Integrated Landscape Management for Agricultural Production and Water Quality Midwest

Final Report for Minnesota Department of Agriculture
Project Contract #123945

Prepared by

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July 2020

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Acknowledgements

First and foremost, we would like to thank Heather Johnson (MDA), Brad Redlin (MDA) and William Fitzgerald, (MDA) for their support throughout the project. We thank Mark Coulter (UMN Technician) for his experienced help with the field research and laboratory analyses. Lastly, thanks to Doug Moody, Ivan Hovland and Nolan Hovland (UMN Researcher Assistants) for their invaluable help with field maintenance and sample collection.

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Executive Summary

In this project our team set out to quantify the individual and cumulative impacts of select in-field, edge-of-field and beyond-the-field/in-stream management practices on water quantity and water quality for a small watershed and upscale these results to watershed scale (HUC 8). To accomplish this task we designed an experiment within a small watershed at the University of Minnesota Southwest Research and Outreach Center near Lamberton, MN. Our goals were to measure individual and cumulative response of cover cropping, bioreactors, constructed wetlands and drainage ditch management on water and nutrient (nitrogen and phosphorus) loss within a small agricultural watershed. Another goal was to use data from our experimental site coupled with actual on-farm management data from farms in the Cottonwood River Watershed that are enrolled in the Minnesota Agricultural Water Quality Certification Program (MAWQCP).

This report consists of four chapters plus a chapter that contains supplementary material. The first chapter provides background and context for the experiment. It also briefly reviews the four strategies used to mitigate water and nutrients from agricultural runoff and subsurface drainage in the small watershed. The second chapter provides an extensive summary of the small watershed research. This chapter includes information on the measured source water (surface runoff and subsurface drainage) along with the individual impacts of the practices on water quantity and quality. It should be pointed out that there was additional unmonitored source water (surface runoff and subsurface drainage) that only entered the ditch system so there is a level of uncertainty in the cumulative impacts of the practices. We were unable to monitor all of the source water for every system due to limited funding. However, we can report that based on measurements of surface runoff and subsurface drainage, the practices were effective in increasing permanent (wetlands) and temporary (drainage ditches) storage of water in this landscape. We can also report that based on the flow weighted nitrate-nitrogen concentrations in the subsurface drainage and surface runoff, 15.1 mg L^{-1} and 11.2 mg L^{-1},respectively, that the concentration at the ditch outlet was 6.2 mg L^{-1}, which is the mean of the treatment and control ditch combined. For dissolved reactive phosphorus, although the individual practices showed reductions in load and concentration, overall we observed a higher concentration leaving the ditches, 103 ug L-1, than was observed in the measured surface runoff and subsurface drainage, 72.8 ug L^{-1} and 66.7 ug L^{-1}, respectively. Since we were not able to measure all sources of water entering the ditch system it is difficult to complete a water and nutrient balance due to the inherent uncertainties. What is evident is that phosphorus is being exported from the watershed and additional focus should be put on strategies for mitigating phosphorus losses. Chapter 3 focuses on bioreactor research that was conducted during 2016 and 2017 and is included in the report as additional information from the watershed. The fourth chapter covers the watershed modeling component. This chapter highlights that some practices are very effective at mitigating nutrient losses but that in some cases additional practices are required in order to alleviate unintended or unexpected nutrient loss. The final chapter contains supplemental tables and figures.

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Chapter 1 Background Information and Context of the Project

1.1 Relevance of the project

This proposed research directly addresses the MDA Impaired Water Research priority areas 1 and 2: Priority area 1, '***Address research gaps*** *identified in the 2012 Minnesota Agricultural Best Management Practice (BMP) Handbook…**for common agricultural drainage water management practices**, including one or more of the following:* ***the sediment, nitrogen and phosphorus reduction effectiveness,*** *associated biogeochemical processes,* ***resulting hydrologic impacts****, or construction design criteria'*. Priority area 2: *Evaluate and quantify using* ***field plots****, the* ***impact and effects of agricultural fertilizer*** *BMPs and/or soil health principles* ***on sub-surface drainage water quality, specifically the movement and loss of nitrates, phosphorus, and other contaminants****.*

In order to improve productivity, extensive agricultural areas in the Midwest require drainage systems consisting of subsurface drainage (tile) and open ditches. These drainage systems are known to transport particulate and dissolved phosphorus, nitrate-nitrogen, and sediment to streams and rivers (Randall et al., 1997; Carpenter et al., 1998; Gilliam et al., 1999; Addiscott et al., 2000; Smith et al., 2004). Management practices on agricultural lands have been shown to affect hydrology and water quality (Gilliam et al., 1999; Castillo et al., 2000). It is also widely accepted that in-field, edge of field and in-stream practices can be modified to improve water quality through nutrient and residue management (Dinnes et al., 2002), crop rotation (Randall et al., 1997), the use of biological filters, bioreactors (Jaynes et al., 2008; Robertson and Merkley, 2009; Woli et al., 2010), cover cropping (Feyereisen et al., 2006; Kaspar et al., 2007; Strock et al., 2004) and ditch management (Needelman et al., 2007). While these studies separately illustrate the effects of in-field, edge of field, and in-stream practices on water quality, ***there have been few quantitative studies that have investigated the*** **cumulative impacts of multiple, integrated best practices on water quantity and water quality in order to meet nutrient load reduction goals**. Consequently, decisions concerning soil and water resource management and conservation are often made that only address short-term or single goals, while ignoring the complexity of coupled terrestrial-aquatic ecosystems. As a result, low environmental benefit of management practices have usually been observed. To solve complex environmental problems in Minnesota and to meet the 20% nitrogen and phosphorus load reduction established for the Mississippi River/Gulf of Mexico by 2025 and 45% by 2040 (MRGMWNTF, 2008; MPCA, 2012) it will be necessary to apply a multidimensional approach that incorporates management and conservation of terrestrial and aquatic ecosystems.

Expected results and benefits

The use of agronomic and engineering practices are broadly implemented across the state in order to mitigate water quantity and water quality problems. This proposal establishes a site where the impact of multiple, integrated best practices on water quantity and water quality can be quantified. Specifically, we will monitor water, nitrogen (N), phosphorus (P) and sediment individually and cumulatively from in-field, edge-of-field and in-stream management practices. Research data collected in this study will be used to determine the water, N, P and sediment reduction potential of these integrated strategies and their potential for meeting Minnesota's Nutrient Reduction Strategy goals. Another outcome of the project will be a demonstrated ability to meet the dual goals of maintaining farm productivity while improving watershed conditions and water quality.

The ***goal*** of this proposal is to quantify the individual and cumulative impacts of select in-field, edge-of-field and in-stream management practices on water quantity and water quality for a small watershed and upscale these results to watershed scale (HUC 8). Specific objectives are to:

Objectives

1) Measure individual and cumulative response of cover cropping, bioreactors, constructed wetlands and drainage ditch management on water, N, P and sediment loss within a small agricultural watershed.

2) Predict flow, sediment, and nutrient export by employing a watershed scale modeling approach. Employ the calibrated and validated simulation model in concert with field monitoring data to quantify the individual and combined effects of a suite of alternative agricultural management practices in order to identify optimal combinations for improving water quality.

1.2 Introduction

Nonpoint source pollution control measures are either structural or management-oriented BMPs that are intended to minimize water quality degradation while still permitting productive use of the land. Best Management Practice effectiveness varies from site to site due to spatial and temporal variations in site-specific conditions such as soil type, topography, climate, and land use. In particular, rate and timing of fertilizer and manure applications, subsurface tile drainage on varying soil types, topography, and climatic conditions have a major impact on the pattern and magnitude of nutrient losses.

1.2.1 Cover Crops

Water and nutrients are essential inputs for profitable corn production. Previous research (Randall and Mulla, 2001, Dinnes et al., 2002, King et al., 2015) has shown subsurface tile drainage systems deliver nitrate-N and dissolved reactive phosphorus (DRP) to surface waters and thereby degrade water quality. Row crop agriculture in the Midwest is under scrutiny to reduce nutrient concentrations and loads in tile drainage. The use of cover crops and applying appropriate rates of fertilizer, especially N, for corn are potential management strategies to reduce nutrient losses in tile drainage water (Dinnes et al., 2002). Cereal rye is effective at scavenging N when it is established early and not terminated until spring. Generally, Minnesota farmers who plant cover crops either use cereal rye, or a seed blend of annuals like oat, annual rye, clover and radish. These annuals are terminated by cold temperatures and/or tillage.

1.2.2 Constructed Wetlands

Natural, restored, and constructed wetlands represent a potentially promising practice for treating agricultural drainage whether the influent consists of surface water runoff, subsurface drainage, or a combination of both. Research has demonstrated that wetlands can provide removal of suspended sediments, biological treatment of nitrogen, and storage of carbon and phosphorus (Kadlec and Knight, 1996; McCarty and Ritchie, 2002). Wetlands are very efficient at removing nitrate. They are also considered effective in removing phosphorus associated with suspended sediment; however, they are less effective in retaining phosphorus than nitrogen. Wetlands are generally considered to be phosphorus sinks due to sediment accumulation of bound inorganic and organic phosphorus.

The effectiveness of wetlands depends on several factors including: climate, temperature, hydraulic loading and residence time, wetland to watershed area ratio, type and density of vegetation, the characteristics of the microbial communities, and the distribution and characteristics (concentration) of the influent water. Optimal reduction in nutrients occurs when hydraulic residence time is long and hydraulic loading rate is low. There is considerable potential for restoration of wetlands across the drained U.S. corn belt (Crumpton, et al., 2008). One disadvantage of wetland restoration is that the wetlands often disrupt the continuity of farming practices because of their location, thus the targeted positioning of constructed wetlands to treat agricultural drainage is promising.

1.2.3 Ditches

In watersheds where artificial drainage is practiced, surface and subsurface runoff from agricultural lands is often carried by a network of open-ditches that have replaced headwater streams. These new headwaters function primarily as water and solute transport systems, with little capacity for biogeochemical processing and natural retention of dissolved or particulate materials. Open-ditch systems are typically small (3-6 ft. wide) historical natural streams that have been artificially deepened and straightened, or new channels, constructed to serve as an outlet for field drainage systems. In Minnesota more than 27,000 mi. of open ditch (Helland, J. 1999), act to transport not only water, but also nutrients, sediment, pathogens, and pesticides from agricultural fields to small streams and larger rivers. A modeling study of the effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico demonstrated that the rate of nitrogen removal (via assimilation and denitrification) in streams declined rapidly with increasing channel size (Alexander et al., 2000).

Research on open-ditches is limited. Meulemen and Beltman (1993) reported on the use of vegetated open-ditches for water quality improvement through successful nutrient reduction. Current research in Ohio suggests the potential exists for using vegetated agricultural open-ditches as best management practices (BMP's) for mitigating potential agricultural contaminants. It has been hypothesized that a compound open-ditch channel would create a linear zone of plants and soil within open-ditch geometry for enhanced denitrification potential. Initial efforts on alternative strategies for ditch management focused on improving their hydrologic functioning at low flow by constructing a two-stage ditch (Powell et al., 2007).

1.2.4 Bioreactors

There has been increased interest in developing Best Management Practices (BMPs) for mitigating the effects of subsurface drainage. Ideally, a successful BMP would mitigate the negative impact of subsurface drainage while having minimal consequences on field management and crop production. There have been several design variations of denitrifying bioreactors including, in-field denitrification walls (Jaynes et al., 2008), edge-of-field bioreactors (Woli et al., 2010) and stream bed bioreactors (Robertson and Merkley, 2009).

Field experiments were conducted at the University of Minnesota Southwest Research and Outreach Center (SWROC) in Lamberton, Minnesota to experimentally assess the impact of a novel two phase bioreactor design for removing N and P from agricultural subsurface drainage water (Strock et al., 2017). Modular bioreactors were constructed using mixed woodchips plus corn cobs for facilitating denitrification plus either crushed concrete, steel slag or limestone fragments for P sorption. Nitrate removal was tied to the hydraulic retention time in the bioreactor coupled with the addition of acetate. Longer retention time resulted in a greater removal of nutrients however, acetate improved nitrogen removal efficiency. According to the results of the one-year field experiment, Steel Slag was most efficient for P removal.

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Chapter 2

Integration of In-Field, Edge-of-Field and Beyond-the-Field/In-Stream Practices for Achieving Agricultural Productivity and Environmental Quality.

2.1 Introduction

Members of our group previously conducted field and watershed scale (greater than 133,436 ac) research to investigate the impact of changes in farm level nutrient management, field scale cropping system management, and economic costs of water quality improvement of the Cottonwood River Watershed (Strock et al., 2005). While this research yielded valuable information, it only addressed the single goal of improving water quality through in-field management practices and ignored potential improvements in water quality through edge-of-field and beyond-the-field biogeochemical processes. While there is relatively good understanding of the individual impacts of nutrient management, cover cropping, controlled drainage, ditch management, constructed wetlands and wood chip bioreactors on water quality, the cumulative impacts of multiple practices at a watershed scale are less well understood. The infrastructure in place at the University of Minnesota, Southwest Research and Outreach Center near Lamberton, MN enabled our research team to conduct sub-watershed scale experiments and simulation modeling to investigate the effect of integrated in-field, edge-of-field, and beyond-the-field practices and processes on water quality and the environment. Our site includes a paired drainage ditch system, six constructed wetlands, nine modular bioreactors and a field where we can conduct drainage system management, nutrient management and/or cover crop research (Figure 1).

