Treatment Wetland Vegetation Harvesting for Phosphorus Removal in Upper Midwest Agricultural Watersheds

A THESIS SUBMITTED TO THE FACULTY OF
THE UNIVERSITY OF MINNESOTA
BY

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IN PARTIAL FULFILLMENT OF THE REQUIREMENTS
FOR THE DEGREE OF
MASTER OF SCIENCE

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December 2019

Acknowledgements

I acknowledge my thesis committee for their significant guidance throughout the entirety of this work. Thank you to my advisor Chris Lenhart for his willingness to help me learn. Thank you to Joe Magner for encouraging me to seize opportunities. Thank you to Jacques Finlay for his wide array of water quality knowledge and feedback along the way.

I acknowledge the many University of Minnesota staff who provided support along the way. Thank you to Rebecca Swenson for providing me with direct guidance towards my future career path. Thank you to Nic Jelinski for his willingness to answer any and all soil related questions I had. Thank you to Bruce Wilson and Winnie for their assistance in the statistical analyses necessary for this work.

I acknowledge the graduate and undergraduate students that assisted me along the way. Thank you to Masha Hoy for paving the way for this research. Thank you to Austin Yantes, who made graduate school incredibly less terrifying. Thank you to Katelyn Younger who made field days so much better. Thank you to my labmates for being willing to share your own experiences with me that helped me figure things out along the way.

Thank you to my family and friends for their unconditional support and willingness to always let me talk to them about plants.

Abstract

Midwestern wetlands can provide a variety of ecosystem services to the surrounding landscape including nutrient retention. Although wetlands are often times sinks for phosphorus (P), accumulation of P in wetland soils, referred to as legacy phosphorus, can affect water quality when flushed out of a system. The prevention, management, or removal of stored soil P within treatment wetlands can be challenging for land managers, but P may be stored within plant biomass and removed by harvesting. In Minnesota, a small-scale edge-of-farm treatment wetland (Granada treatment wetland), a micro-scale experiment of 30 mesocosm wetland systems (University of Minnesota, St. Paul campus), and a large-scale flood water storage impoundment (North Ottawa Impoundment) were assessed. In the small scale Granada treatment wetland and the St. Paul mesocosm experiments, harvested wetland vegetation was compared in autumn for P retention within the above ground biomass. A wet prairie vegetation mix from the Granada treatment wetland was sampled from each cell within the wetland and various native Minnesota plants were tested in the mesocosms at different times of the year. The monoculture species in the mesocosm experiments ultimately removed more P per biomass than the wet prairie mixes at the Granada treatment wetland. Biomass was not witnessed to be a direct indicator of P removal per the species studied in our experiments. The time of season of harvest and correlated phosphorus content was found to be an indicative factor for phosphorus removal potential. The Granada treatment wet prairie vegetation mix removed phosphorus each season through harvesting in the fall with approximately 2.3 kg/ha removed by vegetation in 2017 and 3.2 kg/ha removed in 2018. From the 2017-2018 mesocosm experiments, both *Schoenoplectus tabernaemontani* and *Scirpus fluviatilis* removed approximately 1.6 g of P per tank or up to approximately 12 kg/ha of P. Both bulrush species removed more P than *Calamagrostis canadensis, Spartina pectinata, and Carex stricta*. In the large scale North Ottawa Impoundment (NOI), a 2014 *Typha x glauca* harvest was analyzed for P removal potential in which results indicated up to 2,564 kg of P removal from biomass harvesting, or approximately 3.11 kg/ha. Each site was also monitored for soil legacy phosphorus reductions. In the 2018 St. Paul mesocosm soil analysis, reductions of soil P after plant harvest significantly exceeded loading of P. This may indicate potential for legacy phosphorus reduction by removing vegetation. Harvesting vegetation in treatment wetlands based on phosphorus content within the shoots of selected species can be a successful management practice to reduce phosphorus accumulation over time. Currently, treatment wetland vegetation harvesting is not widely practiced in Midwestern agricultural watersheds; if treatment wetland design and harvests were cost-effective and compatible with surrounding farm systems, there may be potential for widespread application of harvesting vegetation for P removal. Harvesting treatment wetland vegetation annually may aid in reducing legacy phosphorus content within soil and may further prevent water quality degradation within agricultural watersheds at different scales.

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1.1 Introduction

There have been many significant attempts at classifying various types of wetlands into specific, understandable categories. As more of the natural world has been explored for developmental purposes, increased interest has been given in assigning specific terminology to occurrences of wetlands, often as it has pertained to matters of the law. Classifications and terminology associated with wetlands can vary depending on which region of the world you are looking at, such as different classification systems for the wetlands of Africa (Thompson and Hamilton (1983) than those in Europe (Palczynski 1984). Cowardin et. al. (1979) developed a classification system that included wetlands as well as deep water habitats; this system includes regions where the water source was of most importance. Lacustrine, riverine, marine, estuarine, and palustrine systems were sub-classified to include marine and vegetated habitats. This classification system was adopted by the US Fish and Wildlife service in 1979 and revised in 1993 (Cowardin & Golet 1995); this was later used for the National Wetlands Inventory. Recent attempts at classifying wetland types have included Holland et. al (1990) in which hydrology and fertility are the main factors to consider. This classification system includes a range of periods of hydration as well as fertility and includes salt flats, permanent or temporary shallow lakes, floodplains, bogs, fens, marshes, and swamps. Frazier (1996) described the Ramsar sites located globally; this classification system describes three large groups of wetlands including coastal, inland, and man-made. The Hydrogeomorphic Approach for Assessing Wetland Functions (Brinson 1993) included considerations largely based on geomorphic setting. In the United States, typical descriptions and classifications appear to be mostly based off of these systems, as well as interpretations presented by Keddy (2010) and Stewart and Kantrud (1971).

In the upper Midwest, there are large seasonal fluctuations in both temperature and precipitation. This area is also the location of recent glacial ice sheet movement and deposition of till which produced fertile soils. Eggers & Reed (2014) developed a classification system of wetlands that occur throughout Minnesota which was adapted from the Circular 39 system, in which vegetation types, depth, and duration of saturation are all included as considerable factors (Figure 1). The southern and western regions of Minnesota are covered in an extensive area of wetlands often described as the Prairie Pothole Region (Galatowitsch & Van der Valk 1996) which extends from the northern area of Iowa into areas of Canada and earned its name through the processes and remains of glacial movement. Wetland percent cover in these areas has been greatly reduced within the last century and within recent decades in part due to the increased demand for agricultural expansion. Dahl (2014) found that within a 12 year study, wetland percent cover in the prairie pothole region had declined by over 1%. Dahl (2000) found that in 1986-1997, approximately 26% of wetland area losses were attributed to agricultural conversions and about 30% of freshwater wetlands in the continental US were lost to urban development. Once dredged and drained, fertile wetland soils in the region are often utilized for their high crop productivity. In some areas of the prairie pothole region, much of the fertile topsoil content can be lost due to plowing practices which can significantly erode over time. Only in recent years has there been more of an interest in legally protecting the wetlands on both federal and state levels.

Since wetland ecosystems have been found to be overall highly productive (Cicek et. al. 2006), there have been many studies conducted to identify the services that they may provide. The ecosystem services provided by wetlands in North America was quantified by WESPUS (Adamus 2011) on a situational basis for the services provided based on their value to that area in particular. WESPUS emphasized that the importance and true value of ecosystem services are conditional upon the needs and issues present; for example, nutrient reduction in wetlands is more valuable to areas that are experiencing excessive nutrient loading into nearby water bodies than areas that may not be experiencing similar loading, although the wetlands may be operating at the same reduction rates. As with classifying different wetlands in different areas, there is often a regional restriction on the services that different wetlands may provide. The MEA (Millennium Ecosystem Assessment) (2005) created four groups of services that may be provided by a natural area including: provisioning, regulating, cultural, and supporting services. The MEA created a list of 17 specific services that fit into each of the 4 groups that range from fish production to nutrient cycling and carbon sequestration. McLaughlin and Cohen (2013) viewed relationships between conditions of wetlands and their corresponding services, with land use in surrounding areas having a larger impact on ecosystem services a wetland may provide at a given time. Rains et. al. (2015) examined geographically isolated wetlands, which are common in North America. These wetlands can be surrounded by uplands but still play large roles in overall groundwater transport to other wetlands over large distances; these isolated wetlands can also be critical for wildlife habitat depending on time of season.

As global climate change has become an increasingly alarming issue, more research has been invested in understanding the carbon sequestration and methane production that occurs in wetland ecosystems, such as with Silvola et al. (1996), Gorham (1991) in regards to northern wetland ecosystems, and Woodwell & Mackenzie (1995) exploring the various feedback loops found in natural systems. It has been noted that different levels of inundation (Busnardo et. al. 1992), varying occurrences of vegetation and composition (Fraser et. al. 2004), and temperature (Mitsch & Ahn 2002) can have significant consequences on the services provided, and therefore may need to be explored further in situational contexts.

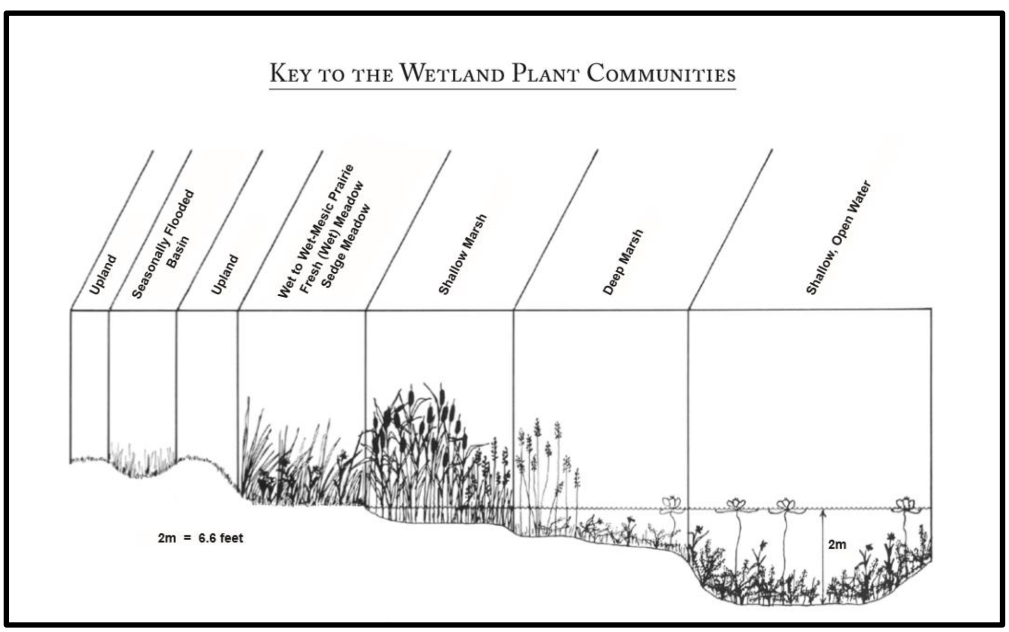

Figure 1: Eggers and Reed key to the wetland plant communities

High productivity in some wetland ecosystems can largely contribute to the nutrient cycling processes (Larkin et. al. 2012 (a)). As some wetland plants are able to grow clonally and rapidly, they are able to utilize a high amount of nutrients that enhance their growth, including nitrogen and phosphorus (Larkin et. al. 2012 (b)). Verhoeven et. al. (1996) concluded that nitrogen and phosphorus are both limiting nutrients of growth within wetland plants; this was a strong conclusion considering past research had decided nitrogen as the primary limiting nutrient (Barko & Smart 1980). They are able to translocate these nutrients from the roots to the shoots of the plant and back again depending on the specimen's individual nutrient demands (Cicek et. al. 2006). This process includes a chemical transformation of the soluble bioavailable forms of phosphorus into plant tissue, therefore removing the excess nutrients from the soil.

Phosphorus accumulation within the sediment is known as legacy phosphorus (Penn et. al. 2015) and can be released due to environmental condition variations or disturbances (Koski-Vahala & Hartikainen 2001; James & Barko 2004; Kadlec et. al. 1999; Bennett et. al. 2001; Phillips et. al. 1994). Phosphorus that has been trapped or bound to sediment through adsorption processes can remain undisturbed within the sediment for long periods of time (Guzner 2016). Agricultural landscapes throughout the Midwest that have been receiving nutrient inputs via fertilizer applications have accumulated and stored phosphorus within the soil (Bennett et al 2001). Soil phosphorus content and storage within soils is important to consider, especially in the consideration of a drawdown in aquatic systems as a management strategy, in which the phosphorus could be released from the sediment (Reddy et. al. 1997; Jordan et. al. 2006). An example of this type of situation in which phosphorus is released from the sediment is the former Great Black Swamp in Ohio (Mitsch 2017), in which much of the wetland area has already been converted to agricultural land (Dahl 2000; Thorslund et. al. 2017; Mitsch & Ahn 2002).

Excess nutrients that have been added to agricultural or urban watersheds can collect in surface water runoff and deposit into a water body, causing excess algae growth and eutrophication, which can present issues for overall water quality (Bennett et al 2001) and associated ecosystem services. Phosphorus is often a limiting nutrient for growth in freshwater ecosystems and therefore is the target of management for upper Midwestern aquatic ecosystems. While nitrogen can be converted to a gas, phosphorus is not commonly found in its gaseous phase; it tends to accumulate in low topographical parts of the landscape, such as wetland areas. Although wetlands have been found to be effective in their ability to remove nitrogen from the system (Nairn & Mitsch 2000; Crumpton et. al. 2006), there is more to be explored in regards to phosphorus movement and removal in agricultural watersheds (Sharpley et. al 1994; Westra et. al. 2007). Under anaerobic conditions, release of stored phosphorus from sediment can occur, in which P is released from Fe, which may impact water quality within a drainage area (Hoffman et. al 2009; Penn et. al. 2015).

Alternative management strategies for preventing or treating eutrophic or low water quality systems caused by high loads of nutrients have been presented (Huser et. al. 2016), but more natural solutions, such as utilizing the nutrient uptake capabilities of wetland plants, have recently been presented as a management strategy. Another strategy that has recently been explored has been to implement and harvest wetland vegetation for nutrient remediation in a particular area (Fraser et. al. 2004; Cicek et. al. 2006; Gottschall et. al. 2007; Grosshans 2014; Wang et. al. 2015; Yang et. al. 2016; Verhofstad 2017; Amarakoon et. al. 2018). Excess nutrients or sediment to a particular wetland ecosystem may cause higher biomass and a decrease in diversity, resulting in monoculture systems (McJannet et. al. 1995). Implementation of edge of farm buffer strips or treatment wetlands are an innovative engineered solution to provide natural removal of excess nutrients by vegetative uptake from a landscape given particular conditions in hydrology and soils.

1.2 Hydrologic and Nutrient Cycling Processes by Wetland Type

Various types of wetland ecosystems will appear globally and appear differently based on their specific hydrology, soils, and vegetation present. In the upper Midwest of the United States, the prairie pothole region, there are a variety of wetlands present that have been classified (Keddy 2010) and interpreted. Wetland nutrient cycling is driven by water sources, soil conditions, and, in some cases, atmospheric deposition, all of which can be analyzed from a landscape perspective.

1.2.1 Bogs

Bogs are a specialized class of wetlands that occur in select climates around the globe, often times found in northern latitudes. These types of wetlands will occur in places where precipitation is the main source of hydrologic and nutrient inputs, which is typically in upland areas. Bogs can occur in small depressions, known as perch bogs, or in flat landscapes, known as raised bogs (Grigal 1985). Bogs will typically have very acidic soils, due to the inputs of carbonic acid from precipitation, as well as the high abundance of *sphagnum spp*. Phosphorus accumulation can be high within bog soils due to the sorption processes in these low pH systems (Kadlec & Knight 1996) and decomposition rates of present organic matter (Bridgham et. al. 1998).

The majority of nutrients that deposit into bogs come from precipitation, so nitrogen and phosphorus presence for plant growth can be naturally very limited. Other nutrients that are available for plants in bog systems come from the organic material that slowly decomposes in these systems due to the typically colder temperatures and high amounts of precipitation. The present species found throughout bogs is highly dependent upon which are adapted to survive and compete with low nutrient availability and in highly acidic conditions (Chapin et. al. 2004). *Sphagnum spp*. mosses are mostly present in bogs and can be used as a key identifier for this type of wetland. *Sphagnum spp.* will often times create hummocks and hollows within the bog, or small depressions or raised formations due to the accumulation of the moss (Grigal 1985) which can be another indicative factor in identifying a bog. Lower elevations in bogs that experience longer durations of inundation are likely to include more diversity in species present, such as specific conifer species (Damman 1986). In the upper Midwest, bogs are mostly found in northern portions of Michigan, Wisconsin, and Minnesota and primarily occur in forested landscapes with little to no cropping history. Therefore, bogs do not tend to have the issues with sediment and phosphorus accumulation that is typical of agricultural-region wetlands.

1.2.2 Inland Meadows

Inland meadows are a common subclass of wetland throughout the upper Midwest. These types of systems are characterized by their dry growing seasons, with ponding infrequently and for short durations (Eggers & Reed 2014) resulting in temporary or ephemeral hydroperiods (Stewart & Kantrud 1971). Sedge meadows, wet prairies, and calcareous fens are just a few of the many types of inland meadows that may be present in certain areas and are often dominated by graminoids.

Sedge meadows are found abundantly throughout the Midwest and, more specifically, in areas that are not as frequently inundated. This includes other types of wetlands or inland meadows. These types of ecosystems are home to a variety of grasses and sedges, including hummock sedge (*Carex stricta*) that remains throughout the entirety of the growing season (Eggers & Reed 2014). Comparatively, wet meadows are another subclass and will often experience saturation for longer periods throughout the season, or more frequently than sedge meadows. An example of a wet meadow/wet prairie would be the Granada treatment wetland in southern Minnesota (Gordon 2019; Lenhart et. al. 2016). There is often high plant diversity in undisturbed meadows due to the saturation periods that occur, typically following a spring snowmelt or flooding such as in Minnesota. Canada blue-joint grass (*Calamagrostis canadensis*) can be frequently found, and reed canary grass (*Phalaris arundinacea*) may be present, especially in cases of disturbances, which will impact and reduce the diversity of plant species present (Galatowitsch et. al. 2000). Reed canary grass is an invasive species in Minnesota and can alter the ability of a wet meadow to provide sufficient space for establishment of other native species following an invasion.

Prescribed burns and herbicide applications are examples of management strategies for restoration in these areas (Galatowitsch & Adams 2006). Calcareous fens are a unique and less abundant subclass of inland meadow. These areas are often times highly protected due to their unique makeup of soils, hydrology, and vegetation that may be present. These systems are typically groundwater fed which alters the chemistry of the water that can enter the system. The inputs of water may be higher in mineral content, which allows for specialized plant species to establish and flourish, often times resulting in marsh vegetation (Mitsch & Gosselink 2000). Sedges may be abundant here, as well as brown mosses in lower elevation areas. Over time, these calcareous fens may develop into bogs due to the accumulation of decomposing vegetation and peat (Eggers & Reed 2014).

1.2.3 Marshes

Typical freshwater marshes in the upper Midwest will experience inundation for the entirety of the growing season beginning with a spring snowmelt. Surface water fed marshes tend to be more nutrient-rich than groundwater-fed wetlands. Marshes may include both sources depending on position throughout the landscape, particularly in the prairie pothole region, though are typically surface-water dominated (Keddy 2010). There are also known subclassifications of both deep and shallow marshes, which are based off of the inundation levels experienced in an average season and resulting vegetation. In upper Midwest settings, many marshes have received increased runoff and tile drainage flow with high amounts of N, P, and sediment.

As marshes can receive surface water or groundwater which both will contribute nutrients to the wetland, the soils present and their subsequent characteristics are highly dependent upon the hydrologic regime. Organic matter is a large contributor to the nutrients available in the soil, including nitrogen and phosphorus, and a longer hydroperiod can indicate a larger available nutrient pool (Craft et. al. 1988). At times of drought, freshwater marshes may have the capacity to hold and store water within their soils for plant uptake, which is dependent upon the soils present, including pH (Weller 1994; Cowardin et. al. 1979; Koski-Vahala & Hartikainen 2001). Marshes can experience nutrient loading from runoff during precipitation or flood events as well as decomposing organic matter including the vegetation present within the wetland (Craft et. al. 1988).

