Constructed Wetland Used to Treat Nitrate Pollution Generated from Agricultural Tile Drainage Waters in Southern Minnesota

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Dedication

To my ever supportive husband Christopher.

Abstract

Nitrate molecules are highly soluble in water and are bioavailable to plants. These properties are why excess nitrates in water are one of the main causes of hypoxia in the northern Gulf of Mexico. Over 90% of these nitrates originate from non-point sources such as agricultural fields. In fields with tile drainage systems nitrates have a swift passageway from field to surface waters. This study focuses on one Midwestern farm field located in southern Minnesota, along Elm Creek, a Blue Earth tributary. Tile drainage water from this field discharges into Elm Creek at a concentration averaging 23.0 mg/L NO_3 as NO_3-N. During the spring of 2013 a three celled treatment wetland was constructed adjacent to Elm Creek. The tile drainage system was re-routed to discharge into the constructed wetland. In the 2013 field season water volumes were monitored continuously and water samples were taken from the inlet, the wetland cells, and the outlet on a periodic basis. During the season the volume of tile drainage water that reached Elm Creek as surface water was reduced by 82%. The concentration of NO_3-N in the water was not significantly reduced. However, the total load of NO_3-N that reached Elm Creek as surface water was reduced by 262 to 332 pounds (14.4-18.2 lbs./acre). Most of the water that did not reach Elm Creek infiltrated into the subsurface soils and still contained NO_3-N. Using the MPCA's (2013) estimates of groundwater denitrification for agroecoregions, a 45% reduction rate was applied at this location. When the 45% reduction rate is applied to the subsurface load it is estimated that 113.0 to 134 lbs. (6.21-7.36 lbs./acre) of NO_3-N were removed from the infiltrated water. Thus a

total of 124 to 172 lbs. (6.81-9.45 lbs./acre) of NO_3-N were removed from the entire wetland system which accounts for 37.1-43.3% of the NO_3-N.

A concurrent laboratory experiment was set up in 2013 to test the effectiveness of different soils and vegetation at removing nitrate loads. Wetland mesocosm experiments were set up with soil collected from the field site and the design vegetation used in the field cells. Three vegetated mesocosm tanks were planted in Coland soils with Switchgrass (*Panicum virgatum*), Fringed Sedge (*Carex crinita*) and a tank with an equal mix of Dark Green Bulrush (*Scirpus atrovirens*), *Panicum virgatum,* and *Carex crinita.* The results showed that the mixed vegetation regime and the *Panicum virgatum* had significantly greater nitrate removal than the control (Coland bare soil). The mixed vegetation mesocosm had the highest amount of nitrate removal after 10 days at 34.9%. There was no significant difference in the nitrate removal rates in the soils tested.

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CHAPTER 1

BACKGROUND: THE INFLUENCE OF AGRICULTURE IN NITRATE POLLUTION

1.1 FORMS OF NITROGEN

There are many different forms of nitrogen compounds found in Earth's atmosphere, water, and soils. There are four basic forms of nitrogen that are common to Earth: dinitrogen, ammonium nitrogen, organic nitrogen and nitrate (Killpack and Buchholz, 1993). Below is brief description of these forms of nitrogen.

Dinitrogen (N_2) is an atmospheric gas which comprises 78% of the Earth's atmosphere. This inorganic form of nitrate is not readily available for plant growth. However through lightening and biological processes (Killpack and Buchholz, 1993) dinitrogen can be "fixed" naturally into a reactive form of nitrogen known as ammonium nitrogen and ammonia. Humans have also devised a way to fix atmospheric nitrogen into ammonia through the Haber-Bosch process. The ammonia created is often used as a fertilizer for crops (Reddy and DeLaune, 2008). The simplified formula below outlines this process (Kozuch and Shaik, 2008):

$$N_2 + 2H_2 + 8^* \rightarrow 2N^* + 6H^* \rightarrow 2NH_3 + 8^*$$

Ammonium nitrogen (NH_4^+) is an inorganic form of nitrogen and is utilized by plants for growth. This nitrogen compound is commonly used as a fertilizer. Ammonium is more common in acidic soils and waters while ammonia (NH_3) is more common in alkaline soil and water conditions (Reddy and DeLaune, 2008; MPCA, 2013). Natural sources of ammonium include human and animal waste, and mineralized organic matter. Ammonium binds onto soil particles and therefore is not very mobile in the soil. Thus it generally bound in the soils and is less likely to seep downward into groundwater (MPCA, 2013). Ammonium can still be found in groundwater and surface waters, and precipitation, but generally not in large concentrations (Magner and Alexander, 2002; Razania, 2011). In soils ammonia can transform into nitrate thorough a process called Nitrification. Under the right temperatures, soil moisture, and aeration ammonia in the soils can be converted to ammonium, then to nitrite (NO_2^-) and finally to nitrate (NO_3) (Finstein and Miller, 1985; MPCA, 2013). This process is facilitated by microbes that exchange the ions on the nitrogen molecule with those found in soils (Magner and Alexander, 2002)

Organic nitrogen ($C\text{-}NH_2$, where C is a complex organic group) exists in many forms and can be transformed by microorganisms. Terrestrial organic nitrogen is mainly found in decaying organic matter such as plant residues (Sprent, 1987). In nature, organic nitrogen decomposes in the soils and is mineralized into plant available forms (Follett, 2008). Recent studies have shown that organic nitrogen can be taken up by plants before mineralization (Näsholm et. al., 1998). In areas of abundant organic material and low levels of nitrogen, such as forests or rangelands, organic nitrogen can make up a significant portion of the total nitrogen content in waters (MPCA, 2013)

Nitrate (NO_3) is the form of nitrogen that is most pertinent to this research. This form of nitrogen is readily dissolved in water and is negatively charged. This allows nitrate to freely move through oxygenated waters and soils. Thus nitrate can move through tile drainage systems, ditches, surface waters, and into groundwater without transforming into another form of nitrogen. Due to the stability of this molecule in water, nitrate is the dominant form of nitrogen in oxic and suboxic groundwater, in fast moving surface waters (such as rivers and streams), and in surface and subsurface drainage systems (MPCA, 2013).

Nitrate also becomes the dominant form of nitrogen in soils where fertilizer is applied. Whether nitrogen fertilizer is applied as ammonium, urea, manure or NO_3, a large majority the nitrogen applied to the soil will rapidly transform to NO_3 (MPCA, 2013). Thus, elevated nitrate levels are often associated with heavy agricultural activity in locations where fertilizer is necessary for good crop yields (Almasri and Kaluarachchi, 2004).

High levels of nitrate in water can cause many health and environmental issues. Exposure to concentrations of nitrate that exceed 10 mg/L of nitrate-N can contribute to a medical issue in infants known as "blue baby syndrome" or methemoglobinemia. This syndrome causes low oxygen transport in the blood which will cause the skin to turn a blue-gray color. Methemoglobinemia can cause breathing issues and can lead to death if it is not treated (Knobeloch et. al., 2000). High concentrations of nitrate can also become toxic to aquatic life. For example there is a 50% mortality rate in the *Bufo bufo* (common toad) tadpoles when they are exposed to water with concentrations of 384.8 mg/L NO_3-N for 96 hours; and an 84.4% mortality rate when exposed to 9.1 mg/L NO_3-N water for 13 days (Baker and Waights, 1993). While each species has different tolerance levels, in general freshwater animal species have a lower tolerance than marine species (Camargo et. al., 2005). The adult *Echinogammarus echinosetosus* (a freshwater invertebrate) experiences a 50% mortality rate when exposed to concentrations of 56.2 mg/L NO_3-N for 120 hours (Camargo et. al., 2005). The juvenile *Penaeus monodon*, a marine invertebrate species, commonly called the giant tiger prawn, experiences a 50% mortality rate when exposed to concentrations of 1449, 1575 and 2316 mg/L NO_3-N, with seawater concentrations of 15%, 25% and 35% for 96 hours (Tsai and Chen, 2002). The growing hypoxic zone located in the northern Gulf of Mexico is another well-documented environmental concern that is largely caused by an influx of nitrate from the Mississippi River (MPCA, 2013). This is one of the major issues caused by nitrates and will be explained in depth later in the paper.

The examples above demonstrate the harm excessive nitrates can cause. Nitrates ability to move freely through the water and its bioavailability to vegetation means that it can greatly impact the plants, animals, and humans it encounters. This is why nitrate is a major concern and an important topic to research.

There are processes where nitrate can be removed from water. When water becomes anoxic, nitrate can be converted back into nitrogen gas through a process called denitrification (Reddy and DeLaune, 2008). During denitrification the nitrate molecule goes through two intermediated forms, nitric oxide and nitrous oxide (N_2O), before it becomes dinitrogen gas (Alexander 1965).

Nitrous oxide (N_2O) is one of the six greenhouse gases (GHG) to be curbed under the Kyoto Protocol and it makes up 7.9% of the global total anthropogenic GHG emissions (IPCC, 2007). Thus in environments where N_2O can be produced there are often concerns. In denitrification N_2O is rarely found at elevated levels because most often the cycles completes with NO_3 as the end product (MPCA, 2013). Nitrous oxides can be produced, but it has been found that <1% of denitrified nitrogen in rivers (Beaulieu, et. al., 2010), and 0.1 to 1.0% of denitrified nitrogen in lentic waters (Seitzinger and Kroeze, 1998) are emitted as NO_2-N. Also, a study on wetlands in the prairie pothole region reported that N_2O only contributed 1% of the net GHG emissions produced by wetland (Gleason et. al., 2009). These low percentages are likely due to the lack of oxygen available during the conversion process (Beaulieu, et. al., 2010). While N_2O emissions are generally low there are factors that can increase its release into the atmosphere. Peak emissions of N_2O have been recorded during times of higher precipitation and higher soil water-filled pore space (between 40-60%) in prairie pothole wetlands (Gleason et. al., 2009). Other factors that promote the production of N_2O include low activities of the enzyme nitrous oxide reductase and low pH levels in the soil (Gale et. al., 1993; Rasmussen, 2000).

The most relevant forms of nitrogen were explained above but, there are many other forms of nitrogen that have not been discussed here.

1.2 NITROGEN IN AGRICULTURE

Nitrogen is a naturally occurring chemical that is essential in plant growth. This chemical is necessary for the development of proteins, nucleic acids, and other cell elements in plants. While nitrogen is abundant in the atmosphere, it is generally not an abundant chemical in the soil. If it is present it is often in a chemical form that is not readily bio-available for plant uptake (Novoa and Lommis, 1981). Consequently nitrogen can often be a limiting nutrient for plant growth.

Farmers recognized the importance of nitrogen fertilizer and began using more throughout the years. The rate of nitrogen fertilizer application intensified from 58 lbs./acre in 1964 to 125 lbs./acre in 2010; which equates to a 116% increase. During this same time period corn yield was increased by 57% (Figure 1). While other factors also contributed to the increase in corn yield, fertilizer inputs were a significant factor.

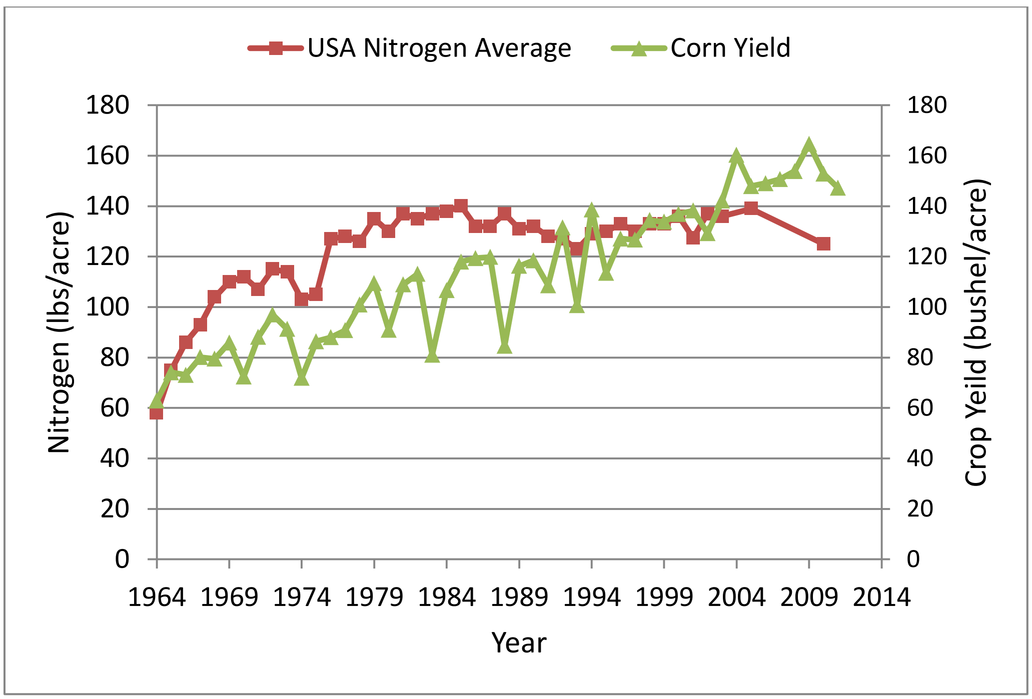


Figure 1. Historical nitrogen rate (pounds/acres) used as fertilizer on agricultural fields for corn compared to historical corn yield.
Data Source: USDA National Agricultural Statistics Service

1.3 EXCESS NUTRIENTS

When excess nitrogen is applied to agricultural fields as fertilizer, nitrate (a dissolved form of nitrogen) will runoff overland or infiltrate into tile drains and discharge into surface waters (Baker et. al.,2006; Follett and Delgado, 2002). These excess nitrates can cause environmental harm.

Eutrophication is an environmental issue that can occur in freshwater lakes and is caused by excess nutrients. In Minnesota’s lakes phosphorous is typically considered the “limiting” nutrient as it often is the main cause of eutrophication. However, nitrogen can also be a limiting nutrient in some Minnesotan lakes. These nutrients can cause large algae blooms. When the biomass dies off and sinks to the lake bottom it decomposes and uses up the available oxygen. The low concentrations of oxygen can harm local aquatic life. These freshwater eutrophic areas in Minnesota area localized areas of concern however; excess nutrients can cause much larger problems (MPCA, 2013)

The hypoxia zone in the Gulf of Mexico is an example of a national environmental concern caused by excess nitrates. Hypoxia is defined as an area of water with dissolved oxygen concentrations that are below 2 mg L^{-2} (Obenour, et. al., 2012). The hypoxic zone in the Northern Gulf of Mexico is driven by the excess nitrates that enter the gulf in the late spring (MPCA, 2013). During this time of year nutrient rich water entering from the Mississippi and Atchafalaya rivers is warmer and less dense than the existing cool ocean saltwater. As a result the incoming freshwater “floats” on top of the ocean saltwater and creates a temporarily vertically stratified system (Rabalais et. al., 2007). This large flux of nutrients allows for a large plume of phytoplankton to thrive.

When the phytoplankton die off naturally a large amount of organic matter sinks to the ocean floor. As this organic material is decomposed oxygen is used up and becomes depleted at the bottom of the ocean (Bianchi, 2010). Consequently a low oxygen “dead zone” in the Gulf of Mexico has formed and is on average 13,500 km^2 (average form 1985-2007). This seasonal phenomenon can last from two to three weeks on the low end, and up to one and a half months on the high end (Rabalais et. al, 2007).

This area of low oxygen is an issue because it can negatively impact coastal fish and invertebrate populations. These organisms often die if they do not escape the area of hypoxia fast enough. Studies have shown that as oxygen levels are depleted the diversity and density of demersal fish populations tend to decline. There are also mortality risks to fish in early life stages. The fish eggs and larvae in this hypoxia area may be abandoned by the male fish making them more exposed to predation. Furthermore, the eggs can also die directly from oxygen depletion. (Breitburg, 2002).

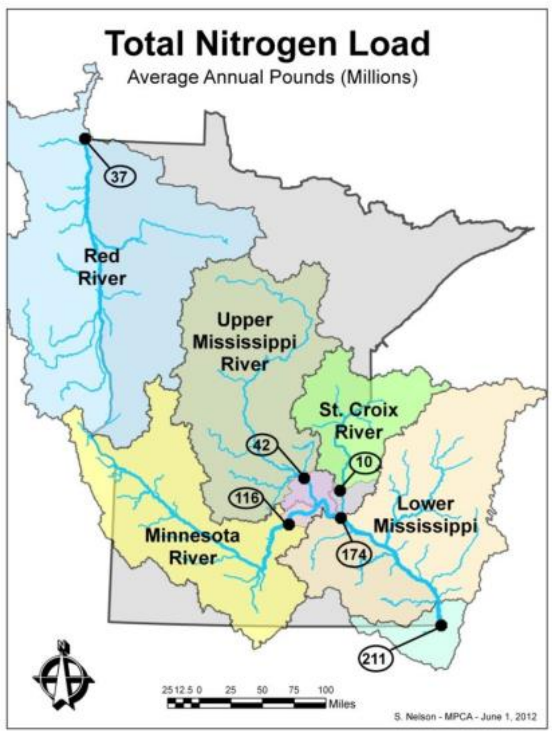


Figure 2. Average (10-15 years) of total nitrogen load in major rivers in Minnesota (Source: MPCA. 2013)

Studies show that 90% of the nitrate that discharges to the gulf originates from non-point sources, (i.e. not a district point or pipe) with the largest contributor being agriculture. Furthermore it was calculated that 50% of nitrogen in the water originates from fertilizers and inorganic nitrogen in soils. The source of naturally mineralized soils could not be easily separated from the unnatural fertilizer inputs because of their strong correlation (Goolsby, 1999).

Minnesota is a large contributor to the nitrate load in the gulf. As water from the Mississippi River flows out of Minnesota it carries an average load of 211 million pounds of total nitrogen per year (Figure 2). The sources of nitrogen are diverse, but an in an average precipitation year an estimated 79% of the nitrogen load comes from croplands. In the Minnesota River Basin an even greater portion, 89%, of the nitrogen load originates from croplands (Figure 3). The Blue Earth River Basin, a tributary to the

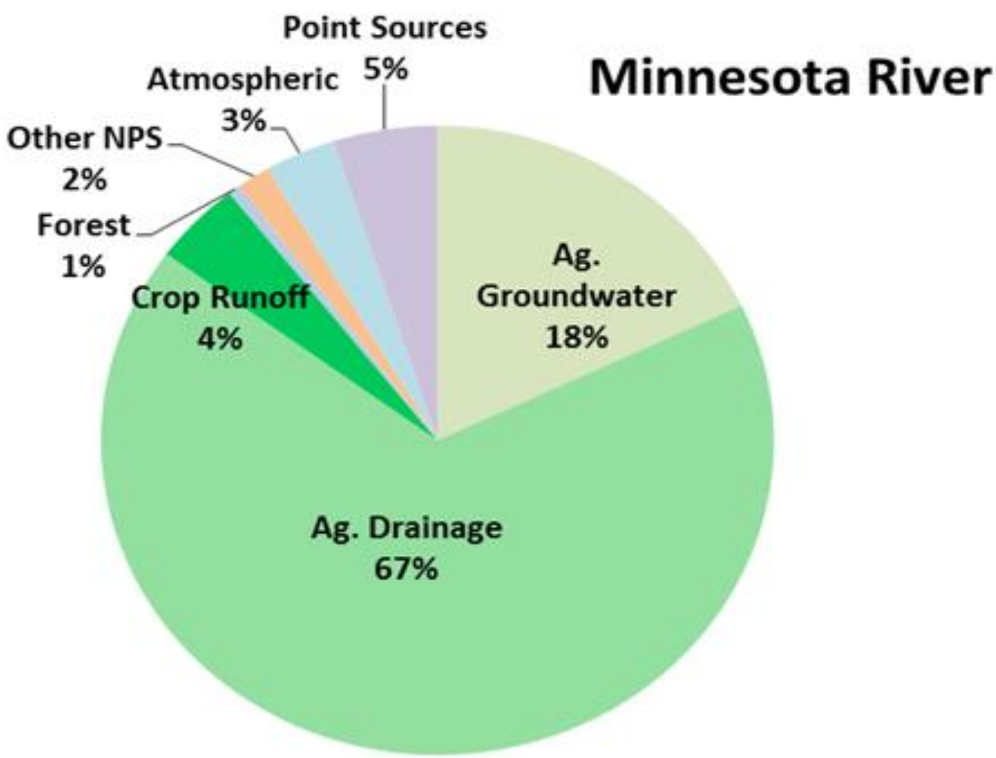


Figure 3. Estimated sources of nitrogen load delivered to surface waters in the Minnesota River Basin (source: MPCA, 2013)

Minnesota River, is the focus region in this research. A study found that this watershed contributes 69% of the nitrate load to the Minnesota River, but only comprised 47% of the stream discharge. Thus this particular area is contributing a disproportionately high percent of the nitrate load (Payne, 1994).

1.4 TILE DRAINAGE

Drainage systems are not new to agriculture. In fact surface ditches were used in 200 B.C. by Egyptians and Greeks (Sutton 1958) and more recently, when colonists came to the America's they brought drainage technology with them. Early settlers even practiced some forms of subsurface drainage. The first clay tile drain was made in 1935 and tile drainage has been evolving ever since (Weaver, 1964). In 1955 a horse drawn revolving-wheel type trenchers, the Pratt Ditch Digger, was introduced. This was followed by many other similar machines all designed for digging trenches (Pavelis, 1987).

Data on farmland drained has been monitored by the U.S. Department of Commerce. According to the 1920 census data 187,232 acres of farmland were drained in Martin County, MN and this drainage area increased to 201,101 acres in 1930. Census data for this Minnesota County was obscured by later surveys when farmers were asked to report land within a drainage project, instead the acres of land drained (Moline, 1933). However, national data from the most recent agricultural survey shows that in 1978, 17% of all agricultural fields in the United States had altered their drainage systems (Pavelis, 1987). Drainage continues today and recent data suggests that approximately 40-70 million acres of subsurface drainage tiles have been laid under the Mississippi basin (Mitsch et. al.,2001).

Modern tile drainage systems consist of perforated tubes that are installed 2-4 feet below the surface. These systems are designed to remove excess water from fields, but cannot prevent saturation. This helps to reduce the risk of waterlogged fields to the farmers, and can increase yield (Eidman, 1997; US EPA, 2012). According to an analysis conducted by Eidman crop yield and profits generally increase 10-15% when tile regimes are installed (1997).

Subsurface drainage helps farmers manage water and can even ameliorate some water quality issues. Drainage can extend the growing season for farmers which can be especially valuable in northern climates. Tile drainage is can be a particularly useful tool for farmers for removing standing water in areas with low topographic relief and in soils with impermeable clay layers (MPCA, 2013). By reducing surface water runoff these systems can reduce soil erosion and associated phosphorous transport (Blann et. al., 2009). However, tile drainage systems can cause some water related issues. By moving water from the fields to the nearest channel quickly peak flows can increase (Eidman, 1997) which can lead to flooding. Tile drainage can also increase the load of nitrates draining from fields into surface water systems. Nitrates are highly soluble and are dissolved in waters, so tile drainage systems can fast track large amounts of nitrates directly into drainage ditches or other waters (US EPA, 2012; Blann et. al., 2009).

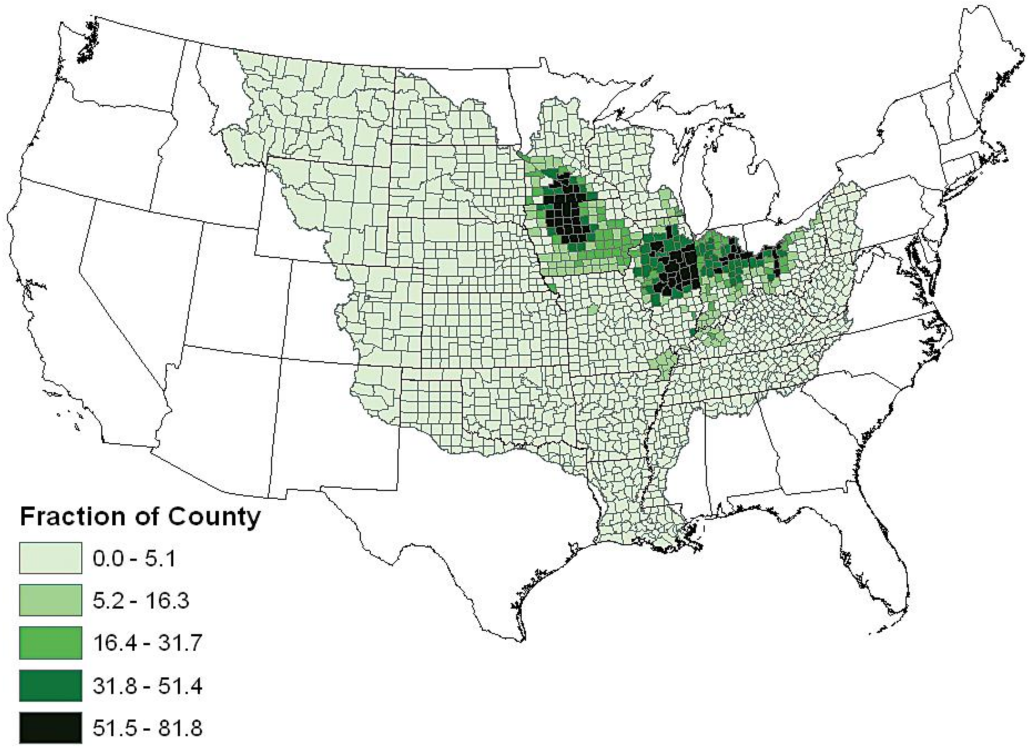


Figure. 4 The estimated fraction of county area that is tile drained in the Mississippi River basin. ArcGIS was utilized to make estimates based on soil survey information and row crop areas. (Source: David, 2010; Original data compiled by: Sugg, 2007).