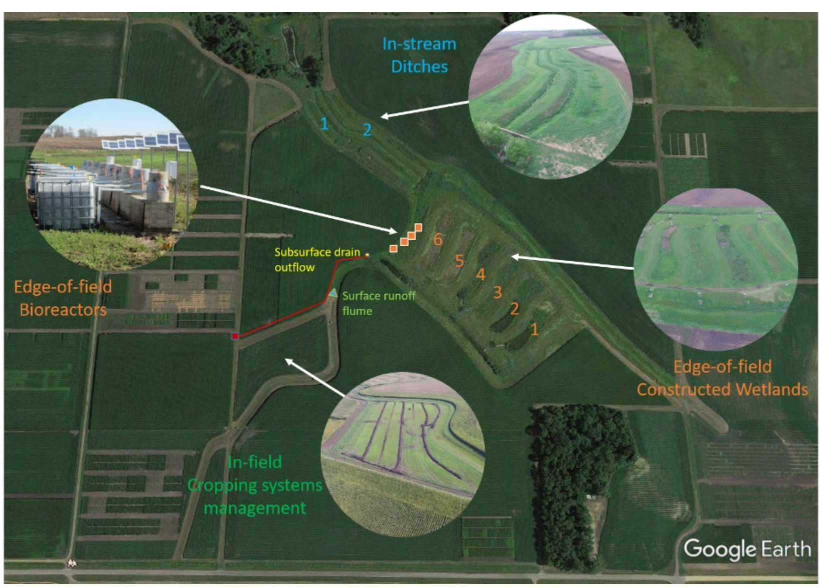

Figure 1. Integrated in-field, edge of field and in-stream infrastructure for conducting research on the cumulative impacts of multiple practices at a watershed scale.

The research described in this chapter will describe how landscape features (e.g ditches, constructed wetlands, bioreactors) and land use management (relay cropping) practices influence water quality in the Cottonwood River Major Watershed in southern Minnesota. Measured experimental results will be used to parameterize and constrain, in-field, edge-of-field, and in-stream water flux and solute transport using the Soil Water Assessment Tool (SWAT). Our methods incorporated detailed small watershed scale research designed to identify best practices for achieving improved water quality, computer simulation modeling of in-field and in-stream water flux and solute transport as related to crop, soil, and water management, and measurement of water quality characteristics. Information from this project will inform farmers and state agencies on the cumulative impacts of multiple, integrated best practices in order to meet nutrient load reduction goals.

RATIONALE AND SIGNIFICANCE

Nutrient losses to surface waters are of great concern regionally and nationally. Eutrophication, caused by inputs of nitrogen and phosphorus, is a common problem in lakes and rivers (Carpenter et al., 1998). Eutrophication causes increased growth of algae and nuisance aquatic plants that interfere with use of the water resource for fisheries, recreation, industry, agriculture, and drinking. Oxygen depletion caused by decomposition of nuisance plants causes fish kills. Eutrophication is also a factor in the loss of aquatic biodiversity. While phosphorus in water is not considered to be directly toxic to humans or animals (Amdur et al., 1991), its toxicity is indirect and caused by toxic algal blooms or anoxic conditions stimulated by phosphorus pollution. In contrast, nitrogen as nitrate in water is toxic to humans and other mammals. Nitrate in water is toxic at high concentrations and has been linked to methemoglobinemia in infants and toxic effects on livestock (Amdur et al., 1991).

Understanding the relationship between management and conservation of terrestrial and aquatic ecosystems is very important to the development and implementation of practices to improve water quality while maintaining farm productivity, profitability, and environmental quality. The greatest opportunities to provide flood control, trap sediments, recycle nutrients, create and maintain biological diversity, and sustain the biological productivity of downstream lakes and rivers begin in headwater watersheds. A combination of in-field, edge-of-field and in-stream management practices within should be coordinated to efficiently improve water quality in headwater watersheds.

The research we conducted is potentially transferable to millions of hectares of agricultural cropland throughout the United States. This represents an opportunity for adoption and implementation of management systems that consist of integrated soil and water resource management practices that will satisfy agricultural productivity and water quality objectives.

2.2 Material and Methods

2.2.1 Water Quality Analysis

Water samples were analyzed for total nitrogen, nitrate-nitrogen, ammonia, total phosphorus and dissolved reactive phosphorus concentration. All chemical analyses were completed using Lachat 8500 Flow Injection Analyzer (Hach, Loveland, CO). Samples were typically collected within 24 h of a storm event and stored at 4^o C until analysis. A well-mixed sample was filtered through a standard glass fiber filter and the filtered sample was analyzed for nitrate-nitrogen, ammonia, and dissolved reactive phosphorus. Dissolved reactive P was analyzed using colorimetric analysis by the molybdate-blue/ascorbic acid method (10-115-01-1-Q) at 880 nm. Nitrate-nitrogen was analyzed using colorimetric analysis by the cadmium reduction method (10-107-04-1-A) at 520 nm. Ammonia was analyzed using colorimetric analysis by the salicylate method (10-107-06-2-O) at 660 nm. Total P and total N were simultaneously autoclaved using a persulfate digest on unfiltered water samples. All forms of nitrogen are converted to nitrate nitrogen during the persulfate digestion and phosphorus compounds are liberated as ortho-phosphate. The digested sample is analyzed colorimetrically for TP as orthophosphate and TN as nitrate.

2.2.2 Description of research site

The field research site was located in the Cottonwood River Major Watershed. Farming is the principal segment of the economy in this watershed consisting primarily of row crop production of corn and soybeans. Land cover in the watershed consists of approximately 85% cultivated cropland, 8% grassland (including CRP land), 1% woodland, and 6% other land. The watershed landscape is characterized as having a complex mixture of gently sloping (2-6% slope) well drained loamy soils and nearly level (0-2% slope) poorly drained loamy soils formed in glacial till. Artificial drainage to remove ponded-water from flat and depressional areas is extensive. Poorly drained soils are highly productive due to an extensive network of subsurface tile drainage and open-ditches. The climate of the region is characterized as interior continental with cold winters and moderately hot summers with occasional cool periods. Average annual temperature is 7˚C (45˚F), with monthly extremes ranging from 21˚C (70˚F) in July to -9˚C (16˚F) in January. The experiment was conducted at the University of Minnesota Southwest Research and Outreach Center (SWROC) located near Lamberton, MN (lat. N: 44^{o}14’36”, lon. W: 95^{o}18’20”*)*. The long-term total annual precipitation at SWROC is 695 mm, considered adequate to grow crops without irrigation because most of it (69%) falls during the growing season from April to September.

2.2.3 Ditch System

Strock et al. (2007) described the development of a vegetated ditch research facility in the glacial till plain within the Northern Corn Belt Plain region of Minnesota. This site represents physiographic features and land use typical of southwest and south-central Minnesota. Two side-by-side, 200 m (656 ft) long, experimental vegetated ditch channels receive surface runoff and subsurface drain flow from 113 ha (280 ac). These experimental channels discharge into the Cottonwood River. This experimental site was established to identify the effectiveness of ditch management strategies to increase water storage and decrease nutrient discharge from an agricultural landscape. Water flow in the channels is seasonal, with higher flows from April through June when spring snowmelt combines with spring rainfall and seasonally high subsurface drainage flow. The contributing watershed comprises 74% cropland (row crops), 20% pasture, and 6% farmstead.

A paired watershed design was used to evaluate the impact of drainage system management on water quality and quantity. The use of paired analysis experimental design has gained in popularity as well as credibility among the scientific community particularly where classical replicated statistical designs are difficult to implement. The paired analysis technique has been successfully applied by Clausen et al., (1996) to evaluate tillage impact on surface water quality at the sub-watershed scale.

Paired analysis experimental design requires a minimum of two similar experimental units, in this case ditches. The design for this project consists of a control and treatment ditch. The control ditch serves as a check for climatic and other variations during the year. In this project, the variables of interest were water discharge, phosphorus and nitrogen. The treatment received the prescribed treatment, installation of 30 cm (12 inches) of low-grade weirs, while the control ditch remained unchanged.

The ditch system consists of five key components: water collection area, ditch channels, overflow channels, flow control structures, and flow measurement structures. Prior to construction in 2002, it was necessary to relocate four subsurface drainage lines. Three subsurface lines were rerouted and allowed to empty into the water collection area. One line was rerouted to bypass the system completely. In order to achieve the desired 0.10% channel slopes, a process of cut-and-fill was used during excavation. Soil from the inlet end of the system was removed to bring the channels to the appropriate grade while at the outlet end of the system fill was used to bring the channels up to the proper grade.

The water collection area was designed to allow mixing of surface and subsurface runoff before entering the two experimental ditch channels. The volume of the water collection area measures 1.1 m (3.75 foot) wide by 18.3 m (60 foot) long by 0.06 m (0.2 foot) deep and has a storage volume of 1.3 m^3 (45 ft^3). The collection area accumulates water from three pre-existing subsurface drainage lines and two pre-existing grassed waterways. The subsurface drainage system consists of plastic and concrete drainage tile. Approximately 40.5 ha (100 acres) of land drain by subsurface drainage and contribute to the total flow of the ditch system. Surface water runoff, directed to the collection area by the grassed waterways, contributes to the overall flow of the ditch system. Potentially 111.3 ha (275 acres) of land contribute surface runoff to the ditch system through surface runoff.

The ditch system consists of two experimental channels and two overflow/expansion channels. The ditch channels were designed for evaluating the quantity and quality of water transported through a ditch system. The purpose of the overflow channels is to divert water from the water collection area during times of excessive flow in order to prevent uncontrolled discharge of water into the experimental ditch channels. The channel bottom grade of the experimental channels and the overflow channels is 0.10%. The ditch channel geometry is trapezoidal. The bottom width of the experimental ditch channels is 1.5 m (5 foot) and 3.0 m (10 foot) for the overflow channels. The side slopes for the experimental channels is 2:1, typical of drainage ditches constructed in southern Minnesota. The side slopes of the overflow channels are 6:1 and 4:1. The maximum water depth in the experimental ditch channels is 1.2 m (4 foot) and the maximum depth in the overflow channels is 0.67 m (2.2 foot). The center berm between the two open-ditches is 0.5 m (1.5 foot) above the maximum channel depth in order to prevent mixing between the two experimental channels if the channels reach maximum capacity. Water is delivered from the collection area to the experimental ditch channels through Palmer-Bowlus flumes installed at the head of each experimental ditch channel. Water flows through 0.76 m

(2.5 foot) H-flumes at the outlets of the experimental channels. A 1.5 m (5 foot) wide by 3.0 m (10 foot) long by 0.76 m (2.5 foot) deep polyvinylchloride (PVC) approach sections come before the H-flume in each channel. Water level in the treatment ditch is regulated using PVC weirs. Each weir was manufactured with a rubber seal to assure a tight fit to prevent leakage. Weirs measure a minimum of 0.15 m (6 inches) in height. Multiple weirs were stacked to achieve the desired water level control height, 0.3 m (12 inches) in this experiment. Daily mean values of drainage volume were calculated from continuously recorded stage data using stage-discharge relationships for the H-flumes. Each H-flume was instrumented with an OTT bubble level sensor (OTT Hydromet, Germany). The sensor detects water level by measuring the pressure required to force a gas bubble from a submerged tube. An ISCO 3700C compact portable sampler (Teledyne ISCO, Inc., Lincoln, NE) was installed after each water metering H-flume. The 3700 samplers were used collect composite and discrete water samples from each experimental ditch based on base flow or storm flow hydrograph conditions. A Campbell Scientific CR1000 data logger (Campbell Scientific, Logan UT) was used to control the portable sampler and to collect the water level data. A program was developed to monitor ditch hydrograph response to storm flow. The program was designed to collect a composite daily sample during base flow conditions and discrete samples during the rising, peak and recessional components of the hydrograph. The combination of water flow measurement and nutrient concentration allows for estimation of contaminant loading to surface water. The arrangement of this drainage system will allow for monitoring and evaluation of changes in water quantity and quality as a result of ditch management. Discharge was continuously measured and water samples collected between ice-free periods in order to minimize potential damage to instrumentation. A YSI 556 Multi-Probe System (YSI Inc. Ohio) was used to simultaneously measure dissolved oxygen, pH, conductivity, temperature and oxidation-reduction potential on a weekly basis during ice-free periods.

A commonly used practice in drained agricultural landscapes is known as controlled drainage which involves the use of a variable height riser in the drainage system outlet to conserve water and nutrients in the soil profile (Strock et al., 2011). Extending the concept of controlled drainage to ditch channels involves the installation of adjustable low-grade weirs (aka risers), in a strategically arranged pattern within a drainage ditch. Adjustable low-grade weirs act as impedance structures that function like check dams within drainage ditches (Figure 2). They reduce the effective slope of the channel, thereby reducing the velocity of flowing water and create pools of water which in turn creates multiple, enhanced wetland systems within the larger drainage ditch. With the presence of weirs, hydraulic residence times (HRT) within the ditch system are longer, and thus are hypothesized to create conducive conditions for denitrification to occur. The low-grade weir in the treatment ditch is 0.30 m (12 in.) tall and is located at the ditch outlet. The low grade weir also has two 1.3 cm (0.5 in.) orifices in it located about 2.52 cm (1 in.) up from the bottom of the weir.

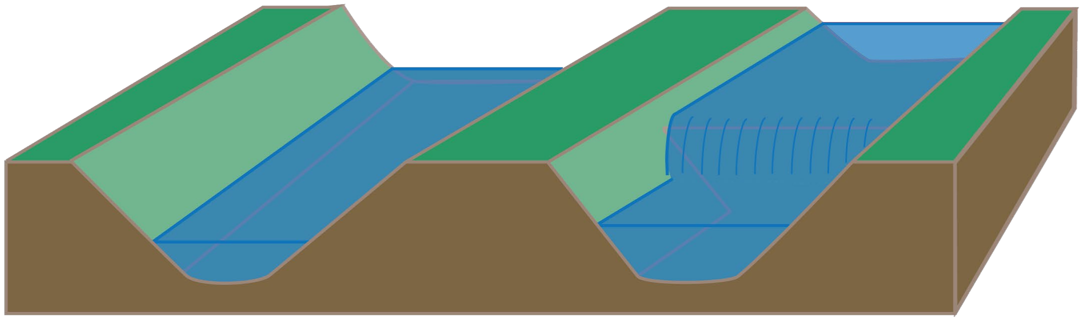

Figure 2. Illustration of ditch without weir (left) and ditch with a weir (right).