Internal loading of phosphorus from the sediment within the wetland is a possibility that should be considered by land managers; phosphorus release from the sediment can be a large contributor to aquatic systems based on a variety of factors including pH level, soil type, and fluctuating hydroperiod (Khalid et. al. 1976; Kadlec & Knight 1996). A longer hydroperiod or longer saturation period has been witnessed to decrease the risk of phosphorus release from the sediment at times of flooding occurrence due to anaerobic conditions (Dunne et. al. 2010). Drawdown followed by flooding can mobilize stored phosphorus from within the wetland, which can cause further problems downstream (Mitsch 2017; Reddy et. al. 1997). Vegetation found to be present in these marsh systems have adapted to the lack of oxygen. Aerenchyma tissue can be found in abundance throughout many of the species; this is a special adaptation to provide oxygen to the root systems at times of inundation (Gurevitch et. al. 2006). Emergent vegetation dominates in marsh systems. This can include bulrushes (*Schoenoplectus lacustris, Schoenoplectus fluviatilis, Schoenoplectus tabernaemontani)*, graminoids such as *Phragmites australis, Typha spp.*, and sedges *Carex spp.* Emergent vegetation well adapted to withstand varying inundation levels will dominate throughout these systems (Van der Valk & Squires 1992; Mitsch & Gosselink 2000; Rothrock 2009; Lishawa et. al. 2010; Eggers & Reed 2014).

1.2.4 Swamps

Swamps are unique in their distribution and naturally occurring vegetation. Typically found in low parts of the landscape, swamps can be dominated by shrubs and forbs as well as hardwood and softwood trees. *Thuja occidentalis, Pinus strobus, Salix spp., Alnus, spp., Taxodium distichum, Acer rubrum*, and various spruces and other species that can withstand varied levels of inundation may be present within swamp systems (Wallace & Kadlec 2008; Eggers & Reed 2014). Their distinct vegetation mixes of included hardwoods can be one of the key identifiers of this class of wetland, as inundation levels vary throughout the season as well as in different locations. Red maple swamps may occur more abundantly in the northeastern United States; southeastern United States will contain more baldcypress swamps (Mitsch & Gosselink 2000). Freshwater swamps, also identified as forested wetlands, are often times considered temporary or seasonally flooded wetland systems (Stewart & Kantrud 1971). The duration of saturation can be determined by a swamps drainage capacity and soil profile (Eggers & Reed 2014). Frequent drawdowns throughout a season caused by periods of drought and flooding can contribute to phosphorus loading due to resuspension of deposited sediment. (Koski-Vahala & Hartikainen 2001; Thorslund et. al. 2017). A significant portion of nutrients to freshwater swamps, including phosphorus, will be deposited through sediment left from flood occurrences or alluvial deposition (Mitsch et. al. 1979) or through litter from on-site vegetation (Mitsch & Gosselink 2000).

1.2.5 Floodplain Forests

Found adjacent to rivers and streams, floodplains are natural hotspots for loading of sediment and nutrients due to recurring flooding and deposition. Overflow runoff from flood events to lakes and rivers can produce floodplain forests over long periods of time. Flooding can cause issues for plant establishment and growth, so plants have to be well adapted to fluctuations of inundation and drought periods. Plant injury, inhibited seed germination, and premature senescence or mortality can be some of the consequences for floodplain forest plants that may not be as well adapted to these seasonal changes (Kozlowski 1997). The Intermediate Disturbance Hypothesis indicates a relationship between the frequency of disturbance and species diversity present throughout floodplains; species diversity and richness was found to be highest in areas that experienced intermediate levels of disturbances over time (Townsend et. al. 1997). The infrequent or weather dependent duration of saturation in these systems would indicate a temporary or ephemeral hydroperiod (Stewart & Kantrud 1971).

Floodplain soils can appear different than other wetland soils due to their frequent cycling of sediment deposition and removal through flooding and runoff events (Hughes 1997; Koski-Vahala & Hartikainen 2001). Phosphorus that enters into floodplain forests usually is deposited attached to the sediment that is deposited in runoff events. Leaf litter from the above canopy can also be a large contributor to the wetland, as decomposition can contribute nitrogen and phosphorus (Lockaby et. al. 1996) but depends largely on the type of vegetation cover present. Floodplain forests have been successful in their nitrogen removal (Kadlec & Knight 1996), including the annual herbaceous vegetation significantly contributing to seasonal nutrient cycling (Peterson & Rolfe 1982). Floodplain forests are equipped with well adapted woody vegetation that can withstand a variety of environmental conditions including varied hydroperiods and related soil chemistry (Kozlowski 1997) (Hughes 1997). Some species present and adapted to floodplain conditions can include *Quercus nigra* , which produces heavy acorns that have a high survival rate, or *Acer rubrum*, which produces a high amount of seeds to aid in increasing the survival rate.

1.2.6 Shallow Open Water Basins

Shallow open water wetlands will have higher water levels for longer periods of time than other types of wetlands; they receive surface water, ground water, and precipitation. These are considered wetland systems due to the variation in water level and possible drawdown events in which seed banks may regenerate, which may classify them as permanent or semi-permanent wetland systems. These systems can be productive for waterfowl habitat as well as nutrient retention services (Eggers & Reed 2014). Periphyton can play a critical role for phosphorus retention in shallow freshwater systems (Dodds 2003), as they provide phosphorus uptake and filtering particulate phosphorus throughout the water column. Phosphorus release through resuspension of sediment can be one of the leading contributors of excess nutrients to these systems and can cause algal blooms in some cases (Phillips et. al. 1994). Common vegetation will include floating or submerged vegetation, such as *Typha spp.*, *Ruppiaceae spp.*, and *Salicornia rubra* (Stewart & Kantrud 1971).

Freshwater Wetland Classification				
Wetland Type	**Landscape Setting**	**Hydrologic Regime**	**Characteristic Plant Communities**	**Source(s) of Phosphorus**
Bogs	Northern latitudes, colder climates, high precipitation present	Permanent/Semi-permanent	*Sphagnum spp.* dominates Conifer species possible	Vegetation litter accumulation and decomposition
Meadows	Various; common in prairie pothole region	Temporary/Seasonal	Graminoids dominate Sedges (*Cyperaceae)*, prairie cordgrass (*Spartina pectinata*), canada blue joint grass (*Calamagrostis canadensis)*, switchgrass (*Panicum virgatum*)	Runoff from uplands during rainfall/snowmelt events Vegetation litter accumulation and decomposition Surface runoff Tile drainage
Marshes	Riparian or coastal regions	Seasonal	Emergent aquatic Sedges (*Cyperaceae*), bulrushes (*Scirpus spp.*), cattails (*Typha spp.*), common reed (*Phragmites spp.*) pickerelweed (*Pontederia cordata*)	Re-supension of nutrients following drawdown events Decaying vegetation Surface runoff
Swamps	Riparian or lacustrine zones, high precipitation present	Seasonal	Hardwoods and emergent vegetation present Willow (*Salix spp.*), alder (*Alnus spp.*), white pine (*Pinus strobus*)	Flood deposition of sediment Surface runoff
Floodplain Forests	Riparian or lacustrine zones	Temporary/Seasonal	Hardwood vegetation present Oaks and maple species possible	Sediment/Erosion Vegetation litter accumulation and decomposition

Shallow Open Water Basins	Varied; can be drawn down or inundated	Permanent/semi-permanent	Floating, submerged and emergent aquatic vegetation present Cattail (*Typha spp*.)	Sediment/Erosion Surface runoff

Table 1: Freshwater wetland ecosystems in the northern Midwest area

1.2.7 Synthesis

Nitrogen and phosphorus reductions in wetlands may be achieved using various processes. For example, nitrogen is most effectively removed by establishing a set residence time for denitrification to occur. Residence time of water directly relates to nitrogen within the wetland. Some wetlands, such as the North Ottawa Impoundment in the Red River Valley of Minnesota, have incorporated different residence time settings within each of the cells of the treatment wetland. Some cells may be designed for water storage holding only to achieve denitrification, while other cells are designated specifically for plant harvesting. Phosphorus removal may be more challenging to remove than nitrogen due to its primarily solid and liquid phases. Particulate phosphorus may settle within a wetland or bind to sediment, or be utilized by wetland plants. Treatment wetlands that are designed to target phosphorus loading within a system will often consider the establishment of wetland plants and harvesting so there is less of a risk of flushing of phosphorus. Maintaining partially aerated soils for sustained vegetative life is recommended for nutrient removal success by process of harvesting (Kadlec & Knight 1996; Lenhart et. al. 2017; Fransen 2012).

Constructing treatment wetlands in agricultural watersheds should largely depend on the location within the watershed for targeted nutrient reductions. A restored wetland may already have high amounts of phosphorus stored within the soil which may become mobilized following a disturbance such as construction of the wetland or the re-saturation of the wetland for treatment. Discharge of phosphorus from a treatment wetland has been witnessed in pre-existing peatlands (Kadlec & Knight 1996), which suggests these wetlands may not be an optimal choice for restoration. Large drainage areas with higher hydraulic loading rates (HLR's) are recommended to be selected for establishing constructed treatment wetlands for optimized nutrient removal. For example, a large agricultural watershed may choose to incorporate a small scale treatment wetland with emergent vegetation that can be harvested annually in a targeted location to maximize the amount of loading to the wetland which would remove more phosphorus. The Blue Earth River Basin with various wetlands located throughout it in the Prairie Pothole Region has experienced high loads of nutrients in the months of April-June in recent years. The restored wetland basins located throughout the basin effectively captured and reduced nitrogen based on outlet samples, but were less effective at phosphorus removal (Fransen 2012). These results indicate further potential for vegetation reservoirs and harvesting as a nutrient remediation strategy in agricultural watersheds experiencing high loads of phosphorus.

1.3 Treatment Wetlands

1.3.1 Overview

Treatment wetlands are an engineered and relatively new management strategy for agricultural watersheds (Levy 2018) that has been introduced in a collaborative effort by land managers and scientists. In regards to urban waste, treatment wetlands have been widely adopted as a management strategy. Constructed treatment wetlands are designed with the intention of replicating the ecosystem services, such as nutrient reduction or removal, provided by naturally occurring wetlands. Recent legislation and incentive programs have aided in encouraging landowners to implement treatment wetlands on their property (CREP and BWSR credit program). In more recent cases, treatment wetlands have been implemented in areas that were once wetlands; restored treatment wetlands are becoming a more desirable management strategy than constructed treatment wetlands. Some subclasses of treatment wetlands include those with subsurface flow, in which underground networks are constructed to capture groundwater runoff or tile drainage, and free water surface flow wetlands, which are designed in open basins (Vymazal 2010). Throughout time, treatment wetlands have been utilized for a variety of purposes. In the 1970’s and 1980’s, treatment wetlands were utilized specifically for targeted treatment of wastewater. In the past 30 years, many multi-purpose wetlands have been restored with water quality treatment as the main goal. Only within the last 10-20 years have treatment wetlands been designed or restored primarily for water quality purposes. In Iowa, the CREP program was designed specifically for nitrogen removal; the CP-39 system was implemented in Minnesota for nutrient removal and treatment (Lenhart et. al 2017).

Although constructed treatment wetlands allow for flexibility and control over factors including hydraulics and soil bed content, which influence the effectiveness of the wetland, restored wetlands are desirable due to their antecedent locations within watersheds. A study by Land et. al (2016) analyzed approximately 203 created or restored wetlands for nutrient removal effectiveness. The wetland systems that were selected were located in northern climates across Europe and North America and varied in experimental design which was a fundamental flaw in understanding and comparing effective nutrient removal. Median phosphorus removal efficiency was found to be 46% (confidence interval of 37-55%). Similarly, a study by Richardson & Bruland (2006) concluded that constructed wetlands did not perform as well as naturally occurring wetlands and had lower overall soil organic matter, which can be directly correlated to phosphorus retention. Instead, restored wetlands were the optimal choice for a variety of services (Kusler & Lentula 1990) including retaining soil organic matter and improving wetland ecosystem services such as phosphorus retention and denitrification.

Treatment wetlands can be designed to capture water from subsurface flow or surface water flow and are placed strategically within a watershed or drainage area for specific purposes. These carefully designed solutions are typically implemented in areas that can experience high loads of nutrients or pollutants, including sources of mine drainage, wastewater treatment and capture (Mitsch & Wise 1998; Robertson et. al. 2018; Vymazal 2010), within urbanized watersheds (Janke 2014), or agricultural landscapes (Guzner 2016; Current et. al. 2016). Treatment wetlands are engineered with specific hydrology, soil, nutrient loads, and plant compositions and can vary in size, ranging from small scale mesocosm treatment experiments (Mitsch & Gosselink 2000; Kadlec & Knight 1996) to large scale agricultural watershed treatment wetlands (Current et. al. 2016; Lenhart et. al. 2016; Guzner 2016; Gordon 2019). In areas experiencing high loads of nitrogen and/or phosphorus, factors including hydraulic loading, wetland age, soil fertility and drainage, seasonality, and temperature shifts, (both in constructed and restored treatment wetlands) are also important to consider when determining the potential for an effective nutrient reduction management strategy (Ziegler 2016; Land 2016; Schilling et. al. 2014; Kadlec and Knight 1996). Some management considerations in man-made treatment wetlands can include subsidence of the land surface from compaction, adjacent land use, and management of water control structures in regards to habitat for wildlife (Kusler & Kentula 1990). In the upper Midwest prairie pothole region, management techniques must also consider the scope of the target watershed in order to be effective in the intended goal of the project (Galatowitsch et. al. 2000; Schilling et. al. 2014; Gordon 2019). Alternative nutrient removal strategies have been explored, including costly alum treatments for phosphorus loading management, particularly in lakes (Huser et. al. 2016), which have made lower cost options like treatment wetlands a desirable solution for preventing high nutrient loads to aquatic systems.

1.3.2 Treatment Wetland Vegetation

Treatment wetlands throughout the upper Midwest must primarily be designed around the intended hydroperiod which will directly influence the vegetation present (Kadlec & Knight 1996). Treatment wetlands will typically receive higher nutrient loads on average than a naturally occurring wetland (Wallace & Kadlec 2008). As a result, nitrogen, phosphorus, and other nutrients in surplus can cause a high initial production rate for existing vegetation (McJannet et. al. 1995). Vegetation that may be present in constructed or naturally occurring treatment wetlands may include floating or submerged macrophytes and may utilize aerenchyma tissue as an adaptation to the flooded soils for much of the growing season. Wetland macrophytes can largely alter the arrangement and chemical composition of the upper portion of the soil, referred to as the rhizosphere, due to their rapid growth and spread of rhizomes (Kadlec & Knight 1996). Selected treatment wetland vegetation is highly dependent upon what nutrients the wetland is intended to treat or remove as well as the overall goal of the project itself. Many treatment wetlands will incorporate numerous types of plant communities in different portions of the wetland, such as including some areas of open water, deep or shallow marsh, and wet prairie (Ziegler 2016). Implementing wetland vegetation mixes is usually done by planting a seed mix, which can be lower cost but may take more time to fully establish. Similarly, the site may be more susceptible to invasive species due to competition with the establishing seed mix. Plugs are another way to establish vegetation within a treatment wetland. These can be more costly and time consuming to implement within the wetland, but the land manager may achieve more desirable plant establishment.

Plants of the grass (*Gramineae*) and sedge *(Cyperaceae*) families have been found to be some of the most productive wetland plants which can remove the most nutrients while also surviving repeated harvests over several growing seasons. Tanner (1996) found *Phragmites australis* was more effective at removing nitrogen than phosphorus when harvested for nutrient removal. *Phragmites australis australis* is also a known invasive species strain that is capable of spreading rapidly. As a result, it has become a less desirable plant for treatment wetlands due to the high risk of spread to unintended areas. Similarly, cattail (*Typha spp.*) has been explored as well due to high biomass and rates of success in a variety of conditions (Jeke et. al. 2018; Cicek et. al. 2006; Lishawa et. al. 2010; Grosshans 2014). Although some species are invasive, such as hybrid cattail (*Typha x glauca)* in North America (Galatowitsch et. al. 1999), this species has displayed high nutrient uptake capabilities and has also been more resistant to predation by muskrats and other fauna (Wallace & Kadlec 2008). Duckweed (*Lemna spp.*), a floating type of vegetation, and coontail (*Certaophyllum spp*.), a submerged aquatic plant, have both been utilized within wastewater treatment wetlands in the upper Midwest (Wallace & Kadlec 2008). Softstem bulrush (*Schoenoplectus tabernamontani*) is a native plant to the continental United States and may be selected for constructed wetlands due to its high seed production, which can be a valuable food source for some species of waterfowl (United States Forest Service). Some species of plants that have minimal tendency and risk towards becoming an invasive species to an ecosystem in the upper Midwest include *Scirpus fluviatilis, Carex comosa, S. tabernaemontani,* and *E. palustris* (Rothrock 2009) and are more desirable for treatment wetland phytoremediation practices.

1.3.3 Wetland Vegetation Harvesting

a. Issues

Harvesting of wetland macrophytes can be an effective nutrient removal strategy but is largely dependent upon the accessibility of the plants throughout the season as well as equipment that is available to the land manager (Grosshans & Grieger 2013). Wetland macrophytes may also have dense cover in some areas; thick stems in above ground tissue for some species may cause problems for some land managers with inadequate equipment. Harvesting of treatment wetland vegetation may not be a practical option if the wetland has high water levels in late fall, which is when harvests are typically conducted in the upper Midwest. Drainage of the wetland using water control structures would be necessary in these cases.

b. Costs

Although maintenance of most wetlands usually only has associated costs with monitoring of about $0.03 to $.09 per cubic meter (Kadlec & Knight 1996), other costs associated with harvests may include the labor, equipment, and disposal of material. Equipment that is used for treatment wetland vegetation harvests may include large mowers or other farm equipment that is available to the land manager (Cicek et. al. 2006; Grosshans & Grieger 2013). However, large equipment such as mowers may cause compaction of the soil throughout the wetland, which may lead to unintended issues throughout the area (Lenhart et. al. 2017). Ditch sites or buffer strips can be significantly cheaper to harvest biomass from compared to lake sites or other fully saturated areas. Ditch site cattail removal can cost approximately $20-$30 per bale at a wetland site or $52-$78/1000kg of dry biomass (Grosshans & Grieger 2013).

c. Harvested Material Disposal

The harvested vegetation may be re-used in some cases. Instances of landowners re-using the harvested material for feed for their livestock or burning it are just some alternative uses for the discarded material instead of directly disposing of it. Some land managers may choose to utilize the harvested biomass from their treatment wetland or buffer strip for energy or biocarbon (Cicek et. al. 2006; Grosshans & Grieger 2013). Direct uses for the produced biocarbon can include crop fertilizer in agricultural areas for improved sustainability (Grosshans & Grieger 2013). Sustainable practices and uses for harvested biomass is an area with room for further research potential.