Today extensive tile drainage systems are installed throughout the Midwest. While the exact amount of tile drainage is unknown studies (Sugg, 2007) have been done to estimate how much tile drained land exists (Figure 4). The areas with the most estimated tile drainage are located in southern Minnesota, Iowa, Illinois, Indiana, and Ohio (David, 2010).

The study area for this research project is located in Martin County which is in southwest Minnesota. In this county it is estimated that there are over 200 drainage systems and thousands of miles of subsurface drainage tile (Meschke and Perrine, 2006). According to David (2010) a majority of the land in this county is likely to be drained by tile. In the process of draining land for agricultural production many wetlands have been lost.

1.5 THE ALTERATION OF WETLANDS

Before European settlement, much of the upper Midwest was classified as swamp (Eidem, et. al., 1999) making these lands difficult to utilize. Living near swamps was not an easy task. There were epidemics of malaria, swathes of mosquitoes and flies, transportation was difficult at best, and these poorly drained lands were hard to farm (Hall and Stall, 1976). Due to these issues and more the Swamp Land Act of 1849 and 1850 (and the additional provision in 1860) were enacted, granting lands to states with the condition that they use the money they earn from the sales to drain the land making it useable for settlement (Shaw and Fredine, 1956).

As more wetlands were lost over time and the government began to realize the values of these lands and changes to the laws were made. In an attempt to preseve some of these lands the federal government enacted the Wetland Reserve Program and the Swampbuster provisions of the 1985 and 1990 Farm Bills (Dahl, 1990, 2000). However many of the wetlands had already been lost.

Since 1850 over 50% of Minnesota's wetlands have been lost because they were drained, filled, or dredged. Regions in the south and west part of the state have seen losses of up to 90% (Freshwater Society, 2007). The region this research paper is focused on has seen a loss of >50% of pre-settlement wetlands (BWSR, 2004). These high percentages are mainly due to the impacts of agriculture (Freshwater Society, 2007).

Unfortunately, the Midwest United States is experiencing issues from nitrate pollution that could have been mitigated or resolved entirely by wetlands. With the passage of the Clean Water Act in 1972 wetlands were finally protected by law (33 U.S.C. §1251 et seq. (1972)). Recently, there has been a push to restore and preserve wetland areas. In 1991 the Minnesota Wetland Conservation Act was approved by legislature. The goal of this act is to have no net loss of wetlands and wetland functions. When a wetland is potentially impacted by activities persons are required to first try to avoid impacting the wetland(s), secondly attempt to minimize impacts, and if an area of the wetland is impacted then it must be replaced with a new wetland of similar ecological value and functions (BWSR, 2004). It will take time to know if wetlands are being replaced and restored successfully (Kentula, 2000).

Recently, there has been a surge in research focused on the effectiveness of treatment wetlands on nitrate removal. Thus, wetlands are being restored and constructed for the purposes of researching nutrient reductions. While this is not a new area of study the success of restored or created wetlands still needs to be researched in different locations, and soil types. In Minnesota, north of Hawk Creek, a successfully designed interception-wetland significantly reduced NO_3-N concentrations from shallow tile drainage water in the cattail and willow portion of the wetland. This research was conducted over 100 miles away from the study in this paper, but it provides insight on the type of work that has been done in this state (Magner and Alexander, 2008). Also, promising studies conducted in southern Minnesota have revealed that restoring natural wetlands can be very successful at removing nitrates. A restored wetland located along

Elm Creek, in Martin County, Minnesota reduced NO_3-N loads by at least 60% and often dropped NO_3-N concentrations below detectable levels (Lenhart, 2008). Also, in Iowa several studies on treatment wetlands receiving tile drainage water have been conducted and conclusions from these studies show significant NO_3-N removal (Iovanna et al. 2008).

However, *constructed* wetlands that are designed to treat tile drainage water are a rare occurrence in south central Minnesota; where this research is located (MPCA, 2013).

1.6 NITRATE TREATMENT IN AGRICULTURE

Besides wetlands there are many other great management practices that can be used to remove nitrates before entering surface water. One best management practice (BMP) being implemented in agricultural fields is a bioreactor. Bioreactors are typically subsurface trenches filled with woodchips or other carbons sources that receive tile drainage waters from the fields. The woodchips are a carbon source for bacteria that drive the denitrification process (MDA, 2012). A study in Boone County, Iowa found denitrification rate potentials ranged from 8.2 to 34 mg N kg^{-1} wood. During the 9 year experiment 75% of the woodchips in the shallowest layer (90-100cm) decomposed and <20% of the woodchips in the deeper layers (155-170cm) decomposed (Moorman, 2010). Thus, over the years the carbon sources in a bioreactor must be replenished to maintain performance. This BMP can be advantageous as it has many benefits such as high rates of nitrate removal, low maintenance, and relatively low installation cost (MDA, 2012)

Controlled drainage is a method used to control the water table level by holding water at a certain level at a water outlet structure. A literature review conducted by Fabrizzi and Mulla (2012), shows that this practice can create reductions in nitrate levels to surface waters ranging from 14-96%. The main source of reduction is not from the nitrate removal processes, but rather by reducing the amount of water that flows into local surface waters. Water will instead be held in the anoxic subsurface waters and allowed time to denitrify before moving into oxic surface waters (MDA, 2012).

Other management practices include: reducing application rates of nitrogen, using cover crops, creating riparian buffers, saturated buffers, 2-stage ditches, and optimizing fertilizer application timing. Not all practices work in every situation, and the best practices can differ greatly depending on variables such as location, soil type, current practices, economics, and local climate (MPCA, 2013). Choosing the best practice can be complicated task. Tools such as the Agriculture BMP Handbook for Minnesota (2012) or the nitrogen BMP watershed planning spreadsheet (NBMP) can be helpful guides for choosing the right BMP (Lazarus et. al., 2013).

CHAPTER 2

EFFECTIVENESS OF NITRATE REMOVAL IN A CONSTRUCTED TREATMENT WETLAND RECEIVING TILE DRAINAGE WATER FROM AN AGRICULTURAL FIELD ALONG ELM CREEK, A BLUE EARTH RIVER TRIBUTARY IN SOUTH CENTRAL MINNESOTA, USA

2.1 EXECUTIVE SUMMARY

Wetlands have the natural ability to remove excess nutrients from water such as nitrate. Extensive tile drainage systems in agricultural fields allow nitrates to be discharged directly into public surface waters. This project focuses on the capability of a treatment wetland to remove excess nitrates from tile drainage water by redirecting the tile drainage through a three celled constructed wetland before the water is discharged into the adjacent Elm Creek. The wetland captures sub-surface tile drainage from a 30 acre agricultural field in south central Minnesota, within the Blue Earth River watershed. Typically, in this region subsurface tile drainage flow is greatest in the spring months and lower flows occur during the growing season because of higher evapotranspiration rates. Nitrate removal occurs when denitrifying bacteria in soils convert dissolved nitrates in water to di-nitrogen gas (N_2) which is released into the atmosphere. Ideally the local hydric soils and native wet-prairie mix seedlings in the wetland would help facilitate this process. Throughout the 2013 field season nutrient and hydrologic data were taken at the inlet, outlet, and in the throughway between ponding

cells. Nitrate concentrations were expected to be reduced as water traveled though the wetland, but the data on these concentrations did not show a significant reduction. However, the overall volume of surface water from the tile drainage inlet was reduced by 81.9% before it was discharged into the creek.

Hence, the total load of NO_3-N that reached Elm Creek as surface water was reduced by 262 to 332 pounds (14.4-18.2 lbs./acre). Most of the water that did not reach Elm Creek infiltrated into the subsurface soils and still contained NO_3-N. Using the MPCA's estimates of groundwater denitrification for agroecoregions, a 45% reduction rate was applied at this location. When the 45% reduction rate is applied to the subsurface load an estimated 113.0 to 134 lbs. (6.21-7.36 lbs./acre) of NO_3-N were removed from the infiltrated water. Thus, an estimated total of 124 to 172 lbs. (6.81-9.45 lbs./acre) of NO_3-N were removed from the entire wetland system which accounts for 37.1-43.3% of the total NO_3-N. The results of the project show that even in the first year of establishment this constructed wetland was effective at reducing surface water nitrates. The lower than expected denitrification could be contributed to the soils, weather conditions, and the short hydraulic residence time. As organic carbon builds up in this wetland results should only improve. Lessons learned from this treatment wetland will be useful in similar projects around the region, especially in the northern Corn Belt or more certainly in the Des Moines Lobe glacial till plain of Iowa and Minnesota.

2.2. INTRODUCTION

Wetlands provide many valuable ecological services. Some of the benefits include: filtering out pollutants, mitigating flooding effects by slowing runoff water, reducing erosion and providing habitats for birds, aquatic insects, fish, and mammals. Additionally 43% of the United States threatened or endangered species live in or depend on wetlands (Minnesota DNR, 2014).

One of the most important water quality benefits of wetlands is their ability to remove excess nitrogen. There are two main systems that wetlands use to remove nitrates. The first is through plant uptake. Wetland plants can directly uptake nutrients like nitrate into their systems (Silvan, 2004). Harvesting the vegetation periodically can help increase nitrogen removal efficiencies (Hammer, 1992). Additionally late fall harvests in temperate climates can prevent nutrients such as nitrogen and phosphorus from reentering the water (Vymazal, 2006).

The second way wetlands can remove nitrates is through bacterial denitrification. For bacterial denitrification to occur there must be anoxic conditions (<0.5 mg/L oxygen) present (Dubrovsky, 2010). In oxygen saturated environments bacteria will utilize the electrons in the oxygen molecules. However, if the water contains little or no oxygen then nitrates become the more favorable electron donor. Thus the denitrifying bacteria use the electrons from the nitrate and transform nitrate into nitrogen gas (N_2) (Knowles, 1982). During this process the nitrate molecule goes through two intermediated forms, nitric oxide and nitrous oxide (N_2O), before it becomes dinitrogen gas (Alexander 1965). Most gas is released into the atmosphere as N_2 (Knowles, 1982), but a small amount is released as N_2O, which is a greenhouse gas that contributes to ozone depletion (US EPA, 2011). In general the nitrate removal by plant uptake is insignificant compared to the amount removed by bacterial facilitation (Hammer, 1992).

Treatment wetlands have been constructed to mimic the functions of natural wetlands. These wetlands are used to treat wastewater from industrial and agricultural settings, and other types of wastewaters. Studies have been conducted on various aspects of treatment wetlands including: the diversity of microbial communities (Faulwetter et. al., 2009), microbial processes (Sleytr et. al., 2009), and the role of Macrophytes (Weisner et. al., 1994 and Zhao et. al., 2009). A review of studies has shown that treatment wetlands can be effective at removing pollutants, with total nitrate load removal ranging from 40-55%, and phophorus removal ranging from 40-60% (Vymazal, 2006).

While research has been conducted on the effectiveness of this technology it is an area that needs to be further studied as much of the work has been conducted in laboratories or controlled settings. There is still much to be learned about the complex wetland systems, such as how we can best recreate their natural systems to remove pollutants from waters, and how they can be utilized most effectively.

The main goal of this project is to construct a treatment wetland that will successfully remove nitrate from agricultural tile drainage water. A secondary goal is to remove phosphorus from the water as well. On the research farm field in this study the tile drainage system was previously discharging nutrient rich water directly into the nearby Elm Creek (Figure 5). In the spring of 2013 a three celled treatment wetland was constructed and tile drainage water was re-routed to flow into it. The main research objectives are to (1) quantify nitrate removal efficiencies and (2) assess the water budget within the confines of the wetland.

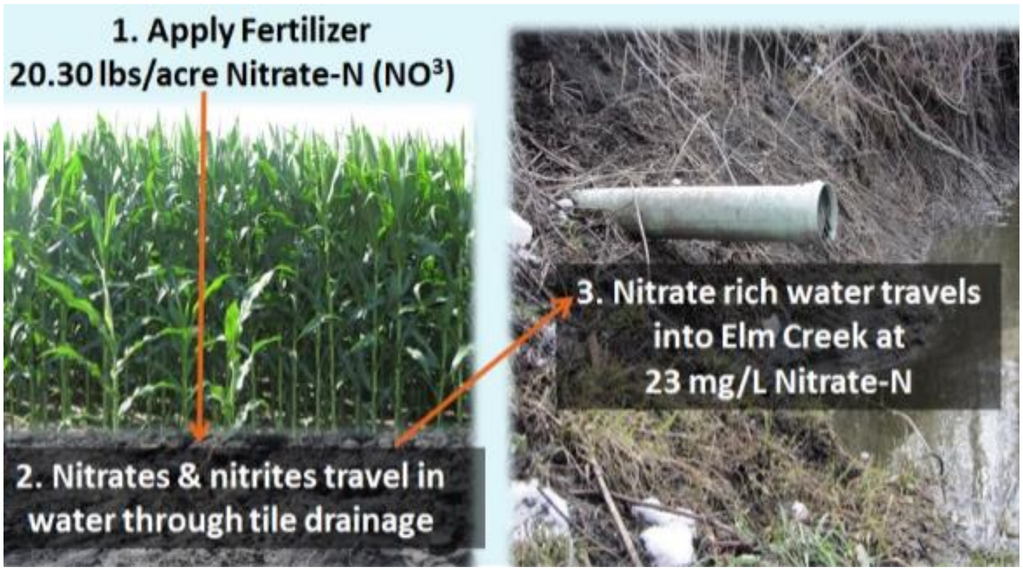
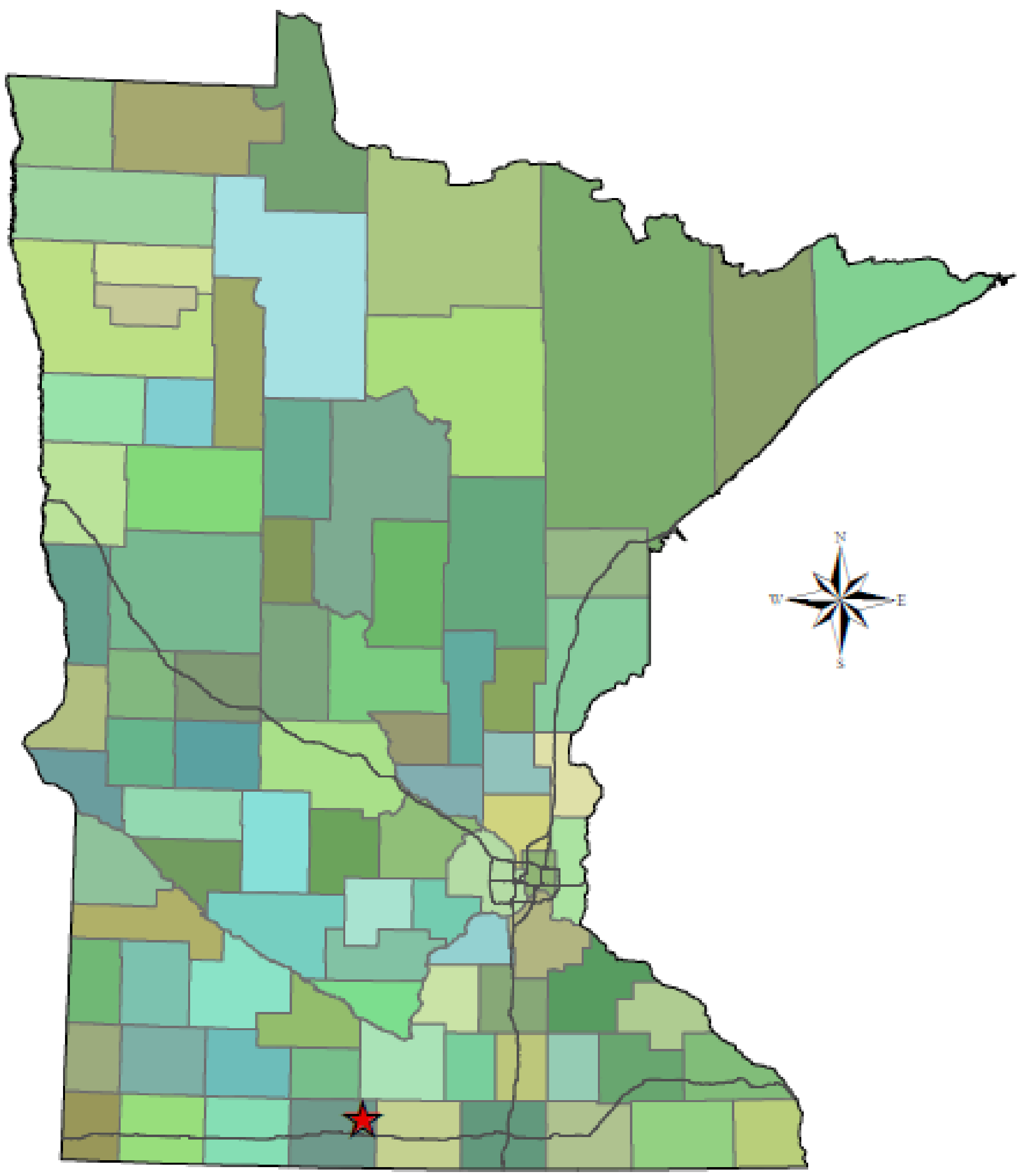


Figure 5. The nitrate path in the tile drainage system on the field site before tile was rerouted into the treatment wetland.

2.3 MATERIALS AND METHODS

2.3.1 Study Site

A treatment wetland was constructed at the agricultural farm located in Granada, MN (43 45’ 4’’N, 94 20’ 51’’W) (Figure 6). The constructed wetland is located between Elm Creek and the northern edge of the row crop field. Elm Creek, a stream that is utilized as a drainage ditch for many local farms, meanders through this agricultural property. A strip of riparian perennials, 67’9” wide, separates the wetland from the creek. The west and south sides of the wetland are abutting row crops, and the east side is adjacent to a sparsely vegetated hill (Figure 7). The field was planted with corn for the growing season of 2013 and the farmer practices rotational farming and will rotate between soybeans and corn. This cropland is approximately 20.2 hectares (50.0 acres) and is divided into three drainage systems. This wetland receives tile drainage water from approximately 7.4 hectares (18.2 acres) of cropland.



Data retrieved from the Minnesota DNR Deli

Figure 6. General project location for the constructed treatment wetland

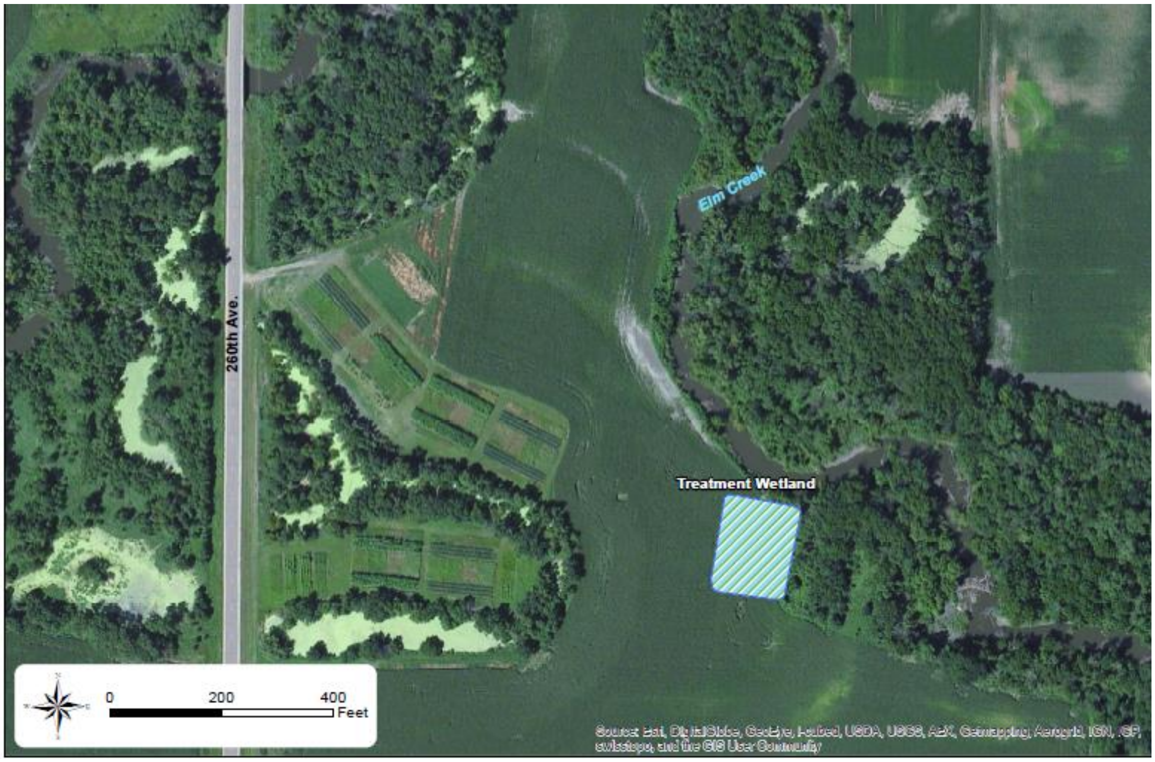


Figure 7. Treatment wetland research site

The excess water in the constructed wetland discharges into Elm Creek which is a tributary to in the Blue River Basin. Water from Elm Creek first flows into Blue Earth River, merges with the Minnesota River, and then joins with the Mississippi River which eventually outlets into the Gulf of Mexico (Figure 8).

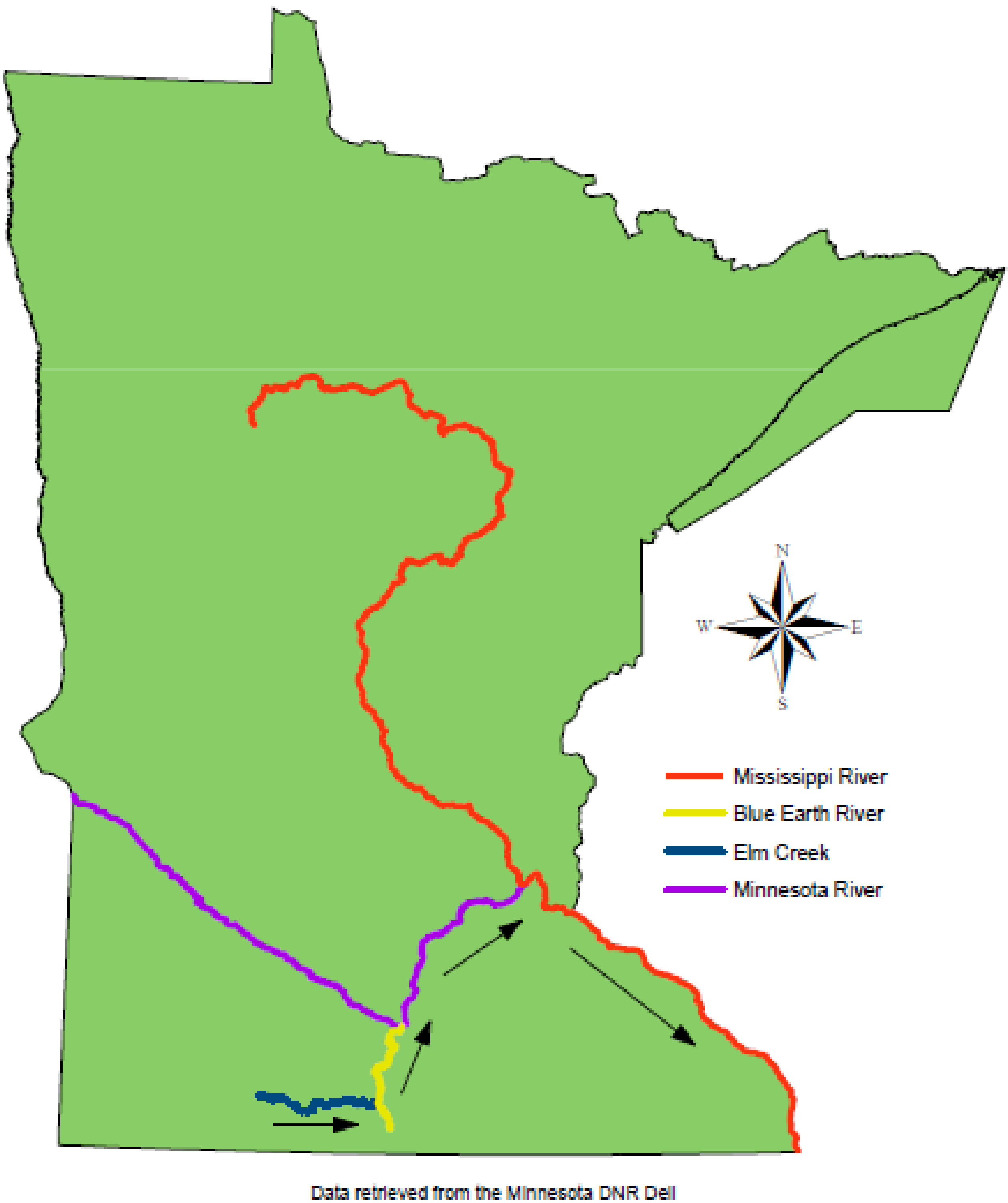


Figure 8. Water flow path from Elm Creek to Mississippi River

→ Flow path

2.3.2 Wetland Design

The treatment wetland was constructed in the early spring of 2013. Including berms the wetland is 135 feet wide and 175 feet long for a total of 23,625 ft^2 (0.219 hectares). The layout includes three separate treatment cells within the wetland; each treatment cell is 45 feet wide by 87.5 feet long for a total of 3,937 ft^2 (0.0365 hectares). The area of active treatment is 11,812.5 ft^2 (0.110 hectares) and includes base and aquatic shelf in all three cell boundaries. Each cell is separated by lower berms that are 1.5 feet in height. The entire wetland is bordered by a larger berm that is 4.5 feet higher than the wetland bottom. At the outlet there is an auxiliary spillway that allows for overflow scenarios (Appendix A).

