statistical ability to draw definitive conclusions. Furthermore, the differences observed in the pairs of cores from center marsh and Lake Francis demonstrated that there can be considerable difference within a single bay; additional cores would validate the ones currently obtained from the other bays, where only one sample was taken. Core samples taken from bays already sampled, for validation purposes, may not require the analysis of as many sample segments which would assist the financial viability of such a study.

    b. Conducting P fractionation along the depth of sediment cores, as opposed to only the surficial sediments, would allow for the assessment of how the P speciation has changed over time, and would provide additional insight to the historical changes in marsh P. This study has identified several distinct periods of change in the sediment

P (pre-and-post-1960, post-1995) so as a cost-saving measure only selected core segments from these periods may need to be analyzed (P fractionation is one of the more expensive analysis).

c. Conducting particle size analysis along the depth of sediment cores, as well as surficial sediments across the marsh, may provide useful information about the sorption dynamics across the marsh and how/why they may have changed over time. It was speculated that a main reason Waterhen Bay was a P source was due to the greater proportion of sand in the sediments, so further information about sediment composition across the marsh may be useful.

d. The collection of deeper sediment cores at Canvasback Bay and Weedy Bay (possibly also Big Lake) may provide additional insight as to the impact of the Portage Diversion on the west marsh. These two sites were the only locations where the bottom of the $^{210}Pb_{ex}$ and ^{137}Cs profiles were not contained within the top 50-cm.

e. The study in point 1b above may easily be done alongside this one to streamline the analysis and reduce overall costs.

3. In my study I suggested that the Portage Diversion was one of the main contributors of nutrients to Delta Marsh, based on the information that I had available. It would be useful to further support this suggestion by quantifying other sources of nutrients to Lake Manitoba and Delta Marsh. The discharge and nutrient export for the Whitemud River, and other inputs, to Lake Manitoba should be determined and compared with the rates of nutrient accumulation in Delta Marsh. Depending on if the increases in discharge of the

Portage Diversion is also represented in other potential sources, this would help to support or disprove my suggestions.

6.2: Recommendations to Managers

This study suggested that disturbance of the marsh sediments may result in an increase in the release of P from the marsh sediments, contributing to eutrophication of the marsh. As a result, any efforts to stabilize the sediments and reduce sediment resuspension would be beneficial in combating eutrophication and therefore support the health of the marsh. Marsh sediment stabilization is one of the expected results of the Common Carp exclusion project currently underway at the marsh, and as such I would suggest that every effort be taken to support this project and ensure its ongoing effectiveness.

The sediments at Delta Marsh would currently be expected to work towards lowering nutrient levels in the water body, reducing eutrophication, and improving marsh health. The most effective way to support this action would likely be through reducing the input of new nutrients to the marsh. This could be achieved through working with landowners within the watershed, particularly farmers, to reduce the nutrients being exported from their lands. It should also be understood that the operation of the Portage Diversion, particularly to the degree that it overflows and breaches the failsafe, will likely result in increasing nutrient levels within the marsh.

It is suggested that the residential development of Delta Beach may be adding nutrients to Delta Marsh due to sewage leaking from aging septic systems. If true, this could represent a considerable, and entirely preventable, source of eutrophication that is negatively affecting the

health of the marsh. As a component of determining and quantifying the sources of nutrient pollution to the marsh these septic systems should be assessed. This could be accomplished in multiple ways; however, the simplest may be implementing record-keeping requirements for either sewage-removal businesses operating in the area, or the residences themselves. Properties that receive these services less often (or never) would likely represent cases where septic systems are not operating to the standards that they should.

This study determined the sediment and water column nutrient levels that represented the inflection point between the marsh sediments acting as a sink or source of P to the water body (EPC). In setting targets for nutrient levels in the marsh, it would be unrealistic to choose levels that are below this EPC value. Below this EPC value the marsh sediments would begin to work against nutrient reduction, as opposed to supporting it. Any efforts aimed at nutrient reduction in the marsh would likely have diminishing returns below this point.

I would recommend that moving forward there should be ongoing monitoring of the nutrient levels in both the marsh sediments, and waters. This is especially important as there is currently ongoing management plans taking place at the marsh as a component of the RTT project. Ongoing monitoring of the marsh, and its nutrient levels, are essential to determine if the current management plans are working towards improvements of the marsh. Without monitoring of P levels in the sediment (ongoing or follow-up) it will not be possible to determine if the sediment sorption capacity, and retention, is improving.

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Appendix

Table A1. Coefficient of determination for goodness of fit for the Freundlich and Two-Surface Langmuir isotherm equations for triplicates of twenty-two sediment samples from Delta Marsh (Figure 1.1), Canada.