Common Treatment Wetland Vegetation Types in the Upper Midwest			
Wetland Vegetation	**Common Locations**	**Biomass**	**Growing Season**
Softstem bulrush *Schoenoplectus tabernaemontani* *(Scirpus validus)*	Marshes Swamps Wet Prairies/Wet Meadows Riparian Zones Treatment Wetlands (OBL)	2-9ft height (Minnesota Wildflowers) 1-3ft height in one season of growth (USFS) Can triple biomass in one season (USFS)	July-September (Minnesota Wildflowers) Varies with latitude (northeast: July-August flowering) (USFS)
River bulrush *Scirpus fluviatilis*	Riparian Zones Marshes Swamps Floodplain Forests Treatment Wetlands (OBL)	3-6ft height (Minnesota Wildflowers)	July-September (Minnesota Wildflowers)
Hybrid Cattail *Typha x glauca*	Invasive species; present with periods of saturation	3-10ft (USFS) Growth via rhizomes	80-117 day growing season (starting in June) (USFS)
Common Reed *Phragmites australis*	Invasive in North America; can establish in most wetlands and disturbed areas	20ft height at maturity (up to 8 years) (USFS)	Dependent on latitude (USFS)

Table 2: Freshwater wetland vegetation commonly found in northern Midwest ecosystems(Sources: http://plants.usda.gov https://www.minnesotawildflowers.info https://www.fs.fed.us/database)

1.4 Conclusion

Freshwater wetlands in the upper Midwest United States can be found in a variety of locations and under a variety of environmental circumstances. Classification systems have sufficiently identified and described these systems for their unique hydrology, soils, vegetation, and services. Identification and classification are necessary for legal protection as well as further environmental management and conservation of these systems. Agricultural watersheds throughout the upper Midwest have been delivering high loads of nitrogen and phosphorus due to routine agricultural practices including repeated fertilizer applications. Phosphorus is a challenging nutrient to remove once it is applied to a landscape; sediment, belowground plant tissue, and above ground plant tissues are the main reservoirs for the nutrient (Kadlec & Knight 1996). Treatment wetlands are an engineered solution that may provide some of the services provided by naturally occurring wetlands to impacted areas. Treatment wetlands can be partnered with other management strategies within a watershed as a treatment train concept in order to achieve a particular goal, such as phosphorus removal in highly impacted areas (Lenhart et. al. 2017). To enhance the abilities of treatment wetlands to capture and remove phosphorus polluted water, natural solutions such as specific plant establishment and subsequent harvesting may be a necessary solution in some cases. This non-invasive, relatively affordable, and seemingly effective solution should be explored in certain cases for further enhancement of the nutrient reduction provided by the naturally occurring or manually engineered wetlands throughout the upper Midwest.

Chapter 2: Harvesting Vegetation in an Edge-of-Farm Treatment Wetland and Mesocosm Experiment for Phosphorus Removal

2.1 Introduction

Treatment wetland vegetation can display high biomass over time as the area becomes more established following installation. Depending on location, hydrology, soils, and other environmental factors, vegetative communities can vary per treatment wetland. However, when designing constructed wetlands, land managers may often times choose seed mixes or selected plug installation throughout the wetland in an effort to control what may establish in the first few years of operation. Vegetative uptake of phosphorus may be one of the main pathways the essential nutrient may take within a constructed wetland system and therefore the vegetation present may be utilized as a nutrient management tool.

2.1.1 Phosphorus Cycling

Particulate forms of phosphorus that is bound to sediment may enter a water body through erosion processes (Kadlec & Knight 1996). Soluble forms of phosphorus can enter a wetland from surface water overflow, tile drainage, decomposition of organic matter, or re-suspension and internal loading of phosphorus stored in sediments (Wallace & Kadlec 2008; Hoffman et. al. 2009; Fransen 2012). Phosphorus can bind to sediment under specific aerobic conditions; low pH systems may bind soluble phosphates to Fe, Al, or Ca. More specifically, in systems with high soluble phosphorus concentrations present and a pH range of 3-7, phosphorus is more likely to experience sorption, in which it binds to Al, Fe, and Ca ions on sediment (Patrick 1974; Khalid et. al. 1976; Schlesinger & Bernhardt 2013). Phosphorus can also bind to Fe^{3+} in aerobic conditions. This may result in particulate aggregates of phosphorus that settle out into the sediment in aquatic systems. However, if the conditions remain anaerobic for a long enough period of time, Fe^{3+} may convert to Fe^{2+} and therefore release the soluble phosphate back into the system (Guzner 2016). Accumulation of phosphorus within the sediment is sometimes referred to as legacy phosphorus and can be released due to environmental condition variations or disturbances (Kadlec et. al. 1999; Koski-Vahala & Hartikainen 2001; James & Barko 2004; VanZomeren et. al. 2019). Over time, this sediment can become disturbed and phosphorus can become mobilized causing the system to become a source of P instead of a sink for P (Reddy et. al. 1997; Koski-Vahala & Hartikainen 2001). Treatment wetlands in areas with high soil phosphorus content due to agricultural practices may become sources of phosphorus (P) instead of a sink for phosphorus and may cause water quality issues (Lenhart et. al. 2019).

There are different forms of soil phosphorus including Total Phosphorus (TP), Olsen Phosphorus (OP), and Bray Phosphorus (BP), which are all analyzed through different types of laboratory testing in which various extraction procedures are used. Olsen-P tests are typically used for alkaline soils (pH above 7.4) whereas more acidic soils can be tested using Bray-P tests (pH below 7.4). Both tests can be used to indicate the bioavailable forms of phosphorus within soil. The Mehlich-3 test is similar to the Bray and Olsen P tests; the Mehlich 3 was developed in order to test for P, K, Ca, Mg, Na, and other micronutrients through a broad scale of pH ranges (Sawyer 1999; Simms 2000). Total phosphorus tests will include all or most forms of phosphorus within the soil and may be more difficult to accurately quantify; TP values may provide a more accurate representation of phosphorus content within soil. Total P values typically range from 300-1000 mg/kg (Lenhart et. al. 2019). Olsen-P values may be more indicative of the amount of soluble or bioavailable phosphorus within the soil that may be at risk for being flushed from the system compared to Total P values (Jordan et. al. 2006), which do not change as rapidly. An Olsen-P value of approximately 10 mg/kg is recommended by Simms (2000) for optimal plant growth; higher values may indicate risk of flushing to the soil water solution and can cause eutrophication issues downstream.

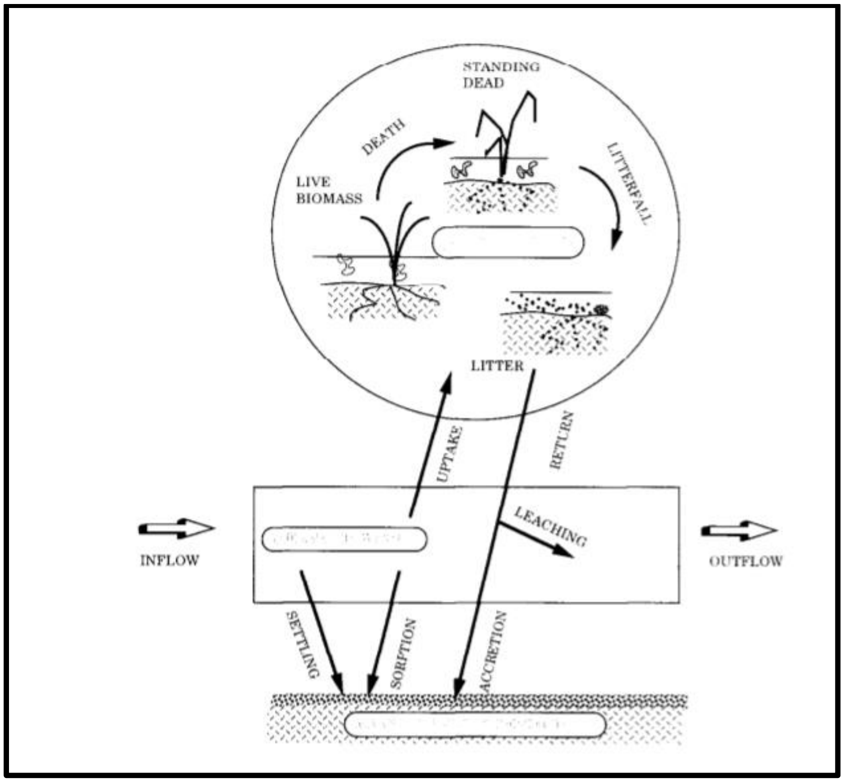

Figure 2: The biomachine model describes pathways of phosphorus within a wetland (Wallace & Kadlec 2008).

Phosphorus may be made available to macrophytes and microbiota by desorption or diffusion through the pore water (Kadlec & Knight 1996). Vegetative uptake of nutrients within the root systems and shoots of a macrophyte (Yang et. al. 2016; Dierberg et. al 2002), can be one of the main pools of bioavailable forms of phosphorus within a wetland or aquatic ecosystem (Gordon 2019). Mitsch (2015) found in a mesocosm experiment in the Florida Everglades that wetland communities that do not include high biomass species (ex: *Typha domingensis*) may be more effective at retaining phosphorus, indicating varied levels of nutrient retention services based on wetland plant community types present. Particulate phosphorus within a wetland may settle out within the sediment; soluble forms of phosphorus will be utilized by biota within the system, bind to sediment within the system, or be exported out of the system (Kadlec & Knight 1996). The uptake of phosphorus for growth by wetland plants can lead to the storage of phosphorus within the above ground biomass of the plant. Considering the large pool of phosphorus within wetland plants, land managers should consider the utilization of these plants for their desired nutrient removal goals in future projects.

2.1.2 Vegetation Harvest for Phosphorus Removal

Although vegetative uptake of phosphorus has been proven to be an effective removal strategy in treatment wetlands (Reddy & Debusk 1985, 1987; Busnardo et. al. 1992; Kadlec & Knight 1996; Tanner 1996; Fraser et. al. 2004; Drizo et. al. 2008; Wang et. al. 2015; Current et al 2016; Bonanno et. al. 2018) there is more to be explored in regards to understanding vegetation for management. Factors to be explored include the time of season, the amount of times to harvest, and what species may optimize phosphorus removal within a treatment wetland. First, there may be variable phosphorus content within the shoot of the plant throughout a season. Harvesting of wetland vegetation is typically done in the fall when water levels have subsided. Further research may be necessary to understand phosphorus content within wetland vegetation throughout a growing season. This knowledge gap may be an opportunity to optimize removal of excess phosphorus by process of vegetation harvesting (Yang et. al. 2016). It has been observed that phosphorus previously taken up within the shoot of the plant can be released back into the system by translocation of nutrients to the rhizomes and flushing (Kadlec & Knight 1996) as well as through decomposition by microbes (Wallace & Kadlec 2008). Harvesting vegetation may aid in minimizing the amount of decomposing vegetation that can add additional phosphorus back into the wetland but may be dependent upon selected species and time of season of harvest. Fraser et. al. (2004) conducted a mesocosm experiment including *Scirpus validus, Carex lacustris, Phalaris arundinacea, and Typha latifolia* and found that *S. tabernaemontani* removed the most amount of TP from the soil when harvested. Harvests in the fall were also less effective at removing phosphorus among all species tested. The low phosphorus content within the shoot of the plant in late-season harvesting indicates a translocation of nutrients to the below ground tissues and surrounding soils (Fraser et. al. 2004). River bulrush (*S. fluviatilis*) has been noted by Tanner (1996) and Mitsch & Gosselink (2000) to have relatively low phosphorus content within the harvestable shoot of the plant around the same time near the end of the growing season (Tanner 1996; Mitsch & Gosselink 2000). These conclusions were subsequently confirmed with data from mesocosm studies (Nairn & Mitsch 2000), in which *S. fluviatilis* was not a recommended plant species for nutrient removal and management strategies.

Similar to timing of harvests in a season, the number of total biomass harvests to conduct in a treatment wetland within a single season may be an additional knowledge gap. A study performed by Jeke et. al. (2018) analyzed the nutrient remediation capabilities by *Typha spp.* in a municipal lagoon by process of harvest. The cattails were harvested at a height of 65 cm above the soil surface as a management technique for flood prevention following the harvest. This study compared the efficiency of nutrient removal from one harvest, a typical management strategy, to performing multiple harvests throughout one growing season. Results showed that performing two harvests within the season reduced the overall nutrient removal rates, indicating one harvest per season as a more effective strategy for phytoremediation practices. Alternate results were found by Keyport et. al. (2018) in which a coastal Great Lakes wetland did not display significant nutrient removal by conducting only one biomass harvest and frequent harvests throughout the season were suggested for optimized nutrient removal. Similarly, a mesocosm study by Amarakoon et. al. (2017) suggested multiple harvests in a single growing season may be a more effective nutrient removal strategy. Multiple harvests may not be a practical management strategy depending on the regrowth of the vegetation present; some species may be well adapted for regeneration following a harvest or disturbance than others. Similarly, research has indicated that harvests in April or November, when macrophytes are typically more accessible with less standing water present, can also be less effective than harvesting in August. However, much of the present research asserts that maintaining water levels for a desired species in wetland systems may be crucial for maximizing P retention for harvests (Cicek et. al. 2006).

Differing conclusions from various studies may be due in part to controlled experiments in mesocosms compared to field site observations. Similarly, conflicting conclusions may be caused by varying environmental conditions in which some species of plants may be well adapted to abundant growth due to weather, soils, and hydrology. Site specific studies and comparisons are necessary in future development of management strategies in order to implement the most effective and efficient nutrient removal by plant harvest. In order to assess these items, research questions were developed with the intention of analysis at an edge-of-farm treatment wetland as well as within a University of Minnesota St. Paul mesocosm experiment. We hypothesized that phosphorus could be removed from the soil by the process of harvesting the vegetation present throughout each site, with decreasing amounts of soluble phosphorus present within the soil after multiple rounds of harvesting. This study aimed to address the following questions:

1. **Can wetland vegetation harvesting remove phosphorus from the soil and reduce legacy phosphorus accumulation?**
2. **What wetland plant species may work best for harvesting in treatment wetlands? Do monocultures or vegetative species mixes achieve higher phosphorus removal rates and efficiencies?**
3. **At what time of the growing season should wetland vegetation be harvested for optimized P removal?**

2.2 Granada Treatment Wetland

2.2.1 Background and Site Description

In the Minnesota River Basin, water quality degradation due to poor agricultural practices has contributed to phosphorus to become a primary concern (Westra et. al. 2007). The Granada, MN treatment wetland is an edge-of-farm constructed subsurface tile drainage wetland that was initially designed and implemented in order to target and reduce the amount of nutrient loading into the nearby tributary, Elm Creek. Nitrogen and phosphorus loads into the creek were primarily from the agricultural fertilizer applications by the landowner. Cover crops and other best management practices were also implemented by the landowner in later years in order to reduce the nutrient loading, as well as allowing the wetland to be built on his land with partnership with the University of Minnesota and the Minnesota Department of Agriculture. Each cell within the treatment wetland is approximately 13.7 m by 26.7 m with wet prairie mixes of vegetation present (Figure 3 and Figure 4). Groundwater wells are located throughout the wetland for continued monitoring in conjunction with the inlet and outlet nutrient concentrations.

Figure 3: Image of the Granada, MN treatment wetland which is an edge of farm treatment wetland operated by the landowner. Surrounding landscape includes cropped land and an adjacent tributary to the Minnesota River, Elm Creek.

Figure 4: Granada, MN treatment wetland vegetation (wet prairie mix) prior to harvest in the fall.

The edge-of-farm treatment wetland implementation strategy proved beneficial in targeting the issue of nutrient loading into Elm Creek, a tributary of the Minnesota River. This design allows for efficient use of the available space at minimal cost, and allows for inflows from the tile drainage present on site (Lenhart et. al. 2016; Gordon 2019). Current (2016) found that phosphorus removal within the wetland improved each year, with vegetation uptake and harvest continuing to increase nutrient removal in each season, with the highest vegetation uptake in 2015. Varied vegetative phosphorus content in some years was likely due to environmental factors such as increased precipitation and flooding (Gordon 2019) which may have resulted in the release of phosphorus from the sediment (Guzner 2016). Phosphorus stored within the sediment of the wetland may be made available or become released, so it is important to understand and study the phosphorus content within the soil. The clay-loam soils present include Coland clay loam and Spillville loam, which are high in clay content with a sandy sub-layer. Inlet and outlet measurements of orthophosphorus allowed for estimates of phosphorus retention within the subsurface of the wetland (Current et al 2016; Lenhart et. al. 2016).

2.2.2 Hydrology

Water volume and related nutrient concentrations and loading rates varied in most years of analysis of the Granada treatment wetland. Inputs to the wetland include precipitation and tile drainage inflows as well as riverbank overflow from Elm Creek in some years. Riverbank overflow from Elm Creek included deposition of particulate forms of phosphorus while the tile drainage gathered through the inlet collected soluble forms of phosphorus. In all years of monitoring, 2018 was the wettest year with approximately 93.3 cm of rainfall. Outflows from the treatment wetland include evaporation, leaching downwards through the soil profile, and through the outlet of the wetland. The wettest years throughout the study were 2015, 2017, and 2018, with the rest of the years experiencing normal to dry conditions (Lenhart et. al. 2019). Water volume going into the treatment wetland was approximately 27,970 m^3 in 2016, 6730 m^3 in 2017, and 21,030 m^3 in 2018. In the outlet of the wetland, approximately 7,680 m^3 was released in 2016, 980 m^3 in 2017, and 13,030 m^3 in 2018. Only about 23% of water inflow flowed through the outlet in a typical year of operation (Lenhart et. al. 2019). Flooding also occurred in some years at the Granada treatment wetland. In 2013, 2014, 2016 and 2018, flooding occurred throughout the wetland due to overflow of Elm Creek which can deliver particulate phosphorus to the treatment wetland as well as upstream concentrations of nitrates and phosphates. In these years, floodwaters were stored within the wetland for approximately 3-4 weeks throughout the season (Lenhart et. al. 2019).

Total phosphorus and orthophosphorus concentrations were measured at the inlet and outlet of the site. In 2017, an average of 0.021 mg/L TP and 0.019 mg/L OP entered the wetland. In 2018, an average of 0.051 mg/L TP and 0.032 mg/L OP entered the site from tile drainage. Nitrate/nitrite load concentrations to the wetland were approximately 5.8 mg/L in 2017 and 8.8 mg/L in 2018 and were contributed to the site via collected tile drainage as well as upstream overflow in some years. Load rates were calculated based on nutrient concentrations and flow volumes to the site. In 2017, TP loading to the treatment wetland was approximately 0.144 kg. In 2018, loading rates at the Granada site were approximately 0.464 kg TP. (Lenhart et. al. 2019).

2.2.3 Methods

In the early years of project development and monitoring efforts, the Granada treatment wetland was intermittently sampled at the inlet, outlet, and at various wells throughout the site. Water samples were measured using ISCO area velocity probe in partnership with pressure transducers. Samples were tested for nitrate and phosphate concentrations (Current et al 2016; Gordon 2019; Lenhart et. al 2019). Previous studies (Current et al 2016; Gordon 2019) analyzed the effectiveness of the treatment wetland and concluded that reductions of both nitrogen and phosphorus were achieved over time as the wetland and wet prairie vegetation became established.

Vegetation in the wetland was sampled at various locations using a single square meter (0.0001 ha) sampling size within each cell. Harvesting of vegetation was attempted at near soil surface level in the fall of each season and tested for nutrient content. Soil samples were gathered within each of the cells each year. Soil augers were used to sample from the top 6” in most years and 12” or more in some years. Soil samples were tested and analyzed for Olsen-P content for analysis of bioavailable P content throughout the site.

2.3 St. Paul Mesocosm Study

2.3.1 Background and Site Description

In the spring of 2016, plans began for a site on the St. Paul campus at the University of Minnesota to begin a mesocosm project for the Department of Bioproducts and Biosystems Engineering (BBE). Similar mesocosm experiments by Mitsch (2002; 2015; 2017) were attempted to be replicated or built similar to for this study, including the use of multiple tanks with specified nutrient loading that replicate treatment wetlands. The site was intended to be used by future undergraduate students, graduate students, and doctoral candidates alike in partnership with research professors for the better understanding of wetlands and their biota. By the end of 2016, 10 mesocosm tanks were set up in the garage of the Biosystems and Agricultural Engineering building. Reed canary grass was of primary concern during this period of time. By May 2017, thirty mesocosm tanks were installed in an excavated plot on the north side of the St. Paul campus. For the 2017 growing season, prairie cordgrass (*Spartina pectinata)*, Canada bluejoint grass (*Calamagrostis canadensis*), and tussock sedge (*Carex stricta*) were the primary focus and so were planted singularly in various tanks via plug installation. Soil samples and plant harvesting allowed for analysis of phosphorus uptake and content for October 2017. Plant phosphorus content was analyzed in one of the BBE labs on the University of Minnesota campus by student workers, and results varied amongst the plants that were tested.