2.2.3.1 Tracer test

In order to obtain ditch flow characteristics, an impulse tracer test was conducted. We dissolved 3000 g of potassium bromide (265.9 g of Br−) in about 6 L of water for a resulting bromide concentration of about 100 mg Br L^{-1}. This bromide solution was poured into each ditch channel at the outlet of the Palmer-Bowlus flume near the head end of each ditch channel in less than 1 min to avoid disturbing the normal flow. Subsequently, samples were collected from the outlet H-flume of each ditch channel using auto-samplers beginning with 5-min intervals during the rise and peak of the hydrograph, and increased to 15 min during the fall of the hydrograph. The last water sample was collected after 24 h from pouring the bromide solution into the inlet flume of each channel. Water samples from the tracer tests were kept in a cooler at 4 °C until analyzed for bromide within 2 weeks by colorimety (Lachat QuikChem 8500 Flow Injection Analysis, Hach Co., Loveland, CO, USA) based on the QuikChem method 10–135-21–2-B. The mean tracer residence time (aka actual hydraulic residence time in hours) is the centroid of the area under residence time distribution curve and was calculated according to the method of Metcalf and Eddy (2014). The percent bromide recovery was calculated as the ratio of the mass of bromide recovered at the outlet to the mass of bromide injected into the inlet of each ditch channel. The mass of bromide recovered (in milligrams) was determined according to the method of Kadlec and Wallace (2009).

2.2.4 Constructed Wetlands

Replicated constructed wetlands were designed and constructed in fall 2007 to mitigate nutrients from agricultural runoff. The area consisted of undrained Canisteo poorly drained soil. The contributing watershed is annually planted to corn and soybean.

Replicated constructed wetland designs included pairs of surface-flow (SF) basins, vertical-flow (VF) basins, and horizontal-flow (HF) basins (Figure 3). The surface-flow wetlands were designed so that incoming water principally flowed above the ground surface, as shallow sheetflow, through a dense growth of terrestrial and emergent aquatic plants. Horizontal-flow wetlands treat water by passing it horizontally through a permeable media, consisting of a layer of coarse gravel (3.8 cm diam.) covered by a layer of fine gravel (1.9 cm diam.) and finally covered with a layer of soil planted with terrestrial and aquatic plants. Vertical flow wetlands are similar to horizontal-flow wetlands except that these systems are designed to treat water by passing it vertically through soil planted with terrestrial and aquatic plants. Each basin occupied an area equal to approximately 0.2 ha (one-half acre). Horizontal-flow wetlands treat water by passing it horizontally through a permeable media, consisting of a layer of coarse gravel (3.8 cm diam.) covered by a layer of fine gravel (1.9 cm diam.) and finally covered with a layer of soil planted with terrestrial and aquatic plants. Nitrogen and phosphorus were the main nutrients of concern from agricultural runoff.

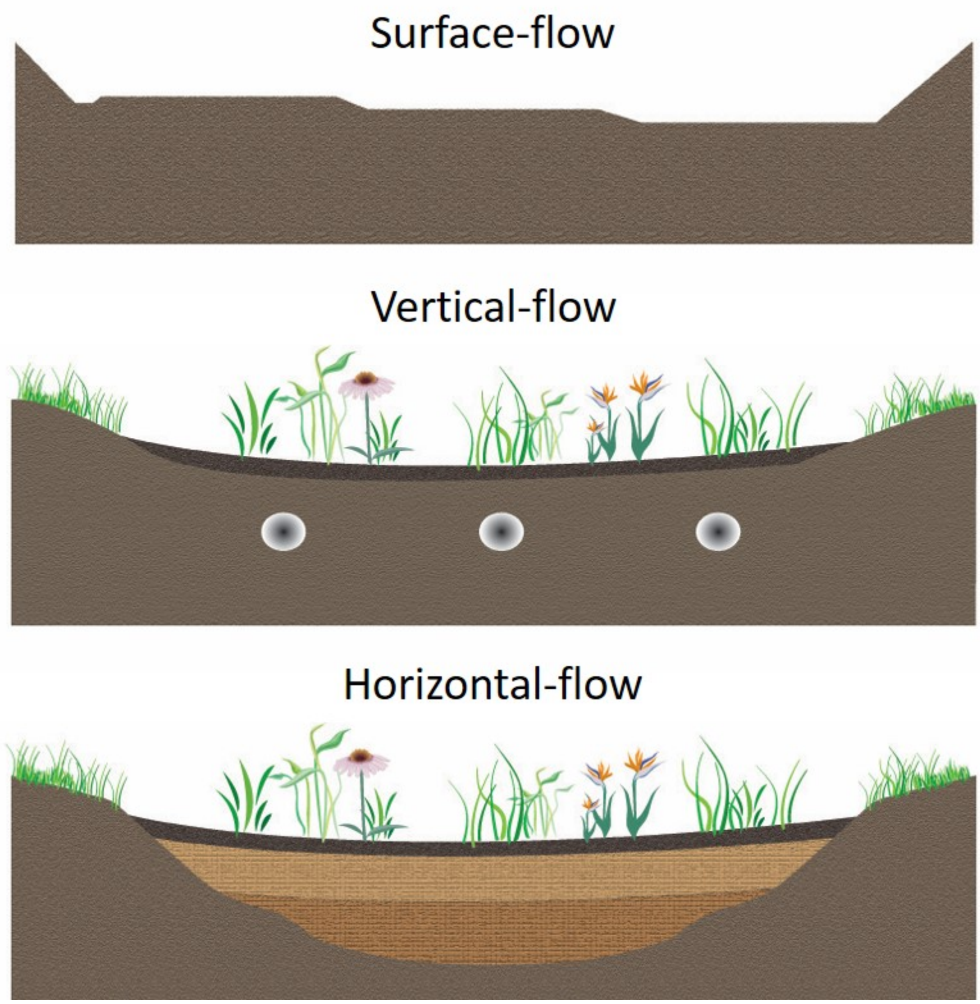

Figure 3. Illustrations of constructed wetlands at the Southwest Research and Outreach Center, near Lamberton, MN.

Each pair of SF and VF wetlands shared an equalization basin for distributing water into the individual basins through H-flumes. The HF nutrient retention basins also share an equalization basin however; water is distributed to the individual basins through two Hickenbottom inlets. A Hickenbottom inlet is a type of above ground surface drainage structure that allows water flow control which promotes sedimentation of soil and particulate materials in surface runoff before the runoff is routed through a subsurface drainage system. Water level and outflow from each basin were controlled by an in-line water level control structure. Instrument shelters located near the outlet water level control structures contain equipment for measuring water level and for water sample collection. Each shelter contains an ISCO 3700 water sampler and a CR1000 data logger used to collect and store stage height (water level) data. Each water level control structure is equipped with an INW, Inc. pressure transducer to record changes in stage height.

Constructed wetland performance and effectiveness are in part based on the temporary storage of water in the basin. The previously described water level control structures located at the outlets of the systems enabled the implementation of basin outlet water level management in order to increase HRT in one of the constructed wetland cells. Each constructed wetland pair was managed with a v-notch weir in order to quantify wetland discharge. In addition, one of the basins was randomly chosen to operate with one additional weir board below the v-notch weir in order to raise the outlet elevation of the wetland (Figure 4).

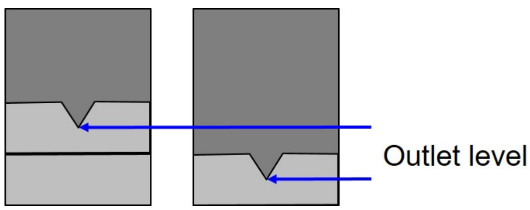

Figure 4. Constructed wetland outlet weir elevation management for the control and treatment wetland pair.

The constructed wetland site received a combination of surface and subsurface agricultural drainage runoff water. An H-flume and accompanying wing-walls for measuring snowmelt runoff and surface water runoff from a 28.7 ha (71 acre) contributing watershed were installed in a grassed waterway leading to the site in late autumn 2010. In March 2011, the flume was equipped with a data logger, OTT bubble sensor and ISCO 3700 sampler. A water level control structure was installed at the end of a subsurface drainage system that collected water from a 46.1 ha (114 acre) contributing area. The water level control structure was instrumented in the same manner as the surface runoff H-flume.

A combination of grab and storm activated discrete samples were collected for each constructed wetland. Water outflow from the basins was monitored in order to quantify nutrient, and hydrologic budgets for each of the three wetland types. Water samples were analyzed at the University of Minnesota SWROC analytical lab for total nitrogen, nitrate-nitrogen, ammonium-nitrogen, total phosphorus, and dissolved reactive phosphorus as previously described.

2.2.5 Modular Bioreactors

The top of a 1.2 m by 1.2 m by 1.2 m reinforced tank was cut off and then the tank/bioreactor constructed with three layers of materials. The bottom, non-reactive layer (outlet) consisted of 0.2 m (0.67 ft.) of lava rock plus a sheet of Brotex cut to fit the tank. The laval rock/Brotex layer was to aid in controlling hydraulic residence time (HRT) and to increase the surface area where additional biofilms could colonize. The lava rock was purchased from a local retailer. The denitrification layer consisted of two 0.2 m (0.67 ft.) thick layers of carbon media. The first layer of denitrification media consisted of mixed species woodchips encased in the honeycomb shaped geotextile cellular containment material. The second layer of denitrification media consisted of coarsely ground corn cobs, also encased in the honeycomb shaped geotextile cellular containment material. The purpose of encasing the denitrifying media in the honeycomb shaped geotextile material was to create multiple "treatment columns" within each bioreactor in order to maximize vertical flow and minimize preferential flow through the bioreactor. The mixed species wood chips were obtained from a supplier near Burnsville, MN. The coarsely ground corn cobs were obtained from the University of Minnesota – Morris. During 2016 and 2017, the top layer of the modular bioreactor consisted of phosphorus sorbing material – steel slag, crushed concrete or limestone layered 0.15 m (0.5 ft.) thick over the denitrification media. The steel slag was obtained from a supplier near Woodbury, MN. Crushed concrete was obtained from a supplier near Lind, MN and the limestone was obtained from a supplier near Shakopee, MN. In this project, potassium acetate (CH_3CO_2K) was used as the external carbon source. The concentration was based on reducing nitrate concentration at 20mg/L for a subsurface drain flow rate of 9.0 gpm in the 2"-PVC water mainline; the target stochiometric ratio of C/N was 0.84 as reported by Lew et al. (2012). The target flow rate of subsurface drainage water delivered to each bioreactor was 4 L per minute (approximately 1 gpm). Injection rates of acetate varied over the course of the experiment. The experiment consisted of three replications of each of the three P sorbing materials for a total of nine individual experimental units.

During 2018 and 2019 the phosphorus sorbing materials were eliminated from the design and a combination of carbon dosing and heat treatments were implemented. Three new treatments were implemented for the bioreactors in 2018. The treatments included woodchips plus corn cobs (WC), woodchips, corn cobs plus acetate (WCA) and woodchips, corn cobs, acetate plus heat (WCAH). The intent of the acetate and heat treatments were to stimulate denitrification during cold periods in early spring and autumn. In addition, thermocouples for measuring the water temperature profiles in the bioreactors were added (top, middle and bottom). This experiment consisted of three replications of each of the three treatments for a total of nine individual experimental units. Heat was supplied to the bioreactors through 15 m (50 ft) of self-regulating water resistant heat cable. The heat cable supplied 5W/ft. and was installed between the woodchip and corncob layers of the heated bioreactor. The heat cable was positioned on top of the woodchip layer in a concentric circular pattern. Power to heat the cables was supplied by a propane fueled generator. Depending on the availability of water, the desired heat treatment periods were March through May and October through November.

Flow measuring devices and water sampling equipment were installed at the experimental site in order to characterize flow rate and volume and to obtain water samples for chemical characterization. Subsurface drainage was measured in an AgriDrain (Adair, IA) flow control structure using a pressure transducer connected to a CR1000 datalogger. The flow control structure was used to divert water to the modular bioreactors. The outlet elevation of the flow control structure was managed at the highest level possible which created a column of water on the upstream side of the flow control structure. A single 5 cm dia. PVC pipe was installed in the bottom, up-stream side of the flow control structure which then diverted subsurface drainage water through an in-line PVC debris Y-filter to the modular bioreactors.

The single PVC pipe from the flow control structure was split into three separate distribution pipes each fitted with a paddlewheel flowmeter which was connected to a CR1000 datalogger. Each of the three distribution pipes routed water to a block of three modular bioreactors. Flow to a block of three bioreactors was split again into three separate distribution pipes and routed to individual bioreactors in which flow was controlled with a small, PVC ball valve. Flow into the top of the bioreactor was delivered through a section of plastic gutter with holes drilled into the bottom in order to attempt to distribute water as evenly as possible across the surface of the bioreactor.

The outlet of the bioreactor consisted of an adjustable standpipe which was used to raise or lower the elevation of water in the bioreactor. The stand pipe was also used to divert bioreactor effluent to a flow gauge and to collect water samples for chemical characterization. A tipping bucket flow gauge with approximately 4 L capacity per tip was used to measure bioreactor discharge volume and rate. Tipping buckets were made from stainless steel. The volume of each tip was calibrated in the laboratory at a series of known flow rates, ranging from 1.2 to 18.9 liters per minute. The calibration process involved applying water to the tipping bucket at a known flow rate for a period of at least 105 tips. During this period, a CR1000 datalogger was used to record the time when the calibration started and stopped as well as the exact time for each tip. Tips were counted using magnetic switches located on the tipping bucket. Each modular bioreactor had its own dedicated tipping bucket. Fixed-time interval samples were collected with automatic water samplers.

An acetate supply system was constructed which consisted of a 1,136 L black plastic tank, a peristaltic pump, CR1000 datalogger and solar panel. Potassium acetate was dissolved in water at a predetermined concentration and then injected into the PVC pipe carrying subsurface drainage water to the bioreactors. The subsurface drain flow rate was used to determine the pumping rate at which acetate was added to the bioreactors.

2.2.6 Cropping Systems

The cropping systems field was previously in an undrained condition before 2017. During 2017, six management zones were created each approximately, 0.13 ha (0.32 ac). Corrugated drain pipe 10 cm in dia. (4 in.) were installed to a depth of 1.0 m (39 in.) in each management zone to simulate a 23 m (75 ft.) drain spacing. Each plot was isolated from adjacent plots by 12 mil plastic installed to a depth of 1.8 m (6 ft.). An inline water level control structure fitted with a v-notch weir was installed at the outlet of each plot for monitoring drain flow. A pressure transducer was connected to a CR1000 data logger for continuous monitoring of water level and discharge. Grab samples were obtained for chemical analysis manually during periods when water was flowing from the system.