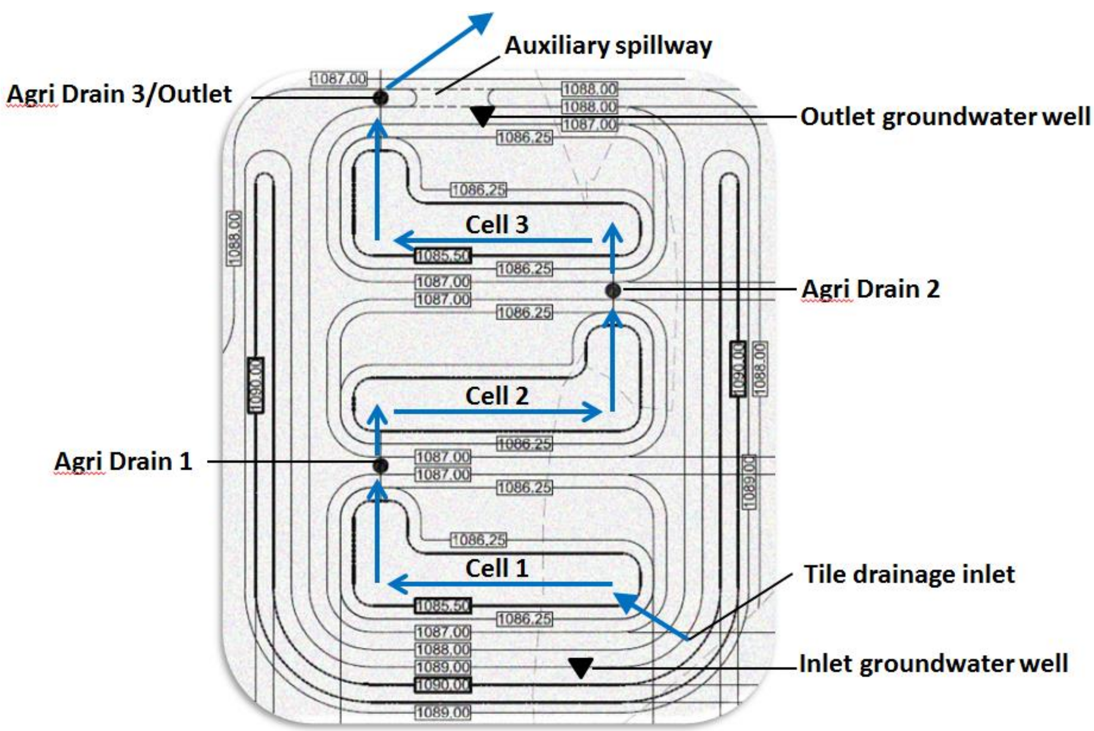


Figure 9. A depiction of the surface water flow through the constructed treatment wetland and the locations water monitoring equipment.

Water from the tile drainage system is directed to a controlled inlet point which pours into the first cell of the wetland. An Agri Drain inline control structure (Appendix B) is located diagonally across from the inlet in the first cell. The Agri Drain controls water levels and the flow into the next cell using stop logs. The control structures also help to prevent back flow and facilitate in flow measurement. The surface water must be pooled above the height of the stop logs to pass through the Agri Drain. For this wetland the height of the all stop logs were held constant at seven inches. More stop logs can be used to force the water to pool even higher within the wetland. The seven inch height was chosen so to ensure that water would flow through all three cells and to allow more data to be collected throughout this first field season. This same setup is repeated for cell two and cell three. Water enters each cell and then it must travel across the cell diagonally to the next control structure, rise above seven inches, and then it discharges into the next cell or the outlet (Figure 9).

Once the water has meandered across all three cells the water flows into the outlet pipe and control structure and then drains into Elm Creek. Previously, the tile drainage water from this system would outlet directly into Elm Creek.

Two ground water wells were installed in the wetland (Figure 9). One was installed on the southern berm nearest the inlet; this well will be referred to as the inlet groundwater well. The inlet groundwater well was installed 52.5 inches (4.38 feet) below ground. Another well was installed on the northern most berm nearest the outlet; this well will be referred to as the outlet groundwater well. The outlet groundwater well was installed 79.0 inches (6.58 feet) below ground. Both wells were screened the across their entire length.

After the grading and major construction was complete a seed mixture, MnDOT lot number 33-261 (Appendix C), was spread on the berms and then erosion control matting was placed over the berms. The first, second, and third wetland cells were seeded with a low, medium, and high diversity wet prairie mix, respectively (Appendix D). The first cell was additionally seeded with Switchgrass (*Panicum virgatum*) and MnDOT lot number 33-261 (Appendix C).

In some cases clay liners are used to prevent excess infiltration. However due to the high clay content of the soil (41.3-46.3%) liners were not used in this design (Table 1).

Sample Name	Sand (%)	Silt (%)	Clay (%)	Soil texture
Cell 1	9.9 / 12.6	43.8 / 41.2	46.3 / 46.2	Silty Clay
Cell 2	5.0	53.8	41.3	Silty Clay
Cell 3	8.7	45.0	46.3	Silty Clay

Table 1. Soil texture analysis results from Research Analytical Laboratory at the University of Minnesota using the Hydrometer method. Soil collected in November, 2013.

Before any construction began predictions on nitrate removal levels were made using a program called DRAINMOD in combination with a level-pool routing, mass balance model. It was predicted that 69% of the surface water nitrate load would be removed by this wetland. These preliminary predictions were based on slightly different parameters than the actual constructed wetland. The model used an area of 0.45 acres of wetland rather than the 0.542 acres it was constructed at. The input for the average inflow nitrate concentration was 15.33 mg/L which was lower than the 23.0 mg/L NO_3-N that was calculated at for the tile drainage to this site. Also, the model assumed that the entire area was densely vegetated, when it reality it was essentially bare soil during most of the 2013 field season (Karlheim, 2012).

2.3.3 Equipment Setup

Monitoring equipment was set up on site to take continuous measurements. At the wetland inlet and outlet ISCO area velocity flow loggers (Appendix B) were mounted. These probes record the velocity and height water in fifteen minute intervals. Solinst Leveloggers (Appendix B) were suspended in the Agri Drain structures that controlled flow from cell one to cell two and from cell two to cell three. Leveloggers were also suspended in both groundwater wells. Leveloggers measured the height of water in ten minute intervals. Temperature was recorded using a weather station and a temperature probe; which were set up less than 50 meters away. Rainfall totals were collected from neighboring cities, Fairmont and Winnebago, Minnesota instead. The rainfall data was collected by the National Weather Service Forecast Office (NOAA, 2014).

2.3.4 Water Sampling

Water sampling was conducted throughout the spring, summer and fall of 2013. Water samples were collected from the inlet, the first and second cell Agri Drains, and the outlet. The parameters tested include: Specific Conductivity, Nitrate and Nitrite as N, Ortho Phosphorous, Total Phosphorus, and Total Suspended Solids (TSS).

2.3.5 Water Volumes Calculations

Water volumes at the inlet and outlet were calculated using data collected from the area velocity probes. The probes recorded both the velocity and depth of water. Both probes were mounted in pipes approximately six inches in diameter. Using the recorded heights, area of flow in the pipes was calculated (Appendix E). The water volumes were then calculated using the recorded velocities and calculated areas of flow (Appendix E). Water coming from direct rainfall was ignored as it was a minor component of the total.

Water volumes discharging into cell two and three were calculated using data collected from leveloggers and a barologger. The water level (m) was recorded on each levelogger and the barometric pressure (kPa) was recorded on the barologger. Barometric pressure corrections were made to the levelogger heights. Each levelogger was suspended in the rectangular Agri Drain structures and the barologger was suspended at the top of a nearby groundwater pipe. Using the corrected height, computations were made to find the velocity (Appendix F) and area of flow. The water volumes were then calculated using the calculated velocities and areas of flow.

2.3.6 Evapotranspiration Rates

A modified Hamon method was used to calculate the potential evapotranspiration rate (PET). It provides good long-term estimates with minimal data requirements. This equation does not respond to wind speed and sporadic weather variations on hourly or daily time scales. This equation was modified to fit conditions at a prairie wetland in the Midwest (Roseberry et al., 2004). Average daily air temperature and day light hours were the only input data required to calculate the rates. Temperature data was collected on site. PET is estimated as:

$$PET = 0.656 \left(\frac{D}{12}\right)^2 \frac{SVD}{100} (24.5)$$

$$SVD = \frac{216.7 \cdot e_s}{T + 273.3}$$

$$e_s = 6.108 \cdot e^{\frac{17.26 \cdot T}{T + 237.7}}$$

<u>Where</u>:
PET = Potential evapotranspiration (mm day^{-1})
D = hours of daylight
SVD = saturated vapor density at mean daily air temperature in (g m^{-3})
T = mean daily air temperature (°C)
e_s = saturated vapor pressure at air temperature (mb)

The PET value was then converted to m day^{-1} and used to estimate the volume of water leaving the system each day through evapotranspiration. The volume was adjusted based on an estimated amount of surface water present in the wetland on a daily basis:

$$ET\ (m^3) = PET\ (m\ day^{-1}) \cdot Surface\ Water\ (m^2)$$

2.3.7 Infiltration Rates

Infiltration rates were estimated using data collected on site. The water infiltration rates were estimated using a Philip-Dunne infiltrometer (Appendix B). Using this equipment the change in water level over time was measured. This rate translated into the infiltration rate of water within the wetland boundaries. Using the estimated rate of infiltration and the estimated area of ponded water the volume of infiltrated water was calculated on a daily basis. The following equation was used to calculate infiltration volume per day:

$$\text{Volume } (m^3 \ day^{-1}) = \text{Infiltration Rate } (m \ hour^{-1}) \times 24 \ (hours \ day^{-1})$$

This equation was adjusted for June 25th and July 6th because those days only had data for part of the day due to the flood that began at 12:30am on June 25th and seceded by 4:30am on July 6th. Rather than using 24 hours in the equation 0.5 hours was used on June 25th and 23.5 hours was used for July 6th. Also, the daily infiltration volume was not allowed to exceed the volume of water available and was adjusted accordingly. The equation for the infiltration limit is below:

$$\text{Infiltration Limit } (m^3) = \text{Inlet Volume } (m^3) - \text{Outlet } (m^3) - \text{ET } (m^3)$$

2.3.8 Water budget

The water volume coming through the inlet, discharging through the outlet, infiltrating into the soils, and leaving through evapotranspiration has been calculated above. The inlet water is considered to be 100% of the water coming into the system. There was some water entering directly as rainfall, but this was a small fraction compared to the amount of tile drainage water. There also is sporadic flooding in this area; water volumes during flooding were emitted. The volume of surface water entering and leaving the wetland system can be accurately measured. The volume of water that infiltrated into the soil or that left the wetland through evapotranspiration was estimated. A water budget is the amount of water coming into a system minus the amount of water leaving a system and the remaining water is the change in water level or in this scenario the amount of ponded water remaining. The following equation is used to depict the water budget:

$$\mathrm{Inlet} - (\mathrm{Outlet} + \mathrm{ET} + \mathrm{Infiltration}) = \mathrm{Ponded\ water}$$

2.3.9 Nitrate Load

The nitrate load in the surface water was determined through data collected. The concentration of nitrate coming into the wetland and the volume of water going through the inlet tile drain will determine the starting load. The outlet load for nitrate in the surface water is determined using the concentration of nitrate in the water samples and the known volume of water passing through the outlet pipe.

To determine the subsurface inlet nitrate load the concentrations of the surface water nitrates and the estimated volume of infiltrated water were used. However, the concentration of subsurface water nitrates is unknown so the nitrate load after infiltration can only be estimated.

2.3.10 Groundwater Well Measurements

Leveloggers suspended in these wells recorded the water level. Barometric pressure readings were used to make pressure corrections for the levelogger heights. A laser level was used to determine the relative heights of the groundwater wells and thus the relative elevation of the water in the wells could be compiled. These water levels were used to e the groundwater flow during the field season.

2.3.11 *Soil Samples*

Soil samples were taken from this farm before any construction was conducted. Additional soil samples were taken from the constructed wetland after the end of the first field season. These samples were analyzed for soil texture, and total organic carbon content at the University of Minnesota. All samples were preserved in a cool container and taken to the University of Minnesota's Research Analytical Laboratory for analysis.

2.4 RESULTS

Data was collected to during the 2013 field season. The wetland was flooded by surface water overflow from Elm Creek during June 25^{th} to July 6^{th} and data collection was disrupted during this time. The field season was effectively broken up into two time periods; "pre-flood" and "post-flood". The pre-flood season occurred during early summer and the post-flood occurred during the mid-summer and fall seasons.

2.4.1 Surface Water Volume

Daily water volumes were calculated at different points along the treatment system for the inlet, Agri Drain#1, Agri Drain#2 and the outlet (Appendix G). These values were used to calculate seasonal surface water volumes in the wetland.

Pre-flood water volumes were collected from May 18^{th} to June 25^{th}. Water did not begin to flow into the wetland from the tile drainage until June 5^{th}. During this time a total of 6.94×10^3 cubic meters (m^3) of water entered the wetland. Of this water, 59.5% discharged into the second cell, 40.1% discharged into the third cell, and 18.9%

discharged from the outlet (Figure 10). Thus, 81.1% of the water during the pre-flood period never surface discharged to Elm Creek.

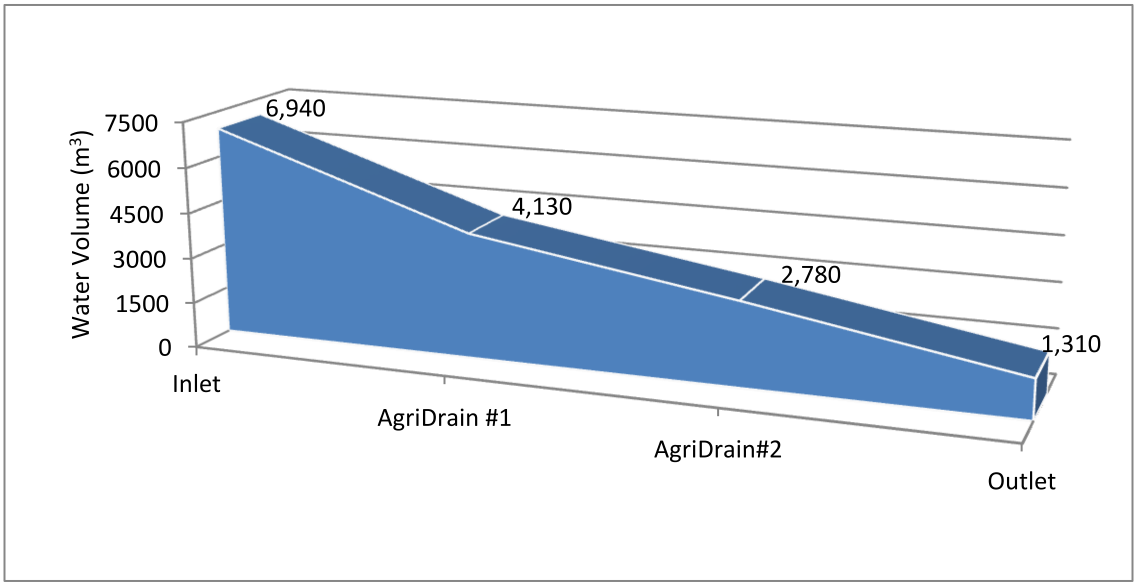


Figure 10. Water volumes at selected sites in the wetland complex during the pre-flood period.

Post-flood water volumes were collected from July 6th through November 7th. Water stopped flowing into the wetland from the tile drainage on September 11th, 2013. Volumes of less than 2.0×10^{-5} m^3 were recorded on September 18th and 22nd, but these volumes were negligible. During this time 3.04×10^2 m^3 of water entered the wetland. Post-flood water accounted for 4.2% of the total water. Of this water, 15.1% discharged into the second cell, 0.3% discharged into the third cell, and 0% discharged from the outlet (Figure 11). Therefore, 100% of the water during the post-flood period never surface discharged to Elm Creek.

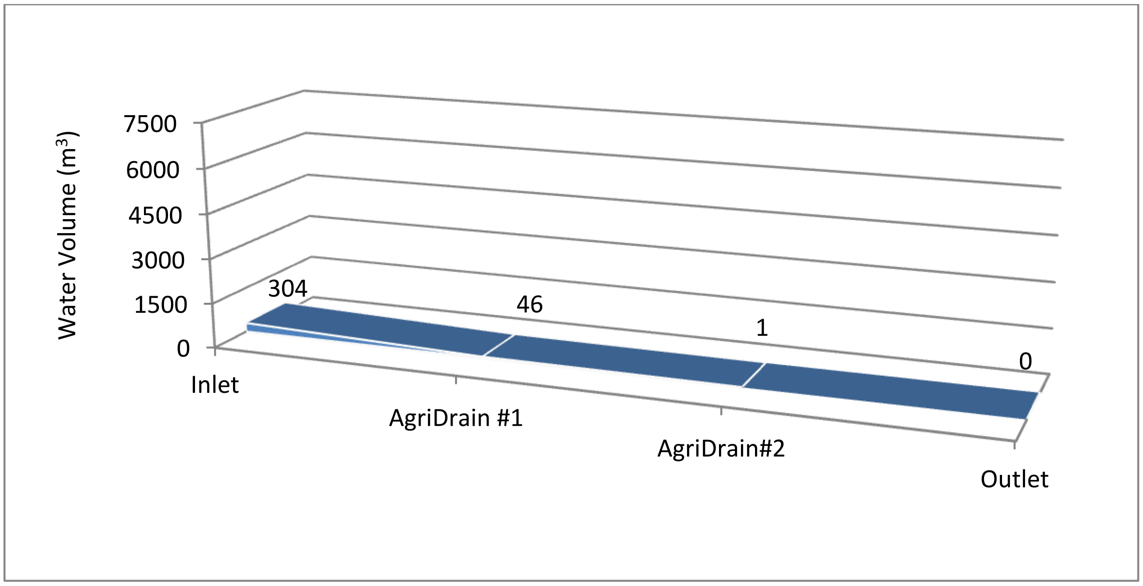


Figure 11. Water volumes at selected sites in the wetland complex during the post-flood period.

Over the entire field season a total of $7.24x10^3$ m^3 (5.9 ac-ft.) of water discharged into the wetland via the tile drainage system (Figure 12). Of this total water 18.1% discharged through the surface water outlet pipe. Water was flowing into the wetland for a total of 50 days and flowing out of the wetland through outlet for a total of 12 days. The water that did not reach the outlet as surface water either infiltrated into the soil, remained ponded, or was removed through evapotranspiration processes.

Velocities were also recorded at the inlet and outlet. The maximum inlet flow rate for the season was 1.98 cubic feet per second (cfs). The maximum outlet flow rate was recorded at 0.940 cfs. Using the drainage acreage a maximum rate of 0.109 cfs/acre was calculated for the inlet and 0.052 cfs/acre for the outlet.

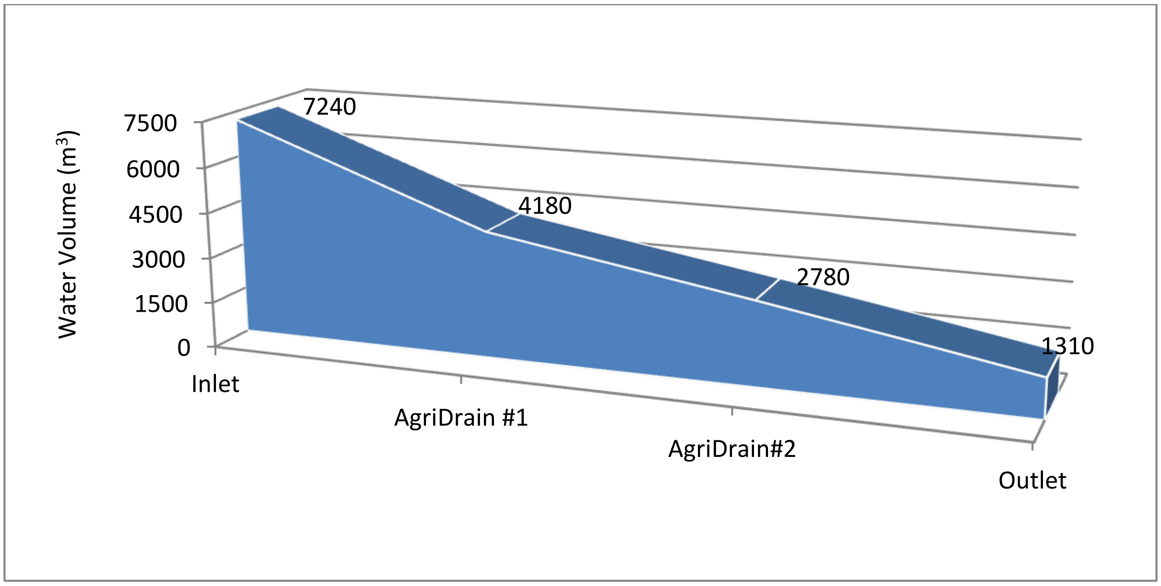


Figure 12. Water volumes at selected sites in the wetland complex during the entire field season.

2.4.2 Evapotranspiration

The average daily temperature at the field site was greatest in the month of July with an air temperature of 22.4°C (72.32°F). Calculated potential evapotranspiration (PET) rates are strongly correlated with average temperatures (Figure 13). A Pearson product-moment correlation coefficient was computed to assess the relationship between the temperatures and PET rates. There was a positive correlation between the two variables ($r = 0.909$, $n=7$, $p= 0.00228$; one-tailed t-test). The day length also factored into the calculations for the PET rates. Temperature and potential evapotranspiration (PET) averages were computed using only days where data collection was occurring. Dates that were not incorporated in the calculations include: days when the wetland was flooded, before data collection began (earlier than May 18th), and later than November 7th when data collected ceased.

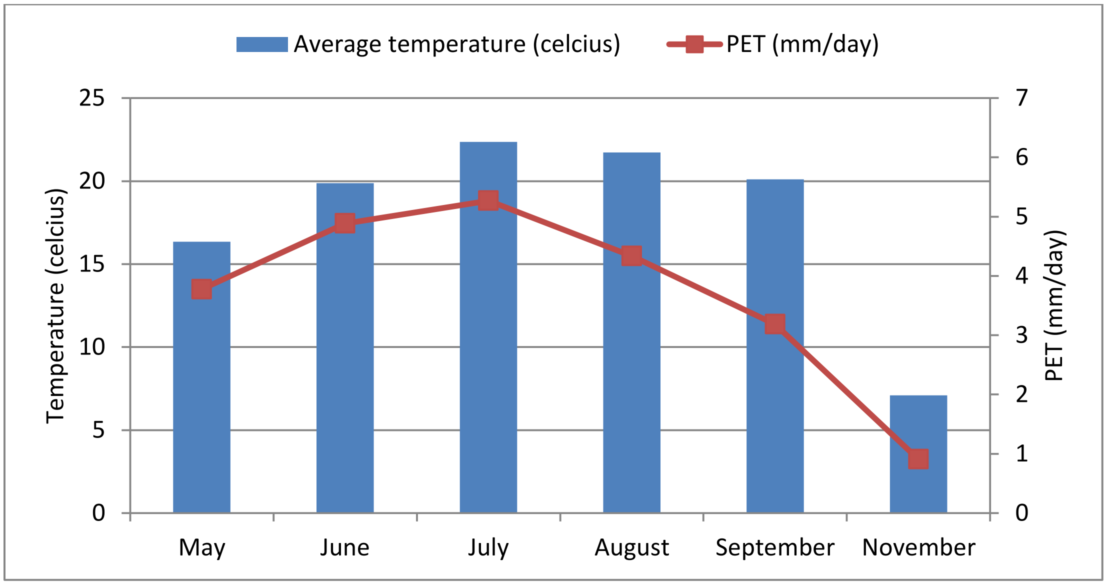


Figure 13. Monthly averages of air temperature and potential evapotranspiration for 2013 field season. Temperatures were collected on site.

The estimated evapotranspiration volumes were also calculated based on the surface area of the water ponded in the wetland (Figure 14). Rainfall data for May-November from nearby Winnebago were uploaded using the Nation Weather Service Forecast Office website, MN (NOAA, 2014). June had the greatest amount of rainfall at 8.41 inches. The month of June had at least 2.8 times more rain than the following months (Appendix H).

Rainfall totals correlated with the amount of evapotranspiration. A Pearson product-moment correlation coefficient was computed to assess the relationship between the amount of rainfall and actual ET volumes. There was a positive correlation between the two variables ($r=0.917$, $n= 7$, $p = 0.00182$; one-tailed t-test). The more water available the more evapotranspiration could occur. Rainfall totals and evapotranspiration (ET) averages were computed using only days where data collection was occurring. Dates that were not incorporated in the calculations include: days when the wetland was flooded, before data collection began (earlier than May 18th), and later than November 7th when data collected ceased.