Site Name	Freundlich	Two-Surface Langmuir
	----------r^2----------	
Big Lake open-water	0.9958	0.9994
	0.9946	0.9985
	0.9941	0.9993
Big Lake Emergent	0.9868	0.9960
	0.9908	0.9995
	0.9850	0.9974
Big Lake Wet Meadow	0.9979	0.9998
	0.9954	0.9998
	0.9956	0.9999
Canvasback Bay open-water	0.9798	0.9930
	0.9826	0.9995
	0.9785	0.9993
Weedy Bay open-water	0.9845	0.9986
	0.9848	0.9983
	0.9842	0.9985
Eaglenest 1 open-water	0.9909	0.9997
	0.9911	0.9990
	0.9887	0.9985
Eaglenest 2 open-water	0.9914	0.9991
	0.9917	0.9974
	0.9972	0.9994
Eaglenest Emergent	0.9913	0.9988
	0.9930	0.9994
	0.9958	0.9997
Eaglenest Wet Meadow	0.9914	0.9998
	0.9898	0.9952
	0.9827	0.9932
Cadham Bay open-water	0.9934	0.9998
	0.9910	0.9997
	0.9861	0.9990
Cadham Bay Emergent	0.9953	0.9997
	0.9869	0.9986
	0.9912	0.9996
Cadham Bay Wet Meadow	0.9960	0.9997
	0.9982	0.9999

	0.9967	0.9997
Lyttle Bay open-water	0.9956	0.9993
	0.9924	0.9991
	0.9924	0.9991
Lyttle Bay Emergent	0.9969	0.9975
	0.9936	0.9987
	0.9968	0.9986
Lyttle Bay Wet Meadow	0.9984	0.9991
	0.9942	0.9996
	0.9906	0.9997
Waterhen Bay open-water	0.9923	0.9993
	0.9938	0.9985
	0.9943	0.9996
Waterhen Bay Emergent	0.994	0.9995
	0.9985	0.9970
	0.9983	0.9988
Waterhen Bay Wet Meadow	0.9992	0.9995
	0.9988	0.9987
	0.9938	0.9976
Lake Francis 1 open-water	0.9876	0.9976
	0.9967	0.9985
	0.9950	0.9994
Lake Francis 2 open-water	0.9950	0.9997
	0.9893	0.9991
	0.9925	0.9995
Lake Francis Emergent	0.9982	0.9979
	0.9984	0.9999
	0.9929	0.9999
Lake Francis Wet Meadow	0.9887	0.9997
	0.9855	0.9991
	0.9847	0.9996

Table A2: Coordinates of sample sites at Delta Marsh (Figure 1.1), Canada, and corresponding marsh units

Site	Latitude	Longitude	Marsh Unit
Big Lake Open-Water	50°10'19.72"N	98°25'49.70"W	West Marsh
Big Lake Emergent	50° 9'16.34"N	98°25'44.27"W	West Marsh
Big Lake Wet Meadow	50° 9'14.77"N	98°25'49.97"W	West Marsh
Canvasback Bay Open-Water	50°10'26.93"N	98°24'51.61"W	West Marsh
Weedy Bay Open-Water	50°10'48.05"N	98°24'14.42"W	West Marsh
Northeast Eaglenest	50°10'19.44"N	98°20'10.76"W	Centre Marsh
Northwest Eaglenest	50°10'24.90"N	98°20'45.28"W	Centre Marsh
Eaglenest Emergent	50° 9'48.76"N	98°20'2.90"W	Centre Marsh
Eaglenest Wet Meadow	50° 9'43.88"N	98°20'4.41"W	Centre Marsh
Cadham Bay Open-Water	50°10'28.94"N	98°17'11.64"W	East Marsh
Cadham Emergent	50° 9'44.08"N	98°16'50.23"W	East Marsh
Cadham Wet Meadow	50° 9'40.18"N	98°16'56.12"W	East Marsh
Lyttle Bay Open-Water	50°10'44.44"N	98°11'35.47"W	East Marsh
Lyttle Emergent	50°10'14.10"N	98°11'34.77"W	East Marsh
Lyttle Wet Meadow	50°10'9.78"N	98°11'32.96"W	East Marsh
Waterhen Bay Open-Water	50°12'57.41"N	98° 6'16.08"W	East Marsh
Waterhen Emergent	50°12'43.07"N	98° 5'47.62"W	East Marsh
Waterhen Wet Meadow	50°12'43.53"N	98° 5'26.79"W	East Marsh
Lake Francis 1 Open-Water	50°18'15.95"N	97°57'52.67"W	Lake Francis
Lake Francis 2 Open-Water	50°18'50.49"N	97°57'33.00"W	Lake Francis
Lake Francis Emergent	50°19'41.09"N	97°56'32.95"W	Lake Francis
Lake Francis Wet Meadow	50°19'45.21"N	97°56'31.20"W	Lake Francis