The mesocosm site is located on the University of Minnesota Twin Cities St. Paul campus. It is located on the northern side of the campus near agricultural and experimental research stations. The 100-gallon mesocosms tanks are made of black high-density structural resin stock tanks measuring 53” x 31” x 25” on the outside. The tanks were placed inside an excavated portion of the plot (Figure 5). Following excavation, weeds became extremely prevalent. For the 2018 season, the site was hand weeded, spot sprayed, and then covered with weed fabric surrounding the entirety of the mesocosms. The weed fabric was put in place using ground staples and gravel, and a wire fence was put in place in order to protect from wildlife interference. The bottom 11 inches of each mesocosm were filled with crushed sandstone in previous years for previous studies. For the 2018 season, the top 5-6 inches of the tanks were shoveled out to remove the top layers of plant material and debris, as well as to create a fresh start for the new plants. Additional soil from on-site was placed into the top layers of the mesocosms.

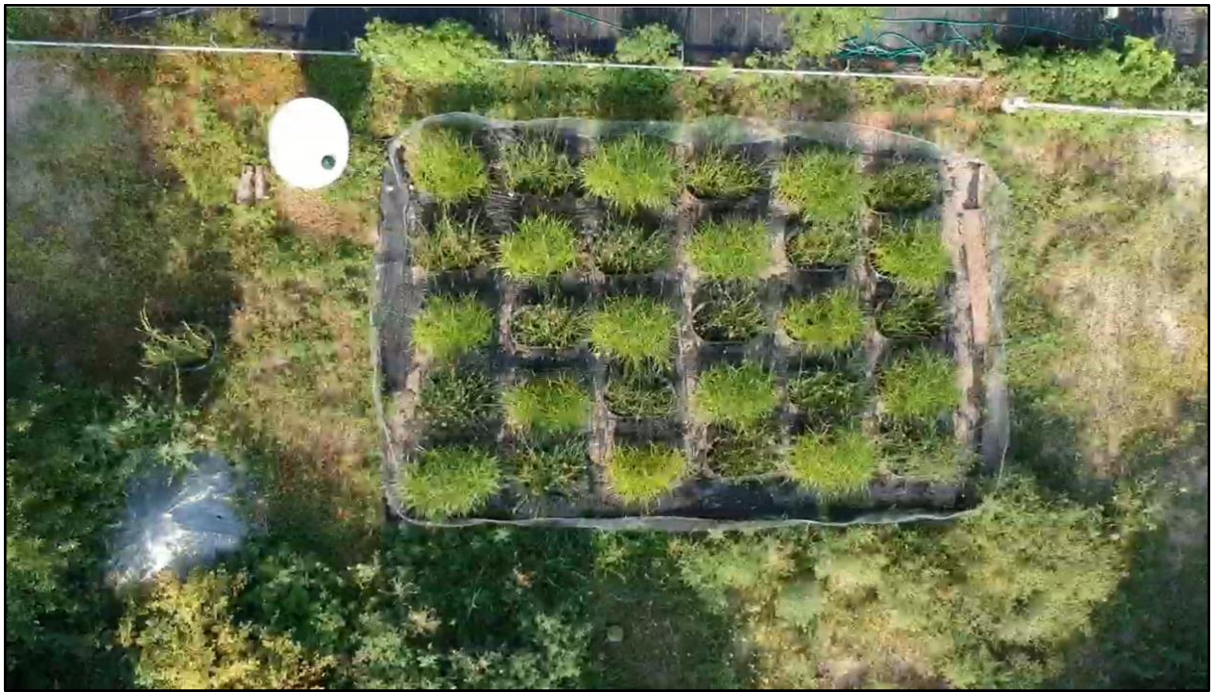

Figure 5: 2018 aerial view of the mesocosm site located in St. Paul, MN. This site includes 30 tanks; of those displayed, 15 are river bulrush, 15 are softstem bulrush.

Selection of particular wetland species for this study relied heavily on commonly occurring wetland plants that have little to no known major threats to survival, ability to grow rapidly and successfully in Minnesota, and tolerant of prolonged inundation in wetland settings. River bulrush (*Bolboschoenus fluviatilis*) and softstem bulrush (*Schoenoplectus tabernaemontani*) were ultimately chosen for the 2018 mesocosm experiment. For this study, 180 softstem bulrush and 180 river bulrush plugs were ordered from Prairie Restorations Inc. 12 plugs were placed in 3 rows of 4 inside the mesocosms at equal distances apart. 15 tanks contain softstem bulrush and 15 tanks contain river bulrush. Following the installation of soil, plants, and water to the mesocosm tanks, each tank had a 0.5 cm-1.0cm hole drilled into the side of the tank. This allowed for a maintained water level of approximately 1”-2” above the soil line. If tanks experienced drawdowns for any reason, water was added to the tanks from an on-site hose to meet the required water level standards. Daily checks were performed on the mesocosms to aid in the maintenance of proper water levels. To provide consistency and prevent potential interference from a variety of environmental factors that may influence outside mesocosm experiments, every other tank was planted with river bulrush (*Bolboschoenus fluviatilis*) or softstem bulrush (*Schoenoplectus tabernaemontani*). Consequently, river bulrushes were placed in odd numbered tanks and softstem bulrush were placed in even numbered tanks (Figure 6).

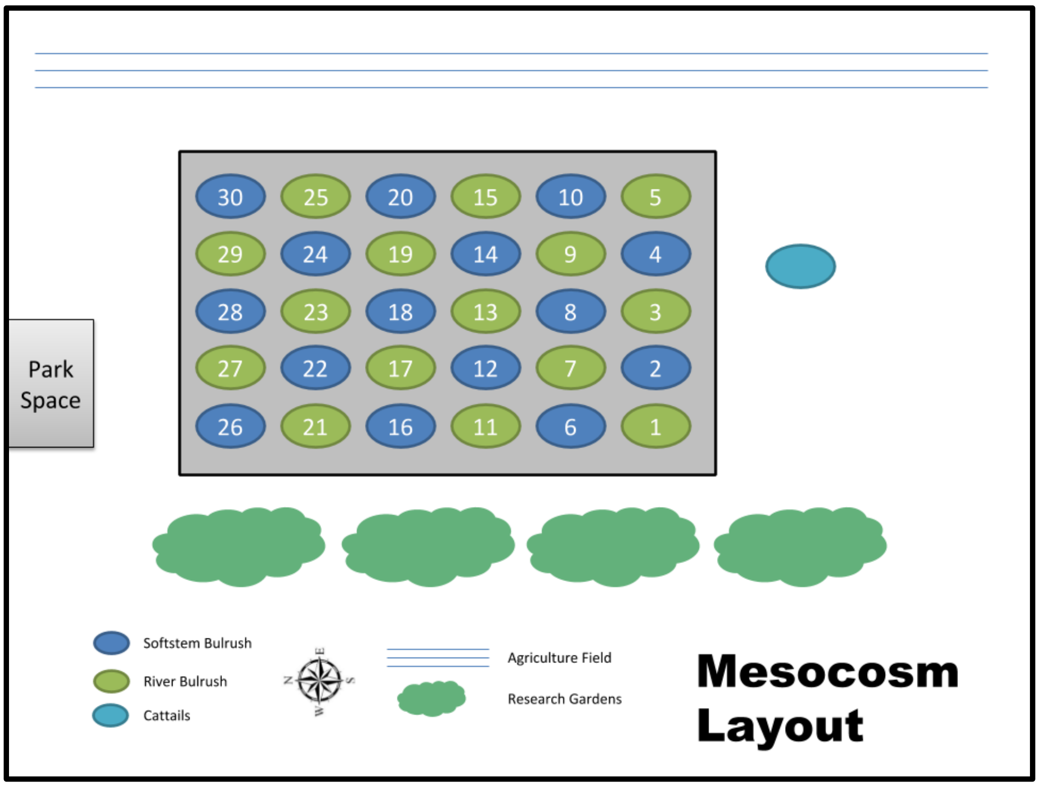

Figure 6: 2018-2019 mesocosm experiment in St. Paul, MN that displays the layout of the plant distribution throughout the site.

2.3.2 Methods

Following the installation of the bulrush plugs in each of the tanks, weekly applications of fertilizer enriched water were supplied to each individual mesocosm tank beginning in June each year. Weekly fertilizer additions lasted for the entire season (approximately 12 weeks). Using water soluble MiracleGro© All-Purpose Plant Food, a calculation was determined in order to supply .06 mg/L of soluble phosphorus to the tanks, which mimics actual agricultural drainage and nutrient loading to the Granada treatment wetland and other agricultural watersheds. A 210-gallon tank on site was filled weekly with the fertilizer mixture. Using approximately 120 gallons of water, 0.7568 grams of the plant food was mixed to create a homogenous solution. The fertilizer solution was created and applied each week during the growing season to all 30 mesocosm tanks. By process of using and filling 5 gallon buckets, 3.75 gallons of the solution were applied evenly to each of the mesocosms by graduate and undergraduate workers. Loading concentrations were approximately 0.72 mg/L per season, which resulted in approximately 10.22 mg of P applied to each mesocosm tank per season.

Plant samples were selected at 5 different dates throughout the growing season of 2018 and 2019 (see Appendix I). 6 tanks were selected at random each week including 3 tanks of each of the types of plant species present. Within each tank, 3 individual samples were chosen at random throughout the tanks and analyzed at RAL labs on the University of Minnesota campus. Similar experimental design was implemented for the 2019 season; both softstem bulrush and river bulrush grew rapidly beginning in early June 2019 following the 2018 fall harvest. Soil samples were taken throughout the mesocosm tanks in the fall of 2017 and tested for Olsen-P. In the 2018-2019 seasons, soil samples were taken at the beginning (June) and end (October) of each season within each tank and tested for Olsen-P content. Samples were gathered within the upper 6 inches of soil within the tanks.

2.4 Data Analysis

2.4.1 Granada Treatment Wetland

The early seasons of monitoring at the Granada, MN treatment wetland, beginning in 2013, assessed nutrient reduction throughout the site (Gordon 2019). Nutrient retention, including nitrogen and phosphorus, increased as the wetland became more established over time with 2015 including the highest retention rate (Current et. al. 2016). Inputs to the wetland included an average loading rate of TP into the wetland of approximately 0.144 kg in 2017 and 0.464 kg in 2018 (Gordon 2019). In 2017-2018, results of harvested biomass from each individual cell were compared to understand which cell removed the most amount of P from the wetland. In 2018, Cell 1 and Cell 3 removed similar amounts of P per sample (DW). Phosphorus removal per biomass was highest within Cell 1 in 2017 but there was no significant difference in P removal within each individual cell between both seasons (Table 3). The 2018 season vegetation removed more P on average than the 2017 season. Estimated total phosphorus removed from the wetland area in 2017 by vegetative harvest was approximately 2.3 kg/ha; in 2018, estimated phosphorus removal within the wetland was approximately 3.2 kg/ha. In all of the seasons of sampling, 2014 had the highest estimated removal rate by harvested biomass with approximately 4.5 kg/ha of P (Figure 7) and approximately 0.44 kg of P potentially removed by vegetation throughout the wetland area (0.1 ha). In 2017, harvested vegetation in the Granada site removed approximately 0.24 kg of P. In 2018, the harvested vegetation removed approximately 0.32 kg of P. Following establishment of the wet prairie seed mix in 2013, vegetative uptake of P was a sufficient pool of P within the treatment wetland.

Granada Treatment Wetland Vegetation Harvests				
Year	**Cell**	**Dry Weight (DW) (kg)**	**P %**	**P(kg)/DW(kg)**
2017	**Cell 1**	0.33407	0.08	0.03
	Cell 2	0.14048	0.07	0.01
	Cell 3	0.53433	0.06	0.03
	Total	**1.00888**		**0.07**
2018	**Cell 1**	0.32024	0.13	0.04
	Cell 2	0.29302	0.09	0.03
	Cell 3	0.23133	0.13	0.03
	Total	**0.84459**		**0.10**

Table 3: Granada treatment wetland vegetation harvests in 2017 and 2018.

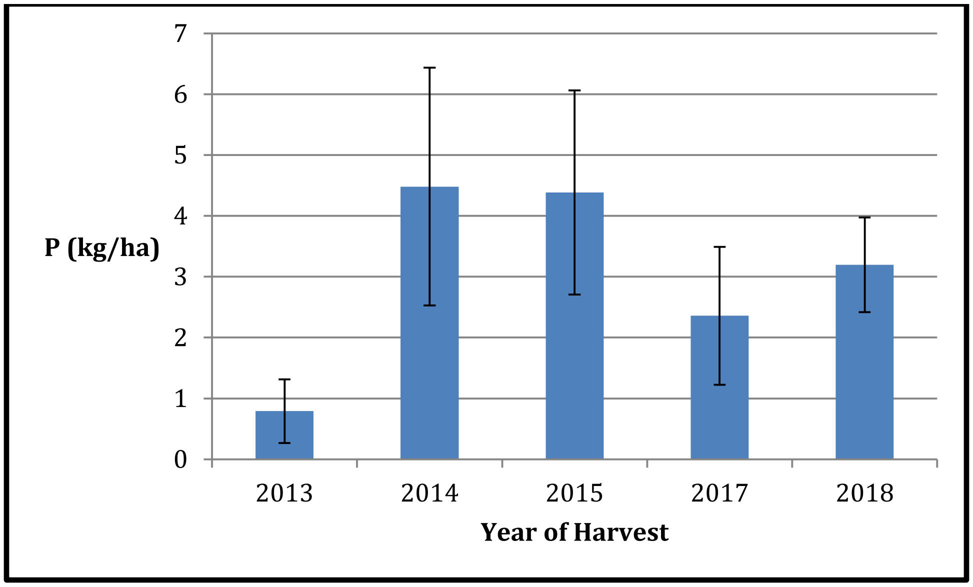

Figure 7: Estimated Granada treatment wetland end of season harvest removal rate (kg) of P per hectare.

2.4.2 Soil Phosphorus in Granada Treatment Wetland

Loading of P to the wetland was directly sourced from tile drainage and associated fertilizer inputs to the adjacent farm. Although the treatment wetland was not initially designed for phosphorus retention, P removal was witnessed within the cells since wetland establishment. In 2012, soil samples within the wetland contained approximately 10.6 mg/kg. In 2018, soil P was reduced to approximately 1.6 mg/kg. Similarly, Olsen P measurements were collected to assess the bioavailable forms of P. In 2014, average Olsen P was 3.9 mg/kg. In 2019, soil Olsen P was reduced to an average of 2.8 mg/kg. Shallow soil samples (upper 6”) within the wetland displayed higher Olsen P values than deeper soil samples (below 6”) (Table 4). In 2014, Cell 3 had higher phosphorus content (6.25 mg/kg) compared to Cell 1 (2 mg/kg). In 2019, Cell 3 continued to display higher phosphorus content (4 mg/kg) than Cell 1 (2 mg/kg).

Average Olsen P (mg/kg) Granada Treatment Wetland			
2014	***Cell 1***	***Cell 2***	***Cell 3***
Shallow (upper 6")	2	3.5	6.25
Deep (6" or lower)	2.5	3	3
2019	***Cell 1***	***Cell 2***	***Cell 3***
Unharvested	3	3	4
Harvested	2	2	4

Table 4: Granada, MN treatment wetland was sampled in 2014 for phosphorus content at varying depths. Upper portions of the soil sampled displayed higher overall Olsen P content than deeper regions. Unharvested portions of the wetland in 2019 displayed higher Olsen P content than the harvested portions.

Granada Treatment Wetland Soil P	
Year	**Mean Soil Olsen Phosphorus (mg/kg)**
2012	**10.6**
2014	**6**
2018	**1.6**

Table 5: Granada treatment wetland soil P average throughout years of treatment wetland operation. Data from Gordon 2019.

2.4.3 Mesocosm Experiment

In the fall of 2017 and 2018, the entire mesocosm tanks were harvested and measured for biomass and phosphorus content. In 2017, Canada blue joint grass (*Calamagrostis canadensis)* had the lowest biomass overall but had higher phosphorus content (2575.5 ppm) compared to the other species, with almost twice the amount of phosphorus as prairie cordgrass *(Spartina pectinata)*. In 2018, softstem bulrush (*Schoenoplectus tabernaemontani)* had higher P content but a lower overall biomass. This resulted in similar amounts of P removed between softstem and river bulrush species; approximately 0.0016 kg (1.6 g) of P was harvested per tank on average (Figure 8). Phosphorus removal per harvested biomass varied per year and per species; a range from 0.00022 kg (0.22 g) of P to 0.00111 kg (1.11 g) of P were removed within the years of sampling.

Sampling of plants within the tanks at various times throughout the season displayed a visual representation of when phosphorus content within the shoots of the plants was at a maximum, which may indicate an optimal time for harvesting. Biomass and phosphorus content may both influence the amount of phosphorus that may be removed per harvest (Figure 11). In 2018, phosphorus content within the shoots of both species of plants was highest in the early season (June-July) and slowly decreased over time towards the fall (September-October) (Figure 9). River bulrush initially contained higher P content in early season samples with an average 0.82% P content compared to 0.76% P in softstem, but softstem bulrush retained more P later in the season with .3% phosphorus content compared to 0.25% in river bulrush.

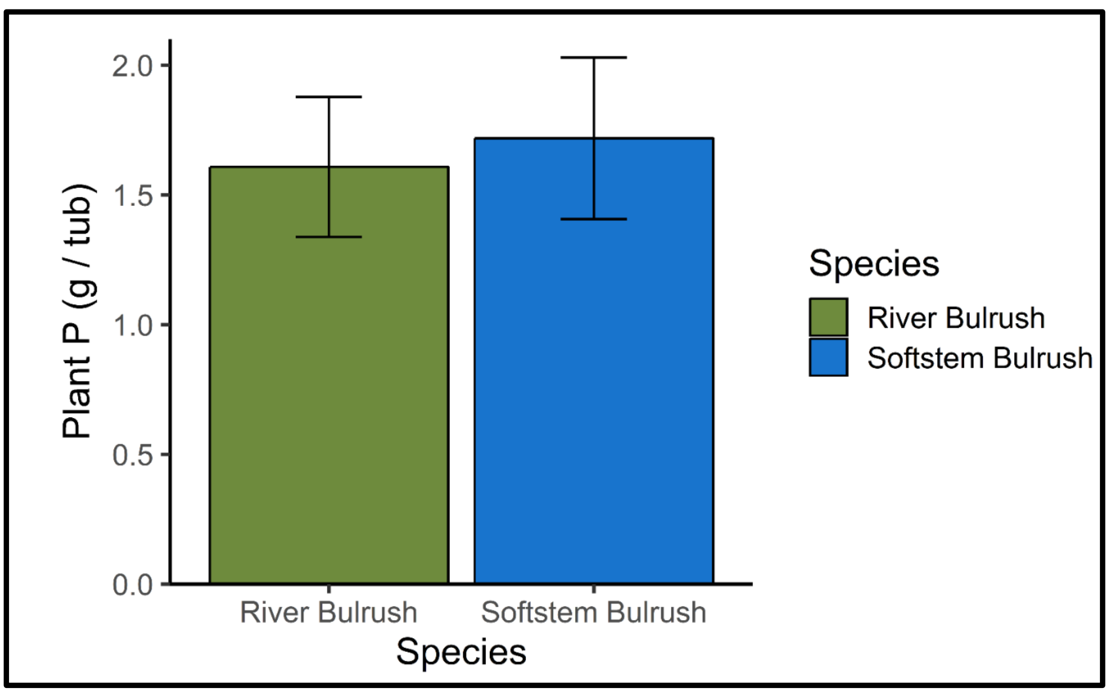

Figure 8: Comparison including ANOVA analysis indicating the plant uptake of the 2018 mesocosm experiment (softstem and river bulrush).