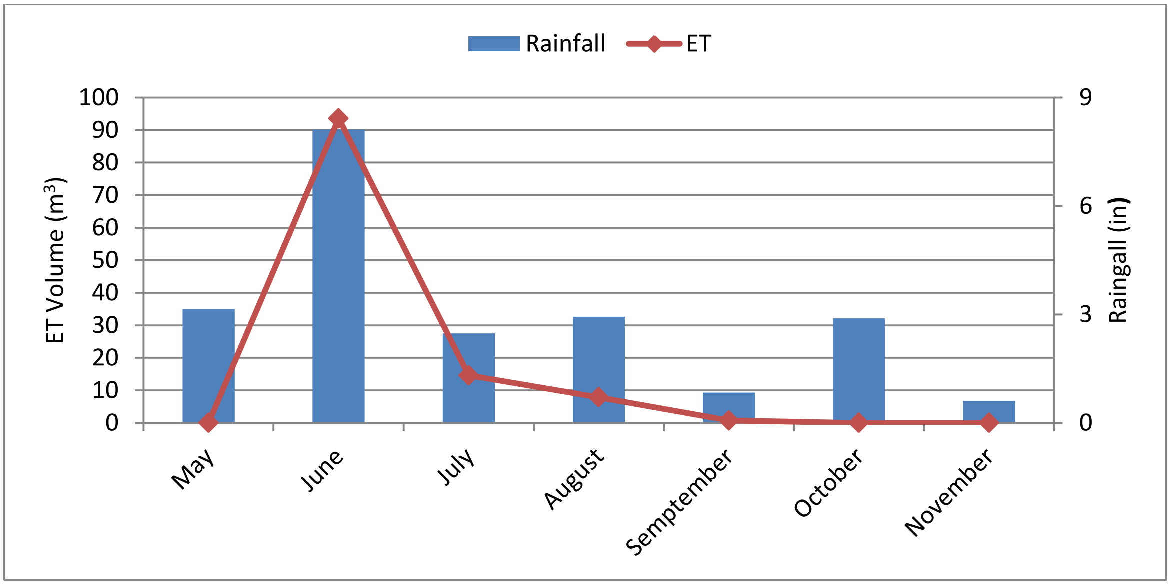


Figure 14. Monthly Evapotranspiration (ET) volumes and rainfall totals in 2013 at the wetland field site. ET was estimated using on site temperatures and rainfall was estimated using totals collected from Winnebago, MN.

2.4.3 Water Budget

Water that entered through the tile drainage pipe is considered to be the inlet water. Rainfall water volume was insignificant in comparison and difficult to estimate accurately and was not considered in the calculations.

For the pre-flood season a total of 6,936.6 m^3 (5.62 ac-ft.) of water entered the wetland inlet from the tile drainage. Approximately 1,313.6 m^3 (1.07 ac-ft.) of water flowed through the wetland and discharge through the outlet which leads into Elm Creek. An estimated 93.6 m^3 (0.08 ac-ft.) of the water was removed from the system through evapotranspiration. Another estimated 5,112.3 m^3 (4.15 ac-ft.) of water infiltrated into the subsurface. The excess water, 417.2 m^3 (0.34 ac-ft.), remained ponded in the wetland (Figure 15).

Pre-flood water budget (m^3)

Inlet (6,936.6) – [Outlet (1,313.6) + ET (93.6) + Infiltrated (5,112.3)] = Ponded water (417.2)

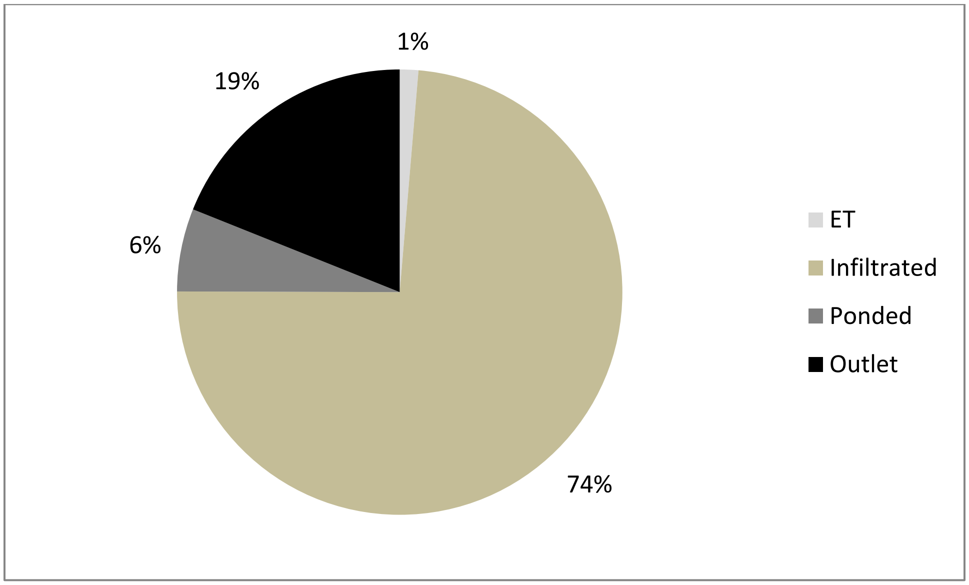


Figure 15. Water volume distribution of outflow for the treatment wetland in the pre - flood season.

For the post-flood season a total of 304.4 m^3 (0.25 ac-ft.) of water entered the wetland inlet from the tile drainage. Water that remained ponded from the flood is not accounted for in the inlet total. No volume of water discharged through the outlet into Elm Creek. An estimated 19.4 m^3 (0.02 ac-ft.) of the water was removed from the system through evapotranspiration. The remaining volume of water, 285.0 m^3 (0.23 ac-ft.), infiltrated into the soils. By the end of this data collection season no water remained ponded in the wetland. Therefore, 6% of the inlet water was removed through ET and 94% of the inlet water infiltrated into the subsurface (Figure 16).

Post flood water budget (m^3)

Inlet (304.4) – [Outlet (0) + ET (19.4) + Infiltrated (285)] = Ponded water (0)

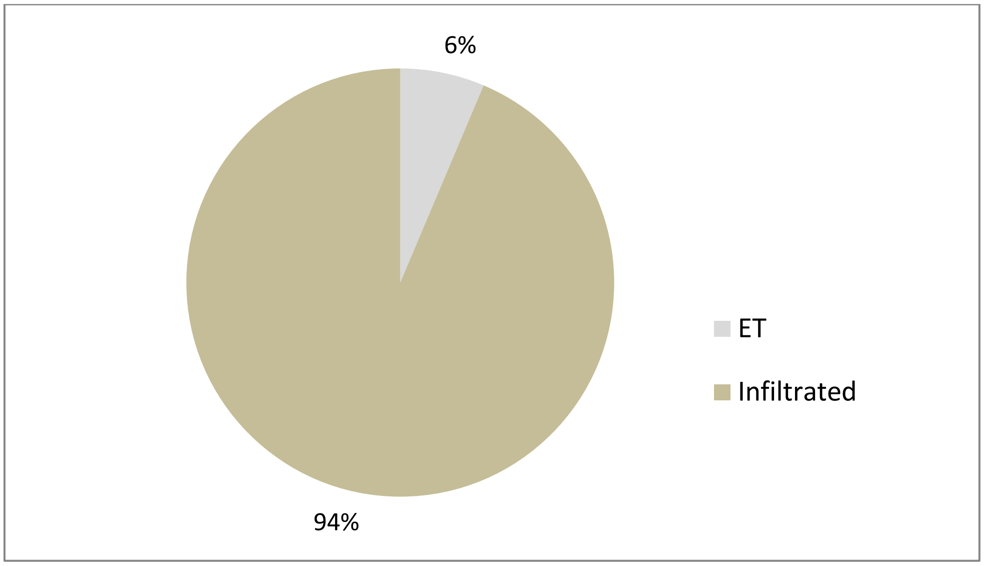


Figure 16. Water volume distribution of outflows for the treatment wetland in the post flood season.

For the entire field season a total of 7,241.0 m^3 (5.87 ac-ft.) of water flowed into the wetland inlet from the tile drainage. Approximately 1,313.6 m^3 (1.07 ac-ft.) of water flowed through the wetland and discharged to the outlet which leads into Elm Creek. An estimated 112.8 m^3 (0.09 ac-ft.) of the water was removed from the system through evapotranspiration. Another estimated 5,397.2 m^3 (4.38 ac-ft.) of water either infiltrated into the subsurface. The excess water, 417.2 m^3 (0.34 ac-ft.), remained ponded in the wetland (Figure 17).

Seasonal water budget (m^3)

Inlet (7,241.0) – [Outlet (1,313.6) + ET (113.0) + Infiltrated (5,397.3)] = Ponded water (417.2)

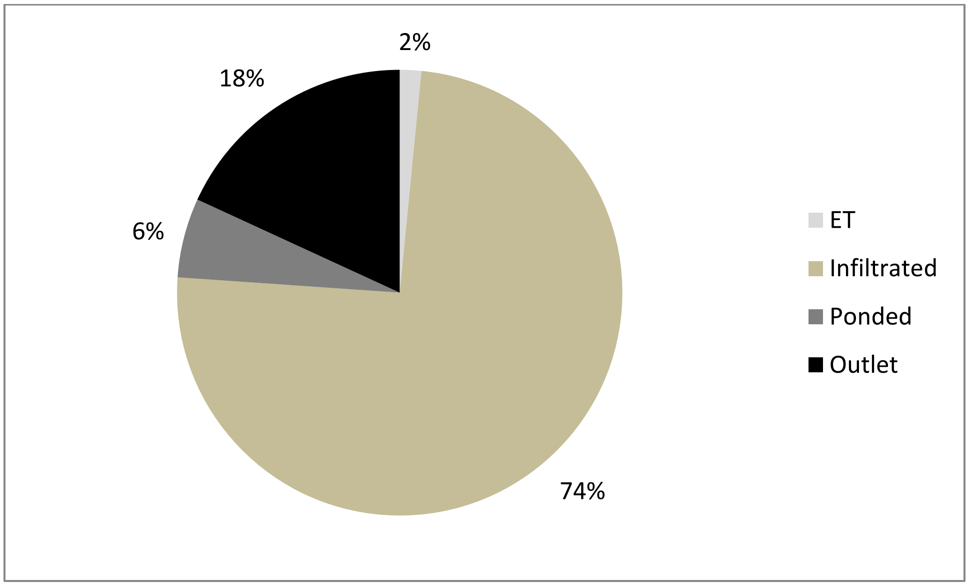


Figure 17. Water volume distribution of outflows for the treatment wetland during the entire season.

2.4.4 Nitrate Load

Surface water samples were collected throughout the field season. A total of eight samples from the inlet, seven samples from Agri Drain#1, seven Agri Drain#2, and three samples from the outlet were collected (Appendix I).

The average water inlet nitrate concentration was calculated at 23.0 mg/L Nitrate-N (n=8). A total of three paired inlet and outlet samples were taken. Results from a paired, two-tailed t-test revealed that there was no significant difference between inlet and outlet water samples for Nitrite and Nitrate as N (p=0.78, n=3). Using the water volumes and average nitrate concentrations from the inlet and outlet a confidence interval was calculated. Approximately, 334 to 397 lbs. of nitrate traveled through the inlet into the wetland. Only 61.2 to 72.6 lbs. of nitrate in the surface water traveled through the outlet pipe.

There is a 95% confidence level that 261.4 to 335.8 lbs. of nitrate were removed from the surface water. However, Nitrates are still present in the infiltrated waters and must be accounted for.

The nitrate load for the volume of water infiltrated can be estimated. The average concentration of nitrates in the water throughout the cells was 23.0 mg/L Nitrate-N (n=25). The standard deviation is 1.0 mg/L Nitrate-N and two standard deviations from the mean yield 95% of the values surrounding the mean (21.0 – 25.0 ml/L Nitrate-N). This range multiplied by the estimated volume of water to calculate the estimated range of the nitrate load entering the subsurface. There is a 95% confidence level that the initial subsurface load that infiltrated into the subsurface is between 250 lbs. and 298 lbs.

2.4.5 Additional Water Quality Parameters

Additional water quality parameters were collected in the wetland. These parameters include: specific conductance, soluble ortho phosphorus, total phosphorus, and total suspended solids (Appendix I). Results from a paired, two-tailed t-test revealed that there were no significant differences between the inlet and outlet values for any of the additional parameters ($\alpha = 0.05$). However specific conductance did have a p-value near $\alpha = 0.05$ (p-value =0.078) and indicated a weak trend of lower levels for outlet samples when compared to inlet water samples.

Confidence intervals can also be calculated for the other parameters. First the average values are multiplied by two standard deviations of means to find a range of data points. For this calculation all inlet values (n=8) and all outlet values (n=3) for each parameter were used. Then the high and low end of the range is multiplied by the by the known water volumes.

The average total phosphorus measurements for the outlet (0.063 mg/L) were higher than the inlet (0.050 mg/L) total phosphorus concentrations. If the water volumes and total phosphorus load are taken into account, there is a 95% confidence that 0.005 to 1.158 lbs. of total phosphorus was removed from surface waters. This equates to an 18.0% to 100% reduction. The standard deviation of the outlet was adjusted for the higher value (1.158 lbs.) because it was estimated to be below zero, which is not possible.

The average ortho-phosphorus measurements for the inlet (0.038 mg/L) were higher than the outlet (0.022 mg/L) ortho-phosphorus concentrations. The there is a 95% confidence that 0.8 lbs. to 0.1 lbs. of soluble ortho phosphorus was removed from surface waters. This equates to a 35.1% to 100% reduction. The standard deviation of the outlet was adjusted for the higher value (0.8 lbs.) because it was estimated to be below zero, which is not possible.

The inlet water samples for the total suspended solids ranged from <2 to 5 mg/L and outlet samples ranged from <2 to 37 mg/L. Hence the standard deviation of the means yielded negative values and no meaningful confidence interval could be calculated.

2.4.6 Groundwater Flow

The relative water levels measured in the inlet and outlet groundwater wells were compiled through the entire season (Appendix J). These values were used to find a ground water gradient at specific times of the season (Table 2). The slopes were calculated by taking the average change in depth of water (Δh) over the horizontal distance between the two wells (Δl). Thus,

$$\text{Slope} = \frac{\Delta h}{\Delta l}$$

During the pre-flood and post-flood seasons there were some slope values that were negative. A negative slope indicates that the groundwater levels nearest Elm Creek (outlet groundwater well) were higher than that the groundwater levels near the wetland inlet (inlet groundwater well). If all of those negative values were removed the new slopes would be 0.00521 and 0.0153 for the pre-flood and post-flood seasons, respectively. These values will not be used in calculations, but show that there was some reverse of flow occurring throughout the season. Most of these slopes throughout the season could not be calculated because there was no water detected in the inlet groundwater well. As a result the average slopes are not complete data sets.

Season	Average Slope (Δh/l)
Pre-flood	-0.000482
Flood	-0.00718
Post-flood	0.01315

Table 2. Seasonal groundwater gradients. The Δh was calculated as the inlet groundwater well – the outlet groundwater well. The length (l) was calculated as the distance horizontal between the wells (38.1 meters).

Darcy's law can be used to calculate the average velocity of the water below the wetland floor. This velocity is assumed to extend beyond the wetland boundaries and until the groundwater reaches the creek. The calculated average slopes shown above were used in this equation. Values for the hydraulic conductivity and effective porosity were collected from the NRCS web soil survey. There is a range of 0.57 to 1.98 in/hour for the hydraulic conductivity so an upper and lower range of velocity (q) was calculated for (Table 3). Below are the equations used to calculate the flow of groundwater water through the subsurface:

$$q = -K\frac{\Delta h}{\Delta l}$$

$$v = \frac{q}{f}$$

Where,

q = Darcy's flux (in/hour)

v = seepage velocity (in/hour)

f = porosity of the soil

Season	Upper "q" (in/hour)	Lower "q" (in/hour)
Pre-flood	0.00466	0.00134
Flood	-0.00849	-0.00244
Post-flood	0.0317	0.00911

Table 3. Seasonal groundwater velocities between the inlet groundwater well and the outlet groundwater well.

Using the range of estimated velocities (q) and the distance to the creek, the time it takes for water to travel through the subsurface to Elm Creek can be estimated. Values for the time period during the pre-flood and during the flood will not be used because the groundwater dynamics were reversed (Table 4).

Time to Elm Creek from:	Cell 1 (years)	Cell 2 (years)	Cell 3 (years)
Pre-flood	NA	NA	NA
Flood	NA	NA	NA
Post flood	30.6 – 8.8	22.7 – 6.5	15.4 – 4.4

Table 4. Average groundwater travel time based on seasonal groundwater gradients and estimated velocities.

2.5 DISCUSSION

2.5.1 Surface Water Volumes

The water volumes throughout the wetland varied greatly. Of the total inlet surface water 57.7% flowed into the second cell, 38.4% flowed into the third cell, and only 18.1% flowed through the outlet. These large reductions from cell to cell are a result of some evapotranspiration, but mainly infiltration processes.

These findings have implications of future treatment wetland designs. In order to keep water at the surface clay liners should be used. For this wetland it was assumed that the large clay content in the sampled soils would be enough to keep the water from infiltrating quickly and keep the water ponded for longer periods of time. Due to large amount of infiltration a lack of surface water ponding this wetland acted more like a saturated buffer. These buffers are edge of field treatments that allow polluted water to filter through a buffer zone before discharging into nearby surface waters. Similar to the treatment wetland in this research, the majority of water flow in saturated buffers is in the shallow subsurface (MPCA, 2013). Factors such as soil pores, macropores, soil moisture content, and deeper layers composed of highly permeable materials can all affect the infiltration rates of the soils. In this wetland soil cracks and macropores were visually observed, which could be one of the main factors leading to a high volume of infiltration.

In areas with similar topography and soil profiles, if no clay liner is utilized then infiltration can be expected. In these designs it can be expected that natural subsurface processes will remove some nitrate. Experimenters can also capitalize on this downward water movement and construct subsurface denitrification systems in combination with the surface treatment provided by the wetland. These dual purpose systems could have a greater nitrate removal capacity.

The tile drainage flow rates in this study were 0- 1.98 cfs. Another study conducted during different years, but in the same region, had flow rates of 0-1.3 cfs (Lenhart, 2008). These two systems had similar rates of flow which is not surprising considering that subsurface tile drainage systems tend to discharge water at relatively consistent flow rates. This consistency can help in the design of future wetlands and other water treatment systems.

2.5.2 Rainfall

This season rainfall totals were determined from values reported in Winnebago, MN. While this city is only about 10 miles away rainfall totals could be significantly different directly over the field site. Due to this potential source of error direct rainfall into the wetland was not accounted for in the inlet water volume totals. Also, rainfall totals were only a small fraction of the inlet volume. If accurate on-site measurements are taken in the next field seasons the rainfall volume can be added in to get a more accurate depiction of the water flow.

2.5.3 Evapotranspiration

The evapotranspiration rates were calculated using a modified Hamon equation. This equation was calibrated to a prairie wetland located in east-central North Dakota (Rosenberry, 2004). While this equation cannot perfectly estimate the actual evapotranspiration rates that occur in the treatment wetland in Martin County, MN it is a good estimate of the potential evapotranspiration rate.

The temperatures used as inputs for the evapotranspiration rate equation were collected on-site and therefore should be as accurate as possible. When the PET rates were converted to ET volumes the estimated surface area of ponded water was used in the calculations. These estimates of ponded surface area were inexact because they were based the dimensions of the wetland, visual observations and data from leveloggers. Since the surface area of ponded water was constantly fluctuating it was impracticable to get an exact measurement.

The calculated ET values were rather low this season. The low values were mainly due to the fact that during most of the season the wetland consisted of bare soil or sparsely vegetated and infiltration rates were high. In succeeding years the wetland will become more densely vegetated. The vegetation will likely increase the ET rates.

2.5.4 Water Budget

After the flood only a small fraction (4.2%) of the total inlet water flowed through the tile drainage. While this was partly due to the lower rainfall total, evapotranspiration in the row crop fields also factored into the lower tile drainage volume. Later in the season as the corn matured more evapotranspiration occurred throughout the tile drained region. Also, due to higher temperatures more evapotranspiration occurred in the wetland itself in the post-flood period. This resulted in a higher percent outflow of ET during the post-flood than during the pre-flood season.

When water was flowing through the tile drainage pipe the flows were low. Approximately, 7.5% of the monitoring period (11 days) accounted for over 75% of the total inlet flow. This demonstrates the relatively flashy nature of tile drainage in this field. The tile drainage system creates a fast track for seepage water to move from the field to the end of the pipe. One of the benefits of the treatment wetland is to allow water to discharge into Elm Creek at a slower rate and can provide a pathway for of NO_3-N reduction. This one wetland will not make a significant impact on Elm Creek. However, multiple best management practices (BMPs) all along this watercourse could help to lower the overall nitrate load.

The tile drainage discharge fluctuated seasonally. For this project all of the highest volume discharge days all occurred in June. According to a fifteen year study located in southern Minnesota, tile drainage systems discharge 68-71% of their annual water budget and 71-73% of their annual nitrate load from April-June (Randall, 2004). This finding is consistent with the seasonal nature found in the tile drainage discharge in this research.

In the wetland a high percentage of inlet water left the system through infiltration. Over the season approximately 74% of the water exited the surface water system through infiltration. Similar tile drainage fed wetlands had much lower infiltration rates, ranging from 47-27% of the inlet flow. The soils that lined those wetlands were compacted silty-clay loams (Larson et. al., 2000). The Elm Creek constructed wetland had higher clay content that the soils seen in the Larson experiment and yet the infiltration rate was still much higher.

2.5.5 Nitrate Load

There was no significant change in the concentration of nitrates for inlet and outlet surface water for this wetland. More reduction in the surface water was expected in this first field season. The main factors that can reduce NO_3-N are vegetation uptake, denitrifying microbes, organic carbon content, and hydraulic residence time.

The construction occurred in March 2013 and most of the wetland samples were collected in June 2013. During June the wetland was a recently constructed bare soil depression with little to no plant growth. The lack of vegetation during tile flow could explain the lack of nitrate reduction in surface waters.

Vegetation is an essential component to the success of a treatment wetland for several reasons. First, plants can directly uptake some of the nitrates and consequently aid in nitrate removal. However, this may only account for a very small percent of the nitrate load, rarely exceeding 10%, and often much lower based on values reported in the literature (Vymazal, 2006, Hammer 1992, Kantawanichkul, 2009).

Secondly, vegetation can provide a stable habitat for microbes to thrive. Compared to unplanted wetlands, vegetated wetlands have a greater diversity of microbes and the microbial community is more stable throughout the season (June-December)

(Zhao, 2010). Additionally, ammonia-oxidizing bacteria (*Nitrosomonas*) populations have been shown to be two to three magnitudes greater in vegetated wetlands (Kantawanichkul, 2009).

Lastly, vegetation can create organic carbon matter in the soil. It is important to have a large source of organic carbon because it aids in bacterial denitrification (Triska, 2007). In natural wetlands often times there is a layer of muck in the first six inches of soil. This muck contains a minimum of 12-18% of organic carbon (USDA-NRCS, 2010). Blue Earth mucky silty clay loam is an example of a muck soil located within Martin County. This muck has an organic matter content ranging from 10-25% down to 60 inches (NRCS-USDA web soil survey).

Soil samples taken from the first six inches in the wetland cells had a total organic carbon content of 2.0-2.77% by the end of the 2013 season (Appendix K). A larger organic carbon source could have aided in a higher rate of bacterial denitrification within the wetland. Organic carbon tends to accumulate in saturated soils because the microbes do not consume carbon as quickly in these conditions (USDA-NRCS, 2010). If the soils in the treatment wetland can remain saturated then there is a potential for higher rates of organic matter accumulation. For example, in Canadian peat lands organic accumulation rates range from 10 to 35 g C m^{-2} yr^{-1}(Ovenden, 1990). Even if soils do not remain saturated the lack of tilling and native plantings will help organic carbon to accumulate at faster rates. A study on organic carbon accumulation rates in the top 5 cm of soil found that early successional native ecosystems in Michigan have accumulation rates of 31.6 g C m^{-2} y^{-1}. These were the highest rates in the study which analyzed row crop systems, perennial cropping systems, and native ecosystems (Grady and Robertson, 2007). As organic carbon accumulates in the treatment wetland it can aid in the denitrification process.

The soil texture and structure may have contributed to the lower than expected denitrification rates as well. The soils from the surface of the wetland (0-6 inches) are all classified as silty clay. The water drained through the soil pores rapidly, which reduced the hydraulic residence time of the surface water. Soils with higher clay contents would allow water to pond easier and provide a longer hydraulic residence time. The longer residence time allows the NO_3-N rich water to interact with the organic matter, and denitrifying microbes in the soil for longer periods of time; allow denitrification to occur (MPCA 2013).

Although the concentrations of nitrates did not significantly decrease in the surface water samples there still was a loss of nitrate from the surface water. A low percentage (18.1%) of the total tile inlet water flowed to Elm Creek as surface water. As a result a large load of nitrate was removed from the surface tile water inflow.

The wetland received a total load of 334 to 397 lbs. of NO_3-N. An estimated 261 to 336 lbs. of NO_3-N were removed from the *surface* water. However, NO_3-N is still present in the infiltrated waters and must be accounted for. An estimated 250 to 298 lbs. of the nitrate load infiltrated into the soils.