Opportunities for cover crops are often limited in temperate cropping systems that are dominated by corn (*Zea mays*) or soybeans (*Glycine max (L.) Merr.*), as both these crops leave only a short growing period following harvest with limited degree-days and possible lack of soil moisture. Winter Camelina (*Camelina sativa L.*) inclusion in such rotations provides a “niche” opportunity to include winter hardy cover crops into cropping systems after harvest. There are two main requirements for establishing a profitable multiple cropping sequence: i) adequate time is available for production of a second crop and ii) sufficient water is available to produce two crops, whether from stored soil moisture, precipitation, or irrigation. Winter Camelina, an industrial oilseed, has been identified as an excellent crop for relay-cropping in the North Central U.S. (Gesch et al., 2014).

This experiment was designed as a randomized block design with a split-plot arrangement and three blocks and two treatments. The main plot was 1) winter Camelina (cv. Joelle) seeded in the autumn and 2) plots without Camelina only with corn or soybean residue ('fallow'). The split plot consisted of dividing each plot into corn and soybean so each crop was present every year in every plot. In this experiment, corn and soybean were seeded into a growing winter Camelina stand. This cropping sequence is known as relay cropping. Relay-cropping is defined as a method of multiple cropping, where a crop is planted into an already established crop whereby the life cycles of the two crops overlap each other. Winter Camelina was planted in three of the six plots in autumn after main crop harvest using an interseeder (Interseeder Technologies, LLC; Woodward, PA). Three rows of Camelina were drilled with a spacing of 18 cm (7 in.) between rows. The seeding rate was 9 kg ha^{-1} (8 lb ac^{-1}) at a depth of approximately 13 mm. Camelina was fertilized at a rate of 67 kg ha^{-1} (60 lb N ac^{-1}) in spring by broadcasting urea prior to a 10 mm rain event. Corn and soybean were planted in each of the six plots in order to model a corn-soybean production system with and without Camelina under the prevailing weather conditions each year. During 2018 and 2019, Corn and soybean were planted in vacant rows between the Camelina at a 76 cm (30 in.) spacing when the Camelina reached full flower growth stage. The remaining three management zones acted as a control. Camelina was harvested manually from 1 m by 1 m quadrats and corn and soybean grain were mechanically harvested with a plot combine. Soil water content was collected on a weekly basis starting in 2019 using a PR2/6 soil moisture profile probe (Delta-T Devices, Houston, TX). Access tubes were installed in each crop sequence of every management zone for a total of 12 measuring locations. Access tubes were installed with a hydraulic probe.

2.2.7 Statistical analysis

The statistical tests for the following experiments were run on a JMP 15.0 Pro platform. The alpha value level of confidence was set at 0.10 for field experiment. The parameters tested for the following experiments include flow volume, nitrate load, flow-weighted nitrate concentration, dissolved phosphorus load, and flow-weighted dissolved phosphorus concentration.

<u>Bioreactors</u>

The statistical design for this system is based on the randomized complete block (RCBD) in which the treatments include the control, the acetate addition, and the acetate plus heat. In this case, the blocking system gives some control over the influence of the flow volume on each treatment; this is another form of two-way analysis without interaction. Each of the three blocks contained one of the three treatments and each block represented the flow volume that occurred over the experiment period. Each treatment was replicated three times. Each set of annual data (2018-2019) was tested separately and then a combined two-year data was also tested.

Ditches and Wetlands

All sites on this aspect of the experiment were set in pairs. The t-test in the broad context of One-Way Analysis was selected to find any significant difference on any tested parameters. JMP t-test assumes unequal variances and uses the difference of the means to check for statistical difference between each pair of value set. All p-value reported for the means difference refers to lower tailed test (Prob < t).

2.3 Results and Discussion

2.3.1 Precipitation

Water is one of the most common substances on earth and is the principle driving force behind water transport, transformation and storage in terrestrial and aquatic environments that impact water quantity and quality. Overall, the study period was characterized by above average precipitation, especially during the growing season. Precipitation data show that 2017 was the driest year in the study with annual precipitation of 763 mm (30.0 in.). Although 2017 was the driest of the three years, it had approximately 68 mm (2.7 in.) more in total annual precipitation than the long-term mean (695 mm, 1994 to 2019) at the SWROC near Lamberton, MN. Precipitation over the potential growing season (April to October) was 27 mm (1.1 in.) more than the historic average of 551 mm (21.7 in.). During 2017, June and September precipitation was below the long-term mean (Figure 5). Precipitation in 2018 was 977 mm (38.5 in.), which was the next wettest year in the study, while 2019 had total annual precipitation, just over 1010 mm (39.8 in.). Precipitation in the growing seasons of 2018 and 2019 were similar, with 763 and 715 mm (30.1 and 28.2 in.), which were 27.8% and 22.9% above the long-term growing season mean, respectively. Aside from a brief dry period in April, it was apparent from monthly precipitation that 2018 was a wet year (Figure 5). Monthly precipitation was above the long-term average for eight of the 12 months during 2019. During 2019, monthly precipitation was high for the three-month period, February through April. This contributed to surface runoff and subsurface drainage that was observed early in 2019.

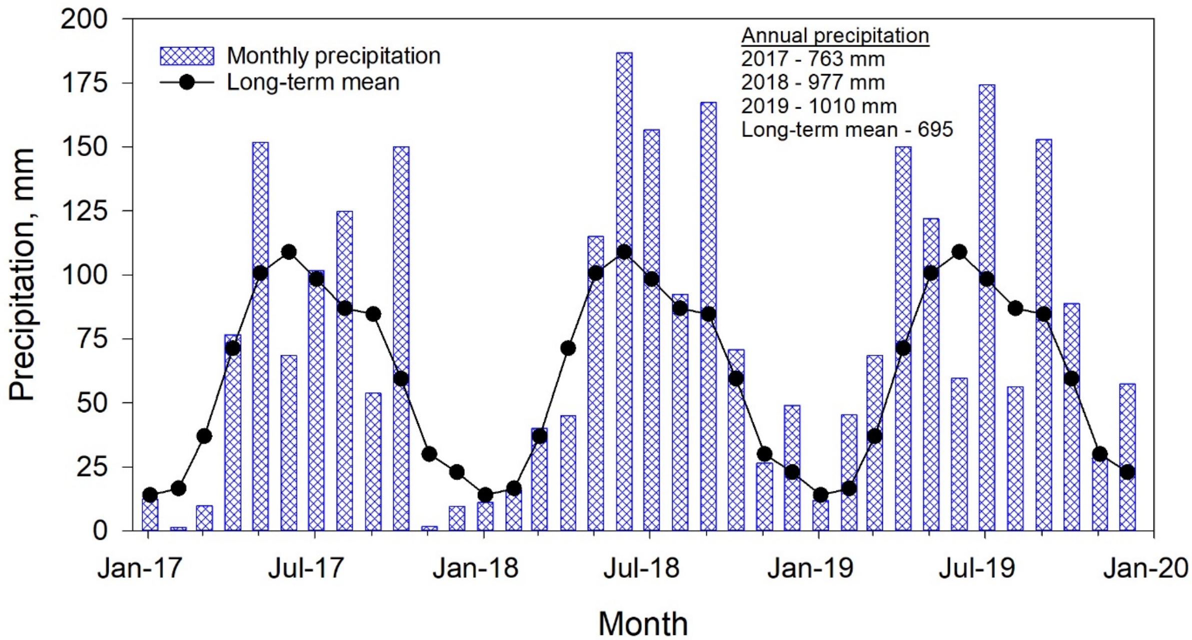

Figure 5. Three-year monthly precipitation for the period 2017-2019 near Lamberton, MN.

2.3.2 Surface runoff and subsurface drainage

Surface runoff can be categorized as either snowmelt runoff or rainfall runoff. Snow melt runoff can occur when temperatures warm rapidly during the spring thaw or when a rain on snow or rain on frozen ground event occurs. Rainfall runoff occurs when high intensity rain occurs or when the surface storage capacity of the soil becomes full to the point of saturation at which point runoff occurs. Soils in southwest Minnesota are prone to rainfall runoff in spring or early summer before the plant canopy closes. Water from surface runoff and subsurface drain locations entered the constructed wetland site and then the ditch site. A portion of the subsurface drain flow was diverted to the bioreactor site and from the bioreactor site to the ditch site. In addition to the surface run off and subsurface drainage that was monitored entering the constructed wetlands and bioreactors, there were also three other unmonitored sources of water that supply water to the ditches (Figure 6). Due to budget constraints, it was not possible to monitor these water sources for flow volume or water quality parameters. There were two unmonitored grassed waterways one of which received discharge from all six constructed wetland cells. Water from these two waterways was combined in the mixing channel at the head of the ditch system. In addition, there was a site where three unmonitored subsurface drains discharge in a common location (Figure 6). This water was also combined with the runoff water in the mixing channel at the head of the ditch system.

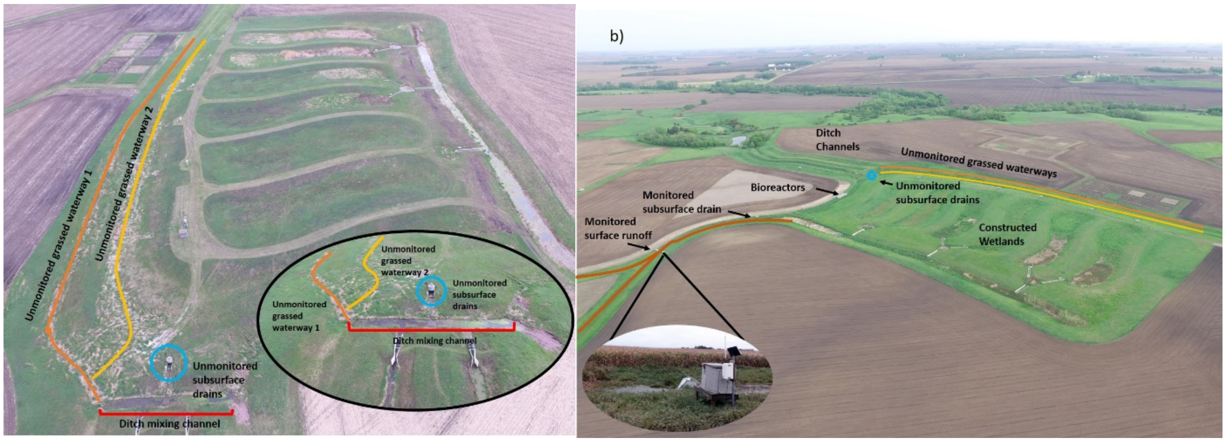

Figure 6. a) Unmonitored and b) monitored source water entering ditch mixing area before redistribution into experimental ditch channels.

Runoff was observed during all three years of data collection. The greatest annual cumulative surface runoff volume, 31,610 m^3, was recorded during 2018. Surface runoff occurred in every month between April and October during 2018 and 2019 (Table 1). As noted previously, precipitation during these two years was above average. The distribution of runoff was variable from year to year. The greatest monthly total runoff volume occurred during May 2017 which accounted for 91% of the runoff that year. In 2018, the maximum monthly flow volume occurred in September and in 2019 during April. The number of runoff events per year were similar in 2018 and 2019, 17 and 16, respectively.

April and May is a period when the soil surface is relatively bare depending on the amount of residue leftover from primary and secondary tillage, if tillage is part of field management. No residue measurements were made on fields in the watershed. In addition, normally annual row crops like corn and soybean are seedlings at this time with very small leaf areas leaving soil exposed to rain drop impacts. Runoff during late summer and early fall often results when the soil surface is wet in part due to a reduction in water uptake by plants as they senesce and cool temperatures that reduce evaporation. Under these conditions frequent or high intensity rainfall events can produce surface runoff.

Observed surface runoff was variable between years and within years during this project. Due to automated sampler problems during 2017 no water samples were collected during April and June. Annual flow weighted mean nitrate-nitrogen concentration in surface runoff was the same during 2018 and 2019 at 8.4 mg L^{-1}. In contrast, the annual nitrate-nitrogen load in 2018 was nearly double that load observed in 2019 (Table 1). This was primarily due to differences in flow volume measured between the two years. Additional factors that could influence N and P in surface runoff include nutrient fertilizer management, mean surface soil temperature, and microbial activity. Soil temperatures and microbial activity were not quantified. Nutrient management is similar from year to year and not expected to affect nutrient concentrations. The volume of surface runoff was 1.8 times greater in 2018 than 2019. Annual flow weighted mean ortho-phosphorus concentration in surface runoff was 94.3 ug L^{-1} in 2018 and 76.5 ug L^{-1} in 2019. Annual ortho-phosphorus loads were nearly 10 times greater in 2018 compared to 2019 (Table 1).

Table 1. Observed surface runoff between 2017 and 2019 at research site in southwest Minnesota, USA.

Month	Number of events	Discharge Volume, m^3	NO3- Load, kg	FWC NO3-, mg/L	Ortho-P Load, g	FWC Ortho-P, ug/L
2017						
Apr		308				
May	4	18,941	320.3	16.9	903.4	47.7
Jun		1,551				
Total/Mean	4	20,800	320.3	16.9	903.4	47.7
2018						
Apr	3	6,041	39.7	6.6	388.4	64.3
May	1	1,768	7.7	4.4	144.5	81.7
Jun	5	8,685	46.5	5.4	1052.2	121.2
Jul	4	3,673	25.8	7.0	374.9	102.1
Aug	1	205	0.9	4.6	14.4	70.4
Sep	1	9,450	128.4	13.6	868.1	91.9
Oct	2	1,789	15.3	8.5	137.8	77.1
Total/Mean	17	31,610	264.2	8.4	2980.3	94.3
2019						
Apr	4	8,676	59.0	6.8	829.2	95.6
May	3	1,388	15.0	10.8	76.3	54.9
Jun	1	4	0.01	3.0	0.20	47.0
Jul	3	2,793	31.2	11.2	171.0	61.2
Aug	1	1	0.01	8.7	0.1	64.8
Sep	2	3,643	21.3	5.9	506.5	139.0
Oct	2	714	1.5	12.4	52.0	72.8
Total/Mean	16	17,219	135.4	8.4	272.5	76.5
3-year total/mean	37	69,629	719.9	11.2	4,156.2	72.8

Based on annual observations over 20 years, on average, subsurface drainage begins between the middle of March and the beginning of April, depending on air temperature, the presence or absence of snow and frost depth. During a normal year, drain flow ends in mid- to late-July. During 2018 subsurface drainage was observed between June and November and during 2019 between May and November (Table 2). Drain flow in 2018 and 2019 was 3.3 and 4.1 times greater than drain flow in 2017. Drain flow in 2019 was 1.3 times great than 2018. As noted previously, precipitation during these two years was above average. The greatest annual volume of subsurface drainage, 115,042 m^3, was recorded during 2019.