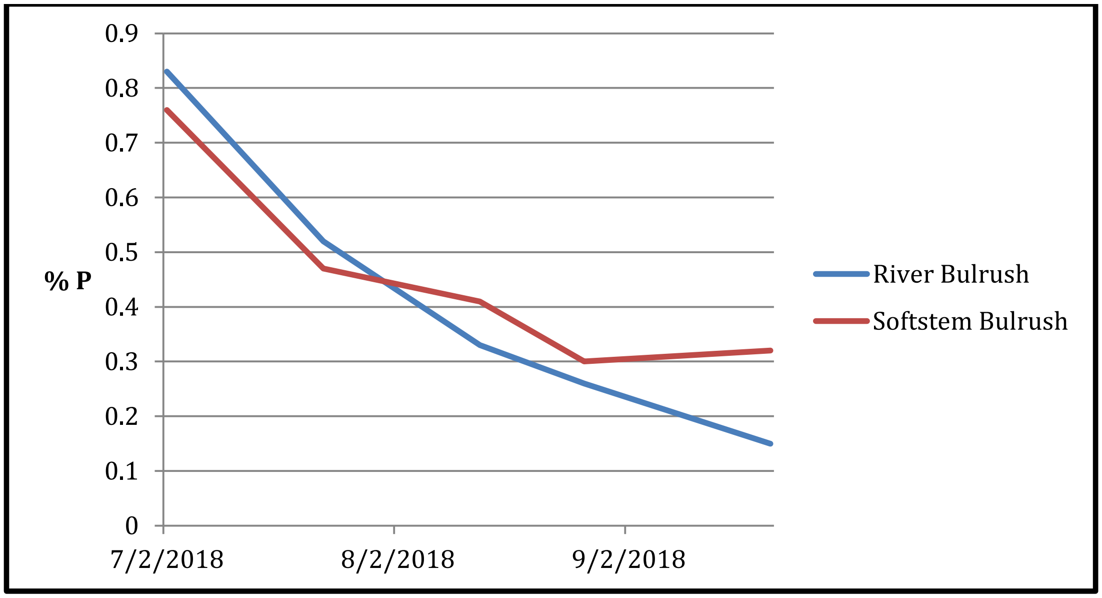

Figure 9: Describes the phosphorus content (%) in the 2018 season mesocosm experiment between the two plant species (softstem bulrush and river bulrush).

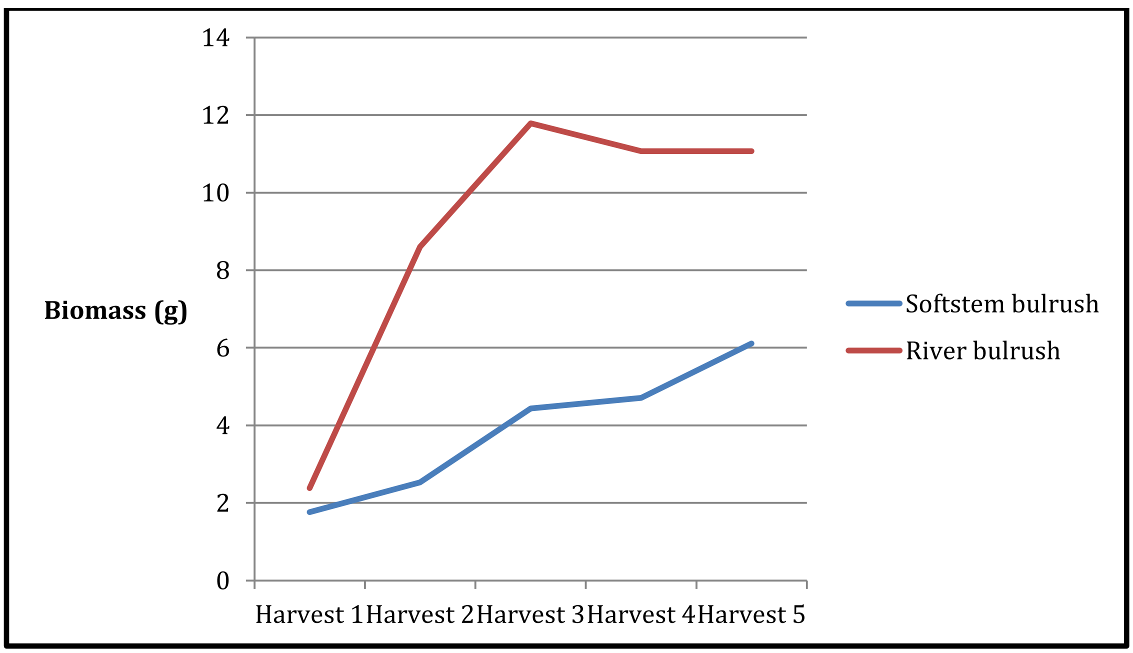

Figure 10: Describes the biomass in the 2018 mesocosm experiment between the two plant species.

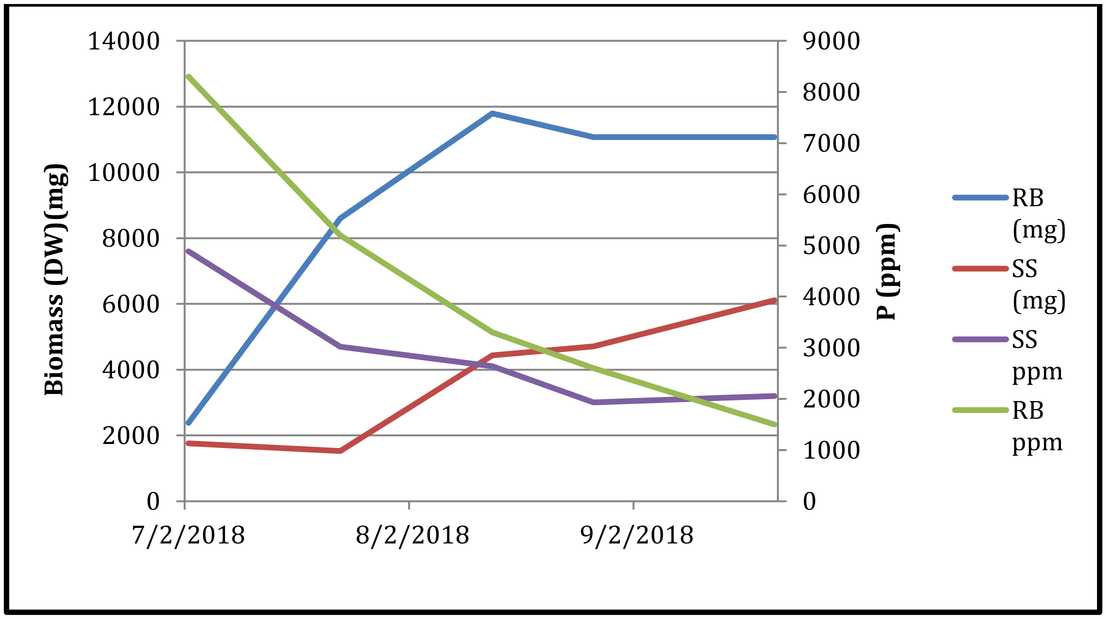

Figure 11: Describes the relationship between the 2018 mesocosm species biomass (mg) and P content (ppm).

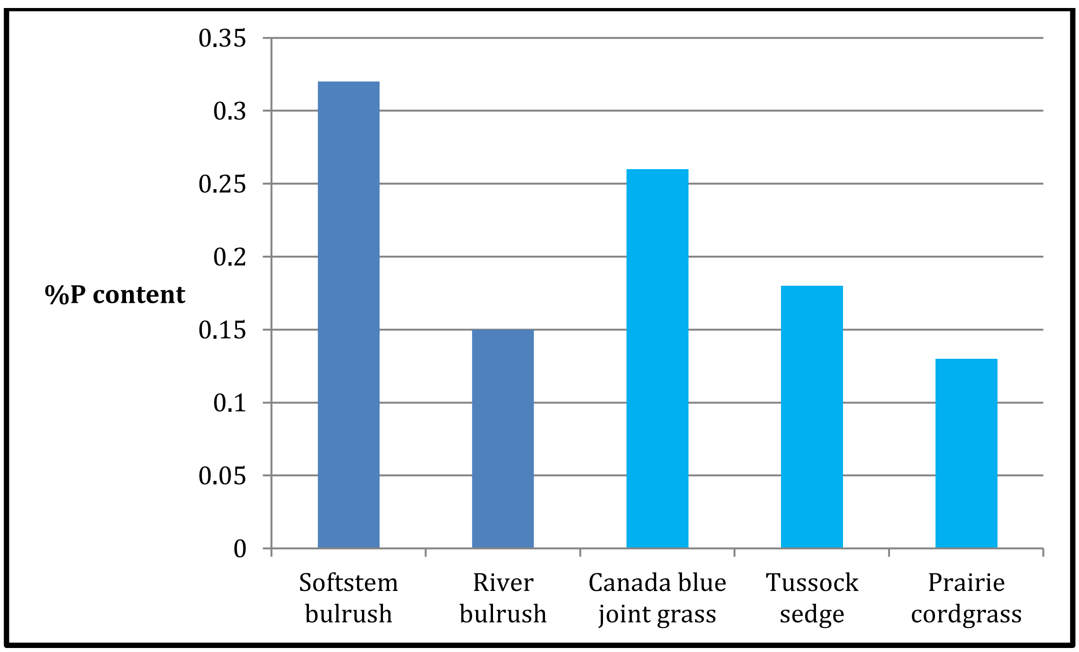

Figure 12: Mesocosm experiment comparison at the end of season harvest within the 2017-2018 seasons with varying phosphorus content at the end of season harvest (fall) between each species.

2018 Mesocosm Experiment – St. Paul, MN						
	River Bulrush *Scirpus fluviatilis*			**Softstem Bulrush** *Schoenoplectus tabernamontani*		
	Dry Weight (DW)(g)	**% P Content**	**P(g)/DW(g)**	**Dry Weight (DW)(g)**	**% P Content**	**P(g)/DW(g)**
1	2.38	0.83	0.02	1.76	0.76	0.01
2	8.6	0.52	0.04	2.53	0.47	0.01
3	11.79	0.33	0.04	4.44	0.41	0.02
4	11.07	0.26	0.03	4.71	0.3	0.01
5	11.07	0.15	0.02	6.11	0.32	0.02

Table 6: The 2018 mesocosm experiment results including dry weight and phosphorus content at each sampling date (June through October).

2.4.4 Soil Phosphorus in Mesocosm Experiment

Mesocosm Olsen-P content was analyzed to determine phosphorus reductions as a result of plant harvesting. Reductions of Olsen-P in the soil were observed from 2017-2018, as well as within the 2018 beginning and end of season sampling. Following the 2017 and 2018 harvests, soil Olsen-P content was reduced in every tank ranging from 1 mg/kg to 20 mg/kg per tank reduction (mean reduction of 7.9 mg/kg of Olsen P with a standard deviation of approximately 5.7 mg/kg), which is approximately a 1%-50% P reduction range (see Appendix II). Loading was approximately 10.2 mg of P in 2018 while removal for the season was on average 33.3 mg/kg P per tank.

Olsen-P reductions were witnessed in the 2017-2018 season within all of the mesocosm tanks by process of plant uptake and harvest, with softstem bulrush tanks removing more Olsen-P on average than river bulrush tanks. However, further statistical analysis including a T-test indicated that there was not a significant statistical difference in Olsen-P reductions at the 5% level. Total phosphorus was analyzed in 2018 soil samples, as well. In 2018, average Total Phosphorus (TP) content in the softstem bulrush tanks was 736.5 mg/kg and 847.9 mg/kg in the river bulrush tanks. Total phosphorus reductions were not measured in the short timeframe of this study.

2.5 Discussion

2.5.1 Vegetation

Successes

The comparable phosphorus content in both the mesocosm and field studies indicate an adequate pool of phosphorus within shoots of wetland vegetation that may be harvested annually from treatment wetlands for enhanced removal of phosphorus. The Granada treatment wetland wet prairie vegetation mixes that were present throughout each of the cells displayed lower phosphorus content at end of season harvest than the mesocosm tanks; this may indicate influence from a controlled mesocosm experiment and ideal conditions provided for the vegetation compared to field sites. Softstem bulrush (*Schoenoplectus tabernaemontani)* displayed lower biomass than river bulrush *(Scirpus fluviatilis)* (Figure 11), but each mesocosm species removed similar amounts of phosphorus from each tank at approximately 0.0016 kg (1.6 g) of P removed per mesocosm tank at the end of season (Figure 8). These results indicate that biomass or dry weight (DW) is not directly correlated to overall phosphorus removal potential when harvested and that phosphorus content within the vegetation may be a more indicative factor. *Scirpus fluviatilis* and *Schoenoplectus tabernaemontani* may be suitable options for treatment wetland vegetation, particularly within treatment wetlands that may maintain anaerobic conditions throughout the growing season.

In 35 wetland species studied, phosphorus concentrations have been witnessed to range from 0.08% to 0.63% (Boyd 1978; Kadlec & Knight 1996), which was comparable to the mesocosm vegetation P content which ranged from 0.13% to 0.32% in the two seasons of analysis (Figure 12). Grosshans & Grieger (2013) assessed the phosphorus removal capacity of cattails (*Typha spp.*) in Canada; their results indicated that phosphorus content per biomass averaged around approximately 0.087% to 0.11% at peak growth in August. At a similar time in the growing season, results showed that the St. Paul mesocosm species contained approximately 0.3% P. Biomass was not witnessed to be a direct indicator of phosphorus content or uptake abilities per the species studied in our experiments, however time of season of harvest was found to be a direct factor corresponding to phosphorus content within the shoots of plants studied (Grosshans & Grieger 2013; Kadlec & Knight 1996). Optimal time of season for harvesting treatment wetland plants in Minnesota was found to be in late summer (August) per the results of our mesocosm experiment.

Each species harvested each year displayed different results. Between the 2017 and 2018 end of season harvests within the mesocosm experiment and the Granada treatment wetland, phosphorus removal ranged from approximately 0.00022 kg (0.22 g) of P to 0.00111 kg (1.11 g) of P, while biomass per collected samples ranged from approximately 0.084 kg (84 g) to 0.53 kg (530 g). In 2017, the loading rate at the Granada site was approximately 0.144 kg TP. Harvesting of vegetation in this year removed approximately 0.24 kg of P. These results indicate a high potential for legacy P removal by harvesting the vegetation and prevention of further accumulation of the nutrient. In 2018, the Granada treatment wetland had a loading rate of phosphorus of approximately 0.46 kg TP. In this season, P removal by harvesting the vegetation was approximately 0.44 kg. The high removal rate compared to the P loading rate at this site indicates that harvesting the wet prairie species mix may aid in reducing legacy phosphorus accumulation over time if harvests are conducted annually.

The Granada treatment wetland is located within an agricultural watershed and could be a suitable environment to implement softstem bulrush within any or all of the cells present. Nutrient uptake or P content by plants throughout a growing season determines growth and survival in wetland ecosystems. Per the Granada treatment wetland area in which 0.11 ha to 0.22 ha are flooded seasonally, softstem bulrush could have the potential to remove between 1.22 kg (2.7 lbs) to 2.45 kg (5.4 lbs) of phosphorus by process of harvesting the vegetation at the end of season (September-October) when water levels may be drawn down. The mesocosm species monocultures could potentially remove up to 12 kg/ha while the wet prairie species mix removal rates were estimated at approximately 3.2 kg/ha. The large difference in removal rates may be related to the assumptions associated with monoculture experiments including ideal conditions for the selected species. This may also be directly related to the varying growth rates among species. The higher phosphorus content per biomass in earlier season sampling (June-July) indicates that earlier season harvests may be more effective for removing higher amounts of phosphorus as compared to typical harvests in the upper Midwest (September-October). The nutrient uptake or P content displayed in the fall indicates a translocation of those essential nutrients back to underground tissues (Tanner 1996). The results from these experiments ultimately provided similar conclusions to Fraser et. al. (2004) in which phosphorus removal was significantly lower in October (fall) than in earlier season harvests.

Challenges

Wet prairie mixes are more commonly used in edge-of-farm treatment wetlands than a monoculture species. The Granada treatment wetland plant species included a variety of native (e.g. prairie cordgrass, *Spartina pectinata*) and invasive species (ex: reed canary grass, *Phalaris arundinacea*). Competition by invasive species and other plant types throughout the wetland may have impacted the nutrient uptake rates each season within this site. Establishment of unintentional or invasive species can be an issue for recently disturbed lands such as constructed treatment wetlands. Maintenance of selected vegetation may be a challenge for land managers throughout edge-of-farm treatment wetlands in future years following the installation of the wetland.

Flooding or prolonged saturation in the spring and throughout some of the growing season may have impacted vegetation growth in some cells which may have resulted in varying phosphorus removal and lower P removal in 2017 (2.3 kg/ha) than in 2018 (3.2 kg/ha). The practicality of harvests in earlier season (June-July) is largely dependent upon the accessibility of the area, which includes water levels. Haying or mowing equipment may only be useful when water levels have subsided; draining of the wetland in earlier season would be a necessary task in some areas in order to gain access to the area for vegetation harvesting. Similarly, dense vegetation or flooding may prevent the harvesting of vegetation by hand, which may be an option for small scale treatment wetlands or buffer areas. Tanner (1996) found that at the end of season harvest (fall), a large portion of phosphorus had translocated to below ground tissue from the shoots of the selected species. This large pool of nutrients within the root systems of wetland plants can be difficult to quantify or harvest and so may not be a practical solution for nutrient remediation practices in most edge-of-farm treatment wetlands. Optimized removal of phosphorus by harvesting vegetation is recommended to take place earlier in the growing season if possible.

2.5.2 Soil

Successes

Soil samples were gathered in 2014 at varying locations throughout the Granada treatment wetland. The northwest portion of the wetland (Cell 3) displayed higher overall Olsen P content at varying depths. Shallow portions of the soil displayed higher overall phosphorus content. In 2019, soil samples were gathered in each of the cells throughout the treatment wetland where results displayed lower average Olsen P content throughout the cells than what was previously observed in 2014; 2019 Total P was estimated to be approximately 1.6 mg/kg which is a large reduction compared to the 2014 samples in which total P was estimated to be 6 mg/kg. The reduced Olsen P and Total P content over multiple years may indicate a reduction through plant harvesting but could also be attributed to the flushing of phosphorus to the creek from the wetland in some years due to flooding. Harvested portions of the Granada treatment wetland displayed lower Olsen P than unharvested portions of the wetland in 2019 (Table 4). In the mesocosm experiment, only approximately 10.22 mg of P were loaded in 2018 while an average removal rate of 33.34 mg/kg Olsen P was achieved that season by harvesting bulrush species. This indicates a removal of stored phosphorus within the soil by harvesting wetland species. The reductions in Olsen-P throughout the mesocosms as well as the reductions in the Granada treatment wetland indicate an effective practice for the removal of legacy and soluble forms of phosphorus from the soil by harvesting the vegetation (Sharpley et. al 1994) throughout the wetland annually. Treatment wetlands that gather tile drainage or surface water runoff from nearby agricultural land may accumulate or gather phosphorus within the soil over time (Penn et. al. 2015). The results from the mesocosm and edge-of-farm treatment wetland studies display significant potential for removal of excess phosphorus within the soil in agricultural watersheds by harvesting treatment wetland vegetation. Annual harvesting of vegetation over multiple seasons may reduce phosphorus content, both soluble forms and legacy forms over time, within soil and decrease the risk of flushing phosphorus at times of flooding.

Challenges

Phosphorus loading from the wetland was witnessed in 2016-2017 to the adjacent Elm Creek due to high rainfall and flooding in the area. A possible cause for flushing of P from the wetland to Elm Creek could be the high amounts of soluble forms of phosphorus (Olsen P) (Jordan et. al. 2006; Mallarino et. al. 2013) found previously throughout the area from fertilizer inputs sourced from the adjacent farm. The total hydraulic loading rate to the Granada treatment wetland from 2013-2016 was approximately 12.6 $m/y^{-}1$(Gordon 2019); in partnership with the continued flooding in some years and fertilizer inputs to the field, the wetland released phosphorus to Elm Creek in some years of monitoring may have also contributed to the flushing of P from the wetland to Elm Creek.

Kim & Geary (2001) found in a nutrient uptake study of *Schoenoplectus mucronatus* (bog bulrush) and *Baumea spp.* that *S. macronatus* had higher P uptake, which displayed similar results to our study bulrush comparison. However, this study also concluded that harvesting of these plants may not be a practical option when considering all factors per site. The harvesting of the vegetation may mean an unintentional disturbance of the soil which can release soluble forms of phosphorus back into the system (Koski-Vahala & Hartikainen 2001). Soil disturbance and erosion are factors that can lead to release of phosphorus and can ultimately negatively impact water quality or alter the intended remediation efficiency of the treatment wetland, therefore it is essential to consider this factor when determining if treatment wetland vegetation harvests are an optimal solution for nutrient removal.