Denitrification processes still occur once the water has infiltrated. While some additional subsurface denitrification is expected the actual rates for the infiltrated water this particular site are unknown. Denitrification rates in subsurface zones are highly variable and are dependent on the conditions. For microbial denitrification to occur there must be anoxic conditions (<0.5 mg/L oxygen) present (Dubrovsky, 2010). When water remained ponded in the wetland anoxic conditions were assumed to occur, thus allowing for microbial denitrification. Other beneficial factors are warm temperatures and plentiful organic carbon sources (Triska, 2007). Due to the nature of the experimental site the temperatures varied greatly and could not be controlled. Organic carbon sources were present in the soil but they were not elevated to the levels of a natural wetland. Additionally the hydraulic loading rate can impact denitrification rates (Lin et. al., 2008). Since the source water is a tile drainage system the flow of water is not consistent in timing or in discharge rates. All of these factors make it difficult to estimate a probable subsurface denitrification rate.

Studies on the rate of denitrification in the field can vary greatly. A managed aquifer recharge pond in California estimated denitrification at an average of 56 mg/L/day-N. This pond likely has such high denitrification rates due to the high concentrations and steady supply of nitrates and dissolved carbons to the system (Schmidt et. al., 2011). In a study of four sandy aquifers with low organic matter a denitrification rate ranging from 0 to 0.20 mg/L/day-N was found (Green, et al., 2008).

No research was found with conditions exactly matching those at the experimental wetland in this study. Therefore calculating an exact denitrification rate for this specific location is improbable with the current information. Although the exact amount of nitrate removal will not be known for this site, evaluations from previous studies can be used to estimate how much nitrate was removed from the subsurface load.

An extensive literature review of 27 studies was conducted by the MPCA to estimate denitrification percentages for different agroecoregions. After each of these agroecoregions were analyzed the presumed groundwater denitrification was 25% Karst agroecoregions, 40% for Sand Plain and Alluvial agroecoregions, 60% for finer textured soil agroecoregions and drained soils, and 50% for all other agroecoregions (Table 5).

Agroecoregion	Denitrification factor
Buffers, Rochester Plateau	0.25
Anoka Sand Plains, Alluvium and Outwash, Inter-Beach Sand Bars, Steep Valley Walls, Steeper Alluvium	0.40
Forested Lake Sediments, Mahnomen Lake Sediments, Poorly Drained BE Till, Poorly Drained Lake Sediments, Red Lake Loams, Somewhat Poorly Drained Lake, Swelling Clay Lake Sediments, Very Poorly Drained Lake Sediments	0.60
Other agroecoregions	0.50
Drained soils	0.60

Table 5. Groundwater denitrification percentages for all agroecoregions of Minnesota.
Data source: MPCA, 2013

This watershed's agroecoregions include Wetter Blue Earth (BE) Till, Wetter Clays and Silts, and Rolling Moraine and all are classified under "Other Agroecoregions". The site itself sits adjacent to fine alluvium sand (Appendix L). "Other Agroecoregions", and Alluvium are estimated to have a 50%, and 40% loss of nitrates by the time their groundwater water reaches the nearest surface water.

Thus to account for both agroecoregion types a 45% reduction rate was applied to the subsurface load. This resulted in approximately 113 to 134 lbs. of nitrate removed from the infiltrated water. Thus a total of 124 to 172 lbs. of nitrate was removed from the entire wetland system, which accounts for 37.1-43.3% of the nitrates.

In preliminary modeling conducted by Karlheim (2012) it was estimated that 69% of the surface nitrate load would be removed. However, the calculated 41-47% nitrate removal efficiency falls short of this prediction. There are many reasons why the modeled percent was higher. The model used slightly different inputs than the actual wetland parameters. The nitrate concentration was assumed to be lower at 15.33 mg/L NO_3-N rather than 23.0 mg/L NO_3-N. The wetland was estimated to be slightly smaller at 0.45 acres rather than the actual 0.542 acres. However, the author of this paper recognizes that biggest differences were that the model and equations used did not account for factors such as dissolved oxygen, pH, and organic carbon content, and temperature fluctuations. Also, the model assumed a densely vegetated wetland when in reality the wetland remained bare soil and sparsely vegetated during most of the data collection in 2013 (Karlheim, 2012). While the model did not accurately predict the outcomes it was a good starting point for this project and it serves as a reference point on what the results could be with more ideal conditions.

Numerous water samples from Elm Creek have been collected at a location less than a mile downstream of research site, by the Martin County Soil and Water Conservation District (SWCD). The average concentration of nitrate and nitrite as N in Elm Creek itself is 6.67 mg/L (Appendix M). From this data the creek on average is below the drinking water standard for nitrate of 10 mg/L (MPCA, 2004). However, approximately 27% of the samples did exceed the drinking standard; with the highest concentration at 20 mg/L. While the average is low the nitrate concentration can be exceptionally high and this usually occurs in the spring and early summer. This drinking water standard does not apply to this stream as it is not a municipal drinking source. Nonetheless, this is a good metric for water quality evaluation. With an average concentration 23.0 mg/L nitrate and nitrite as N the tile drainage discharge into the wetland is considerably over this standard. The treatment wetland prevents this entire load from reaching Elm Creek surface waters, but the concentration of the wetland discharge (22.8 mg/L nitrate and nitrite as N) is still above 10 mg/L. BMPs such as this wetland help to prevent raising the concentration of NO_3-N in Elm Creek. If more systems like this are put into practice then the concentration of Elm Creek could be lowered and thus overall load delivered to the Blue Earth River would be reduced.

2.5.6 Additional Water Quality Parameters

For all of these proceeding parameters only three paired inlet and outlet samples were taken. For that reason any trends seen were not robust. Additional water samples collected in subsequent field seasons will help to make a more robust data set.

Specific conductance is the measure of the ability of a material to conduct an electrical current at 25°C. Thus the more ions present in the water the higher the specific conductance measurement (Hem, 1985). The data sets for specific conductivity at the inlet and outlet do not show a statistically significant difference (p-value = 0.078, $\alpha = 0.05$, paired, two-tailed t-test). However there was a weak trend of lower specific conductance at the outlet. This difference could be attributed to the input of rainwater into the system. Rainwater typically has values below 100 (umhos/cm) (Hem, 1985) and the average inlet and outlet from the paired sample was 786.7 (umhos/cm) and 761 (umhos/cm), respectively. So rainwater could dilute the tile drainage water to some extent.

Soluble orthophosphate (PO_4-P) measurements did not have any significant change between the inlet and outlet values. This chemical compound is a readily available source of phosphorus for aquatic and terrestrial plants (Bieleski, 1973). It is not surprising that this measurement did not decrease. There was no evidence of algae growth in the wetland and during the water sampling period little terrestrial growth occurred. Also, there was little plant growth when the water samples were taken. The majority of the vegetation was still less than 12 inches in height. Other factors such as light availability, and plant species can affect the plant uptake rates of orthophosphate (Zhai, 2013). There was a weak trend of ortho-phosphorus reduction from the inlet to the outlet. Thus some uptake by vegetation may have occurred.

Total phosphorus was another parameter measured in the wetland. This measurement includes all forms of phosphate including soluble ortho-phosphorous, condensed phosphorous, and organic phosphorous. Phosphorus (P) is commonly measured as total phosphorus (TP); including both organic and inorganic forms of P (Kadlec and Wallace, 2008). Water quality of lacustrine systems (lakes and wetlands) is highly influenced by soluble reactive P (SRP) (Jarvie, 2006; North et. al., 2014). Dissolved orthophosphate (DPO_4-P) comprises most of the SRP along with other condensed pyro-P, meta-P, and poly-phosphates that include acid hydrolysable P. P can be tied up in organic P complexes, but released upon oxidative digestion (Kadlec and Wallace, 2008). P bound by inorganic minerals and buried will be removed from the open water column (Reddy et. al., 1995); however, exchange site P saturation can lead to imbalances in the equilibrium P concentration (EPC) releasing P back into the open water

(Jarvie et. al., 2005). Additionally, P will be extracted by plant roots and temporarily stored in plant material only to be released back to the water column during stem and leaf decomposition unless removed from the system (Reddy et. al., 1995). The total phosphorus concentrations at the inlet were low, averaging 0.0486 mg/L (n=8, sd = 0.0077). There was not a statistically significant decrease in total phosphorus. In fact, the average phosphorus concentration was greater at the outlet (0.064 mg/L) than the inlet (0.050 mg/L) (inlet = 0.05, outlet = 0.0636, paired water samples).

The total suspended solids were mainly tested to see if solids were settling out as water passed through the wetland. The velocities of water recorded in the wetland were rather fast at times (inlet high velocity: 1.68 m/s or 5.51 f/s, outlet highest velocity: 0.71m/s or 2.32 f/s) and therefore did not allow for ideal conditions for solids to settle. There was no trend detected for this parameter. The average inlet value was 1.3 mg/L and the average outlet value was 15 mg/L. Levels below 2 mg/L could not be detected at the laboratory so they were assumed to be 0 mg/L for purposes of analysis.

In the future all of these additional parameters will have more data values collected. Hence stronger trends can be established in time.

2.5.7 Groundwater Levels

Frequently, the groundwater was too low to detect any water in the inlet groundwater well. Over 20% of the season's groundwater was below the inlet groundwater well and no analysis could be applied to groundwater dynamics during this time. The groundwater wells were not installed deep enough to capture all the possible measurements and a complete depiction of relative water levels and the corresponding slopes could not be captured. Due to the lack of adequate information the water levels and slopes would not be representative of the groundwater flow occurring below the wetland. The addition of a deeper inlet well, and more wells within the wetland would help to create a better profile and capture the groundwater dynamic occurring in future seasons.

2.6 CONCLUSIONS AND MANAGEMENT IMPLICATIONS

This season presented some challenges. The spring started exceptionally wet and this led to a flood that poured into the wetland area. This was followed by a dry summer and fall which left little opportunity for additional data collection. There will be two more years of research conducted on this wetland. Additional data will help establish a more robust trend for water volumes, nitrates and other water quality parameters. With each season the vegetation should become more established which will create a larger potential source of organic carbon for denitrifying bacteria.

The wetland was successful in reducing total nitrate loads although denitrification in the surface water was very limited. The main objective of this research was to design a wetland that successfully removes nitrates from agricultural tile drainage water. Nitrates were removed from the surface water and water that infiltrated had further denitrification opportunities in the subsoil. While the concentration of nitrates did not significantly differ between the outlet and inlet the reduction in water volume and the infiltration opportunities make this a success.

The secondary goal of reducing total phosphorus loads was not achieved this year. As the vegetation establishes it may begin to uptake some of this load. Phosphorus can continuously cycle through a wetland, thus harvesting the wetland vegetation could help to remove some of this load from the system permanently.

Besides the water quality benefits, this wetland provides many other potential benefits to the local region. It can serve as a natural habitat for aquatic insects, mammals, amphibians, and birds such as: mayflies, stoneflies, caddisflies, muskrats, minks, turtles, snakes, frogs, waterfowl, songbirds, and pheasants (Minnesota DNR, 2002). In fact, throughout the first season several wildlife species including Killdeer, American Goldfinch, spiders, white-tailed deer, field mice, damselflies, butterflies, moths and tracks of small mammal were spotted utilizing the constructed wetland habitat. Another benefit is that the wetland can provide flood control by slowing down runoff water. The wetland can also become a living classroom where local citizens can learn about water quality and innovative agricultural conservation practices. Projects of this nature have the potential to raise awareness about local conservation efforts. In fact during a field day local farmers, SWCD personnel, and researchers gathered to discuss the constructed wetland project. This event was also showcased in an article written in the local paper; the Fairmont Sentinel.

This project could have additional significance due to the location in the Elm Creek watershed. Elm Creek is one of four pilot watersheds that the Minnesota Department of Agriculture has chosen to participate in the Minnesota Agricultural Water Quality Certification Program (MAWQCP). This is a voluntary certification that promotes farmers to adopt conservation practices (MDA, 2013). Thus the results from this project and others along Elm Creek could provide more robust information to the MDA on effective nutrient management practices in this watershed.

The constructed wetland results can also help guide future water quality projects, even if tile drainage is not the source of water. The capabilities of this wetland should translate to other Midwest locations with hydric soils and a source of high nitrate water. Certain factors such as organic carbon content, water volume, soil texture, weather and other components could create differences in results between this and other constructed wetlands. The results found in this study do not necessarily translate into restored wetlands because the historical hydrology differs greatly than that of a constructed wetland.

There have been several lessons learned from this first season that can be applied to the proceeding seasons. First, it is important to synchronize vegetation growth and tile drainage. Having the vegetation growing during the water inflow increases the overall nitrate removal potential. This issue should self-correct because vegetation that was planted in 2013 should re-emerge in the spring of 2014. Secondly, infiltration was a major water outflow component. Thus infiltrated water contains a large proportion of the nitrate load. Additional groundwater wells should be installed in order to better describe the subsurface water flow. Also, water samples should be taken from each of these wells. The results from these wells can be used to help quantify potential nitrate loss in the subsurface. Third, local weather conditions should be monitored for more accurate water budget information. In 2013 accurate rainfalls were not taken on site. For 2014 a weather station and a separate rain gauge have already been setup. These measurements can help to calculate a more accurate ET rate. Also, local rainfall measurements can help to quantify the inlet totals more precisely. Although direct rainfall was a small percent of the wetland water it would be better to include it if possible.

There are other discoveries that can be applied to future treatment wetland research. First, organic carbon is an important resource in bacterial denitrification (Triska, 2007). Thus, it would be beneficial to construct wetlands in areas that already have high organic carbon content or simply restore more wetlands because they typically have higher organic carbon content in the soils. Secondly, to establish an effective wetland it is best not to place it in an area that floods frequently. Flooding due to high river levels can confound data and interpretation of data gathered. However, as flooding is a natural process that many wetlands experience this is a complexity that will likely be seen again in other experiments and in future years.

One last thing that should be emphasized is that time is essential when conducting research of this nature. Many studies only conduct monitoring for one season or a few years. The current funding allows for two more years of monitoring. However, with relatively little additional funding much more about the dynamics of this treatment wetland could be discovered. Over time carbon supplies will build, denitrifying bacteria populations can grow, data trends will become clearer and the effectiveness of the treatment wetland should improve. While data from the first few years is important, it would be interesting to see how this wetland functions five or ten years down the line. Since the wetland has already been built, the equipment has already been purchased the succeeding years should be relatively cost effective I recommend continued funding for this projected and extending the monitoring period.

In conclusion, there were significant reductions of nitrates from the surface water. However, the water that is entering Elm Creek still has a higher concentration of nitrates than the water in Elm Creek itself. The rate of nitrate removal in the subsurface waters remains unknown; but was presumed to be significant. Thus the main goal of removing nitrate from the tile drainage water was met. This effort alone will not change the water quality in the Gulf of Mexico, the Blue Earth River watershed or even Elm Creek. With over 1,000,000 metric tons of Nitrate flowing into the Gulf of Mexico each year (USGS, 2013) a few hundred pounds does not make a significant difference. However, if successful nutrient removal systems became a trend in this region significant changes might be seen in the future. As for the secondary goal, data did not show that the phosphorus load was reduced, and thus this goal was not met in the first season. The extent of the success or failure of these objectives will become clearer with more water quality data, and additional groundwater monitoring. The effectiveness of this wetland should improve with time as vegetation becomes established, bacteria populations grow, and more organic carbon is added to the system. The data collected during the next two field seasons should answer many of the lingering questions.

CHAPTER 3

INFLUENCE OF VEGETATION AND SOIL TYPES ON NITRATE REDUCTIONS IN WETLAND MESOCOSMS

3.1 EXECUTIVE SUMMARY

Wetlands are complex systems that can be difficult to understand. Wetland mesocosms are important tools that can allow researchers to focus on one component of wetland dynamics at a time. Utilizing wetland mesocosms this research aims to quantify the nitrate removal capacity of different vegetation types and soil types. The soils were collected from an agricultural land where a treatment wetland was constructed. The experiment included four vegetated mesocosms and four bare soil mesocosms. Nitrate (23.0 mg/L) rich water was pumped into each mesocosm and the nitrate levels within the mesocosm were monitored over a period of 10 days. After 10 days all vegetated mesocosms had removed more nitrates compared to the bare soil control. The mixed vegetation wetland removed the most nitrates (34.9%). The bare soil plots had low and consistent levels of reduction (11.2-15.6%). Vegetation can facilitate nitrate removal from water and should be utilized in treatment wetlands and any other waterways that have excess nitrate.

3.2 INTRODUCTION

3.2.1 Previous Mesocosm Experiments

There are a variety of water quality issues that can be remediated by wetlands. However not every type of wetland can solve every water quality issue. Often times small scale mesocosms are used to test the effectiveness of different components of wetland functions. Mesocosms have been set up to research a multitude of water quality improvement strategies such as: treatment for acidic water and metals runoff from coal storage piles (Collins et al., 2004), the mitigation of pesticides (Lizotte et. al., 2011), and the reduction of nutrients such as phosphorus (White et. al., 2006).

One recent focus of wetlands has been on using wetlands to reduce nitrate levels from tile drainage wastewater. Nitrate rich water flowing from farms can cause environmental issues downstream (MPCA, 2013). Wetland mesocosm studies on reducing nitrates have been conducted and have found encouraging results (Isenhart, 1992; Tyler et. al., 2012). Further research could help landowners and conservationists learn even more about the capabilities of treatment wetlands on farm fields.

These small scale experiments have the advantage of being able to control environment variables, use less complex systems, and take measurements at a smaller scale. However, there are many limitations when using mesocosms as a surrogate for wetland analysis. One of the greatest limitations is the ability to mimic the hydrology of a wetland. The hydrology in these mesocosms is often drastically different than natural systems. Groundwater dynamics and flow rates are not often mimicked in these studies. Also, while it can be an advantage to control environmental factors it does mean that the experiment will not fully capture what would occur in natural settings. Despite the limitations these types of experiments can provide useful information that can be applied to natural systems.

3.2.2 Vegetation Impacts

Several studies have shown vegetation to be more successful at removing nitrogen from water than bare soils. However, the types of vegetation used could significantly affect the amount of nitrogen that was removed from water. A mesocosm study conducted by Tyler et. al. (2012) found that burreed (*Sparganium americanum*) did not have a significant reduction compared to unvegetated mesocosms. Cutgrass (*Leersia oryziodes*) and cattail (*Typha latifolia*) reduced nitrate loads in water by more than 57% which was significantly more than the bare soil controls,. The unvegetated plots removed 18-40% of nitrate meaning that as little as 17% of the reduction from *L. oryiodes* and *T. latifolia* might have been facilitated by vegetation. While vegetation proved to be a significant component in nitrate removal it was not the main removal mechanism in this experiment. Isenhart (1992) found that cattail (*Typha latifolia*) mesocosms could reduce 10 mg/L nitrate rich water to undetectable levels within a five day span. This study also showed that the vegetation only accounted for a small fraction of the nitrate removal. However, a field study conducted by Silvan et. al. (2004) found that in some instances vegetation can be the main driver in nitrate removal from water. In this study tussock cottongrass (*Eriophorum vaginatum*) removed approximately 70% of the nitrogen load.

Thus vegetation has shown to reduce nitrate concentrations in water, but the amount is not always consistent.

This experiment aims to uncover what type or mixture of vegetation will contribute to nitrate reduction in conditions similar to a constructed treatment wetland on an agricultural field. It is expected that the vegetation will have a significant effect on nitrate loads, but it may not be the main driver behind nitrate removal processes.

3.2.3 Soil Impacts

It is expected that denitrification in the soils by denitrifying bacteria will contribute a significant amount of nitrate reduction. Other research has indicated that denitrification processes contribute significantly more to nitrate removal than plant uptake (Isenhart, 1992; Hammer, 1992). Denitrification is the process where bacteria convert nitrates into nitrogen gas (Dubrovsky, 2010). However, the soil composition can affect the rate of denitrification. Thus three different soil types from the same agricultural field are being compared. While the compositional differences are slight, there could be some effect on denitrification rates.

3.2.4 Research Focus

The focus of this research is on nitrate removal in wetland mesocosms. The goal is to determine the best vegetation regime and soil type to facilitate nitrate removal processes for agricultural tile drainage wastewater. It is hypothesized that the vegetation will significantly reduce the nitrate load in the mesocosms compared to the unvegetated control.

An associated field study, located in an agricultural field in southern Minnesota, is examining the effectiveness of a constructed treatment wetland on removing nitrate loads. All of the soils used in this experiment were collected from this field, referred to as the Robert's Farm. Results from the mesocosm experiment could help determine what type of vegetation and soil types would be best for a treatment wetland in this general area.

3.3 MATERIALS AND METHODS

3.3.1 Mesocosm Wetland Setup

Wetland mesocosms were set up indoors in the University of Minnesota's basement of the Bioproducts and Biosystems Engineering building. Eight separate 110 gallon LDPE recycled plastic tanks (36 x 53 x 20 inches) were used as the containers. The tanks were filled with approximately eleven inches of screened sand and then a six inch layer of soil was placed on top. There are three to four inches remaining at the top of the tank to allow for water ponding (Figure 18).

Four of these tanks were utilized to conduct bare soil tests on four different soils. All of the soils were collected from the Robert's Farm near the wetland site. These soil types are Spillville loam, Coland loam and Clarion-Storden loam (Table 6). A fourth soil (Coland

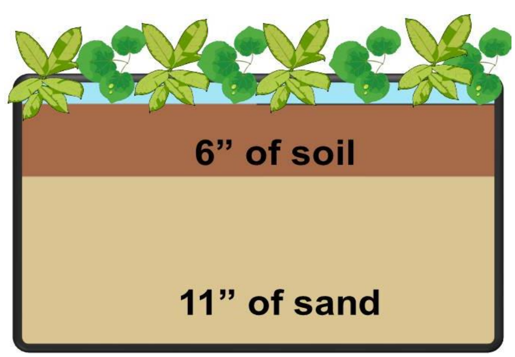


Figure 18. Profile view of vegetated mesocosm layout.

Loam) was collected from nearby, in the same farm field, only a little farther southeast. For clarity the Coland soil collected from near the wetland will be referred to as Coland A and the soil collected further southeast will be referred to as Coland B. These soils have similar properties, but there are some differences between each soil type (Table 6).

Soil Characteristics	Clarion-Storden Loam	Coland Loam A & B	Spillville Loam
Hydrologic soil group	*B	**B/D	*B/D
Drainage Class	Well Drained	Poorly Drained	Moderately well drained
Capacity of the most limiting layer to transmit water (Ksat)	0.57-1.98 in/hr.	0.57 – 1.98 in/hr.	0.57 – 1.98 in/hr.
Available water capacity	High (about 11.1 inches)	Very high (12.6 inches)	High (about 11.1 inches)
Slope	6-12%	0-2%	0-2%
Landform	Hills on moraines	Flood plains	Flood plains
Frequency of flooding	None	Occasional	Occasional
Parent Material	Fine-loamy till	Fine-loamy alluvium	Fine-loamy alluvium
Texture	Loam	Loam	Loam
Wetland (Hydric) Soil Rating %	10	97	5
Organic Matter	2.0-4.0 (0-10 inches) 0.5-2.0 (10-18 inches) 0.0-1.0 (18-60 inches)	5.0-7.0 (0-10 inches) 3.0-5.0 (10-25 inches) 0.0-2.0 (25-60 inches)	4.0-6.0 (0-19 inches) 3.0-5.0 (19-51 inches) 1.0-3.0 (51-60 inches)

Table 6. Properties of soil types used in mesocosm experiments. *B – Moderately low runoff potential when thoroughly wet, **B/D – moderately low runoff potential when drained, high runoff potential when undrained.
Data source: USDA-NRCS, 2009 and NRCS's Web Soil Survey, http://websoilsurvey.nrcs.usda.gov

The remaining three mesocosm tanks were used to test the effectiveness of different wetland plant types at removing nitrate. Each of these mesocosms was filled with the Coland B soil; and then was planted with 32 plants per tank. One tank was planted with Switchgrass (*Panicum virgatum*), another was planted with Fringed Sedge (*Carex crinita*) and the other tank was planted with an equal mix of Dark Green Bulrush (*Scirpus atrovirens*), *Panicum virgatum*, and *Carex crinita*. Each of these tanks was watered regularly and equipped with a grow lamp to maintain strong plant growth and root establishment.

Nitrate tests were conducted on all eight wetlands. The experiments were conducted during the month of September, 2013. The mesocosms were stored indoors and the loading dock garage door was opened while measurements were taken; exposing the mesocosms to ambient temperatures, wind and natural sunlight.

3.3.2 Preparing Wetland Mesocosms for Testing

The soils were collected from row crop farm. The field was sprayed with nitrogen enriched fertilizer several times a year. Due to this practice the soil could potentially have excess nitrogen. Therefore, each mesocosm was prepared in the proceeding manner to filter out any excess nutrients and prevent nitrate from leaching out later in the experiment. Low nitrate water (3.3-3.8 mg/L NO_3-N) was pumped into each mesocosm for three hours at a rate of 121 Liters/hour (32 gallons/hour). Water was discharged out of the wetland through the outlet tubing for at least one hour. Then the pumping was discontinued and the water remained stagnant in the wetland for a period of 45 hours. The low dose nitrate water was pumped into the wetland again for an additional hour.

After the preparation, each mesocosm was allowed to sit stagnant for at least 12 days in order to allow the nitrate level in the mesocosm to reduce below 4.0 NO_3-N.