Table 2. Observed subsurface drainage between 2017 and 2019 at research site in southwest Minnesota, USA.

Month	Flow volume, m^3	NO3- Load, kg	FWC NO3-, mg/L	Ortho-P Load, g	FWC Ortho-P, ug/L
2017					
Apr	3,054	54.0	17.2	41.6	13.8
May	19,813	358.0	18.3	450.8	23.6
Jun	5,085	91.9	18.2	427.4	55.0
Jul	-	0.0	17.6	0.0	91.3
Total/Mean	27,951	503.9	18.0	919.9	34.9
2018					
Jun	24,568	341.7	13.9	2,061.7	83.9
Jul	23,933	380.6	15.9	2,018.8	84.4
Aug	3,529	51.7	14.7	341.6	96.8
Sep	13,168	158.3	12.0	1,624.9	123.4
Oct	25,195	367.7	14.6	2,385.2	94.6
Nov	898	12.9	14.3	85.0	94.6
Total/Mean	91,292	1,312.8	14.4	8,517.3	93.3
2019					
Apr	-	-	-	-	-
May	20,087	284.9	14.2	857.6	42.7
Jun	19,620	303.3	15.5	1,333.0	67.9
Jul	30,731	430.2	14.0	2,178.3	70.9
Aug	184	1.4	7.8	24.3	132.0
Sep	13,669	140.8	10.3	1,114.4	81.5
Oct	27,320	281.2	10.3	2,439.4	89.3
Nov	3,412	37.8	11.1	329.0	96.4
Total/Mean	115,024	1,479.6	12.9	8,276.1	72.0
3-year total/mean	234,267	3,296.3	15.1	17,713.3	66.7

For common months during 2018 and 2019 when data were collected, which included June through November, drain flow was numerically smaller in 2018 compared with 2019 at 91,291 m^3 and 94,936 m^3, respectively. In contrast, the flow-weighted nitrate-nitrogen concentration was numerically greater during 2018 at 14.2 mg L^{-1} compared with 11.5 mg L^{-1} in 2019. Consequently, the nitrate-nitrogen load was greater in 2018 than 2019, 1,313 kg compared to 1,195 kg, respectively. Similar to nitrate-nitrogen, the flow-weighted ortho-phosphorus concentration was numerically greater during 2018 at 96.3 ug L^{-1} compared with 89.7 ug L^{-1} in 2019. Consequently, the ortho-phosphorus load was greater in 2018 than 2019, 8,517 g compared to 7,418 g, respectively.

2.3.3 Ditches

Overall ditch management was quite effective in mitigating discharge and nutrient loads from runoff and drainage water (Table 3). The comparison between the control and the treatment ditch, that contained the low-grade weir, resulted in a three-year average reduction in discharge of 57%. Likewise, on average, N was reduced by 67%. The three-year mean reduction in P was 27%. This value was strongly influenced by P loss in 2017. During October 2017, a pulse of P was detected at the outlet of the treatment ditch. There was no reason to believe that this flush of P was an outlier. The most likely plausible explanation was that P was released from dead plant material during a high flow event. No pulse of P was measured from the control channel. It is possible that since this was a short duration event that the sampling intensity was not precise enough to capture a pulse of P in the control channel.

Table 3. Summary of ditch annual load percent change for discharge, nitrate-nitrogen and dissolved reactive phosphorus between 2017 and 2019 near Lamberton, MN.

	Cumulative Discharge	Nitrate-N load	Dissolved reactive phosphorus load
		%	
2017	58	76	-47
2018	61	65	67
2019	53	61	61
3-yr mean	**57**	**67**	**27**

In order to obtain ditch flow characteristics, an impulse tracer test was conducted May 20, 2020. Tracer recovery near 100% is important for validating the conservancy of the tracer used (Kadlec and Wallace, 2009; Metcalf and Eddy, 2014). In our tracer experiments, we calculated a narrow range of bromide mass recovery from 75.0% to 75.3%. The average bromide recovery of 75% reveals that approximately 25% of the bromide was retarded in the ditch channels. One possible explanation for the bromide retardation under field conditions could be sorption to biofilm, bacteria and/or sediment organic matter. Another possible explanation could be bromide entrapment into the interior pores of ditch sediments following tracer testing. Another reason could be entrapment of bromide in zones of stagnant water in the treatment channel. The mean tracer residence times ranged from approximately 1.0 to 5.0 h. However, the low bromide recoveries suggest retardation of bromide in the channels. When retardation is present in a tracer test, mean tracer residence times will be overestimated.

Flow through the ditches is seasonal and generally occurs between March and September. Snow melt generally occurs during early- to mid-March in most years and subsurface drainage systems generally begin to flow from mid- to late-March in most years in southwest Minnesota. Rapid warming and/or rain on frozen ground generally contribute to surface runoff events. Subsurface drainage systems generally begin to flow as soon as the frost is out of the ground. Flow data from the ditch systems shows that ditch flow was observed between March and November (Figure 7). However, data from 2017 was two orders of magnitude smaller than in 2019.

On an annual basis, flow was 1,014 m^3 for the treatment channel compared to 2,401 m^3 for the control channel during 2017 resulting in an overall reduction in discharge of 58% for the treatment channel compared to the control channel (Table 4). Data from 2017 also showed that discharge was attenuated in the treatment ditch that contained the low-grade weir in every month except October (Figure S1). The low grade weir could partition water into storage, seepage to shallow groundwater or seepage from shallow groundwater, uptake into ditch vegetation and/or loss to evaporation. Additional data collection to investigate ditch-shallow ground water interactions took place during the summer of 2020. That results were not available at the time this report was prepared. During 2017, maximum flow occurred during May for the control ditch and October for the treatment ditch which also corresponded with periods of above average monthly precipitation (Figure 5). Flow was relatively stable during June through October for the control ditch during 2017 whereas flow in the treatment ditch incrementally decreased to a low point in September.

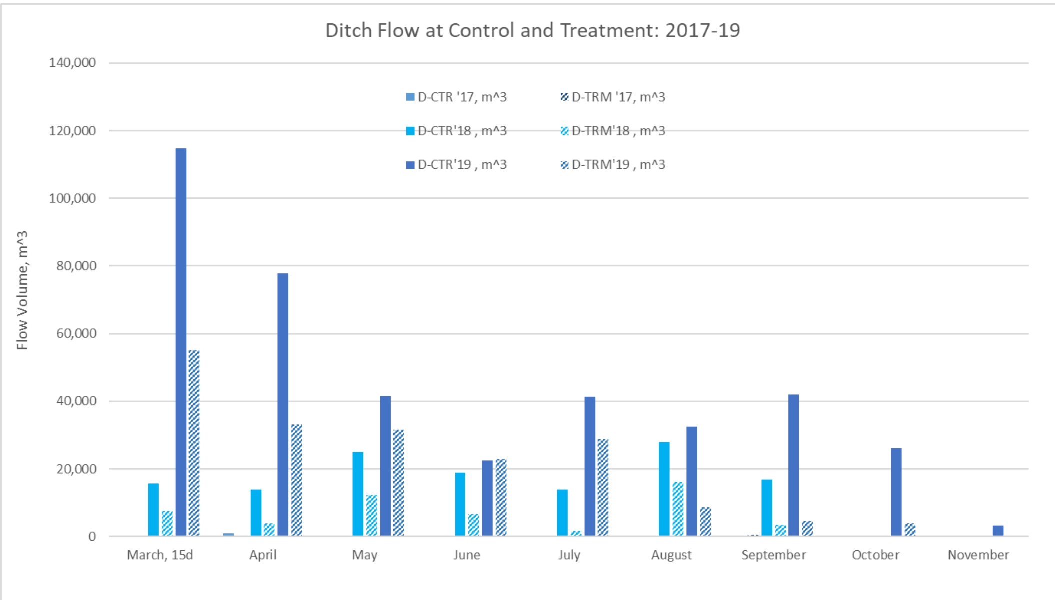

Figure 7. Three year monthly ditch discharge volume for the period 2017-2019 near Lamberton, MN.

Table 4. Summary of annual ditch discharge, load and flow weighted concentration nitrogen and dissolved reactive phosphorus between 2017 and 2019 near Lamberton, MN.

			Nitrate-N				Dissolved reactive phosphorus			
	Cumulative Discharge		Load		Flow Weighted Concentration		Load		Flow Weighted Concentration	
	m^3		kg		mg L^{-1}		g		ug L^{-1}g	
	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control
2017	1,014	2,401	6.3	25.8	4.5	10.8	286	194	176	90
2018	51,577	132,106	354	997	2.7	7.6	5,642	17,041	43	129
2019	189,249	401,979	958	2,451	5.0	6.4	15,327	39,733	78	96
mean	**80,613**	**178,829**	**439**	**1,158**	**4.1**	**8.3**	**7,085**	**18,989**	**99**	**105**

Although 2018 and 2019 were both extremely wet years, the distribution of precipitation affected discharge from the ditch channels. During 2018, above average precipitation occurred mainly during the growing season between May and September whereas in 2019, above average precipitation was more evenly distributed across the entire calendar year. There was 33 mm (1.3 in.) more precipitation during 2019 compared to 2018. The influence of growing plants on surface runoff and drain flow can be significant depending on the frequency, intensity and duration of precipitation. Ditch discharge was relatively constant and consistent during 2018 (Figure S4). On an annual basis, flow was 51,577 m^3 for the treatment channel compared to 132,106 m^3 for the control channel during 2018 resulting in an overall reduction in discharge of 61% for the treatment channel compared to the control channel (Table 4). The maximum amount of flow from both channels occurred during August. Consistently wetter conditions during 2019 resulted in greater flows from both the treatment and control channels during 2019. During 2019, above average temperatures in late February and early March resulted in rapid snow melt that caused surface runoff. Flow recorded from the ditch channels in March was dominated by multiple surface runoff events. Flow through the control channel, no treatment, was consistently higher during every month except June when the discharge observed was nearly equal. The impact of slow drawdown of water from behind the low-grade weirs between precipitation events was apparent in August through October (Figure S4). Additional storage capacity behind the low-grade weirs resulted in reduced flow from the channel compared to the unmanaged control.

Nutrient cycling in flowing systems is conceptualized and modeled as a “spiral” to describe the cycling of nutrients as they are assimilated from the water column into benthic biomass, temporarily retained, and mineralized back into the water column (Webster, 1975; Newbold et al., 1981; Mulholland et al., 2002). The downstream distance which a nutrient molecule moves, while being cycled through the various biological and physical compartments, is considered the “spiral length”. Short spiral lengths are expected where rapid uptake and release occurred or hydraulic retention of the reach is long. We would expect newly formed ditches or stream channels to have long nutrient spiral lengths because of the lack of biological communities and straight channel morphology. Conversely, mature ditches and streams should have shorter spiral lengths because of a more complex channel morphology, heterogenous water flow paths, and greater biologic production and diversity.

Over the three-year study period, 2017 through 2019, at no time was the monthly nitrate-N load from the treatment channel greater than the control channel. On an annual basis, nitrate-N load was 6.3 kg for the treatment channel compared to 26 kg for the control channel during 2017 resulting in an overall reduction in discharge of 76% for the treatment channel compared to the control channel (Table 4). Data from 2017 also showed that nitrate-N load was attenuated in the treatment ditch that contained the low-grade weir in every month (Figure S2). Nitrate-N load peaked for both channels in May 2017. Nitrate-N load gradually declined for both control and treatment ditches although the rate of decline was greater for the treatment ditch. At first glance, it might be tempting to assume that the reduction in nitrate-N is only due to the differences in flow between the control and the treatment channels. If one were to assume that the concentrations of nitrate-N were the same in both channels, it would make sense that the treatment ditch would result on less nitrate-N load than the control because the flow was 58% lower. However, our data indicated that the annual flow weight nitrate-N concentration from the control channel was 10.8 mg L^{-1} whereas it was 4.5 mg L^{-1} from the treatment channel during 2017 (Table 4). The 58% reduction in nitrate-N concentration indicated that the increased residence time in the treatment channel coupled with a larger wetted perimeter and deeper channel behind the low-grade weirs likely resulted in denitrification. Biological denitrification is expected to be the main driver of nitrate-N reduction in the treatment channel although uptake into emergent vegetation also likely occurred as air temperatures increased and vegetation matured. Another possible explanation for nitrate-N differences would be that the treatment channel was influenced by shallow groundwater – ditch channel interactions in which the ditch channel was gaining water from shallow groundwater that was lower in nitrate-N than the drainage water in the ditch resulting in a dilution effect. The impulse tracer test conducted in May 2020 occurred during a period of relatively high flow. A second test was planned to be repeated during a period of relatively low flow during 2020. Data were not available at the time this report was prepared.