Wetland phosphorus processes can be complex and dependent upon many variables. Sorption of phosphorus and release or desorption of phosphorus may unintentionally occur at different times of the season or under specific environmental conditions (ex: flooding) which may provide different results for reduction rates and effectiveness in treatment wetlands. Similarly, the overall phosphorus balance may be difficult to quantify. Various reservoirs including the soil, water, and vegetation within a constructed treatment wetland may all result in different levels of phosphorus. It is necessary to include monitoring schemes and analysis for all forms of phosphorus within all reservoirs in order to accurately describe the functions and efficiency of nutrient capture and removal by vegetation harvesting.

2.5.3 Economic Analysis

Although effective removal of phosphorus can be achieved through treatment wetland implementation and harvesting of biomass, the practicality and associated costs may be a primary concern for land managers. Treatment wetlands may be more efficient at nitrogen removal than phosphorus removal or sediment retention (Lenhart et. al. 2017). Targeted nutrient or sediment removal is often attributed to the ratio of watershed basin to treatment wetland size. Denitrification rates within treatment wetlands may be dependent upon denitrifying bacteria while phosphorus removal may be dependent upon plant uptake abilities and if harvests are conducted (Lenhart et. al. 2017). Although nutrient retention can be achieved, the initial cost of implementing a treatment wetland can be costly and can range from $5,886.46 to $8,191.45/acre of wetland treatment area (Lenhart et. al. 2017). Another BMP that land managers may consider is the implementation of saturated buffer strips which are another useful practice for treating agricultural drainage. Saturated buffer strips are often used to target areas of high nitrogen loading but can also contain wetland species that may also contribute to phosphorus removal by plant uptake. Costs associated with saturated buffer strips can include the initial cost of implementation which can be up to $3,000 to $5,000 and an additional $10-12 per linear foot of tile pipe installed (Lenhart et. al. 2017). Although initial costs of implementing a saturated buffer strip may be lower than installing a treatment wetland, other services provided by treatment wetlands may make them the optimal choice for land managers. Additional services that may be challenging to quantify in a treatment wetland can include flood retention services, habitat for wildlife, as well as sediment retention at times of flooding or overflow. Land managers should consider initial implementation costs, maintenance costs, and nutrients of high concern for the selected watershed area when considering BMP placement on their property.

2.5.4 Future Research Potential

The St. Paul mesocosm study and Granada treatment wetland study provided useful information for distinguishing potential vegetation types and harvest date suggestions that can be implemented in treatment wetlands or buffer strips throughout agricultural settings in the upper Midwest. Unforeseen results from the mesocosm experiment included the invasion of weeds throughout the mesocosm tanks; *S. fluviatilis* displayed an increased perceptibility to invasion by other weed species, as well as seed dispersal and establishment of *Schoenoplectus tabernaemontani* within a few of the river bulrush tanks by the second growing season. This indicates that river bulrush may be a less successful monoculture system if selected for establishment in treatment wetlands. Accessibility of wetland vegetation also proved to be a challenging consideration within these studies. Increased precipitation in recent years throughout parts of Minnesota resulted in higher water levels into the fall season. Accessibility of wetland plants for harvesting could be a challenge in future years due to the unpredictability of a changing climate. Higher water levels or flooding for longer periods of time throughout the upper Midwest in recent years may cause further issues for treatment wetland effectiveness (Casanova and Brock 2000).

Tanner (1996) noted the large nutrient accumulations in below ground tissue as well as above ground tissue at different times of the season. End of season nutrient allocation in below ground tissue and nutrient translocation with the soil may be areas of research that need to be further explored in future years. Currently, accessing the below ground tissue for harvest can be a challenging and costly practice that not all land managers may consider for their treatment wetland. However, accessing this large pool of nutrients in the fall when water levels have subsided may be an area for researchers to explore for effectiveness of nutrient remediation in constructed treatment wetlands.

Research gaps which future studies should consider the difficulties associated with scaling up mesocosm experiments to in-field situations due to external or environmental factors (Mitsch 2002). Environmental factors such as shading, interference from animals or other people, flooding or drought, and many others may result in differing conclusions from mesocosm and in-field studies such as edge-of-farm treatment wetlands. It may be difficult to replicate or mimic field conditions in some mesocosm experiments; for example, flooding in field conditions in early spring may inhibit growth for some wetland plants (Casanova & Brock 2000). In mesocosm experiments, protection from flooding may be provided and may allow growth of the wetland plant species at an earlier date. Mitsch & Ahn (2002) and Mitsch et. al. (2015) conducted mesocosm experiments in Ohio and Florida in regions where there is a demand for nutrient remediation strategies in impaired areas. Some challenges with mesocosm experiments include the initial cost of implementation as well as relating the findings to larger field study sites. Mesocosm experiments aid in the ability to closely monitor a small set of variables under specified conditions. Field conditions that treatment wetlands are placed in can be highly variable and difficult to replicate in some cases and should be considered when assessing mesocosm experiments in future studies.

Chapter 3: Nutrient Reduction by Treatment Wetlands at Different Scales in Agricultural Watersheds of Minnesota

3.1 Introduction

The North Ottawa Impoundment (NOI) is a large-scale storage impoundment located in northwestern Minnesota that was initially constructed for capturing flood water from the Red River. Treatment wetland services were monitored within the impoundment over the years following installation of the 776.99 ha (8 km^2.) impoundment. Further assessment of nutrient removal abilities within large scale floodwater storage structures such as the NOI may aid in future management decisions. Similarly, smaller scale treatment wetlands such as the Granada treatment wetland may be assessed for nutrient reduction benefits and native vegetation nutrient removal potential. Comparison and further understanding of numerous wetland areas of differing scales across the state may be beneficial for land managers.

Floodwater storage impoundments are constructed cells designed to store water and minimize flood damages in low-gradient landscapes. Understanding the water quality impacts of these storage systems is important today because there are large monetary investments and allocations of land for the construction of impoundments throughout Minnesota. As floodwater storage impoundments are becoming increasingly more common as a tool for water management in the Red River Basin, there has been an increased concern and interest over the potential for additional water quality benefits provided by these structures. An assessment by Ellis (2011) of freshwater storage impoundments noted that these systems can be less stable than natural freshwater lakes or wetlands and may be due in part to the high rates of nutrient capture and susceptibility to pollution. Similarly, floodwater storage impoundments are more susceptible to variation in water levels throughout a given season. Floodwater storage impoundments have been noted to serve many purposes including habitat for appropriate wetland vegetation and waterfowl species while also contributing to the stagnation of floodwaters and collection of sediment (Ellis 2011). Constructed wetland ponds historically have been utilized to capture floodwaters and have also displayed the potential to capture sediment and phosphorus, including the Granada treatment wetland (Nairn & Mitsch 2000; Lenhart et. al. 2019). Phosphorus export in agricultural watersheds may be highest during storm events or flooding due to the increased rates of erosion to stream flow as noted by Sharpley et. al. (2007). Agricultural watersheds may deliver high amounts of phosphorus during storm events (Sharpley et. al. 2007) which can accumulate within sediment and cause water quality issues when eroded or flushed out of a system (Heathwaite et. al. 2000). Many impoundments and wetlands, including the Granada treatment wetland, have been explored for their water quality benefits, but the NOI may be unique in its size and location. The purpose of this study was to analyze P removal efficiency in treatment wetlands and impoundments at different spatial scales through the NOI and the Granada treatment wetland.

3.2 Background and Site Description

Historically, the Red River of the North, beginning in northwestern Minnesota, has flooded frequently in the spring following snowmelt events. Major flood events, such as the 1997 Red River flood, have been caused by increased precipitation and snowfall as well as a north flowing river. The 1997 Red River Flood sparked increased interest in flood damage reduction structures; in the 20 years following this event, almost $1 billion has been spent in the last 20 years in flood damage reduction projects (Gunderson 2017). The melting snow in the spring season has also caused significant movement of agricultural runoff which can cause be a problem for downstream Lake Winnipeg. The yearly snowmelt and flat landscape has historically caused flood damages to the area and has resulted in a collaborative effort to address the flooding issues. Other flood damage reduction projects along the Red River that have recently been implemented include a large scale levee that cost approximately $409 million (Gunderson 2017). The Minnesota Department of Natural Resources (DNR), the Red River Basin Commission (RRBC) and the Bois de Sioux Watershed District (BdSWD), as well as various other stakeholders, collaborated to create a flood water storage impoundment to better address these issues. The NOI cost approximately $19 million to construct and was completed in 2014. The

NOI was implemented in Grant County, MN as an effort to capture and retain potential floodwaters to prevent further damage to people, property, and surrounding agricultural land.

The Red River Basin spans across multiple states including Minnesota and North Dakota and into provinces of Canada. Due to the magnitude and interdependent uses of the region by various stakeholders including the Red River Basin Commission and Bois de Sioux watershed district, cooperative land management decisions were deployed to better address multiple environmental concerns such as reducing flood damages, capturing excess nutrients, and providing habitat for waterfowl. The NOI is located in the lake plain of ancient glacial Lake Agassiz which contains soils with high clay content throughout the entire area. The topography has very low relief, with only a 0.6 m gradient across the entire impoundment area. The large, low-gradient watershed surrounding the NOI indicates low erosion potential from possible hillslopes (Sharpley et. al. 2002) but can indicate high potential for sediment and P transport at times of flooding. The specific design of the NOI provides the capacity to store 75% of a 100-year flood for the area (Guzner 2016). Flood water within the impoundment is either held for storage or can be transferred to other cells with the management of internal stoplog structures. The NOI is managed between 9 individual cells (A1, A2, A3, A4, B1, B2, B3, B4, C) and gathers water from a large agricultural watershed through the inlet (Figure 13).

Agricultural watersheds that utilize tile drainage systems have historically contributed significant amounts of TP that may be a leading cause for harmful algal blooms (King et. al. 2014). The Granada treatment wetland is a small scale edge-of-farm constructed solution to target agricultural runoff (Gordon 2019). The Granada treatment wetland is only about 0.1 ha in size and is located in southern Minnesota. The small scale treatment wetland collects agricultural drainage from approximately 10 ha, which is a significantly smaller area than the NOI watershed. Inputs to the NOI include drainage from the 50,245.8 ha watershed transported via the inlet. It is important to assess constructed wetlands of differing sizes and drainage area in order to fully understand nutrient retention services.

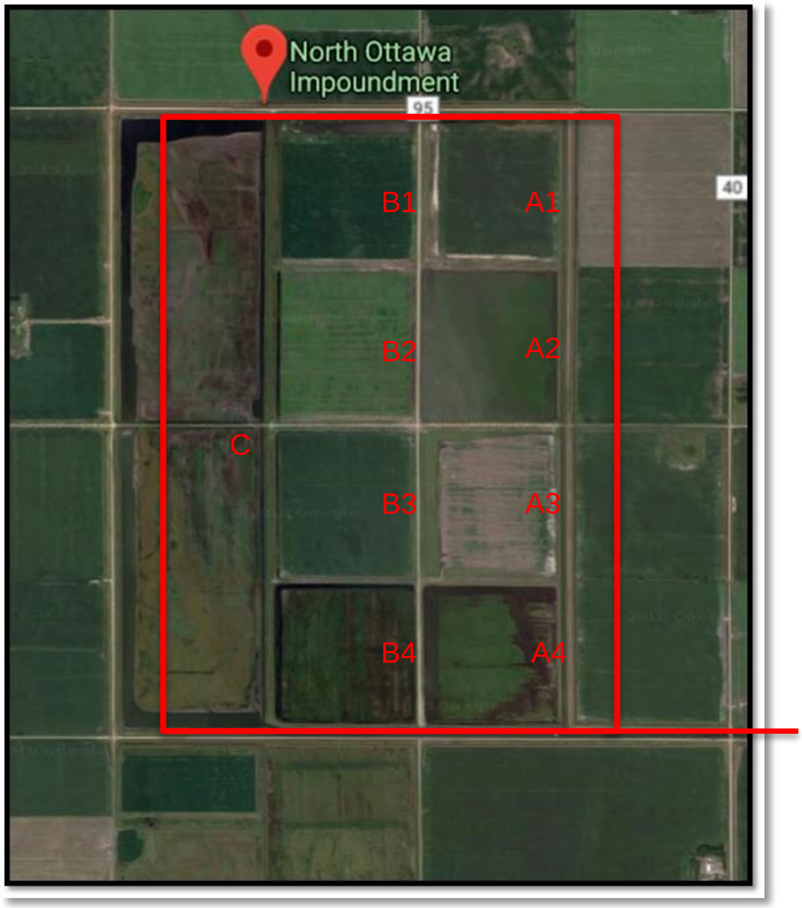

Figure 13: Map of the North Ottawa Impoundment with 9 included cells. The North Ottawa Impoundment is located within the Red River Basin in northwest Minnesota.

3.2.1 Land Uses and Management

Both the NOI and the Granada treatment wetland are surrounded by highly managed agricultural areas. The NOI watershed and the Elm Creek watershed have been primarily utilized for agricultural purposes in which point and nonpoint source pollution of nitrogen, phosphorus, and sediment may present an issue for water quality.

Land use in the NOI varied since its construction and can be found in Appendix III. Management of each cell varied every year an included renting the land to farmers for crop production while other cells were designated for water quality benefits and included cattail (*Typha spp.)* vegetation throughout. The C cell was left unmanaged for water storage purposes and contained abundant cattail growth throughout each growing season. In 2017 and 2018, land use in the NOI shifted to include more cropped cells. Individual cells A4, B4, and C were designated for water quality storage and treatment purposes and included cattail vegetation. Other cells in the impoundment were utilized for crop production including soy and wheat and were managed by local farmers.

3.2.2 Phosphorus and Soil Sorption

As phosphorus has been noted to be a nutrient of concern due to its contribution to the degradation of water bodies (Mitsch et. al. 1995), and has a high rate of soil sorption (Richardson 1985), it is important to consider the ability of the NOI to store phosphorus within soil. Sorption capacity and equilibrium phosphorus content were analyzed in previous studies by Guzner (2016) at the University of Minnesota. Phosphorus can be heavily stored within soil, and at higher rates within aquatic ecosystems such as lakes, wetlands, or the impoundments such as the NOI, which is high in clay content. Clay soils may be more likely to bind to ions including phosphate and are more likely to display sorption properties than sandy or silt soils. Similarly, organic matter may be high in some natural wetlands (Kusler and Lentula 1990). However, newly constructed systems like the NOI or the Granada treatment wetland may have lower soil organic matter than naturally occurring wetlands (Atkinson and Cairns 2001). Soil organic matter and clay content may contribute to P cycling depending on availability throughout the NOI, Granada treatment wetland, and other constructed sites. As organic material decomposes, P may be released and may contribute to the bioavailable phosphorus content within the impoundment. Phosphorus may bind to humus and accumulate which may result in higher soil P storage over time throughout constructed wetland areas (Borie and Zunino 1983). Desorption or release of P from sediment may occur within treatment wetlands, such as the Granada site which can result in a large outflow of phosphorus from the system.

3.2.3 Vegetation Harvesting for Nutrient Removal

Harvesting of vegetation within treatment wetlands has been shown to be an effective nutrient removal tool (Cicek et. al. 2006; Mitsch 2017). Harvesting cattail may aid in nutrient removal management practices as displayed by mesocosm experiments (Larkin et. al. 2012) and constructed wastewater treatment wetland sites (Jeke et. al. 2018). An experiment by Larkin et. al. (2012) assessed mesocosm tanks of cattail (*Typha* *x glauca*) and native marsh species for N uptake and removal potential; harvesting of the cattail removed more N than the native marsh species. This study indicates potential for removal of nitrogen by harvesting cattail located throughout the NOI or other constructed wetland systems. A study by Jeke et. al. (2018) analyzed an end-of-life municipal lagoon for cattail nutrient content over a 4-year study. Their results indicated that harvesting *Typha spp*. in August instead of November and only harvesting once per season may optimize nutrient removal. Similarly, Grosshans & Grieger (2013) found that harvesting cattails in August during peak growth from lake systems and ditches may allow for up to 0.11% P content within the biomass.

The NOI is home to a variety of aquatic vegetation, such as *Typha x glauca*, as well as crops within various cells. Harvesting of vegetation throughout the NOI has been considered in some seasons of operation. In 2014, the Red River Basin Commission harvested cattail material (*Typha spp.*) and bulrush (*Scirpus spp.*) from the NOI. Harvested vegetation was analyzed for nutrient content to estimate phosphorus removal potential. The harvesting of crops or other vegetation within various cells within the NOI is typically conducted near the end of season. Harvesting of the present cattails in various locations throughout the NOI, including the inlet, outlet, and some cells, has been attempted but challenges to this management strategy included accessibility of harvestable wetland plants and time of season for optimized nutrient removal. For vegetation harvesting at the NOI, cattail harvesters (Rotochoppers) were considered for utilization if available but can be approximately $85,000 - $400,000 to purchase. The Granada treatment wetland utilizes a wet prairie species mix that is native to Minnesota. Harvesting of the Granada vegetation has also been considered as a management strategy in recent years for P removal. Repeated seasonal harvesting of vegetation throughout the NOI and the Granada treatment wetland at optimal time of the season may reduce P accumulation within the soil if selected in future management decisions.

Removal of the harvested material from a constructed wetland can decrease the amount of decomposing material over time and may reduce accumulation of organic material and associated phosphorus content. Once vegetation is harvested from a small scale treatment wetland such as the Granada treatment wetland or large scale impoundment such as the NOI, there are a variety of methods that can be utilized for efficient use of the material. Shredding the harvested cattails and utilizing the material for compost and soil amendments has been a management practice attempted by the Red River Basin Commission at the NOI in some seasons. Alternative methods have included drying the harvested vegetation and utilizing it as a feedstock (Mercil 2014).

3.3 Methods and Procedures

The North Ottawa Impoundment has been monitored for nutrient retention services since operations began in 2014 by Masha Guzner (2016). At the Granada treatment wetland, water quality data gathered through the inlet was collected in previous years by Gordon (2019). At the NOI and at the Granada treatment wetland, vegetation samples were gathered in some years through harvesting the above ground vegetation. The Granada site was sampled for phosphorus content in wet prairie species mix (native mix) while the NOI was sampled for hybrid cattail (invasive). Each site varied in its hydrology, vegetation, and services provided which included varying monitoring schemes in some years.

3.3.1 Hydrology and Water Quality Assessment in the Impoundment

The NOI was equipped with a variety of instruments that allowed for water quality monitoring throughout the site. Pressure transducers were placed in the inlet weir for monitoring water levels through the inlet channel. For inlet and outlet water monitoring, samples were collected from the two automated ISCO samplers (3700 Portable Sampler Compact Model). In 2015, the ISCO was programmed to collect one sample every 24 hours and data was assessed by Guzner (2016). Selected samples in all years were sent to RMB Environmental Laboratories in Detroit Lakes, MN for analysis of nutrient concentrations.

In 2016, monitoring efforts by Guzner (2016) included connecting the ISCO to a pressure transducer and set to trigger sampling any time the stage increased by 1.6” in 2 hours, or 2.4” in 1 hour. In 2018, the lack of precipitation in northwestern Minnesota did not allow for a stage increase to trigger the ISCO for sampling during the entire season. Instead, grab samples were taken at the inlet throughout the summer. Similarly, in 2018 grab samples could not be collected at the outlet due to lack of water flow, however this is not expected to be a common occurrence for this site.

Vegetation samples were gathered in some years throughout the NOI beginning in 2014. Samples included bulrush (*Scirpus spp.)* and cattail (*Typha spp.)* gathered through the inlet and some cells of the impoundment. Vegetation sampling was gathered through harvesting above ground tissue and utilizing a dry ash method for P analysis.