3.3.3 Simulated Tile Drainage Runoff

During testing all eight wetland mesocosms were treated in a similar manner. As a starting condition each mesocosm was ponded with low nitrate water. Each mesocosm was treated with nitrate enriched water to simulate a tile drainage runoff event. Water with a concentration of 23.0 mg/L NO_3-N was prepared (see Appendix N for mixing calculations) by mixing solid sodium nitrate pellets with water to simulate normal tile drain runoff. Starting around eight o'clock in the morning, this nitrate enriched water was pumped into the saturated mesocosm for one hour to simulate a tile drainage runoff event. The nitrate water was pumped in at a rate of 32 gallons per hour. Then the pumping was discontinued and the water remained stagnant in the wetland to simulate a dry period or no precipitation event.

Water quality measurements from the wetland mesocosm were taken after the pumping ceased at hours: 0, 4, 24, 48, 72, 96, 120, 144, 168, 192, 216, and 240. Measurements were also taken from the initial nitrate water before it was pumped into the mesocosms. All measurements were collected from within the mesocosm near the outlet. Nitrate levels were measured using a Hach Nitratax sc, UV Nitrate sensor. Water temperature was measured using a HANNA Institutes HI 9828 Multi-parameter probe.

3.4 RESULTS AND DISCUSSION

Percent nitrate reduction levels were calculated for each mesocosm at specific time intervals (Appendix O). All wetland mesocosms experienced some reduction in the duration of the experiment. Reductions seen in the vegetated mesocosms were compared to the Coland B bare soil mesocosm. The reduction levels in the Coland B mesocosm were used as control conditions because all vegetated mesocosms were constructed with the Coland soil. Using Coland B as a control allows for the research to account for any extraneous variable that may have affected the results.

As seen in figure 19 all of the vegetated mesocosm had greater reductions than the Coland B reduction levels on day 10 of the experiment. Furthermore, the mixed vegetation and *Panicum virgatum* mesocosms had greater reductions at every measured data point. *Carex crinita* had greater nitrate reductions for every time point exept at the one day mark when Coland B soil had a 0.0786% greater reduction than the *Carex crinita* mesocosm. Besides this one data point the vegetated mesocosms always had a higher percent reduction of nitrate than the Coland Loam B mesocosm (Appendix O).

Paired, two-tailed, t-tests were conducted comparing the reductions in each vegetated mesocosm to the Coland B bare soil mesocosm. All of the percent nitrate reductions in vegetated mesocosms were statistically greater than the Coland B nitrate reductions (Mixed vegetation: p-value = 0.000160, $\alpha = 0.05$, n=11; *Carex crinita* p-value = 0.01792, $\alpha = 0.05$, n=11; *Panicum virgatum* p-value = 0.01721, $\alpha = 0.05$, n=11).

Previous studies have shown that vegetated wetland mesocosms remove nitrates more effectively than non-vegetated wetland mesocosms (Zhu and Sikora, 1995; Tyler et al., 2012). There are several explanations as to why vegetation may aid in nitrate removal in wetland mesocosms. The most direct way the plants facilitate nitrate removal is through direct plant uptake (Silvan, 2004). Besides direct uptake plants can provide an environment that facilitates additional bacterial denitrification processes. A study conducted by Martin et. al. (2003) found that the more transpiration occurs in plants the greater the nitrate removal percentage is. The theory behind this extra nitrate removal is that during transpiration more water movement near the plant roots occurs which allows the nitrates to move between the anaerobic and aerobic zones in the soil and promotes nitrate removal. Another way in which plants contribute to nitrate removal is by providing a source of organic carbon. Studies have found that wetland plant species release dissolved organic carbon (DOC) from their roots (Brix and Headly, 2007; Soto et. al., 1999; Zhai et. al, 2013). Organic carbon is an important component in bacterial denitrification (Burford and Bremner, 1975; Triska, 2007) and higher release rates could help further bacterial denitrification.

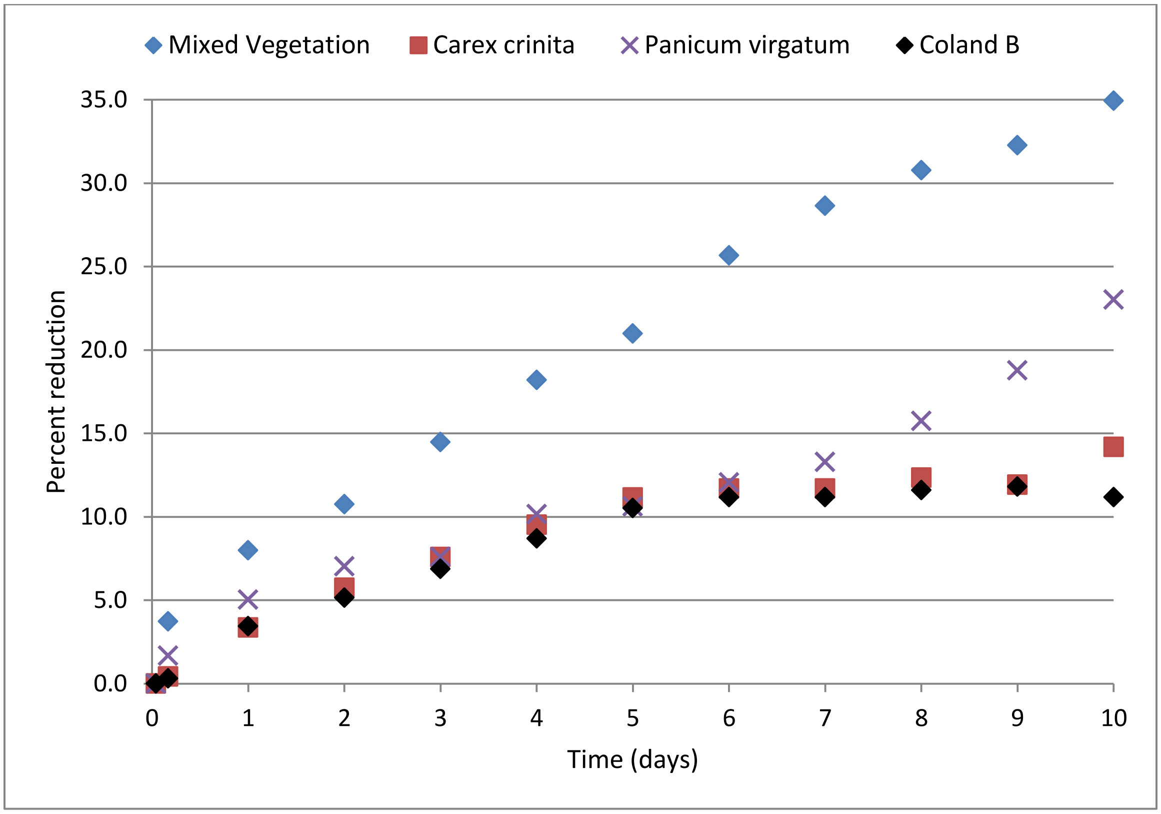


Figure 19. Percent reduction in nitrate concentrations for mesocosms over time.

The effectiveness of nitrate reductions for different soil types was tested. The results from each of the bare soil mesocosms were compared to each other. The results of these tests have much less variation between the soil types compared to the group of vegetated wetland mesocosms. The overall spread of reduction at the 10 day marker was 4.0% for the bare soil mesocosms and 20.7% for the vegetated mesocosms.

Only two soil types were statistically different from each other. The p-value from a paired, two-tailed t-test indicates a strong significant difference between the Coland loam B bare soil mesocosm and the Spillville loam bare soil mesocosm (p-value = 0.03289, $\alpha = 0.05$, n=11). As seen in figure 20, this difference is not due to one soil having significantly higher denitrification throughout the 10 days. In fact, after 10 days the Spillville mesocosm only had 0.32% more reduction. The reason why these two sets of data differed was due to the fact that these two mesocosms had differing rates of reduction throughout the monitoring period. Thus the overall amount of denitrification does not differ, and the difference in rates of denitrification does not have bearing on this study.

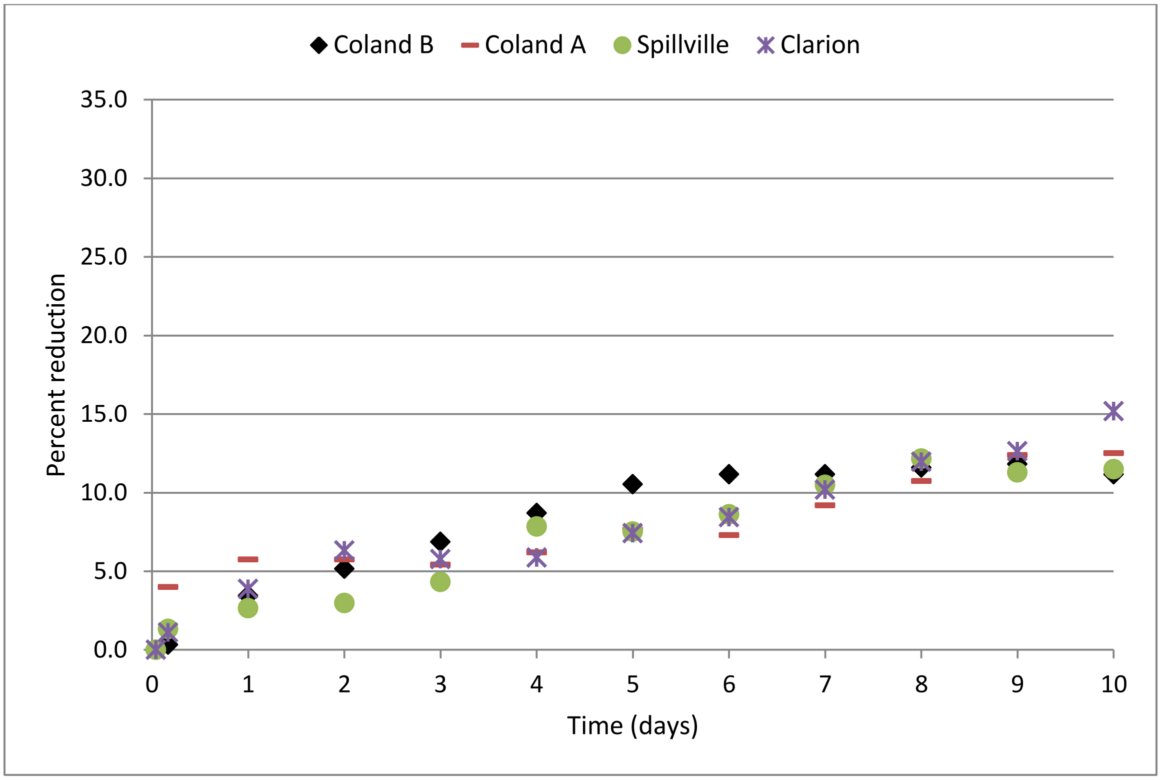


Figure 20. Percent reduction in nitrate concentrations for mesocosms over time (bare soils only)

It is likely that the similarities within these soil types were the reason why the results were so similar. While each soil was classified as a unique soil type all of these soils originated from the same agricultural field and all soils were within several hundred yards of each other. Although the soil was not analyzed for organic carbon it can be presumed that the organic carbon content for each of these soils is very similar. As discussed previously organic carbon content is an important component in soil denitrification (Burford and Bremner, 1975; Triska, 2007).

Oxygen levels in the soils are also another important component. Denitrifying bacteria convert nitrate to nitrogen gas (N_2) in anoxic conditions (Dubrovsky, 2010). All of the mesocosms were ponded for the same length of time, and water was pumped into each mesocosm at the same rate and thus the oxygen content in the water and soils should be similar.

Temperature can also affect denitrification rates and warmer temperatures are beneficial for denitrification (Crumpton et. al., 2008; Isenhart, 1992; Triska, 2007). All of these mesocosms were conducted during the month of September, 2013. The temperature levels should not have significantly impacted daily readings during this experiment as the temperature remained relatively stable during this time. The average water temperatures taken during the testing ranged from 18.25-22.40 °C.

3.5 CONCLUSION AND FUTURE RESEARCH

These mesocosm experiments shed some light on the effectiveness of vegetation and upland agriculture soil types. The knowledge gained through this research is designed to help future researchers and conservationists make informed decisions on vegetation compositions and soil structures for treatment wetlands in agricultural areas.

From the evidence gathered during this experiment it is clear the vegetation has a positive effect on nitrate removal. All of the vegetated mesocosms removed more nitrate than the control Coland B bare soil mesocosm. However, the *Carex crinita* mesocosm revealed only a 3% greater reduction than the Coland B bare soil and at the 10 day mark this vegetated plot had less reduction than the Clarion bare soil mesocosm. Thus, results from this experiment indicate that choosing a monoculture of *Carex crinita* as the vegetation for a treatment wetland would not be the best choice. It was determined that the mixed vegetation was the most effective at removing nitrate from the wetland mesocosm waters. Thus, when determining vegetation to plant in treatment wetlands such as the wetland on the site in southern, Minnesota a mixed composition of native vegetation should be used.

This experiment only focused on upland soils that were historically cultivated. These soils are not typically ideal for treatment wetlands because they contain lower organic carbon than natural wetlands. However, if a wetland is going to be constructed ideal conditions are not always available. Treating tile drainage water from fields means using the soils and field space that exist and these are typically upland soils with low carbon contents. Wetlands constructed in floodplain or alluvial soils are also not always ideal either. These soils may have lower clay contents, which would allow the water to flow more quickly through the wetland surface, into the subsurface and into the nearby surface water. This quicker movement would reduce the residence time of the surface water in the wetland and of the water that seeps into the subsurface. Long residence times are needed to allow the nitrates be converted by bacteria.

The soils in this experiment were tested to determine if one of these upland soils were better suited for treatment wetlands. None of these soils had high reduction levels or were significantly more efficient at removing nitrate from the mesocosm. Thus, none of these soils proved to be a more effective aid in nitrate removal for the treatment wetland constructed in southern Minnesota.

Future improvements should build upon the finding from this paper and should attempt to expand the knowledge behind nitrate removal processes.

In order to create more conclusive results future studies should create more replicates. Several replicates of each mesocosm with the same soils and vegetation would help to create more robust results. The experiments could also be repeated during different times of the year to develop a better understanding of how temperature and light effects nitrate removal. The mesocosm tanks have already been set up so repeated measures with the same soil regime in future months and years would be a relatively straightforward undertaking. Extending the length of the experiments past 10 days could yield stronger trends. The current issue with running the experiment longer is that water could evaporate out of the mesocosm too quickly leaving the ponded water too shallow to measure in and less dilute than before. This issue can be ameliorated by removing some of the lower layers of sand and creating a deeper ponding area.

Another improvement would be to take a wider array of measurements. The nitrate concentrations and temperatures of the water were focused on for this study. Taking dissolved oxygen measurements and redox values could help determine if the water is becoming anoxic, as denitrification only occurs in anoxic conditions (Dubrovsky, 2010). In future studies it would be valuable to conduct a soil texture analysis, and to take measurements of the organic carbon content, and nitrate content before and after the experiment. Many additional parameters would be useful, but there often must be a balance of how many parameters are looked at because it can become costly.

A future area of study would be to investigate hydric soils. Wetland soils could be collected and tested to determine the difference in nitrate removal between the upland farm soils and lowland hydric soils. Due to their high organic carbon content it is more than likely that the hydric wetland soils would achieve greater nitrate reductions.

More can always be done, but there are always factors limiting scientific developments. As more research in this field becomes financed, more academics become interested, and as technologies improve; original and noteworthy results could be forthcoming on nitrate removal dynamics in mesocosms.

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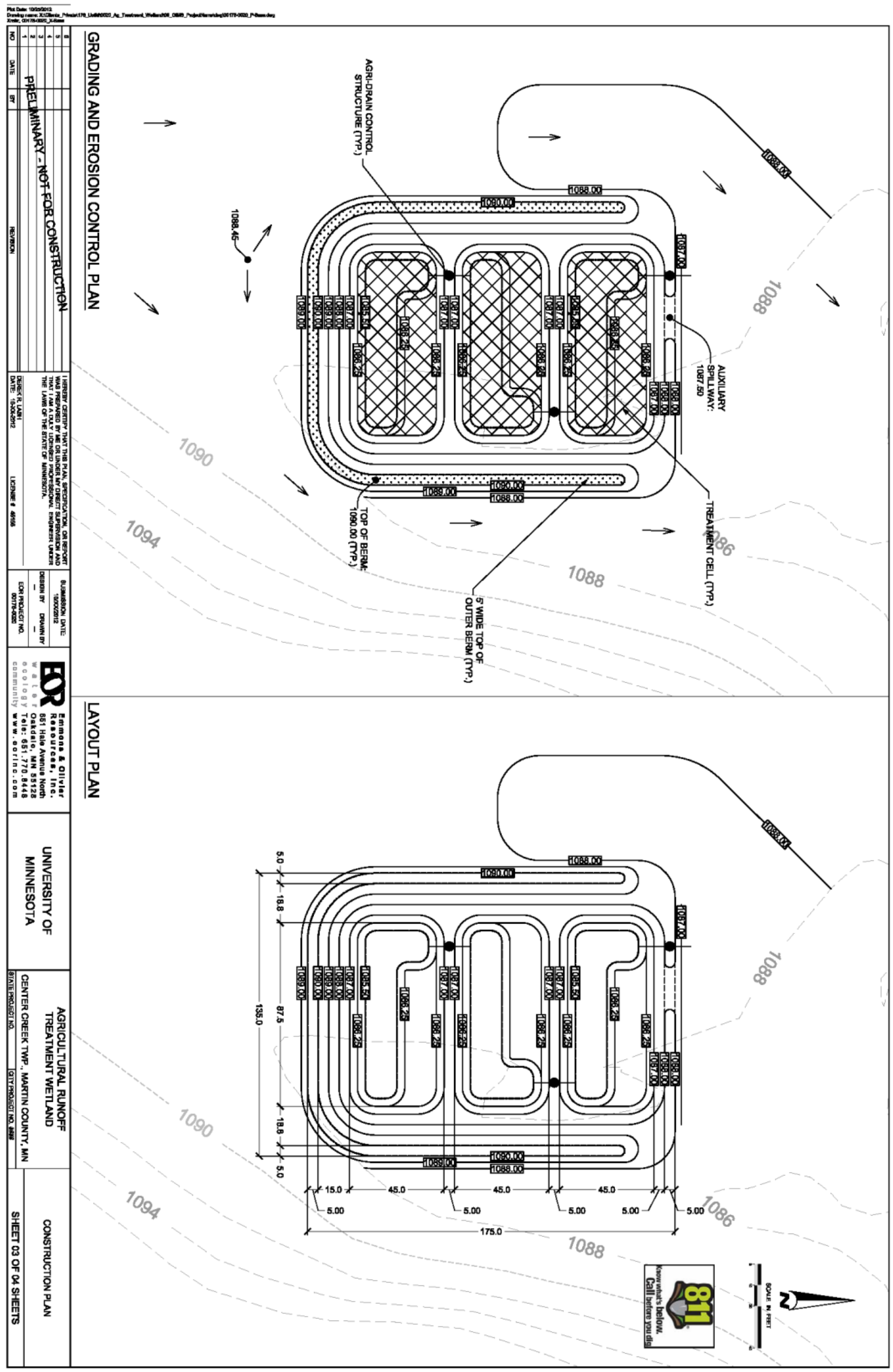
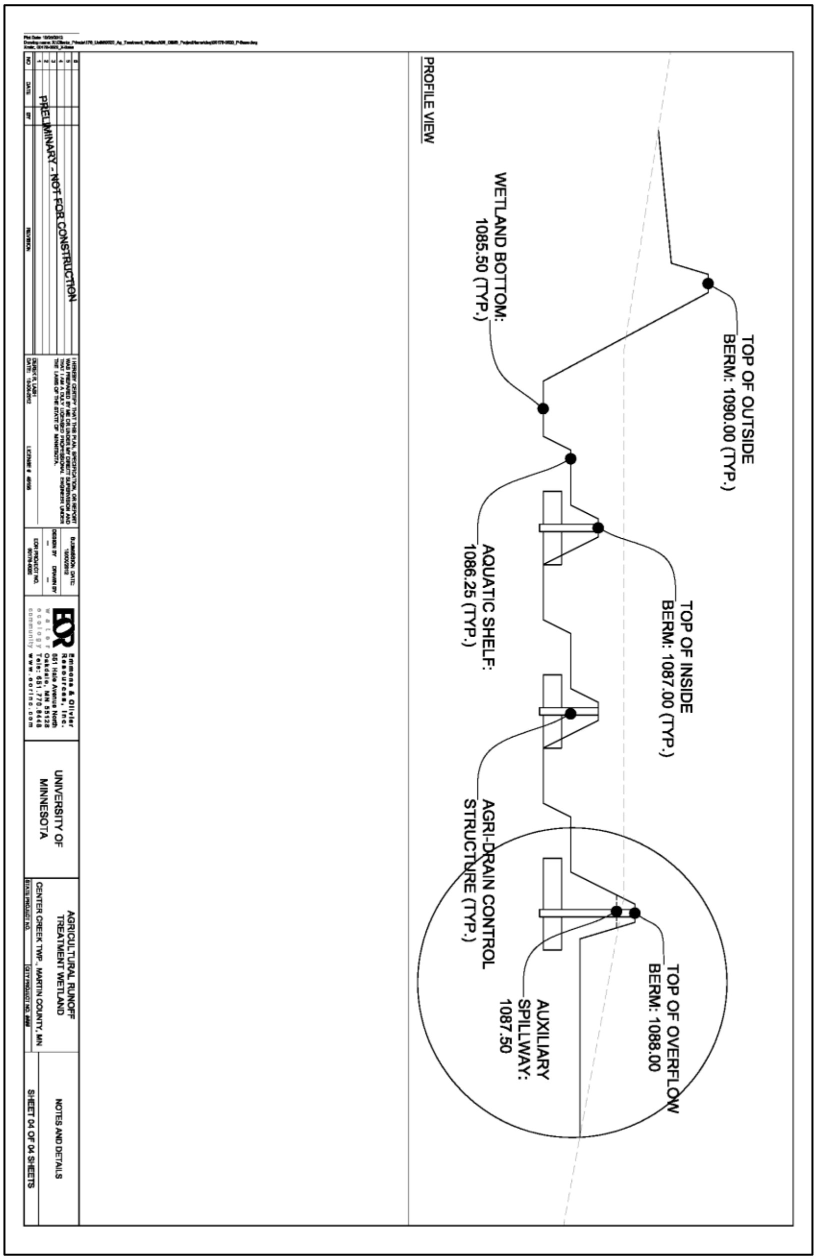
APPENDICES

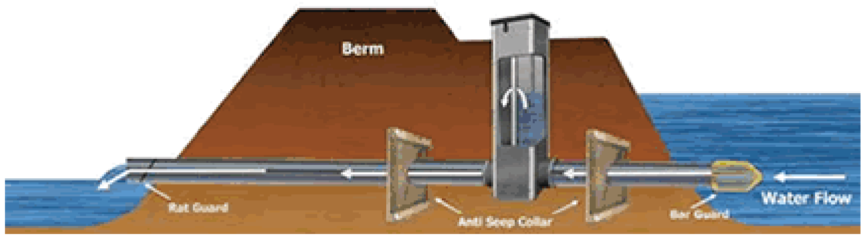
APPENDIX A: Wetland CONSTRUCTION PLAN PREPARED BY EMMONS AND OLIVIER RESOURCES




PROFILE VIEW
TOP OF OUTSIDE BERM: 1090.00 (TYP.)
WETLAND BOTTOM: 1085.50 (TYP.)
AQUATIC SHELF: 1086.25 (TYP.)
TOP OF INSIDE BERM: 1087.00 (TYP.)
AGRI-DRAIN CONTROL STRUCTURE (TYP.)
TOP OF OVERFLOW BERM: 1088.00
AUXILIARY SPILLWAY: 1087.50
PRELIMINARY - NOT FOR CONSTRUCTION
UNIVERSITY OF MINNESOTA
AGRICULTURAL RUNOFF TREATMENT WETLAND
CENTER CREEK TWP., MARTIN COUNTY, MN
NOTES AND DETAILS
SHEET 04 OF 04 SHEETS

APPENDIX B: WETLAND MONITORING EQUIPMENT

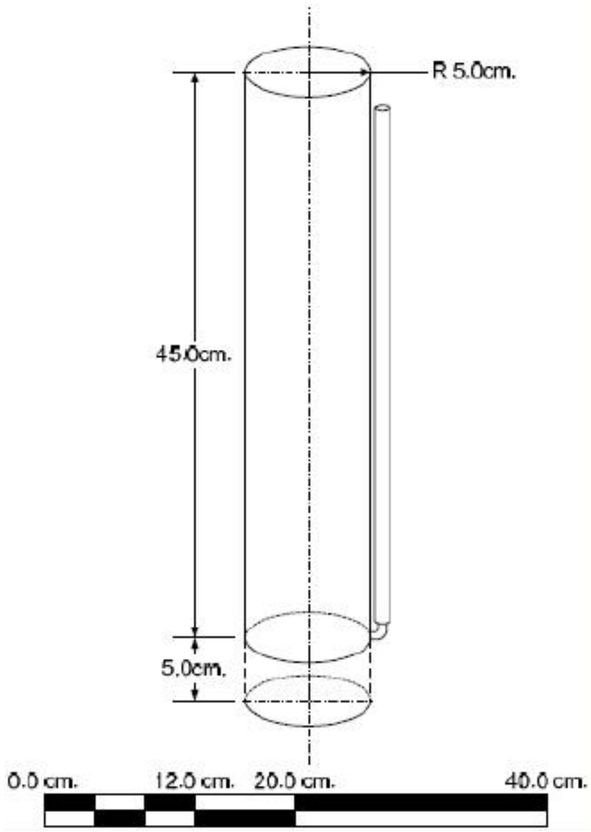
<u>Agri Drain inline water level control structure</u>:
Picture source: http://www.agridrain.com/watercontrolproductsinline.asp