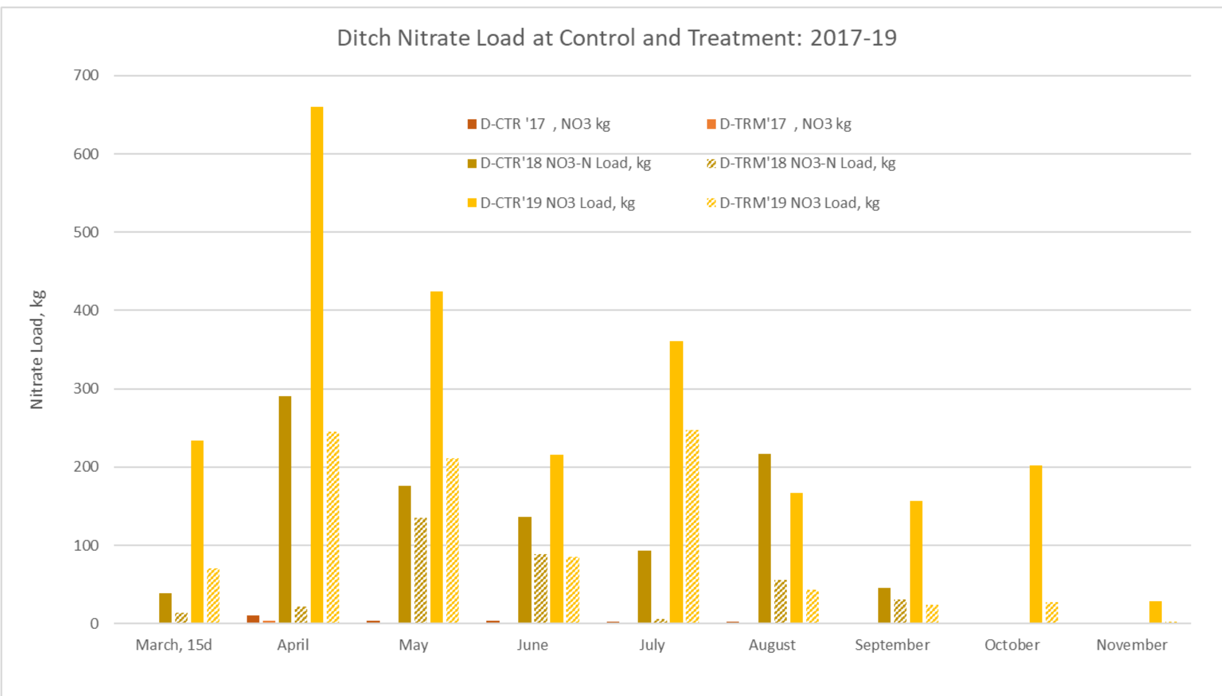

Figure 8. Monthly nitrate-N load from control and treatment ditches between 2017-2019 near Lamberton, MN.

Increased monthly and annual precipitation resulted in greater nitrate-N loads during 2018 compared to 2017. On an annual basis, nitrate-N load was 354 kg for the treatment channel compared to 997 kg for the control channel during 2018 resulting in an overall reduction in discharge of 65% for the treatment channel compared to the control channel. Data from 2018 indicated that the annual flow weighted nitrate-N concentration from the control channel was 7.6 mg L^{-1} whereas it was 2.7 mg L^{-1} from the treatment channel (Table 4). This is a 64% reduction in flow weighted nitrate-N concentration from the control to the treatment channel. It is assumed that similar processes were involved in nitrate-N load reductions in 2018 as were in 2017. The maximum monthly nitrate-N load occurred during April 2018 in the control channel. Although monthly discharge was similar for both channels in March and April 2018, nitrate-N load was about 10 times greater in April at nearly 300 kg compared to March. This is likely due to contributions of nitrate-N from subsurface drainage. Although not specifically measured in 2018, based on historical observation, subsurface drain flow increases from March to April. The treatment channel maximum monthly load was observed in May, however, the load was less than that observed from the control channel for the month (Figure 8). In general, nitrate-N load asymptotically declined for both channels beginning in May. Above-average precipitation in late August and early September resulted in a second maximum nitrate load peak for the control channel although it was smaller in magnitude than the May peak (Figure 8). During July, the treatment channel reached its lowest observed nitrate-N load. This result is not unexpected as corn and soybean are vigorously growing at this point actively taking up water and nutrients.

As previously reported, 2019 was the wettest of the three years of the project and the second wettest on record for the SWROC. It was also a very active year for surface runoff, resulting in 16 runoff events between April 1st and October 30th. On an annual basis, nitrate-N load was 958 kg for the treatment channel compared to 2,451 kg for the control channel during 2019 resulting in an overall reduction in discharge of 61% for the treatment channel compared to the control channel. Data from 2019 indicated that the annual flow weighted nitrate-N concentration from the control channel was 6.4 mg L^{-1} whereas it was 5.0 mg L^{-1} from the treatment channel (Table 4). This is a 22% reduction in flow weighted nitrate-N concentration from the control to the treatment channel. Although about 40% lower than the previous two years, this is not a surprising result due to the consistently high flows through the channels during 2019. Consequently, although not measured, the HRT would have been relatively short even in the treatment channel given the nature of the flow conditions. Similar to 2018, nitrate load was lower in March than in April. This was primarily due to the amount of surface runoff that occurred during March. Surface runoff tends to exhibit lower nitrate concentrations and subsequently loads than subsurface drainage.

Although phosphorus and nitrogen are both added to the soil as fertilizer, the amount of phosphorus that is released in a given amount of time is very low compared with movement of nitrogen. In the environment, nitrogen transformation is microbially mediated process. On the other hand, phosphorus cycling is a result of a combination of physical and chemical process e.g. weather, adsorption/desorption reactions and is strongly affected by pH. Phosphorus is a very dynamic, biologically active element. Phosphorus cycling in aquatic systems is complex and affected by P concentration, oxygen status, pH, the presence of iron and sulfur bearing minerals and turbidity. Turbidity has been widely used as a surrogate to predict concentrations of suspended particles (Bilotta and Brazier, 2008). It is a relative measure of light diffraction in water caused by suspended particles. The suspended particles may include suspended inorganic or organic particles, including clay colloids and algae. Turbidity has been used as a surrogate for total P (TP) based on the premise that most of the TP transported in streams is in particulate form (Settle et al., 2007; Stubblefield et al., 2007; Ruegner et al., 2013). The relationship between turbidity and suspended solids may be confounded by factors such as variations in particle size distribution, particle composition, or water color (Ruzycki et al., 2014; Bright et al., 2018).

In water, under aerobic conditions phosphorous can be bound into particulate iron-humic complexes both in the water column and sediments, while during anoxic conditions iron oxides are reduced and phosphorus is released into the water column. In sediments or the water column, various P compounds maybe chemically or enzymatically hydrolyzed to orthophosphate, which is the only form of P that can be assimilated by bacteria, algae, and plants. In bottom sediments, microbial communities will consume many of the organic constituents of the sediments, ultimately releasing much of their P contents back to the water column as orthophosphate. Oxygenation status is important in aquatic systems. If waters remain oxygenated throughout the year most P will be stored in the bottom sediments. However, if waters become anoxic much of the P in bottom sediments will be released and diffused back into the water column. If the retention time is sufficiently long, a given volume of water moving downstream in a ditch should behave much like water in river or stream. Phosphorus spiraling downstream is the result of uptake of P by attached bacteria and algae (periphyton) and vascular plants and the binding of compounds in bottom sediments. When these P compounds are released back into the water column, either from bottom sediments or attached biota, they move further downstream, before becoming attached again as the P is cycled among the system components. Each such P movement downstream in the water column is referred to as a "spiral."

On an annual basis, the magnitude in the load of dissolved reactive P (DRP) exported through the ditches showed an increasing trend by year with the smallest export amounts in 2017 and the largest in 2019 (Table 4). Dissolve reactive-P load was 286 g for the treatment channel compared to 194 g for the control channel during 2017 resulting in an overall increase in discharge of 47% for the treatment channel compared to the control channel (Table 4). This was not an expected result. The monthly data showed that there was a spike in DRP in October from the treatment channel (Figure 9). This behavior could be due to release of phosphorus from dead or decaying plant materials. In addition, DRP was also greater from the treatment channel and the control in July 2017. Before the October DRP spike, the data showed that DRP was attenuated in the treatment ditch that contained the low-grade weir in every month (Figure S3). Dissolved reactive P load was greatest in May and June for the control channel during 2017. Monthly dissolved reactive P load was highly variable for both control and treatment ditches and closely mirrored ditch discharge which indicates a connection to hydrologic processes (Figure 9). Loss of DRP showed a bimodal distribution with a first peak during the April/May period and a secondary peak, of smaller magnitude, that occurred between July and September (Figure 9). A uniform or unimodal distribution of P would suggest that the processes affecting P loss from the ditch systems were similar from year to year. A bimodal distribution suggests that there may be more than one process or mechanism involved in P loss to the ditches. The initial surge of P observed from the ditch system occur in early spring when snow melt and surface runoff are common. The load of dissolved P was shown to be associated with surface runoff on frozen soil from a remnant prairie in southwest Minnesota (Tollefson et al., 2014). Losses of phosphorus from cultivated land are greatest when high rainfall coincides with low crop cover (Schuman et al., 1973; Burwell et al., 1975; Truman et al., 1993). This can be during the planting season, a time of phosphorus fertilizer application, minimal crop cover and often intense rainfall (Burwell et al., 1975). The second surge in dissolved P from the ditch systems occurred in September and October. Heathwaite and Dils (2000) measured the highest concentrations of P loss during autumn storms following dry summer months. Phosphorus losses at those times were dominated by dissolved forms. Autumn P loss could be associated with preferential loss of P from artificial drainage beneath cultivated fields in the contributing watershed. Subsurface flow includes the movement of water through the soil matrix, through preferential flow pathways, i.e. macropores, biopores, earthworm burrows, and through artificial drainage networks (Stamm et al., 1998; Simard et al., 2000). Artificial drainage systems, together with preferential flow pathways in the soil, can form a rapid, direct hydrological link between field drainage systems and surface waters (Stamm et al., 1998; Dils and Heathwaite, 1999). The importance of subsurface flow depends on soil and catchment characteristics. Soil type, soil phosphorus concentration, antecedent soil water content and the ability of percolating water to mobilize soil particles (Sims et al., 1998; Djodjic et al., 2000; Hooda et al., 2000; Simard et al., 2000; Chapman et al., 2001). The highest loss rates occur in areas with a high water table (Grant et al., 1997). Another possible source of P could be from dead and decaying plant materials mobilized by precipitation and transported with surface runoff. Alternatively, the second surge in dissolved P could be attributed to the die off of phytoplankton species within the ditch channel.

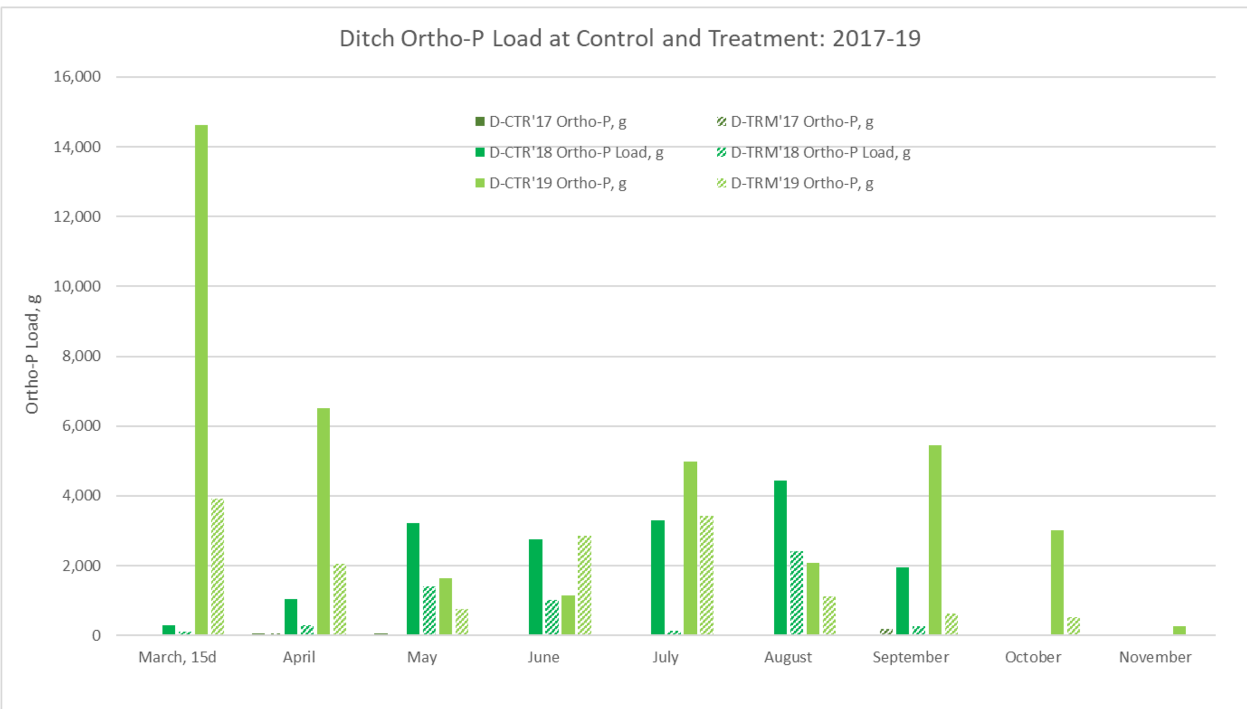

Figure 9. Monthly dissolved reactive phosphorus load from control and treatment ditches between 2017-2019 near Lamberton, MN.

During 2018 and 2019, monthly dissolved phosphorus load indicated that loads were greater from the treatment compared to the control with one exception during June 2019 (Figure 9). During 2018, the monthly dissolve P load gradually increased between the lowest load in March to the highest load in August (Figure 9). In contrast, during 2019 the maximum dissolved P flux occurred during March which coincides with snowmelt runoff. There was also an observable increase in dissolved P loss during July and again in September, 2019 (Figure 9).

2.3.4 Constructed Wetlands

The data in Table 5 summarizes the three-year mean annual discharge, nitrate-nitrogen and dissolved reactive phosphorus load and flow weighted concentrations for the three pairs of constructed wetlands. The data are presented by the constructed wetland design and treatment type. Two water management treatments were compared: a control with no water management treatment and a treatment which consisted of a weir in the outlet flow control structure that was elevated in order to reduce discharged from the system and result in more water storage within a wetland. The three-year summary indicated that there were no statistically significant differences among any parameters for the surface or vertical flow constructed wetlands. The horizontal flow constructed wetlands showed statistically significant differences in mean annual discharge, nitrate-nitrogen load and dissolved reactive phosphorus load between the treatment and control wetlands (Table 5). The data indicated that flow and nutrient loads from the treatment were statistically lower than the control.