3.3.2 Watershed Sites

The NOI drainage area and the Granada treatment wetland drainage area differed in their overall spatial scale. The NOI collected drainage from a large scale watershed (50,245.8 ha) to the approximately 776.9 ha impoundment. The Granada treatment wetland is only about 0.1 ha in size and collects drainage from approximately 10 ha. Each watershed was similar in their agricultural management throughout including collection of subsurface tile drainage. Water quality sites were selected for 10-12 locations within the NOI watershed for regular sampling throughout each season (Figure 14). These sites were selected to obtain representative data for analyzing sources of nitrogen (N) and phosphorus (P) into the impoundment from the large agricultural watershed. Phosphorus was of particular interest for this analysis due to the high likelihood of P transport within agricultural watersheds of various scales (Heathwaite et. al. 2000). The selected locations within the watershed include both surface water and tile drainage sources. Watershed sampling was conducted throughout the summers of 2015-2018 by graduate and undergraduate employees of the University of Minnesota. Sampling occurred following rainfall events or on a consistent monthly basis. Varying weather patterns in each season resulted in a variable sampling schedule throughout the entirety of the project.

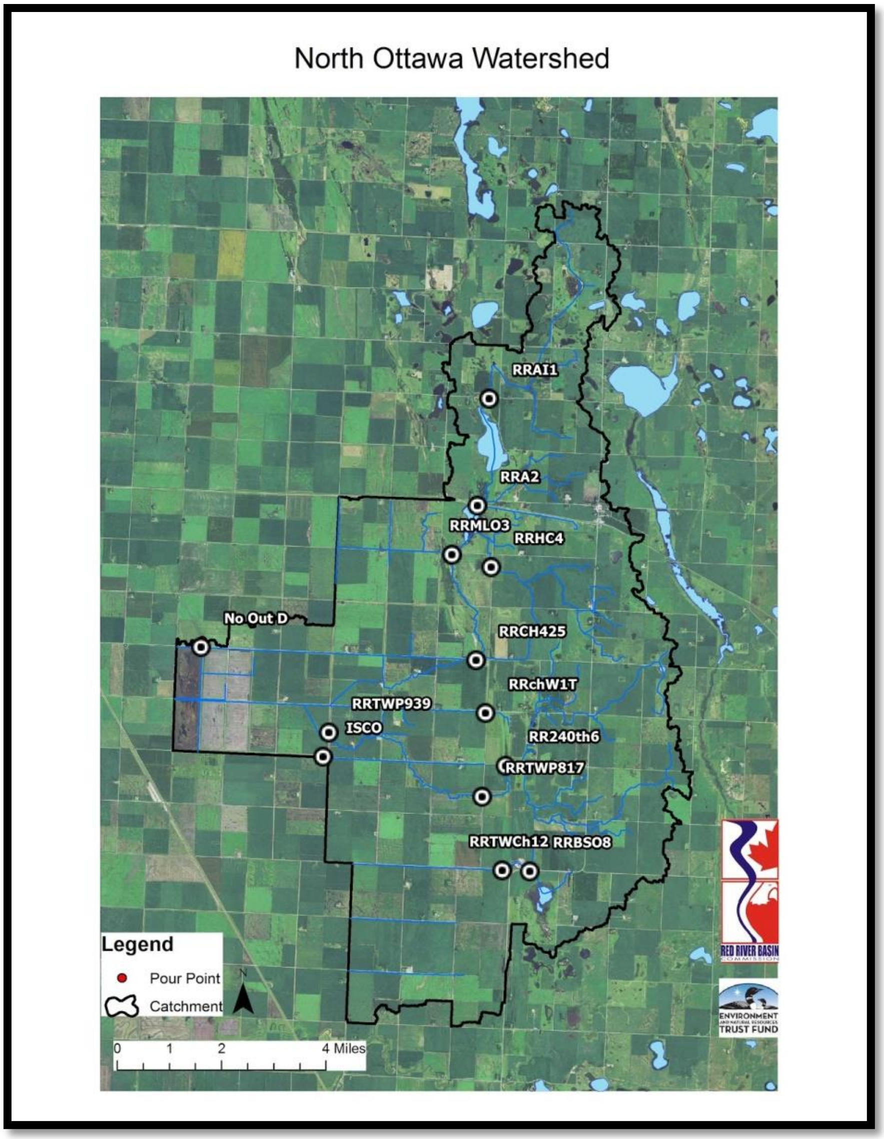

Figure 14: Watershed sites selected for sampling throughout the North Ottawa Impoundment area; drainage throughout the watershed leads directly to the North Ottawa Impoundment (labeled on figure with a star).

Watershed Monitoring Sites 2018		
In Channel Mixed	**Wetland outlet**	**Tile Drainage Outlet**
S008-706 RRAI1	S008-707 RRA2	S008-712 RRTWP817
S008-709 RRHC4	S008-708 RRMLO3	S008-710 RRchW1T
S003-273 RRCh425	S008-713 RRBS08	S008-713 RRTWCh12
S008-711 RR240th6	-	-

Table 7: 2018 watershed sampling sites

3.3.3. Comparison of Treatment Wetlands at Different Spatial Scales

The NOI and the Granada treatment wetland are both constructed areas in agricultural watersheds but differ in the size of their drainage areas as well as their hydrological inputs. Flow, nutrient loading rates including P concentrations, and P content in harvested vegetation per wetland area were all accounted for in comparing the NOI and the Granada site. The NOI watershed is approximately 50,245.8 ha leading to a 776.9 ha impoundment area. The Granada watershed (Elm Creek watershed) is approximately 10 ha and flows to a 0.1 ha watershed area. The wetland area to watershed area is somewhat comparable between each location but the water storage capacity at each site varied. Total flow to the NOI site in 2016 was approximately 3.602 m^3x10^6 (Guzner 2016). At the Granada site, flow to the wetland included 27,970 m^3 (Lenhart et. al. 2019). Nutrient loading from each site included subsurface drainage from each agriculturally managed watershed with varying wetland plant species located at each site.

3.4 Results

3.4.1 Inlet/Outlet Hydrology

Phosphorus loading rates to the impoundment were accounted for through the inlet by Masha Guzner (2016) from 2014-2016. In 2016, estimation of inflow of water was approximately 3.602 m^3x10^6. From 2014 to 2016, average TP concentration at the NOI inlet was approximately 0.27 mg/L of TP. Similarly, average OP concentration was approximately 0.16 mg/L. Average outlet concentrations for 2015-2016 was approximately 0.23 mg/L TP and 0.02 mg/L OP. In 2016, pollutant load reduction was estimated to be approximately 85% of TP throughout the treatment cells through monitoring efforts by Guzner (2016). Inlet and outlet concentrations throughout the NOI can be found in Table 8 and phosphorus concentrations can be found in Figure 15. In 2016, nutrient loading to the NOI included total P loading of approximately 453 kg (1,000 lbs) according to Guzner (2016). At the Granada site, loading of total P was approximately 2.43 kg (5.35 lbs) in 2016 (Lenhart et. al. 2019).

3.4.2 Impoundment Nutrient Reductions

Pollutant concentrations from inflows to outflows were calculated for the impoundment in some years of initial implementation of the NOI (2014-2016) by Guzner (2016). In 2016, the C cell reduced approximately 27% of TP, 51% of TN, and 57% of TSS; the A/B cells reduced 66% of TP, 73% of TN, and 42% of TSS. Similarly, in 2016 an estimated reduction of approximately 457.7 kg (1,009 lbs) was achieved between all cells of the impoundment (Guzner 2016). In 2017, data gathered by Guzner (2016) showed that A/B cells released approximately 264 kg (582 lbs) of TP equating to a -54% capture rate. This indicates that the A/B cells may have acted as a source of phosphorus instead of a sink at this time (Guzner 2016). Organic nitrogen content (TKN) was also measured within the impoundment in some years; TKN increased throughout the impoundment with higher concentrations found at the outlet each season (Figure 15) which may have been impacted by the production of ammonium throughout the impoundment.

North Ottawa Impoundment Nutrient Reduction						
Year	**TP Inlet (mg/L)**	**TP Outlet (mg/L)**	**Average Reduction (mg/L)**	**TN Inlet (mg/L)**	**TN Outlet (mg/L)**	**Average Reduction (mg/L)**
2014	0.46	N/A	**N/A**	5.16	N/A	**N/A**
2015	0.43	0.23	**0.20**	6.44	2.83	**3.61**
2016	0.27	0.22	**0.05**	3.49	1.88	**1.61**
2017	0.22	0.19	**0.03**	6.17	2.81	**3.36**
2018	0.353	N/A	**N/A**	1.595	N/A	**N/A**

Table 8: Inlet and outlet water chemistry (Total Phosphorus and Total Nitrogen) within the North Ottawa Impoundment. Lack of adequate rainfall or sampling in some years is marked by N/A.

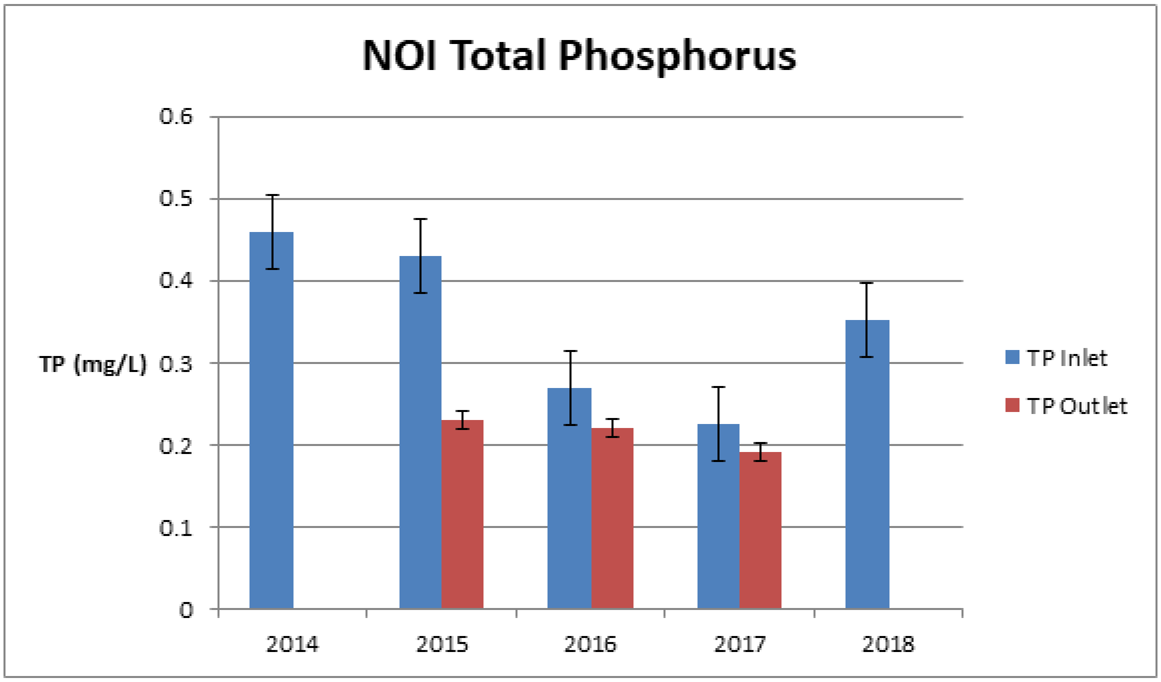

Figure 15: Inlet and outlet TP concentrations (mg/L) at the NOI from 2014-2018. Outlet data was not collected in 2014 or 2018 due to lack of adequate flow as well as end of project contracts.

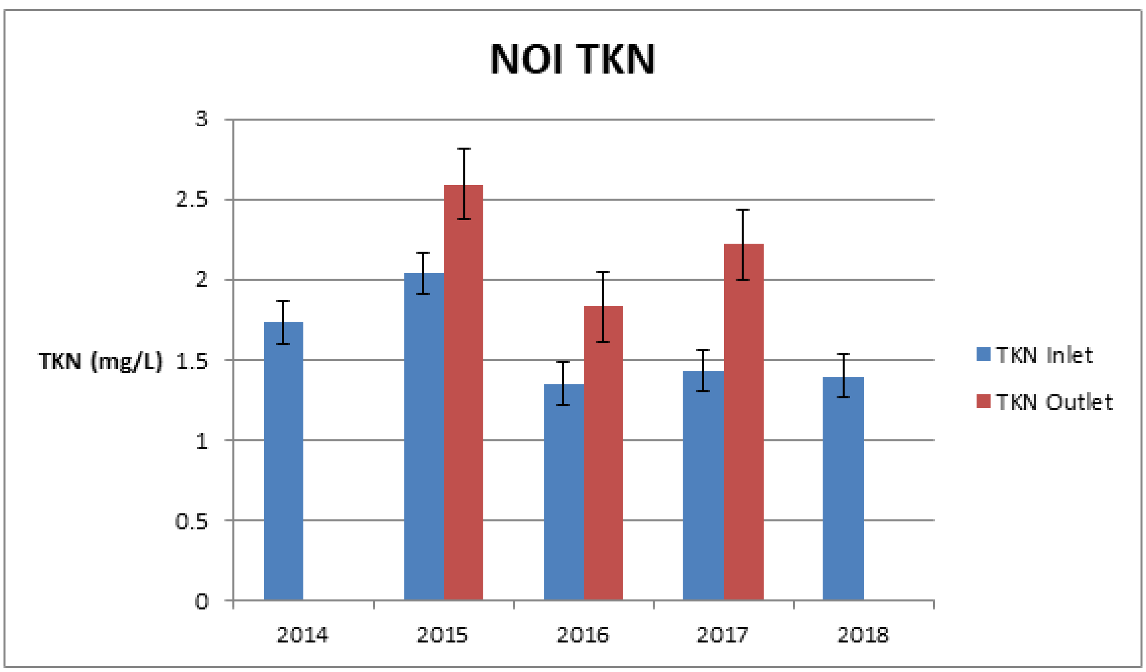

Figure 16: Total Kjeldahl Nitrogen (TKN) measurements at the NOI inlet and outlet for 2014-2018 (mg/L) taken at the inlet and outlet. Outlet data was not collected in 2014 or 2018 due to lack of adequate flow and end of project contracts.

3.4.3 Watershed Results

The contributing drainage to the North Ottawa Impoundment spans approximately 50,245.8 ha. Due to the highly agricultural landscape, there are high nutrient loads within the watershed that can be captured by the impoundment, particularly during runoff events. Watershed sites were selected for sampling in order to determine more direct sources of the nutrients from the contributing watershed. Ten of the watershed sites were divided into three groups for analysis: 1) In-channel mixed sources, 2) Wetland outlet sources, and 3) Tile drainage sources. In-channel mixed sites included RRAI1, RRHC4, RRCH425, and RR240th6. Wetland outlet sites included RRA2, RRMLO3, and RRBS08. The tile drainage sites were RRCHW1T, RRTWP817, and RRTWCH12 (Table 7).

The average TN concentration in the water samples was 4.7 mg/L for the in-channel mixed sites and 3.7 mg/L for the wetland outlet sites (see Table 9, Figure 17, and Figure 18). Average TN concentration for the tile drainage sites was 23.4 mg/L, which was higher than the averages for both of the other source types. In the tile drainage water, the average concentration of TP was actually significantly lower than the other two sources, at 0.16 mg/L.

The average concentration of TP in 2018 was 0.37 mg/L for both the in-channel mixed sites and the wetland outlet sites, which indicates the wetlands throughout the watershed may have acted as a source of P instead of a sink for P (Table 9). Wetlands may act as a source of P to larger water bodies in some agricultural watersheds, particularly with the export of sediment through erosion. As an example, the Granada treatment wetland acted as a source of P to Elm Creek in some years with higher outlet phosphorus concentrations (Lenhart 2019). Nutrient transport throughout the large scale NOI watershed had overall higher concentrations than in the small scale Granada (Elm Creek) watershed (Lenhart et. al. 2019). In 2018, the Granada site only received an average 0.032 mg/L of P while the NOI site received an average of 0.22 mg/L P. This significant difference in loading rates may be accounted for by the large difference in watershed nutrient transport between each site.

NOI Watershed			
Site	**TN (mg/L)**	**TP (mg/L)**	**TSS (mg/L)**
In-Channel Mixed	4.65	0.37	18.57
Wetland Outlet	3.65	0.36	9.10
Tile Drainage	23.35	.016	18.63

Table 9: Watershed site locations (3) throughout the North Ottawa Impoundment (Red River Basin) varied in their average TN, TP, and TSS concentrations in 2018.

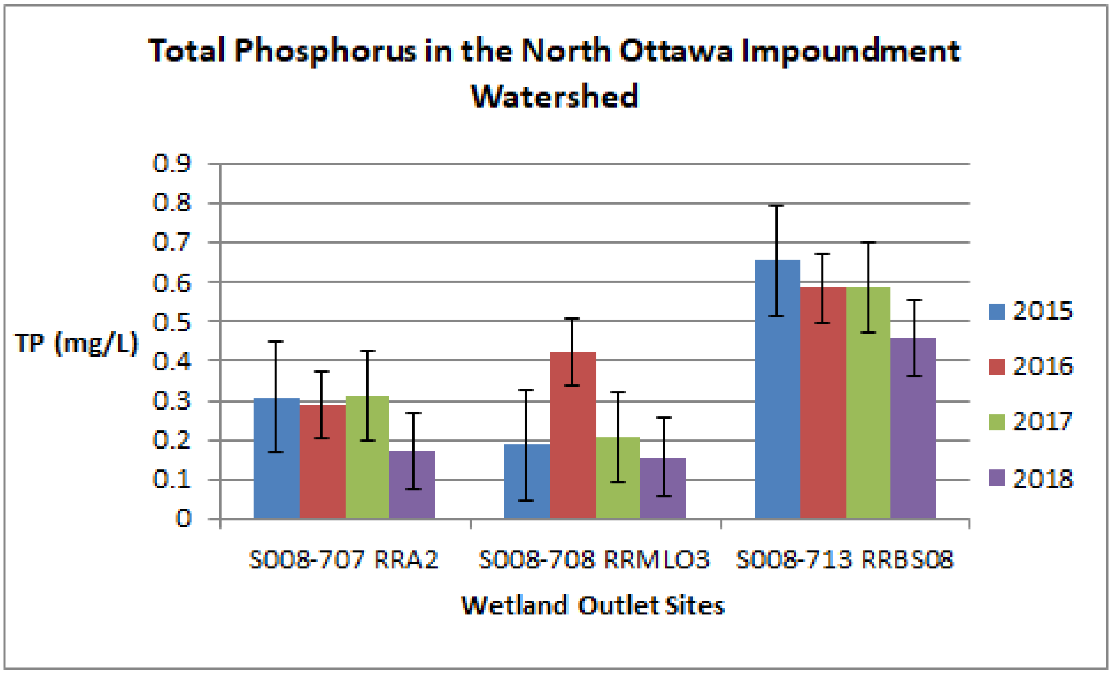

Figure 17: Total Phosphorus (TP) concentrations within the North Ottawa Impoundment watershed sampling site wetland outlet locations. 2018 displayed the lowest overall TP concentration of all the years of sampling.

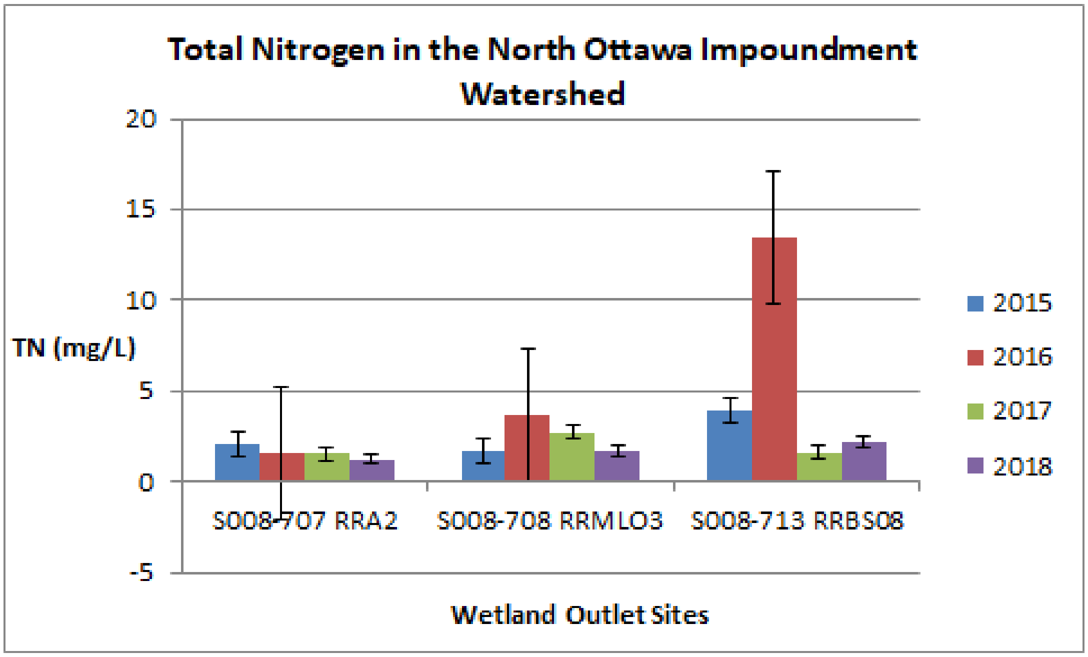

Figure 18: Total Nitrogen (TN) concentrations within the North Ottawa Impoundment watershed sampling site wetland outlet locations.