<u>Area velocity flow loggers (ISCO 4150 & ISCO 2150)</u>:
Picture source: http:/ /www.isco.com

<u>Data loggers</u>:
Picture source: http://www.solinst.com

<u>Philip-Dunne Infiltrometer</u>: (Gulliver et. al.,2010):



APPENDIX C: MNDOT SEED MIXTURE FOR STORMWATER TREATMENT BASINS SOUTH & WEST (LOT NUMBER 33-261)

33-261 **Stormwater South & West**

Common Name	Scientific Name	Rate (kg/ha)	Rate (lb/ac)	% of Mix (% by wt)	Seeds/ sq ft
big bluestem	*Andropogon gerardii*	2.24	2.00	5.72%	7.35
fringed brome	*Bromus ciliatus*	2.24	2.00	5.73%	8.10
bluejoint	*Calamagrostis canadensis*	0.07	0.06	0.18%	6.40
slender wheatgrass	*Elymus trachycaulus*	1.12	1.00	2.85%	2.53
Virginia wild rye	*Elymus virginicus*	1.68	1.50	4.28%	2.31
switchgrass	*Panicum virgatum*	0.43	0.38	1.07%	1.93
fowl bluegrass	*Poa palustris*	1.19	1.06	3.03%	50.70
Indian grass	*Sorghastrum nutans*	0.13	0.12	0.36%	0.55
prairie cordgrass	*Spartina pectinata*	0.43	0.38	1.07%	0.91
	Total Grasses	**9.53**	**8.50**	**24.29%**	**80.78**
awl-fruited sedge	*Carex stipata*	0.28	0.25	0.71%	3.10
dark green bulrush	*Scirpus atrovirens*	0.21	0.19	0.54%	31.70
woolgrass	*Scirpus cyperinus*	0.07	0.06	0.18%	39.00
	Total Sedges and Rushes	**0.56**	**0.50**	**1.43%**	**73.80**
Canada anemone	*Anemone canadensis*	0.08	0.07	0.19%	0.20
marsh milkweed	*Asclepias incarnata*	0.12	0.11	0.32%	0.20
leafy beggarticks	*Bidens frondosa*	0.12	0.11	0.31%	0.20
flat-topped aster	*Doellingeria umbellata*	0.07	0.06	0.17%	1.50
spotted Joe pye weed	*Eutrochium maculatum*	0.07	0.06	0.18%	2.19
autumn sneezeweed	*Helenium autumnale*	0.15	0.13	0.36%	5.97
obedient plant	*Physostegia virginiana*	0.08	0.07	0.21%	0.30
tall coneflower	*Rudbeckia laciniata*	0.08	0.07	0.21%	0.37
New England aster	*Symphyotrichum novae-angliae*	0.08	0.07	0.19%	1.56
blue vervain	*Verbena hastata*	0.06	0.05	0.15%	1.85
golden alexanders	*Zizia aurea*	0.22	0.20	0.56%	0.79
	Total Forbs	**1.12**	**1.00**	**2.85%**	**15.13**
Oats or winter wheat (see note at beginning of list for recommended dates)		28.02	25.00	71.43%	11.14
	Total Cover Crop	**28.02**	**25.00**	**71.43%**	**11.14**
	Totals:	**39.23**	**35.00**	**100.00%**	**180.85**
Purpose:	Stormwater pond edges, temporarily flooded dry ponds, and temporarily flooded ditch bottoms.				
Planting Area:	Tallgrass Aspen Parklands, Prairie Parkland, and Eastern Broadleaf Forest Provinces. Mn/DOT Districts 2(west), 3B, 4, Metro, 6, 7 & 8.				

APPENDIX D: SEED MIXES FOR WETLAND CELLS
(HIGH DIVERSITY, MEDIUM DIVERSITY & LOW DIVERSITY)

Description: High Diversity Wet Prairie
Seeding Rate: 16.55 lb/acre (151.5 seeds/square foot)

Common Name	Scientific Name	% of Mix	Seeds/ft^2	Total lb
Grasses				
Big Bluestem	Andropogon gerardii	7.55%	4.6	0.18 PLS lb
~~Fringed Brome~~	~~Bromus ciliatus~~	~~0.00%~~	~~0.0~~	~~0.00 PLS lb~~
Prairie Cord Grass (sub)	Spartina pectinata	11.33%	4.5	0.26 PLS lb
Blue Joint Grass	Calamagrostis canadensis	0.30%	5.1	0.01 PLS lb
Canada Wild Rye	Elymus canadensis	13.22%	4.2	0.31 PLS lb
Reed Manna Grass	Glyceria grandis	1.13%	4.8	0.03 PLS lb
Fowl Manna Grass	Glyceria striata	0.83%	4.5	0.02 PLS lb
Switchgrass	Panicum virgatum	5.66%	4.8	0.13 PLS lb
Fowl Bluegrass	Poa palustris	1.51%	11.9	0.04 PLS lb
Indiangrass	Sorghastrum nutans	3.78%	2.8	0.09 PLS lb
Prairie Cord Grass	Spartina pectinata	3.78%	1.5	0.09 PLS lb
Sedges & Rushes				
Broad-leaved Woolly Sedge	Carex pellita	0.38%	0.6	0.01 PLS lb
Tussock Sedge	Carex stricta	0.15%	0.5	0.00 PLS lb
Brown Fox Sedge	Carex vulpinoidea	0.76%	4.6	0.02 PLS lb
Green Bulrush	Scirpus atrovirens	0.76%	21.1	0.02 PLS lb
Woolgrass	Scirpus cyperinus	0.23%	23.4	0.01 PLS lb
Forbs				
Canada Anemone	Anemone canadensis	0.23%	0.1	0.01 PLS lb
Swamp Milkweed	Asclepias incarnata	0.60%	0.2	0.01 PLS lb
Swamp Aster	Aster puniceus	0.60%	2.9	0.01 PLS lb
Flat-topped Aster	Aster umbellatus	0.38%	1.5	0.01 PLS lb
Showy Tick Trefoil	Desmodium canadense	3.78%	1.3	0.09 PLS lb
Joe Pye Weed	Eupatorium maculatum	0.30%	1.7	0.01 PLS lb
Boneset	Eupatorium perfoliatum	0.23%	2.2	0.01 PLS lb
Sneezeweed	Helenium autumnale	0.38%	3.0	0.01 PLS lb
Sawtooth Sunflower	Helianthus grosseserratus	0.38%	0.3	0.01 PLS lb
Prairie Blazingstar	Liatris pycnostachya	0.15%	0.1	0.00 PLS lb
Great Blue Lobelia	Lobelia siphilitica	0.08%	2.3	0.00 PLS lb
Monkey Flower	Mimulus ringens	0.08%	10.6	0.00 PLS lb
Mountain Mint	Pycnanthemum virginianum	0.60%	8.1	0.01 PLS lb
Blue Vervain	Verbena hastata	1.13%	6.4	0.03 PLS lb
Common Ironweed	Vernonia fasciculata	0.23%	0.3	0.01 PLS lb
Culver's Root	Veronicastrum virginicum	0.15%	7.3	0.00 PLS lb
Golden Alexanders	Zizia aurea	1.89%	1.3	0.04 PLS lb
Cover Crop				
Oats	Avena sativa	37.46%	2.7	0.87 PLS lb

Description: Medium Diversity Wet Prairie
Seeding Rate: 16.07 lb/acre (154.6 seeds/square foot)

Common Name	Scientific Name	% of Mix	Seeds/ft^2	Total lb
Grasses				
Big Bluestem	Andropogon gerardii	10.89%	6.4	0.25 PLS lb
Canada Wild Rye	Elymus canadensis	19.06%	5.8	0.43 PLS lb
Reed Manna Grass	Glyceria grandis	1.63%	6.7	0.04 PLS lb
Switchgrass	Panicum virgatum	8.17%	6.7	0.18 PLS lb
Fowl Bluegrass	Poa palustris	2.18%	16.7	0.05 PLS lb
Prairie Cord Grass	Spartina pectinata	5.44%	2.1	0.12 PLS lb
Sedges & Rushes				
Broad-leaved Woolly Sedge	Carex pellita	0.54%	0.9	0.01 PLS lb
Brown Fox Sedge	Carex vulpinoidea	1.09%	6.4	0.02 PLS lb
Green Bulrush	Scirpus atrovirens	1.09%	29.6	0.02 PLS lb
Woolgrass	Scirpus cyperinus	0.33%	32.8	0.01 PLS lb
Forbs				
Swamp Milkweed	Asclepias incarnata	0.87%	0.2	0.02 PLS lb
Swamp Aster	Aster puniceus	0.87%	4.1	0.02 PLS lb
Showy Tick Trefoil	Desmodium canadense	5.44%	1.8	0.12 PLS lb
Joe Pye Weed	Eupatorium maculatum	0.44%	2.4	0.01 PLS lb
Sneezeweed	Helenium autumnale	0.54%	4.2	0.01 PLS lb
Sawtooth Sunflower	Helianthus grosseserratus	0.54%	0.5	0.01 PLS lb
Prairie Blazingstar	Liatris pycnostachya	0.22%	0.1	0.00 PLS lb
Monkey Flower	Mimulus ringens	0.11%	14.8	0.00 PLS lb
Blue Vervain	Verbena hastata	1.63%	9.0	0.04 PLS lb
Common Ironweed	Vernonia fasciculata	0.33%	0.5	0.01 PLS lb
Cover Crop				
Oats	Avena sativa	38.58%	2.7	0.87 PLS lb

Description: Low Diversity Wet Prairie
Seeding Rate: 15.01 lb/acre (155.3 seeds/square foot)

Common Name	Scientific Name	% of Mix	Seeds/ft^2	Total lb
Grasses				
Canada Wild Rye	Elymus canadensis	26.82%	7.7	0.56 PLS lb
Switchgrass	Panicum virgatum	11.49%	8.9	0.24 PLS lb
Fowl Bluegrass	Poa palustris	3.06%	22.0	0.06 PLS lb
Sedges & Rushes				
Brown Fox Sedge	Carex vulpinoidea	1.53%	8.4	0.03 PLS lb
Green Bulrush	Scirpus atrovirens	1.53%	38.9	0.03 PLS lb
Woolgrass	Scirpus cyperinus	0.46%	43.1	0.01 PLS lb
Forbs				
Swamp Milkweed	Asclepias incarnata	1.23%	0.3	0.03 PLS lb
Swamp Aster	Aster puniceus	1.23%	5.4	0.03 PLS lb
Showy Tick Trefoil	Desmodium canadense	7.66%	2.3	0.16 PLS lb
Joe Pye Weed	Eupatorium maculatum	0.61%	3.2	0.01 PLS lb
Sawtooth Sunflower	Helianthus grosseserratus	0.77%	0.6	0.02 PLS lb
Blue Vervain	Verbena hastata	2.30%	11.8	0.05 PLS lb
Cover Crop				
Oats	Avena sativa	41.31%	2.7	0.87 PLS lb

APPENDIX E: WATER VOLUME EQUATIONS FOR FLOW THROUGH INLET AND OUTLET PIPES

Partially full pipe flow equations:

Less than ½ full:

$$r = \frac{D}{2}$$

$$h = y$$

$$\theta = 2\arccos\frac{r-h}{r}$$

$$A = \frac{r^2(\theta - \sin\theta)}{2}$$

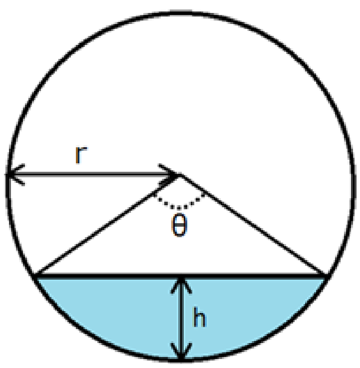
More than ½ full

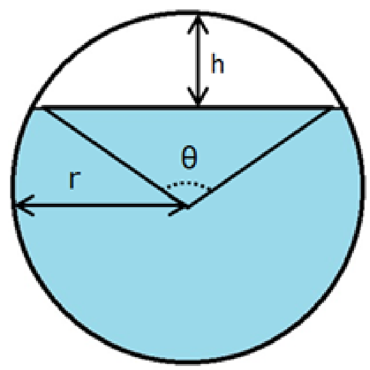
$$r = \frac{D}{2}$$

$$h = 2r-y$$

$$\theta = 2\arccos\frac{r-h}{r}$$

$$A = \pi r^2 - \frac{r^2(\theta - \sin\theta)}{2}$$





$$\text{Water volume} = (V \cdot A)T_{interval}$$

Where:
r = radius
h = circular segment height
y = depth of flow
θ = central angle
A = cross sectional area of flow
V = velocity
$T_{interval}$ = time interval at which velocities and water height are recorded

Data source:

Bengtson, HH, Spreadsheet Use for Partially Full Pipe Flow Calculations [internet]. Stony Point, NY: Continuing Education Continuing Education & Development, Inc.; 2011 [cited 2014 February 8] Available from: <http://www.cedengineering.com/upload/Partially%20Full%20Pipe%20Flow%20Calculations.pdf>

APPENDIX F: WATER VOLUME EQUATIONS FOR FLOW THROUGH AGRI DRAIN STRUCTURES

Step 1. Calculate flow rates (Chun and Cooke, 2008)

$$Q = 0.020(L-0.427H)H^{1.48} \qquad H \leq 0.44L$$

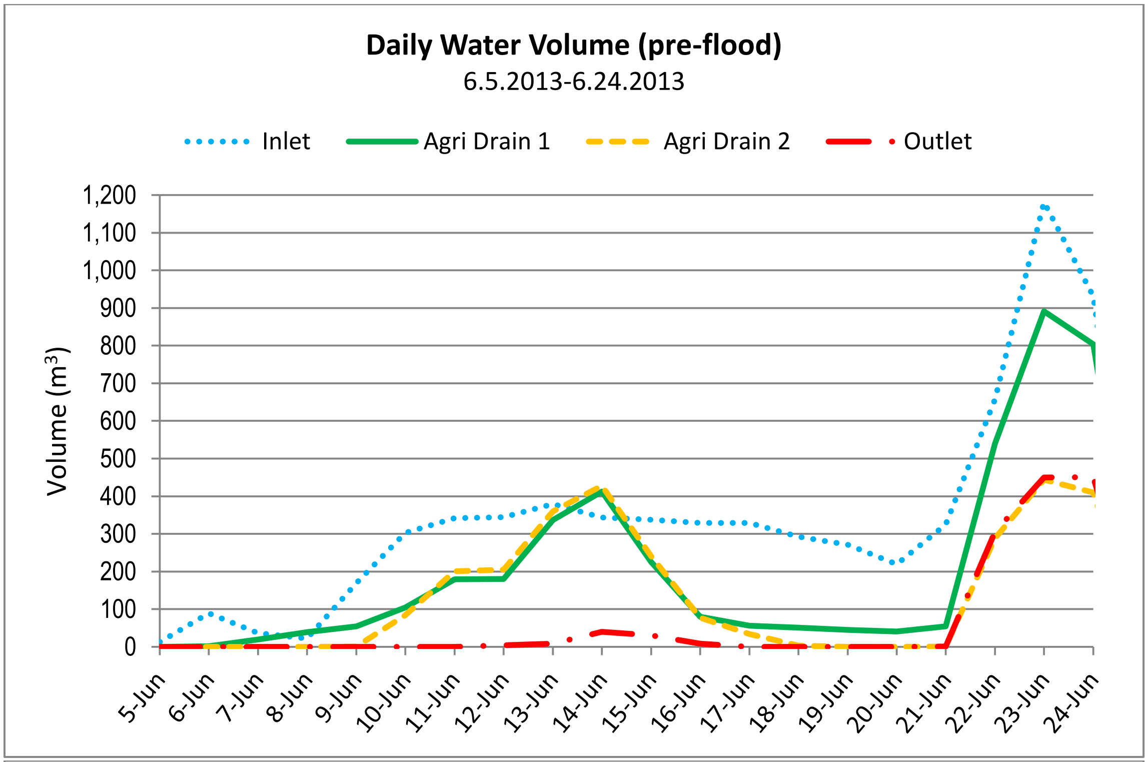
$$Q = 0.027(LH)^{1.2} \qquad H > 0.44L$$

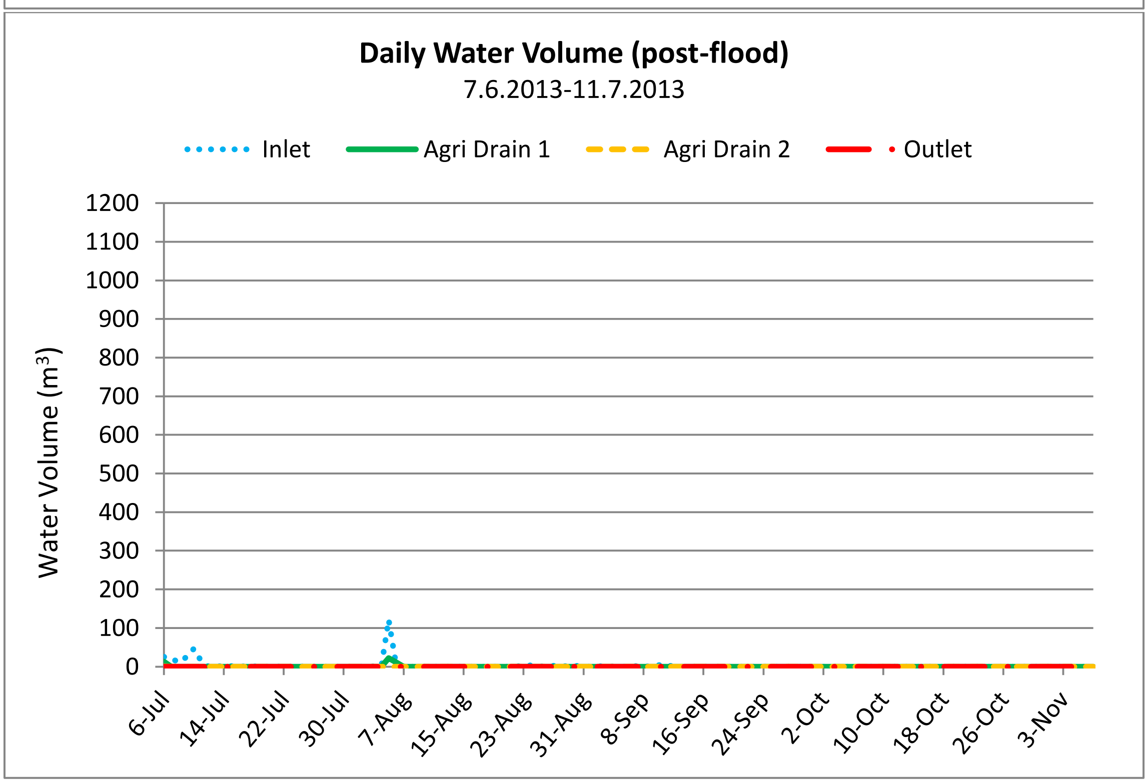
Step 2. Calculate water volume

WATER VOLUME = Q X (L X H)

Where:
Q = velocity (length/s)
L = length of stop logs
H = height above Agri Drain stop logs

APPENDIX G: CALCULATED DAILY WATER VOLUMES THROUGHOUT WETLAND IN 2013





Date	Inlet (m^3)	Agri Drain 1 (m^3)	Agri Drain 2 (m^3)	Outlet (m^3)
18-May	0.00	0.00	0.00	0.00
19-May	0.00	0.00	0.00	0.00
20-May	0.00	0.00	0.00	0.00
21-May	0.00	0.00	0.00	0.00
22-May	0.00	0.00	0.00	0.00
23-May	0.00	0.00	0.00	0.00
24-May	0.00	0.00	0.00	0.00
25-May	0.00	0.00	0.00	0.00
26-May	0.00	0.00	0.00	0.00
27-May	0.00	0.00	0.00	0.00
28-May	0.00	0.00	0.00	0.00
29-May	0.00	0.00	0.00	0.00
30-May	0.00	0.00	0.00	0.00
31-May	0.00	0.00	0.00	0.00
1-Jun	0.00	0.00	0.00	0.00
2-Jun	0.00	0.00	0.00	0.00
3-Jun	0.00	0.00	0.00	0.00
4-Jun	0.00	0.00	0.00	0.00
5-Jun	12.76	0.00	0.00	0.00
6-Jun	89.22	0.53	0.00	0.00
7-Jun	35.98	18.98	0.00	0.00
8-Jun	21.98	39.18	0.00	0.00
9-Jun	168.86	54.05	0.80	0.00
10-Jun	302.23	104.30	83.64	0.04
11-Jun	341.62	179.30	200.30	0.14
12-Jun	344.54	180.00	204.74	3.73
13-Jun	379.35	336.65	359.94	7.97
14-Jun	343.86	412.03	427.28	39.82
15-Jun	337.21	224.90	240.78	30.56
16-Jun	329.36	79.82	76.95	8.26
17-Jun	328.99	55.98	33.75	0.02
18-Jun	292.71	50.97	3.37	0.00
19-Jun	270.75	44.51	0.00	0.00
20-Jun	218.93	40.86	0.00	0.00
21-Jun	325.08	53.80	0.84	0.43
22-Jun	657.16	540.27	287.78	304.93
23-Jun	1181.22	891.81	443.65	449.76

24-Jun	931.09	803.57	410.19	450.33
25-Jun	23.68	14.55	8.39	17.65
26-Jun	Wetland Flooded			
27-Jun				
28-Jun				
29-Jun				
30-Jun				
1-Jul				
2-Jul				
3-Jul				
4-Jul				
5-Jul				
6-Jul	25.71	12.96	0.86	0.00
7-Jul	21.53	0.02	0.00	0.00
8-Jul	11.01	0.00	0.00	0.00
9-Jul	24.82	0.00	0.00	0.00
10-Jul	46.09	0.00	0.00	0.00
11-Jul	13.58	0.00	0.00	0.00
12-Jul	0.00	0.00	0.00	0.00
13-Jul	1.26	0.00	0.00	0.00
14-Jul	0.32	0.00	0.00	0.00
15-Jul	0.95	0.00	0.00	0.00
16-Jul	0.84	0.00	0.00	0.00
17-Jul	0.50	0.00	0.00	0.00
18-Jul	0.16	0.00	0.00	0.00
19-Jul	0.00	0.00	0.00	0.00
20-Jul	0.00	0.00	0.00	0.00
21-Jul	0.00	0.00	0.00	0.00
22-Jul	0.20	0.00	0.00	0.00
23-Jul	0.00	0.00	0.00	0.00
24-Jul	0.00	0.00	0.00	0.00
25-Jul	0.55	0.00	0.00	0.00
26-Jul	0.00	0.00	0.00	0.00
27-Jul	0.00	0.00	0.00	0.00
28-Jul	0.00	0.00	0.00	0.00
29-Jul	0.00	0.00	0.00	0.00
30-Jul	0.00	0.00	0.00	0.00
31-Jul	0.00	0.00	0.00	0.00
1-Aug	0.00	0.00	0.00	0.00

2-Aug	0.00	0.00	0.00	0.00
3-Aug	0.21	0.00	0.00	0.00
4-Aug	0.00	0.00	0.00	0.00
5-Aug	121.04	21.99	0.00	0.00
6-Aug	6.45	10.94	0.00	0.00
7-Aug	0.00	0.00	0.00	0.00
8-Aug	0.00	0.00	0.00	0.00
9-Aug	0.00	0.00	0.00	0.00
10-Aug	0.00	0.00	0.00	0.00
11-Aug	0.00	0.00	0.00	0.00
12-Aug	0.00	0.00	0.00	0.00
13-Aug	0.00	0.00	0.00	0.00
14-Aug	0.00	0.00	0.00	0.00
15-Aug	0.00	0.00	0.00	0.00
16-Aug	0.00	0.00	0.00	0.00
17-Aug	0.00	0.00	0.00	0.00
18-Aug	0.00	0.00	0.00	0.00
19-Aug	0.00	0.00	0.00	0.00
20-Aug	0.00	0.00	0.00	0.00
21-Aug	0.00	0.00	0.00	0.00
22-Aug	0.00	0.00	0.00	0.00
23-Aug	4.17	0.00	0.00	0.00
24-Aug	3.14	0.00	0.00	0.00
25-Aug	0.00	0.00	0.00	0.00
26-Aug	1.50	0.00	0.00	0.00
27-Aug	1.84	0.00	0.00	0.00
28-Aug	2.73	0.00	0.00	0.00
29-Aug	0.00	0.00	0.00	0.00
30-Aug	1.79	0.00	0.00	0.00
31-Aug	0.00	0.00	0.00	0.00
1-Sep	0.00	0.00	0.00	0.00
2-Sep	0.65	0.00	0.00	0.00
3-Sep	2.18	0.00	0.00	0.00
4-Sep	0.00	0.00	0.00	0.00
5-Sep	0.00	0.00	0.00	0.00
6-Sep	0.00	0.00	0.00	0.00
7-Sep	1.18	0.00	0.00	0.00
8-Sep	1.05	0.00	0.00	0.00
9-Sep	0.00	0.00	0.00	0.00