Table 5. Summary of constructed wetland combined mean annual discharge, load and flow weighted concentrations for nitrate-nitrogen and dissolved reactive phosphorus between 2017 and 2019 near Lamberton, MN.

		Nitrate-nitrogen		Dissolved reactive phosphorus	
	Discharge	Load	Flow Weighted Concentration	Load	Flow Weighted Concentration
	m^3	kg	mg L^{-1}	g	ug L^{-1}
	Surface flow				
Control	7,816a	27a	4.2a	1,950a	269a
Treatment	4,152a	16a	3.3a	903a	181a
	Vertical flow				
Control	20,641a	135a	7.5a	1,219a	80a
Treatment	17,590a	133a	7.6a	1,328a	70a
	Horizontal flow				
Control	34,256a	407a	11.2a	2,328a	59a
Treatment	9,999b	95b	10.2a	616b	59a

Means within a column heading pair and constructed wetland type followed by the same lower-case letter are not statistically significant at the $P \leq 0.1$ level of significance.

Year to year variation was evident when examining the yearly data between 2017 and 2019. This variation is mainly attributed to variations in annual and inter-annual precipitation timing, frequency, intensity and duration (Table 6). The data indicate that there were no statistically significant differences for any water quality parameter for any of the constructed wetlands in 2017 (Table 6). Data from the surface flow constructed wetland pair in 2018 showed that the mean annual nitrate-nitrogen load and flow-weighted concentration were statistically lower for the treatment wetland compared to the control. This indicated in 2018, that the water level management treatment increased the hydraulic residence time and water storage volume of the surface flow constructed wetland resulting in water quality improvement for nitrate-nitrogen mitigation. During 2019 there were no statistically significant differences in any parameters from the surface flow wetlands.

Table 6. Summary of constructed wetland mean annual discharge, load and flow weighted concentration of nitrogen and dissolved reactive phosphorus between 2017 and 2019 near Lamberton, MN.

			Nitrate-nitrogen				Dissolved reactive phosphorus			
	Discharge		Load		Flow Weighted Concentration		Load		Flow Weighted Concentration	
	m^3		kg		mg L^{-1}		g		ug L^{-1}g	
	Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment
	Surface flow									
2017	-	-	-	-	-	-	-	-	-	-
2018	16,310	5,076	41a	4.7b	4.7a	1.5b	4,758	2,217	544	434
2019	3,837	4,996	25	23	3.6	3.9	469	113	79b	764a
	Vertical flow									
2017	251	163	2.6	1.4	8.6	8.0	15	8.5	61	108
2018	32,428	16,506	239a	96b	6.5	6.5	2,491	1,399	71	89
2019	32,991	17,817	343a	136b	8.1	7.8	1,845	1,260	53b	75a
	Horizontal flow									
2017	319	279	3.7	4.2	13.6	13.3	13	115	48	35
2018	91,645a	19,730b	1,139a	210b	11.6	10.8	6,380a	1,294b	75a	66b
2019	12,425a	7,562b	110a	55b	9.4	8.2	737a	408b	62	59

Means within a column heading pair, by year and constructed wetland type, followed by the same lower-case letter are not statistically significant at the P ≤ 0.1 level of significance.

Data from the vertical flow constructed wetlands indicated that there was only a statistically significant difference for nitrate-nitrogen load where the control was greater than the treatment (Table 6). This is attributed to a numerical difference in flow volume between these two treatments in 2018 since the flow-weighted concentrations were the same, 6.5 mg L^{-1}. During 2019, there was a similar statistical relationship for nitrate-nitrogen load for the vertical flow constructed wetland. In addition, during 2019, there was a statistically significant difference in flow-weighted dissolved phosphorus concentration, however, in contrast to nitrate-nitrogen where the treatment was less than the control, in this case the dissolved flow weighted phosphorus concentration was statistically greater from the treatment compared to the control (Table 6).

Data from the horizontal flow constructed wetland pair for 2018 and 2019 showed that the mean annual discharge, nitrate-nitrogen and dissolved reactive phosphorus loads were statistically lower for the treatment wetland compared to the control (Table 6). As previously, this is attributed to a numerical difference in discharge flow volume between the two treatments in 2018 and 2019 since the flow-weighted concentrations were statistically similar.

The cumulative annual discharge, nitrate-nitrogen and dissolved reactive phosphorus loads were variable from year-to-year. As previously noted, this is attributed to year-to-year differences in annual precipitation. Other potential factors that could influence nutrient fluxes, particularly for N-cycling, include soil temperature and growing season length. Discharge and load data showed that with one exception, nitrate-nitrogen load in 2019, that the treatment constructed wetland values were always attenuated for each of the three types of constructed wetland systems (Table 7). This is attributed to the presence of a low grade weir in the outlet structure of each treatment. Despite a lack of statistical differences for some water quality parameters it is evident regardless of the type of constructed wetland that they were all effective in mitigating water and nutrient loss from this agricultural landscape as evidenced by the percent reductions loads (Table 8). In almost every case, the wetlands in this study nearly met or exceed the target 45% reduction allocation in the Minnesota Nutrient Reduction Strategy (MPCA, 2012).

Table 7. Summary of constructed wetland cumulative annual discharge and load for nitrate-nitrogen and dissolved reactive phosphorus between 2017 and 2019 near Lamberton, MN.

			Nitrate-nitrogen		Dissolved reactive phosphorus	
	Cumulative Discharge		Load		Load	
	Control	Treatment	Control	Treatment	Control	Treatment
	m^3		kg		g	
	Surface flow					
2017	1,019	551	4.5	0.4	221	152
2018	81,548	10,152	203	9.2	23,789	23,789
2019	34,971	26,857	162	174	5,344	3,285
	Vertical flow					
2017	1,255	654	13	5.7	75	34
2018	194,565	99,037	1,434	576	14,947	8,394
2019	230,936	124,717	2,060	815	11,072	7,558
	Horizontal flow					
2017	1,596	1,114	21	15	66	61
2018	549,872	118,378	6,832	1,258	38,280	7,764
2019	99,401	60,496	879	441	5,894	3,266

Table 8. Constructed wetland percent annual load reductions comparing control and treatment wetlands for nitrogen and phosphorus.

	Cumulative Discharge	Nitrate-N load	Dissolved reactive phosphorus load
		%	
Surface flow			
2017	46	92	31
2018	88	95	81
2019	23	-7.6	39
3-yr mean	**52**	**60**	**50**
Vertical flow			
2017	48	56	55
2018	49	60	44
2019	46	60	32
3-yr mean	**48**	**59**	**43**
Horizontal flow			
2017	30	30	7.4
2018	78	82	80
2019	39	50	45
3-yr mean	**49**	**54**	**44**

2.3.5 Modular, Vertical Flow Bioreactors

The early spring period of the growing season (April and May) presents a serious challenge for water quality during which the water is abundant and cold. It is well known that denitrification depends on temperature, with higher temperature (15°C and above) producing higher denitrification rates (Welander and Mattiasson, 2003). The addition of a carbon source will further increase the performance of the denitrification process even under short hydraulic residence times (HRT) (Roser et al., 2018); the laboratory experiment showed that woodchips with the addition of acetate resulted in the highest nitrate load reduction under warm (80%, 15 °C), cold (80%, 5 °C), and re-warmed (97%, 15 °C) conditions.

The hydraulic residence time of drain water in the modular, vertical flow bioreactor ranged from 2.1 to 2.3 hours in 2018 and from 2.4 hours to 3.3 hours in 2019 (Table 9). Average HRT in 2019 was slightly higher than that of 2018 (2.2 vs. 2.8 hours).

Table 9. Hydraulic residence time (HRT) of modular, vertical bioreactors in 2018 and 2019.

Treatment	2018 HRT, hr	2019 HRT, hr	%Diff.
Control	2.1	2.4	12.4%
Acetate	2.2	3.3	32.0%
Acetate + Heat	2.3	2.8	17.7%
Average	2.2	2.8	19.7%

A summary of the results from the modular, vertical flow bioreactors is given in Table 10. The data represent combined means for 2018 and 2019 for cumulative discharge, nitrate and dissolve reactive phosphorus load and flow weighted concentration. These data indicate that there was no significant difference in mean discharge from the bioreactors nor was there any difference in dissolved reactive phosphorus load or flow weighted concentration. In contrast, nitrate-nitrogen load and flow weighted concentration were statistically smaller in magnitude for the Acetate and Acetate+Heat treatments compared to the control, which only consisted of corn cobs and wood chips (Table 10). These data suggest that there was no statistically significant differences for any parameter between the Acetate and Acetate+Heat treatment. Despite no statistical significance between these treatments a plot of the data indicates an overall trend of decreasing load and flow-weighted concentration (Figure 10). The high degree of variability between replications and the small number of replications (n=3) contribute to the lack of significance between these two treatments.

Table 10. Summary of modular, vertical flow bioreactor cumulative discharge, load and flow weighted concentrations for nitrate-nitrogen and dissolved reactive phosphorus between 2018 and 2019 near Lamberton, MN.					
	Cumulative discharge	Nitrate-N	Flow Weighted Concentration	Dissolved reactive phosphorus	Flow Weighted Concentration
	m^3	kg	mg L^{-1}	g	ug L^{-1}
Control	92a	1,020a	10.6a	6,624a	75a
Acetate	80a	725b	8.4b	6,068a	76a
Acetate+Heat	75a	580b	7.1b	5,269a	75a
Values followed by the same lower-case letter are not statistically significant at the P ≤ 0.1 level of significance.					

A summary of the yearly modular, vertical flow bioreactor data showed similar results as the combined data indicating that there was no significant difference in mean discharge from the bioreactors nor was there any difference in dissolved reactive phosphorus load or flow weighted concentration (Table 11). In 2018, there was a statistically significant difference in flow weighted nitrate-nitrogen concentration but not in nitrate-nitrogen load (Table 11). In 2019, there was a statistically significant difference in flow weighted nitrate-nitrogen concentration and nitrate-nitrogen load (Table 11). It was also apparent from the data that the magnitude in measured values were numerically greater in 2018 for all parameters compared to 2019.

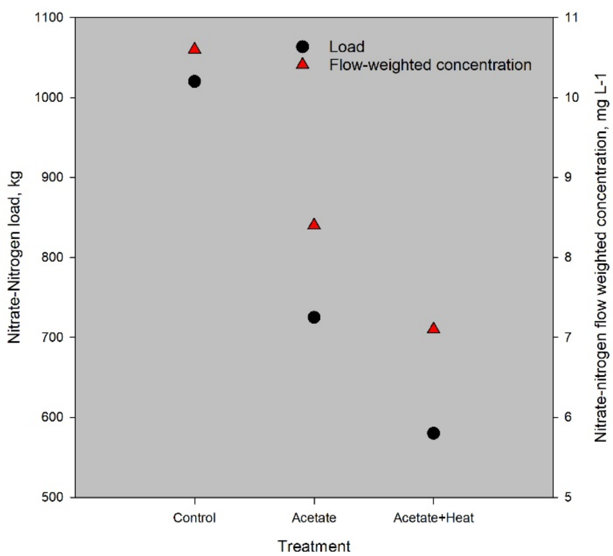

Figure 10. Summary of modular, vertical flow bioreactor nitrate-nitrogen load between 2018 and 2019 near Lamberton, MN.

Table 11. Summary of yearly bioreactor discharge, load and flow weighted concentration nitrate-nitrogen and dissolved reactive phosphorus between 2018 and 2019 near Lamberton, MN.					
	Flow	Nitrate-N Load	Flow Weighted Concentration	Dissolved reactive phosphorus Load	Flow Weighted Concentration
	m^3	kg	mg L^{-1}	g	ug L^{-1}g
			2018		
Control	100a	1,190a	11.3a	7,474a	83.9a
Acetate	93a	959a	9.9a	7,204a	83.0a
Acetate+Heat	85a	787a	8.6b	6,087a	85.0a
			2019		
Control	85a	871a	10.0a	5,880a	67.0a
Acetate	69a	521b	7.2b	5,074a	69.7a
Acetate+Heat	67a	399b	5.7b	4,553a	66.2a
Values followed by the same lower-case letter are not statistically significant at the P ≤ 0.1 level of significance.					

Despite the continuous application of heat to the cubes in 2018, the monthly nitrate load reduction remained moderate for the Acetate+Heat treatment (annual average 35%) with a peak in July at 55% (Table 12). By contrast, nitrate load reduction, compared to the subsurface drainage source water, in 2019 was larger compared to that of 2018; the annual average nitrate load reduction rose from 35% to 49% for the acetate + heat treatment and the peak value occurred in May (Table 12 and Figure 11). The May 2019 nitrate load reduction reached 75%, which is a very unusually high denitrification rate for that month. All things being equal, the temperature applied to the cube bioreactor was the largest contributor to this nitrate load reduction. To a certain extent, a larger HRT in 2019 (average HRT 22% longer compared to that of 2018) also contributed to the nitrate load reduction. The contrast of nitrate load reduction between 2018 and 2019 under the heat treatment can be partly explained by thermal acclimation of the denitrification bacterial community. Acclimation refers to the adaptation of microorganisms to a change in environment (temperature in this case) (Crowther and Bradford, 2013). Such bacterial adjustment may take some amount of time to take effect. The data supported our hypothesis and indicated that the ranked order of percent load reduction was acetate + heat > acetate > control for 2018 and 2019 (Table 12 and 13).