3.4.4 Vegetation Nutrient Analysis

In the 2014 vegetation harvests conducted at the NOI, various cattails and bulrush samples were analyzed for phosphorus content. The harvest occurred in September 2014 utilizing large cattail harvesting equipment. Under a dry ash method, cattail samples averaged approximately 0.21% P content while bulrush samples averaged 0.22% P content. Similarly, results from the Granada treatment wetland indicate a range of P content in the native wet prairie species mix of 0.6% to 0.14% (Table 10). At the small scale treatment wetland, harvesting biomass in 2018 at the Granada wetland may remove up to 3.2 kg/ha P throughout the 0.1 ha wetland area. The Red River Basin Commission conducted a nutrient analysis based on the amount of P content from harvested material and the potential for harvested material from within the NOI. Results indicated an approximate removal rate by harvesting cattails within the NOI of 3.11 kg/ha to 3.30 kg/ha, which could result in up to 2,564 kg of P removed in the 776.99 ha of *Typha x glauca* located within the site (Mercil 2014).

Harvested Vegetation Comparison				
Plant Type	**Mean End of Season Dry Weight (DW)(g)**	**Phosphorus Content (ppm)**	**Phosphorus Removal (gP/g DW)**	**Year and Location**
Canada blue joint grass	84.43	2575.52	0.22	2017 Mesocosm
Tussock Sedge	171.07	1841.51	0.31	2017 Mesocosm
Prairie cordgrass	259.79	1299.04	0.34	2017 Mesocosm
Softstem bulrush	347.41	3205.03	1.11	2018 Mesocosm
River bulrush	409.15	1455.32	0.59	2018 Mesocosm
Cell 1 Mix	334.07	816.22	0.27	2017 Granada
Cell 2 Mix	140.48	725.86	0.10	2017 Granada
Cell 3 Mix	534.33	640.51	0.34	2017 Granada
Cell 1 Mix	320	1263.39	0.40	2018 Granada
Cell 2 Mix	293	859.30	0.25	2018 Granada
Cell 3 Mix	231	1308.24	0.30	2018 Granada
Typha x glauca	N/A	2100	N/A	2014 NOI
Scirpus spp.	N/A	2200	N/A	2014 NOI

Table 10: Vegetation biomass and phosphorus content per site (3)

3.5 Discussion

3.5.1 Impoundment and Watershed

Treatment wetlands are important management tools for capturing and reducing nutrients such as nitrogen and phosphorus. Similarly, sediment content and soil texture can directly correlate to phosphorus content due to the sorption capacity in cohesive soils

(Guzner 2016). There is a large need for sediment capture in agricultural watersheds due to the high potential for soil erosion due to poor agricultural practices (Westra et. al. 2007). Agricultural watersheds with constructed wetlands may experience high loads of nitrates and phosphates, especially within the early months of the growing season when fertilizer applications are common (Fransen 2012).

The NOI had significantly higher loading rates of TP and OP than other smaller scale sites, including the Granada treatment wetland (Table 11). Subsurface drainage was primarily captured in the NOI including P transport from a large agricultural watershed. The volume of water that the NOI collects and is able to store may account for the high nutrient concentrations to the impoundment. In 2016, the NOI total P loading was estimated to be approximately 453 kg (1,000 lbs) (Guzner 2016). Comparatively, primary flow entering the small scale Granada site was through subsurface drainage as well as overflow from Elm Creek. In 2016, approximately 2.43 kg (5.35 lbs) of total P was loaded to the Granada site (Lenhart et. al. 2019). The large difference between sites describes the difference in settings throughout the state of Minnesota. These differences per site include factors such as sources of phosphorus in differing terrains, climate patterns, as well as drainage area scale and overall flow to each site. The variation in phosphorus loading may be largely attributed to the size of the watersheds including a smaller watershed area to the Granada treatment wetland. Soil sorption capacities of the clay soils located within the NOI may have also largely impacted nutrient removal services provided by the impoundment (Guzner 2016). Sorption processes may be more common throughout the NOI than at the Granada site due to the larger area, longer retention period, and clay dominated soils.

As noted by Konopacky (2017), watershed planning and successful BMP implementation may be highly dependent upon the scale of the watershed area assessed. The large scale of the watershed and the impoundment make the NOI a unique site for understanding phosphorus dynamics. Large scale agricultural watersheds may transport high loads of P due to poor soil management practices including increased soil erosion and repeated fertilizer inputs. However, most P processes in watersheds can be scale and time dependent (Sharpley et. al. 2002). Subsurface tile, overland runoff, and overbank riparian flow may all contribute different forms of phosphorus in watershed areas of various scales. Surface water outlets and tile outlets throughout the watershed flow to the NOI each season. Both the NOI watershed and Granada wetland watershed may contribute forms of P through suspended sediment (silt and clay) mobility. The NOI drainage area, although highly managed, is unlikely to transport high amounts of particulate P through the vast, flat landscape. Comparatively, at the Granada treatment wetland, close proximity to Elm Creek and riverbank overflow and impaired outflow in some years may have contributed particulate P to the wetland while soluble P was simultaneously collected via tile drainage throughout the whole season. The Granada treatment wetland in the Elm Creek watershed has a more variable topography and receives more precipitation annually, which may have aided in the flooding and P export from the wetland in some years (Lenhart et. al. 2019). Alternatively, the NOI is located in a low-gradient landscape in the Lake Agassiz Plain and has lower annual precipitation and is therefore less likely to transport P with sediment throughout the watershed (Wilson 2003). Similarly, risk of erosion and sedimentation may be higher near the Granada site due to the proximity of the wetland to the creek, recent bank overflow events, and varying topography throughout the watershed.

Sources of TP, TN, and TSS throughout the NOI watershed were also measured in 2018. Water quality results from throughout the NOI watershed indicated that tile drainage did not have a higher concentration of TP compared to wetland outlet sites. However, high inlet concentrations of TP and OP at the NOI site may be due to the multiple sources of phosphorus throughout a larger drainage area (Table 11). High nitrate concentrations throughout the watershed can likely be attributed to the mobility of nitrates through tile drainage. Subsurface drainage tiles create a direct pathway to carry nitrate from fields to nearby surface water. Although both sites received inputs from agricultural runoff, further considerations of watershed scale and sources of P to constructed wetlands of various sizes are essential factors for understanding P removal capacity. Selected watershed scale for BMP identification and implementation may significantly impact participation within a watershed by various stakeholders, development of data, as well as overall success of the plan towards water quality goals (Konopacky 2017). Smaller scale watersheds such as Elm Creek may be more successful areas for BMP implementation than larger scale watersheds.

Mean Inflow Phosphorus Concentrations				
Year	**Granada OP (mg/L)**	**Granada TP (mg/L)**	**NOI OP (mg/L)**	**NOI TP (mg/L)**
2013	0.040	0.052	N/A	N/A
2014	0.120	0.152	0.034	0.46
2015	0.034	0.048	0.23	0.43
2016	0.004	0.032	0.09	0.22
2017	0.019	0.021	N/A	N/A
2018	0.032	0.051	N/A	N/A

Table 11: Mean inflow of phosphorus (TP and OP) to the NOI and Granada treatment wetland

3.5.2 Harvested Vegetation

Wetland vegetation throughout the NOI may aid in environmental services provided to the area including phosphorus removal capacity at times of harvest. Approximately 3.11 kg/ha P could be removed by harvesting cattail throughout the cells of the impoundment. For the entire impoundment area, phosphorus removal by harvesting cattail could be up to 2,564 kg of P. The removal of P by biomass harvesting may contribute to P removal services provided by the impoundment. The comparable average phosphorus removal rates between harvested species at the NOI (3.11 kg/ha P) and the Granada wet prairie species mix (3.2 kg/ha P) indicate potential for phosphorus retention and removal with both the invasive species monoculture (*Typha x glauca*) and native species.

Equipment used for harvesting vegetation may vary depending on the time of season and associated inundation levels within individual cells and the inlet for accessibility purposes. Rotochoppers are often utilized for harvesting large areas of vegetation but can be costly to purchase or difficult to use in some locations. Due to lack of growth following implementation of the impoundment in 2014, harvesting of available cattail was planned for future project management starting in 2015 of approximately 160 acres under the Red River Basin Commission and Bois de Sioux Watershed District. Future management at the NOI should consider the practice of harvesting cattails and other present vegetation within the wetland area for excess phosphorus removal as well as possibly providing renewable sources of fuel to the area.

Nutrient loading at each site varied and may have an impact on P removal rates by harvesting vegetation; NOI inflow concentration of TP was approximately 0.27 mg/L

(Guzner 2016) in 2016 and the Granada treatment wetland inflow concentration was approximately 0.06 mg/L of TP on average in early years. Phosphorus loads varied per site due to different sized watersheds. Phosphorus loads to the NOI included soluble P while loads to the Granada site included both soluble P and particulate P bound to sediment. The particulate P may settle within the wetland soils while soluble P may be flushed out or utilized by present biota (wet prairie mix). The NOI collected high concentrations of soluble P which is likely to be flushed out of the system, bound to sediment through sorption processes, or utilized by present biota (*Typha x glauca*). Varied hydrology, nutrient loading, and plant communities per site are a challenge for scaling up site comparisons for vegetative phosphorus removal estimates. Results for large scale vegetation harvesting would be dependent upon nutrient loading rates, biomass cover, and plant species present in future NOI management.

3.5.3 Modeling

The NOI and the Granada treatment wetland include potential area for biomass harvesting in future years. At the Granada treatment wetland site, the wet prairie species mix removed approximately 2.3 kg/ha P to 3.2 kg/ha of P in 2017 and 2018. Comparatively, 2014 samples of *Typha spp.* estimate P removal potential to be approximately 3.11 kg/ha to 3.30 kg/ha. Each site varied in size of impoundment and size of watershed. The NOI area is 776.99 ha, therefore the cattail biomass harvest could remove between 2,408 kg P to 2,564 kg P. The Granada species mix is native to Minnesota and could be utilized for future constructed wetlands such as the NOI. It may be beneficial to estimate P removal by planting and harvesting the wet prairie species mix in a large wetland area such as the NOI. For the entire impoundment, we can estimate a total removal of between 1,787 kg P to 2,486 kg of P throughout the entire NOI site utilizing the Granada wet prairie mix P removal rate.

There were comparable estimates of P removal by harvesting hybrid cattail (2,564 kg P) and the estimates of wet prairie species mix (2,486 kg P) at the NOI location. This indicates anticipated successful P removal rates across species, but with a slightly larger P removal with harvesting monocultures of hybrid cattail. The mesocosm experiment also contained monocultures of softstem bulrush and river bulrush and had significantly higher rates of P removal with approximately 12 kg/ha P estimated. The greater P removal potential in the mesocosms compared to the field sites may be partially attributed to the controlled variables of the experiment. Assumptions for phosphorus removal success at each site included consistent cover of vegetation throughout the entire site with little to no interference by environmental factors including competition from other species or lack of adequate rainfall.

3.6 Future Research Potential

The North Ottawa Impoundment project has been useful in recent years in protecting the Red River valley from potential flood damage as well as providing many other services. During the multiple years of monitoring, the NOI captured pollutants and reduced nutrient loads by drainage from a large scale agricultural watershed. Similarly, soil P content was stored within the soil through sorption processes (Guzner 2016). Enhanced nutrient removal within the impoundment was achieved in some years through process of harvesting present cattail (*Typha spp.*) and bulrush (*Scirpus spp.)* species. The extensive wetland vegetation cover throughout the NOI allows for further potential for excess phosphorus removal through harvesting and removal of the biomass, in which the vegetation then may be used for composting or biofuels. Cattail (*Typha x glauca*) harvesting conducted in 2014 suggests that harvesting large areas as a management practice throughout the NOI can aid in phosphorus removal with up to 2,564 kg P from *Typha x glauca* removal. Phosphorus loading may be dependent upon the scale of the watershed area to the size of the watershed. Further research may aid in understanding phosphorus dynamics in large and small scale watersheds in relation to vegetative uptake and removal. Reductions within the impoundment can also be partly attributed to the soil sorption processes as well as vegetative uptake throughout the area. It may be beneficial to further assess the functions of other constructed impoundments or treatment wetlands for nutrient retention services, including P removal by vegetation harvesting. Understanding the capacity of constructed wetland areas of different scales to capture agricultural runoff and remove phosphorus by harvesting vegetation throughout the site may aid in future management decisions. Continued monitoring efforts within the impoundment as well as various watershed sites is recommended for the NOI project for improved understanding and analysis of nutrient loading and reduction in the Red River Basin area. The NOI is a unique large-scale site that can be considered a significant case study in future years for floodwater abatement as well as nitrogen and phosphorus retention.

4. Conclusion

The upper Midwest region of the United States is home to a variety of different types of wetlands that range in their vegetation, specific hydrology, and soils. Treatment wetlands can be an effective management tool for agricultural watersheds for targeting excess nutrients for overall water quality protection (Lenhart et. al. 2017). Constructed treatment wetlands allow for flexibility in factors in a wide range of environmental settings and will often incorporate dense vegetation in both subsurface flow and surface flow designs. Pathways of flow including subsurface drainage, riparian overflow, and surface runoff should be considered for inputs of P while soil texture and legacy P content should be considered for P storage. Topographical considerations including landscape gradient, susceptibility to flood events, and surrounding land use management may impact P pathways within agricultural watersheds of differing scales. Treatment wetland sites analyzed for the purpose of this study included a small scale edge-of-farm treatment wetland in southern Minnesota, a mesocosm experiment of 30 individual wetland tanks on the University of Minnesota campus, and a large scale flood water storage impoundment in the Red River Valley. Harvesting of wetland plants for purposes of enhancing phosphorus removal efficiency in treatment wetlands can be a practical and effective management strategy in areas with high soil legacy phosphorus content. Phosphorus removal by process of harvesting the monocultures and wet prairie mixes was a successful practice at each location for P removal.

Each site assessed had a different primary goal as well as varying watershed sizes but could harvest present vegetation to remove excess nutrients that may be harmful to water quality. The Granada treatment wetland removed approximately 3.2 kg/ha in 2018 through vegetative harvest which was comparable to the removal estimate of approximately 3.11 kg/ha of *Typha x glauca* harvesting at the NOI; the mesocosm species were estimated to remove up to approximately 12 kg/ha of P. Monoculture systems may not provide as many services to the surrounding ecosystem as vegetation mixes, however the monoculture experiments may remove more phosphorus on average than vegetation mixes such as those from the Granada treatment wetland. If monoculture systems are utilized within a treatment wetland, harvests should be conducted at an optimal time in the growing season; our results indicate that an optimal harvest time in Minnesota may be in August when biomass and P content are high to enhance P removal per the species selected. Land managers will inevitably consider the accessibility of the wetland and water levels as well as equipment necessary per size of harvestable area and alternative methods for disposal of the harvested biomass. Management of treatment wetlands located throughout various sized watersheds should consider the presence of bioavailable phosphorus as well as soil legacy phosphorus content throughout the site to understand the potential for flushing of phosphorus. Managers of treatment wetlands of various scales that are located in agricultural watersheds should consider harvesting vegetation as a future management practice for reducing phosphorus accumulation over time.

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Appendix I

2018 Mesocosm Experimental Design					
Mesocosm Tank #	**Plant Name**	**Scientific Name**	**Date planted**	**2018 Harvest Date**	**2019 Harvest Date**
1	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	7/2/2018	6/19/2019
2	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	7/2/2018	9/19/2019
3	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	9/21/2018	8/1/2019
4	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	8/27/2018	7/10/2019
5	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	7/23/2018	6/19/2019
6	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	7/2/2018	8/21/2019
7	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	8/13/2018	7/10/2019
8	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	7/23/2018	8/1/2019
9	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	8/27/2018	8/21/2019
10	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	9/21/2018	8/1/2019
11	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	8/13/2018	9/19/2019
12	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	7/2/2018	6/19/2019
13	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	9/21/2018	9/19/2019
14	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	8/13/2018	8/1/2019
15	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	7/2/2018	8/21/2019
16	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	8/27/2018	9/19/2019
17	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	8/13/2018	9/19/2019
18	**Softstem**	*Schoenoplectus*	5/22/2018	7/23/2018	7/10/2019

	Bulrush	*tabernaemontani*			
19	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	9/21/2018	8/1/2019
20	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	7/23/2018	6/19/2019
21	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	7/23/2018	8/1/2019
22	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	8/27/2018	8/21/2019
23	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	7/2/2018	7/10/2019
24	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	9/21/2018	9/19/2019
25	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	8/27/2018	8/21/2019
26	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	8/13/2018	7/10/2019
27	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	7/23/2018	7/10/2019
28	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	9/21/2018	8/21/2019
29	**River Bulrush**	*Bolboschoenus fluviatilis*	5/31/2018	8/27/2018	6/19/2019
30	**Softstem Bulrush**	*Schoenoplectus tabernaemontani*	5/22/2018	8/13/2018	6/19/2019

Experimental design for 2018-2019 mesocosms on the St. Paul UMN campus.

Appendix II

Mesocosm Soil Analysis				
Tank #	2017 Fall Olsen P (mg/kg)	2018 Fall Olsen P (mg/kg)	Reduction (mg/kg)	% Reduction
1	27	18	-9	33%
2	38	24	-14	36%
3	25	18	-7	28%
4	36	24	-12	33%
5	37	28	-9	24%
6	41	26	-15	36%
7	33	28	-5	15%
8	27	26	-1	3.7%
9	31	25	-6	19%
10	24	23	-1	4.1%
11	27 / 26	25	-1	3.7%
12	35	26	-9	25%
13	38	22	-19	50%
14	29	26	-3	10%
15	42	22	-20	47%
16	36	30	-6	16%
17	27	25	-2	7.4%
18	33	30	-3	9%
19	26	25	-1	3.8%
20	28	24	-4	14%
21	27	22	-5	18%
22	30	24	-6	20%
23	28	22	-6	21%
24	32	23	-9	28%
25	35	26	-9	25%
26	37	23	-14	37%

27	**32**	**26**	**-6**	**18%**
28	**31**	**22**	**-9**	**29%**
29	**31**	**N/A**	**N/A**	**N/A**
30	**41 / 43**	**22**	**-19**	**46%**
Mean	**-**	**-**	**7.93**	**22.7%**

Comparisons of end of season soil bioavailable P following plant harvest for Olsen-P content within the mesocosms in St. Paul, MN.

Appendix III

Land Uses in NOI		
Year	**Cell**	**Use**
2016	A1	Corn
	A2	Water holding
	A3	Wheat
	A4	Water holding
	B1	Soybeans
	B2	Native plants, mudflat
	B3	Corn
	B4	Cattails
	C	Cattails/Unmanaged vegetation
2017	A1	Cropped
	A2	Water holding
	A3	Water holding
	A4	Water holding
	B1	Cropped
	B2	Native plants
	B3	Native plants
	B4	Cattails
	C	Cattails/Unmanaged vegetation
2018	A1	Soy
	A2	Soy
	A3	Wheat

	A4	Flooded; cattails
	B1	Soy
	B2	Soy
	B3	Wheat
	B4	Flooded; cattails
	C	Flooded

Land uses within the North Ottawa Impoundment. Each cell had a different use each season, including for agricultural purposes (cropped), water treatment, water storage, and other uses.