10-Sep	3.80	0.00	0.00	0.00
11-Sep	5.16	0.00	0.00	0.00
12-Sep	0.00	0.00	0.00	0.00
13-Sep	0.00	0.00	0.00	0.00
14-Sep	0.00	0.00	0.00	0.00
15-Sep	0.00	0.00	0.00	0.00
16-Sep	0.00	0.00	0.00	0.00
17-Sep	0.00	0.00	0.00	0.00
18-Sep	0.00	0.00	0.00	0.00
19-Sep	0.00	0.00	0.00	0.00
20-Sep	0.00	0.00	0.00	0.00
21-Sep	0.00	0.00	0.00	0.00
22-Sep	0.00	0.00	0.00	0.00
23-Sep	0.00	0.00	0.00	0.00
24-Sep	0.00	0.00	0.00	0.00
25-Sep	0.00	0.00	0.00	0.00
26-Sep	0.00	0.00	0.00	0.00
27-Sep	0.00	0.00	0.00	0.00
28-Sep	0.00	0.00	0.00	0.00
29-Sep	0.00	0.00	0.00	0.00
30-Sep	0.00	0.00	0.00	0.00
1-Oct	0.00	0.00	0.00	0.00
2-Oct	0.00	0.00	0.00	0.00
3-Oct	0.00	0.00	0.00	0.00
4-Oct	0.00	0.00	0.00	0.00
5-Oct	0.00	0.00	0.00	0.00
6-Oct	0.00	0.00	0.00	0.00
7-Oct	0.00	0.00	0.00	0.00
8-Oct	0.00	0.00	0.00	0.00
9-Oct	0.00	0.00	0.00	0.00
10-Oct	0.00	0.00	0.00	0.00
11-Oct	0.00	0.00	0.00	0.00
12-Oct	0.00	0.00	0.00	0.00
13-Oct	0.00	0.00	0.00	0.00
14-Oct	0.00	0.00	0.00	0.00
15-Oct	0.00	0.00	0.00	0.00
16-Oct	0.00	0.00	0.00	0.00
17-Oct	0.00	0.00	0.00	0.00
18-Oct	0.00	0.00	0.00	0.00

19-Oct	0.00	0.00	0.00	0.00
20-Oct	0.00	0.00	0.00	0.00
21-Oct	0.00	0.00	0.00	0.00
22-Oct	0.00	0.00	0.00	0.00
23-Oct	0.00	0.00	0.00	0.00
24-Oct	0.00	0.00	0.00	0.00
25-Oct	0.00	0.00	0.00	0.00
26-Oct	0.00	0.00	0.00	0.00
27-Oct	0.00	0.00	0.00	0.00
28-Oct	0.00	0.00	0.00	0.00
29-Oct	0.00	0.00	0.00	0.00
30-Oct	0.00	0.00	0.00	0.00
31-Oct	0.00	0.00	0.00	0.00
1-Nov	0.00	0.00	0.00	0.00
2-Nov	0.00	0.00	0.00	0.00
3-Nov	0.00	0.00	0.00	0.00
4-Nov	0.00	0.00	0.00	0.00
5-Nov	0.00	0.00	0.00	0.00
6-Nov	0.00	0.00	0.00	0.00
7-Nov	0.00	0.00	0.00	0.00
Totals:	7241.0	4176.8	2783.3	1313.6

APPENDIX H: 2013 MONTHLY RAINFALL TOTALS NEAR GRANADA, MN

Month	Precipitation (inches)
May	4.95
June	8.41
July	2.48
August	2.94
September	0.84
October	2.89
November	0.73

APPENDIX I: WATER QUALITY MEASUREMENTS FROM WETLAND

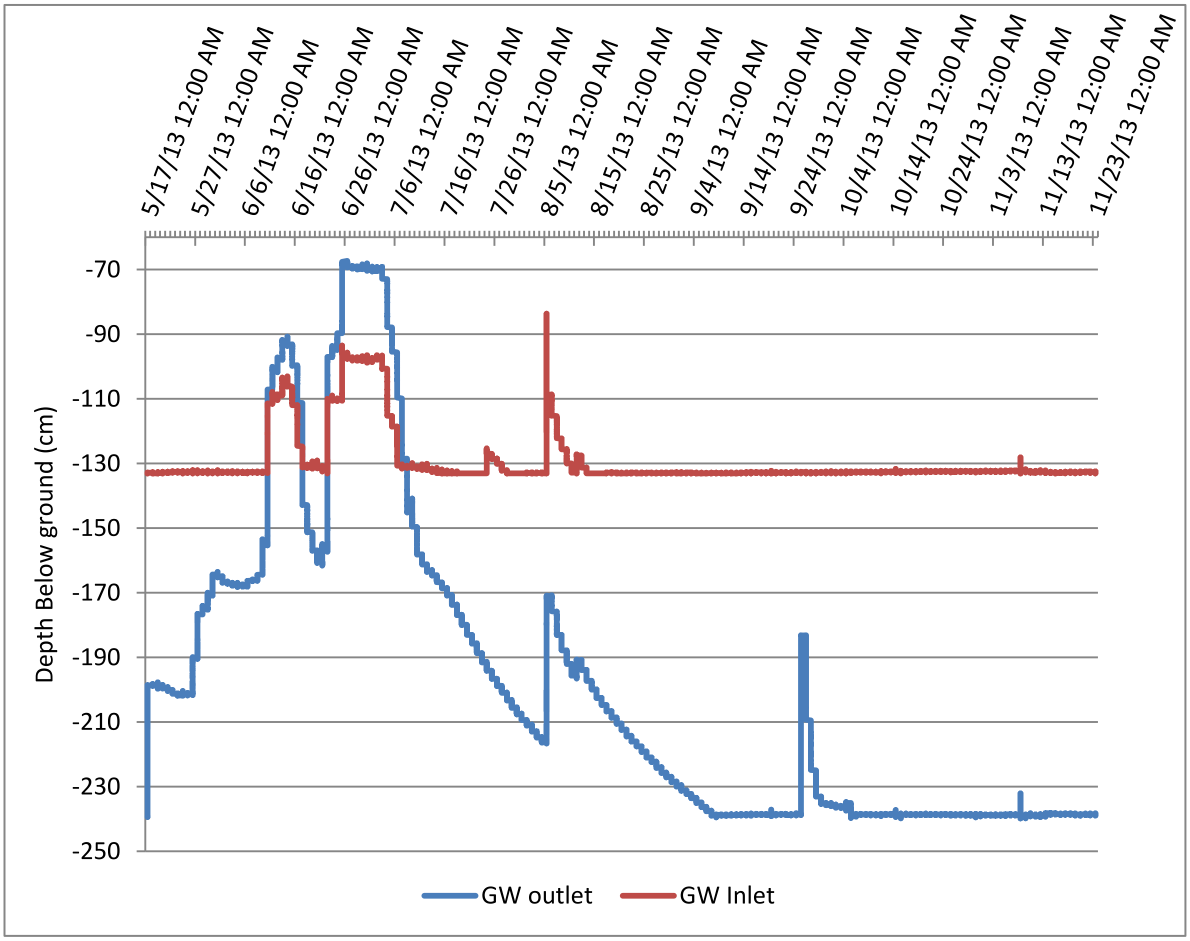
Date	Time	Location	Nitrate + Nitrite (mg/L as N)
6/6/2013	4:20 PM	Inlet	22.4
6/10/2013	10:40 AM	Inlet	22.6
6/10/2013	10:45 AM	Agri-drain01	24.1
6/10/2013	10:50 AM	Agri-drain02	23.9
6/10/2013	12:35 PM	Inlet	24.4
6/10/2013	12:40 PM	Agri-drain01	24.4
6/10/2013	12:50 PM	Agri-drain02	23.7
6/12/2013	4:15 PM	Inlet	21.8
6/12/2013	4:20 PM	Agri-drain01	23.8
6/12/2013	4:25 PM	Agri-drain02	21.7
6/14/2013	9:45 AM	Inlet	22.3
6/14/2013	9:50 AM	Agri-drain01	21.5
6/14/2013	10:00 AM	Agri-drain02	21.5
6/14/2013	10:00 AM	Outlet	21.9
6/14/2013	12:00 PM	Inlet	22.8
6/14/2013	11:50 AM	Agri-drain01	23.2
6/14/2013	12:00 PM	Agri-drain02	22
6/14/2013	12:10 PM	Outlet	24.2
6/17/2013	12:30 PM	Inlet	23.4
6/17/2013	12:40 PM	Agri-drain01	23.4
6/17/2013	12:50 PM	Agri-drain02	24
6/24/2013	10:13 AM	Inlet	24.2
6/24/2013	10:19 AM	Agri-drain01	23.1
6/24/2013	10:26 AM	Agri-drain02	23.1
6/24/2013	10:26 AM	Outlet	22.3

Date	Time	Location	Phosphorus Water Digest Solids, Total Suspended (mg/L)	Specific conductance (umhos/cm)	Phosphorus, Soluble Ortho (mg/L)	Phosphoru s, Total (mg/L)
6/6/2013	4:20 PM	Inlet	5	754.0	*0.043	0.058
6/10/2013	10:40 AM	Inlet	<2	777.0	*0.040	0.049
6/10/2013	10:45 AM	Agri-drain01	4	772.0	*0.033	0.044
6/10/2013	10:50 AM	Agri-drain02	<2	747.0	*0.035	0.053
6/10/2013	12:35 PM	Inlet	<2	780.0	*0.035	0.046
6/10/2013	12:40 PM	Agri-drain01	<2	770.0	*0.032	0.044
6/10/2013	12:50 PM	Agri-drain02	2	753.0	*0.035	0.049
6/12/2013	4:15 PM	Inlet	3	752.0	*0.028	0.042
6/12/2013	4:20 PM	Agri-drain01	4	739.0	*0.023	0.042
6/12/2013	4:25 PM	Agri-drain02	5	729.0	*0.017	0.131
6/14/2013	9:45 AM	Inlet	4	793.0	*0.034	0.044
6/14/2013	9:50 AM	Agri-drain01	3	785.0	*0.024	0.035
6/14/2013	10:00 AM	Agri-drain02	3	783.0	*0.016	0.051
6/14/2013	10:00 AM	Outlet	8	753.0	*0.006	0.044
6/14/2013	12:00 PM	Inlet	<2	782.0	*0.031	0.043
6/14/2013	11:50 AM	Agri-drain01	4	771.0	*0.024	0.038
6/14/2013	12:00 PM	Agri-drain02	6	774.0	*0.019	0.041
6/14/2013	12:10 PM	Outlet	<2	759.0	*0.006	0.034
6/17/2013	12:30 PM	Inlet	<2	790.0	*0.030	0.044

6/17/2013	12:40 PM	Agri-drain01	3	783.0	*0.017	0.038
6/17/2013	12:50 PM	Agri-drain02	3	791.0	*0.008	0.047
6/24/2013	10:13 AM	Inlet	<2	785	*0.050	0.063
6/24/2013	10:19 AM	Agri-drain01	2	783	*0.049	0.062
6/24/2013	10:26 AM	Agri-drain02	2	779	*0.051	0.065
6/24/2013	10:26 AM	Outlet	37	771	*0.053	0.113

APPENDIX J: GROUND WATER LEVELS DURING 2013 FIELD SEASON

- The bottom of the inlet groundwater well is approximately -133 cm and the bottom of the outlet groundwater well is approximately -250 cm. Water level below the bottom of the well could not be estimated or measured.

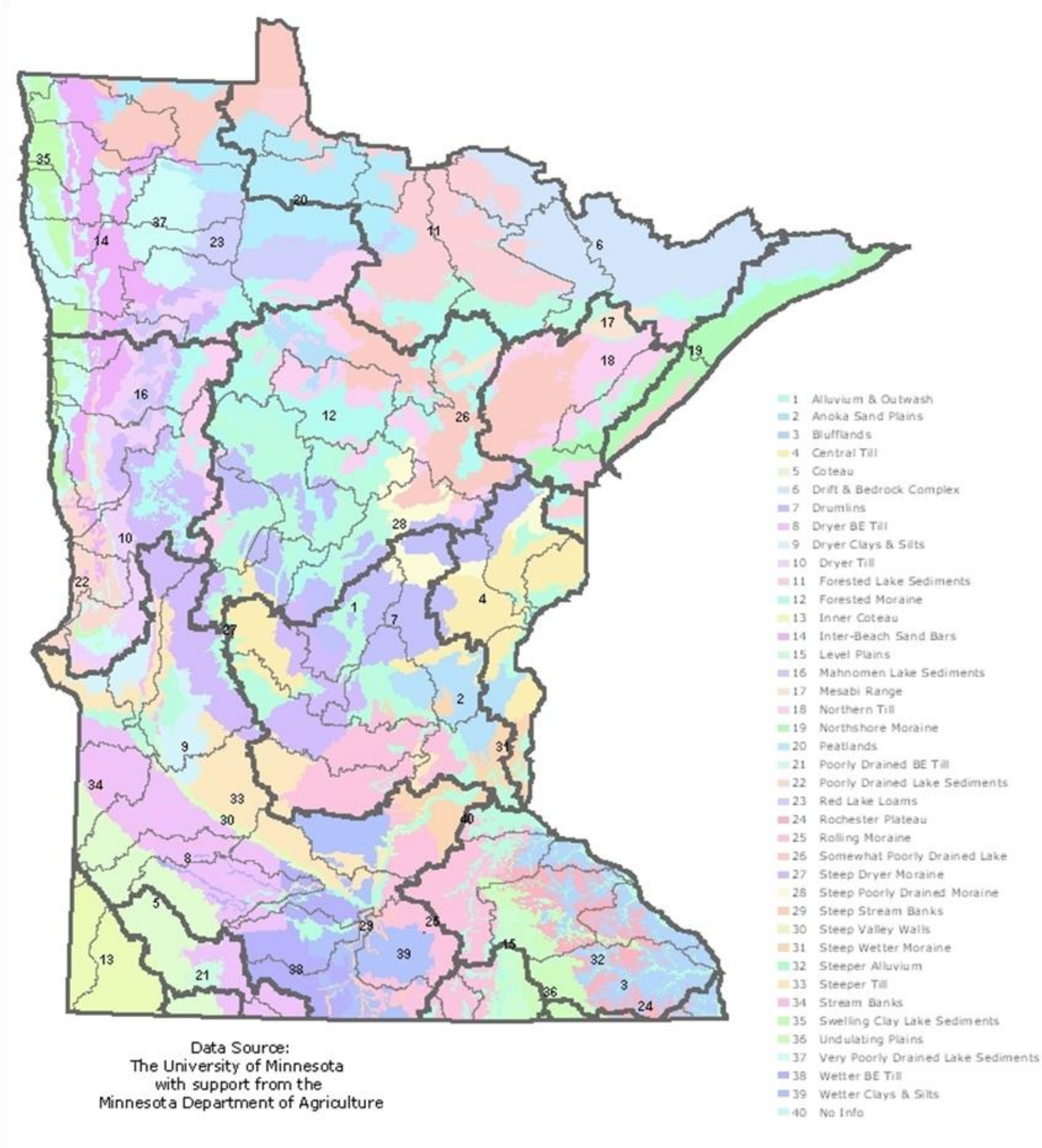


Appendix K: Soil analysis results

Analysis conducted at the Research Analytical Laboratory at the University of Minnesota using the Hydrometer method. Soil collected in November, 2013.

Sample Name	Olsen P (ppm)	LOI OM (%)	TOC (% C)	Total N (% N)
Cell 1	11 / 11	5.0 / 5.0	2.75 / 2.77	0.16 / 0.17
Cell 2	6	4.4	2.39	0.15
Cell 3	6	4.4	2.30 / 2.35	0.10 / 0.13

APPENDIX L: AGROECOREGIONS IN MINNESOTA, USA



APPENDIX M: NITRATE RESULTS FROM WATER SAMPLES FROM ELM CREEK TAKEN AT 290TH AVENUE (MPCA STATION ID S003-025, HUC: 07020009)

DATA SOURCE: http://www.pca.state.mn.us/customPHP/eda/stationInfo.php?ID=S003-025&ORG=MNPCA&wdip=2

Date Sampled	Inorganic nitrogen (nitrate and nitrite) as N (mg/L)	Date Sampled	Inorganic nitrogen (nitrate and nitrite) as N (mg/L)
2/21/2002	4.71	5/16/2006	9.61
4/8/2002	4.22	5/18/2006	9.23
5/14/2002	10.8	5/23/2006	10.1
5/28/2002	8.49	5/25/2006	9.85
6/3/2002	6.08	6/1/2006	8.48
6/13/2002	6.22	6/6/2006	8.18
6/17/2002	9.26	6/8/2006	8.13
7/8/2002	1.04	6/12/2006	8.45
7/22/2002	0.43	6/15/2006	7.16
7/29/2002	0.25	6/19/2006	9.29
8/5/2002	0.2	6/21/2006	10.9
8/7/2002	1.14	6/27/2006	8.62
8/19/2002	0.2	7/5/2006	4.94
8/23/2002	10.9	7/11/2006	4.85
8/26/2002	11	7/13/2006	2.23
9/3/2002	3.12	7/19/2006	0.2
9/23/2002	1.92	7/19/2006	0.2
10/8/2002	13.9	7/20/2006	0.2
4/1/2004	3.68	7/26/2006	0.43
4/14/2004	0.6	8/2/2006	1.68
4/14/2004	0.6	8/3/2006	3.92
4/21/2004	0.68	8/5/2006	4.41
4/21/2004	0.67	8/8/2006	0.51
4/29/2004	1.75	8/15/2006	0.62
5/5/2004	0.2	8/24/2006	0.45
5/13/2004	0.55	8/30/2006	0.86
5/13/2004	0.55	9/13/2006	0.2

5/13/2004	0.56	9/20/2006	1.26
5/19/2004	0.92	9/25/2006	4.63
5/24/2004	20	9/28/2006	5.41
5/25/2004	16.7	3/16/2007	2.92
5/26/2004	14.9	3/17/2007	2.09
6/1/2004	16.7	3/19/2007	3.03
6/2/2004	17.3	3/22/2007	3.94
6/7/2004	13.3	3/26/2007	5.02
6/9/2004	12.7	3/30/2007	6.49
6/10/2004	14.8	4/1/2007	8.43
6/22/2004	13.1	4/2/2007	8.28
7/1/2004	9.49	4/5/2007	8.55
7/6/2004	13	4/13/2007	7.5
7/12/2004	15.2	4/16/2007	8.4
7/12/2004	15.2	4/18/2007	9.03
7/15/2004	15.5	4/23/2007	7.7
7/20/2004	9.32	4/26/2007	7.87
7/22/2004	10	5/1/2007	7.55
7/27/2004	0.9	5/7/2007	8.42
8/4/2004	7.79	5/10/2007	10.8
8/5/2004	3.56	5/14/2007	9.62
8/11/2004	2.39	5/21/2007	9.97
8/16/2004	0.4	5/24/2007	8.9
8/25/2004	6.92	5/31/2007	8.27
8/31/2004	5.49	6/3/2007	7.73
9/7/2004	4.95	6/5/2007	8.13
9/15/2004	5.44	6/7/2007	10.3
9/20/2004	0.58	6/13/2007	8.05
9/23/2004	1.13	6/18/2007	7.55
9/29/2004	0.72	6/27/2007	5.76
10/4/2004	2.53	7/5/2007	0.26
10/6/2004	8.96	7/9/2007	0.2
10/12/2004	11.1	7/17/2007	0.2
4/4/2005	10.3	7/19/2007	0.36
4/4/2005	10.3	7/26/2007	0.2
4/11/2005	9.68	8/7/2007	0.2

4/13/2005	13.1	8/20/2007	0.28
4/18/2005	13.2	8/21/2007	2.54
4/20/2005	12	8/23/2007	2.55
4/27/2005	10.5	9/6/2007	0.2
5/3/2005	10.4	9/10/2007	0.2
5/11/2005	13.9	9/25/2007	0.2
5/16/2005	12.8	10/10/2007	1.11
5/18/2005	12.7	10/23/2007	9.29
5/23/2005	12.6	10/25/2007	9.21
6/1/2005	12.4	10/31/2007	7.61
6/8/2005	19.1	3/25/2008	4.54
6/15/2005	12.9	4/3/2008	3.34
6/21/2005	13	4/8/2008	4.77
7/13/2005	5.32	4/11/2008	13.4
7/20/2005	0.88	4/14/2008	11
7/26/2005	1.49	4/23/2008	8.71
8/3/2005	0.89	4/25/2008	10.8
8/11/2005	1.6	4/28/2008	13.2
8/18/2005	3.96	5/1/2008	12.2
8/23/2005	4.89	5/4/2008	12.7
9/1/2005	3.56	5/8/2008	11.7
9/13/2005	4.81	5/13/2008	10.2
9/20/2005	3.64	5/22/2008	9.9
9/26/2005	3.13	5/30/2008	10.8
9/29/2005	3.75	6/3/2008	12.7
10/3/2005	4.39	6/9/2008	13.8
10/6/2005	4.4	6/11/2008	13.2
10/13/2005	6.99	6/17/2008	10.8
10/26/2005	7.53	6/25/2008	8.37
4/1/2006	12.1	7/1/2008	12
4/4/2006	11.3	7/9/2008	7.81
4/7/2006	9.56	7/15/2008	5.11
4/11/2006	8.18	7/23/2008	3.53
4/13/2006	9.14	7/30/2008	3.25
4/18/2006	9.05	8/5/2008	1.92
4/20/2006	9.78	8/13/2008	0.62

4/25/2006	9.11	8/20/2008	0.2
5/1/2006	12.7	8/28/2008	0.2
5/2/2006	12.02	9/4/2008	0.2
5/4/2006	10.7	9/9/2008	0.2
5/8/2006	9.56	9/16/2008	0.2
5/11/2006	9.65	10/1/2008	0.2

Summary Statistics Inorganic nitrogen (nitrate and nitrite) as N (mg/L)	
Average	6.662619
Variance	23.84926
SD	4.88357
Max	20
Min	0.2

APPENDIX N: NITRATE SOLUTION CALCULATION

A concentration of 23.0mg/L of Nitrate as N (NO_3-N) with Sodium Nitrate pellets ($NaNO_3$) and using a 375 gallon tank to mix the solution:

$$\frac{84.995mol\ \ NaNO_3}{1g}\times\frac{1g}{14.007mol\ \ NO_3\text{-}N}\times\frac{23.0mg\ \ NO_3\text{-}N}{1liter}\times\frac{1g\ \ pellet}{1000mg\ \ pellet}\times\ 375\ \ liter\ =\ 52.3g\ \ pellets$$

APPENDIX O: NITRATE LEVELS IN WETLAND MESOCOSMS

Percent reduction of nitrate (NO_3-N) concentrations in vegetated mesocosms			
Days	Mixed Vegetation (%)	*Carex crinita* (%)	*Panicum virgatum* (%)
0.04 (1 hour)	0	0	0
0.167 (4 hours)	3.73	0.43	1.68
1	7.99	3.36	5.03
2	10.76	5.74	7.04
3	14.48	7.58	7.60
4	18.21	9.53	10.17
5	20.98	11.16	10.61
6	25.67	11.70	12.07
7	28.65	11.70	13.30
8	30.78	12.35	15.75
9	32.27	11.92	18.77
10	34.93	14.19	23.02

Concentrations of nitrate (NO_3-N) in vegetated mesocosms			
Days	Mixed vegetation (mg/L)	*Carex crinita* (mg/L)	*Panicum virgatum* (mg/L)
0.04 (1 hour)	21.2	23.0	23.0
0.1667 (4 hours)	20.4	20.9	20.2
1	19.5	20.8	19.9
2	18.9	20.1	19.2
3	18.1	19.7	18.8
4	17.3	19.3	18.7
5	16.8	18.9	18.2
6	15.8	18.5	18.1
7	15.1	18.4	17.8
8	14.7	18.4	17.5
9	14.4	18.3	17.0
10	13.8	18.4	16.4

Percent reduction of nitrate (NO_3-N) concentrations in bare soil mesocosms				
Days	Coland Loam B (%)	Coland Loam A (%)	Spillville Loam (%)	Clarion-Storgen Loam (%)
0.04 (1 hour)	0	0	0	0
0.167 (4 hours)	0.32	3.98	1.33	1.11
1	3.44	5.75	2.65	3.88
2	5.16	5.75	2.98	6.31
3	6.87	5.42	4.31	5.76
4	8.70	6.19	7.85	5.87
5	10.53	7.63	7.51	7.42
6	11.17	7.30	8.62	8.44
7	11.17	9.18	10.50	10.19
8	11.60	10.73	12.15	11.96
9	11.82	12.39	11.29	12.62
10	11.17	12.50	11.49	15.17

Concentrations of nitrate (NO_3-N) in bare soil mesocosms				
Days	Coland Loam B (mg/L)	Coland Loam A (mg/L)	Spillville Loam (mg/L)	Clarion-Storgen Loam (mg/L)
0.04 (1 hour)	21.0	20.4	20.4	20.4
0.167 (4 hours)	21.0	19.6	20.2	20.2
1	20.3	19.2	19.9	19.6
2	19.9	19.2	19.8	19.1
3	19.6	19.3	19.6	19.2
4	19.2	19.2	18.8	19.2
5	18.8	18.9	18.9	18.9
6	18.7	18.9	18.7	18.7
7	18.7	18.5	18.3	18.3
8	18.6	18.2	18.0	18.0
9	18.5	17.9	18.1	17.8
10	18.7	17.9	18.1	17.3