Table 12. Mean monthly and annual nitrate load reduction from modular, vertical flow bioreactors in 2018 compared to subsurface drain source water. Annual nitrate load was as follows: control (8.85 kg), acetate (6.42 kg), and Acetate+Heat (5.74 kg).				
Month	**Source water**	**Control**	**Acetate**	**Acetate + Heat**
	kg	%		
May	0.012	5.2	0.8	18.0
June	1.81	22.6	27.4	34.1
July	2.19	15.5	42.1	55.0
August	0.80	20.2	29.0	31.3
September	2.01	21.3	23.8	32.3
October	1.86	4.5	14.1	21.8
November	0.43	2.0	14.5	15.2
Average	9.11	15.5	26.6	35.1

Table 13. Mean monthly nitrate load reduction from modular, vertical flow bioreactors in 2019 compared to subsurface drain source water. Annual nitrate load was as follows: control (6.96 kg), acetate (4.16 kg), and Acetate+Heat (3.19 kg).				
Month	**Source water**	**Control**	**Acetate**	**Acetate + Heat**
	kg	%		
April	0.684	-2.3	10.9	28.6
May	0.862	22.2	62.6	74.9
June	1.245	17.1	35.3	48.9
July	1.101	24.4	60.9	70.5
August	0.701	23.5	53.0	51.4
September	0.805	16.5	38.3	56.6
October	1.499	9.7	23.2	40.2
November	0.238	3.7	4.7	19.3
Average	7.134	14.3	36.1	48.8

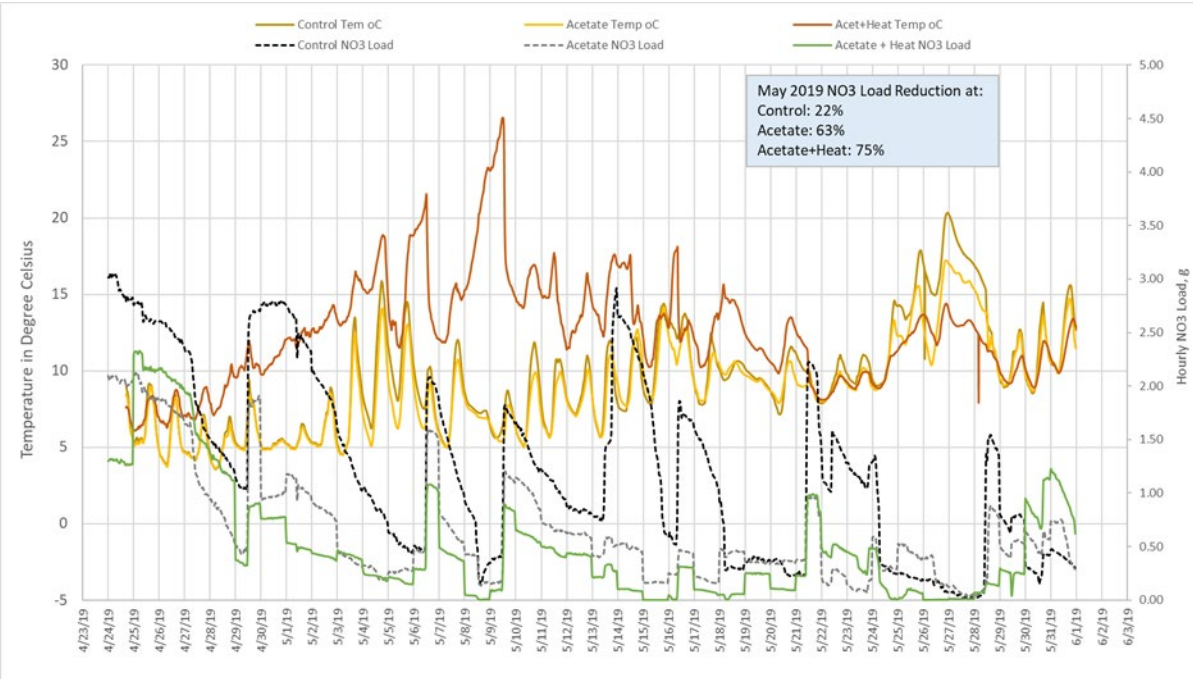

Figure 11. Nitrate load for the control, acetate, and acetate + heat treatments during May 2019 and temperature record from the thermocouple located in the middle of the bioreactor. Heat was applied to the bioreactors during April and May in 2019.

2.3.6 Cropping Systems

We used a relay-cropping strategy to grow winter Camelina as a scavenger crop and as an oilseed crop to reduce nitrogen loss from the drained experimental plots plus produce an oilseed crop to help offset planting costs while allowing two crops to be grown in the same season. Relay cropping is a form of double cropping where the second crop, in this case corn or soybean, is planted into the first crop, in this case winter Camelina, before the first crop is harvested, rather than waiting until after harvest as in a true double-cropping system. In general, for the years of this experiment, the relay cropping system with winter Camelina resulted in poor Camelina stand establishment as well as poor corn and soybean stand establishment mainly due to conservation tillage management and delayed planting due to wet conditions. Poor stand establishment resulted in poor yields in plots where Camelina was grown. Camelina yields were also rather poor.

The drainage system was installed in late summer 2017 and the winter Camelina cover crop, planted in every plot (six total) into wheat residue, during fall 2017 survived the winter and grew quite vigorously in spring 2018 despite wet conditions. No tillage was performed in the plots in the spring before planting. Corn and soybean crops were harvested from the experiment in fall 2018 and a new stand of winter Camelina was planted in selected plots (three total). Despite a rather wet growing season and no-till conditions, the mean corn yield from the six plots was 150 bu ac^{-1} with a range of 134 to 156 bu ac^{-1}. This average corn yield was less than expected for this region even with somewhat wetter than normal conditions. The mean soybean yield from the six plots was 55 bu ac^{-1} with a range of 49 to 58 bu ac^{-1}. This average soybean yield was also slightly less than expected for this region even with somewhat wetter than normal conditions.

Delayed planting due to wet conditions resulted in very poor corn and soybean growth in 2019. Despite a rather wet growing season, the mean corn yield from the three plots with Camelina was 43 bu ac^{-1} with a range of 40 to 52 bu ac^{-1}. This average corn yield is much less than expected for this region even with somewhat wetter than normal conditions. The mean corn yield from the three plots without Camelina was 204 bu ac^{-1} with a range of 185 to 217 bu ac^{-1}. This average corn yield is more than expected given the wetter than normal conditions. The mean soybean yield from the three plots with Camelina was 27 bu ac^{-1} with a range of 21 to 35 bu ac^{-1}. This average soybean yield is less than expected for this region even with somewhat wetter than normal conditions. The mean soybean yield from the three plots without Camelina was 48 bu ac^{-1} with a range of 43 to 54 bu ac^{-1}. This average soybean yield is about what would be expected for this region given the wetter than normal conditions. Camelina yield.

Cameilina was planted in every plot rather than only half of the blocks in the fall of 2017. Consequently, no comparison could be made between treatments with and without Camelina in 2017. The winter Camelina cover crop planted into no-till corn and soybean residue in fall 2018 survived the winter. Camelina growth was highly variable both within and between plots. We hypothesize that the variability was due to a combination of no-till residue, cool temperatures, wet conditions and relatively late broadcast nitrogen application. Camelina growth was visually better where it was planted into soybean residue compared to corn residue. As temperatures warmed in spring, both air and soil, and growing degree units accumulated, Camelina growth became more uniform within and between plots. Camelina was harvest in mid-July in 2018 and 2019. Camelina yield under all plots in 2018 ranged from 1.5 to 2.6 bu ac^{-1} with a mean of 1.9 bu ac^{-1} for corn. Camelina was planted to half the plots as planned in fall of 2018. Similar weather and growing conditions persisted in 2019. Camelina yield in 2019 ranged from 3.9 to 9.4 bu ac^{-1} with a mean of 6.5 bu ac^{-1} for corn. Camelina yield under soybean ranged from 2.3 to 4.8 bu ac^{-1} with a mean of 3.3 bu ac^{-1}.

Data for drain flow volume by month for treatments with and without Camelina are presented in Table 14. The data indicated that there was only drain flow recorded in September 2018 for the plots where Camelina was planted whereas drain flow was observed in May and June from plots without Camelina. This suggests that there may have been more water storage capacity in treatments with Camelina due to water use by the crop. In 2019, there was no drain flow recorded. This was not expected but could be the result of the loss of water from the plots with surface runoff which was not monitored during 2018 or 2019. This is under consideration for the future given that there is some natural topographic differences among plots.

Table 14. Cropping systems field monthly drainage discharge summary for plots with and without winter Camelina.

Year/Treatment	April	May	June	July	August	September	October
2018							
With Camelina		-	0.0	0.0	0.0	4.0	0.0
Without Camelina	0.0	10.6	54.5	0.0	0.0	0.0	0.0

2019							
With Camelina	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Without Camelina	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Figure 12 is a plot of soil water content, by month, measured under corn and soybean with and without Camelina. The variability observed in the plots reflects water use be the primary crops, corn or soybean, water use by the secondary crop, Camelina, precipitation and evapotranspiration. During June, the data indicated the treatments with Camelina were drier at all depths compared to the treatments without Camelina. July soil water content data were highly variable between crops and depths. Then during August and September there was much less variability between treatments and trends among depths were similar for both crops. This is likely because Camelina is normally harvested during mid-to late-July and soil water content became less variable. It is likely that under the wetter than normal conditions during 2018 and 2019 root growth was limited due to near saturated conditions below 60 cm during the growing season.

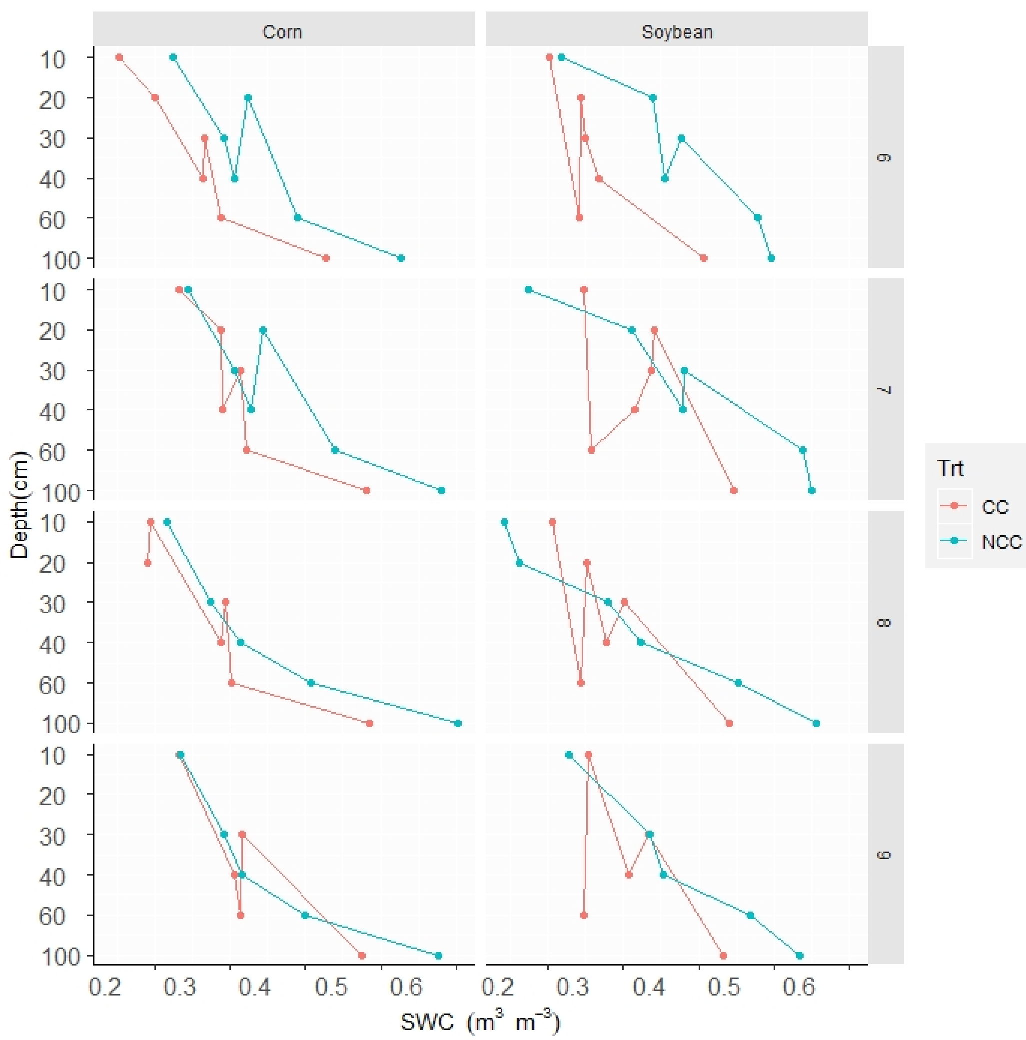

Figure 12. Mean soil water content, by month and depth, for treatments with and without winter Camelina. Legend definitions: Camelina crop (CC), no Camelina crop (NCC)

2.4 Conclusions

Our team experienced numerous challenges over the course of the project mainly due to an extended period of wet conditions. We are not able to assess how these systems respond under different conditions, for example, during normal nor during or following a dry period. It is difficult to assess the project's success since we were unable to measure all sources of subsurface drainage and surface runoff nor were we able to measure water and nutrients at the inlets and outlets of each practice, mainly the wetlands and ditches, due to funding limitations because we underestimated our needs. In spite of these challenges, our team was pleased with our accomplishments. Our data support that structural practices like bioreactors, constructed wetlands and managed ditches can have a positive effect on water quality and can impact water storage and export from a watershed. Our data support that for nitrogen, the practices we implemented and studied facilitated the reduction in flow-weighted concentration and load leaving our small watershed. In contrast, dissolved reactive phosphorus appeared to be exported from our watershed despite our mitigation efforts. There could be a number of explanations for this including phosphorus loss from the unmonitored source waters mentioned previously, legacy phosphorus from the constructed wetlands and ditches being released into the water column under certain conditions, and finally losses from vegetation located in the constructed wetlands and ditches. What this component of the project indicates is that through the implementation of a combination of in-field, edge-of-field, and beyond-the-field/in-stream practices, water quantity leaving the landscape can be managed along with dissolved nitrogen. These combined practices can have a major impact in reaching the 45% reduction in nitrogen leaving our agricultural landscapes. The project also indicates that there is more work to do in understanding and managing phosphorus loss from agricultural landscapes.

Although we have shown positive impacts from implementing multiple strategies to mitigate agricultures impact on water quantity and water quality, we realize that the scale of this research has been conducted on a relatively small scale. One question that still remains to be answered is what impact will these and other combined practices have at larger watershed scales? Another question is what are the mechanism controlling nitrogen and phosphorus loss in a cold climate? Are there ways that nutrient reductions could be further augmented?

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2.6 Supplementary Material

Year 2017

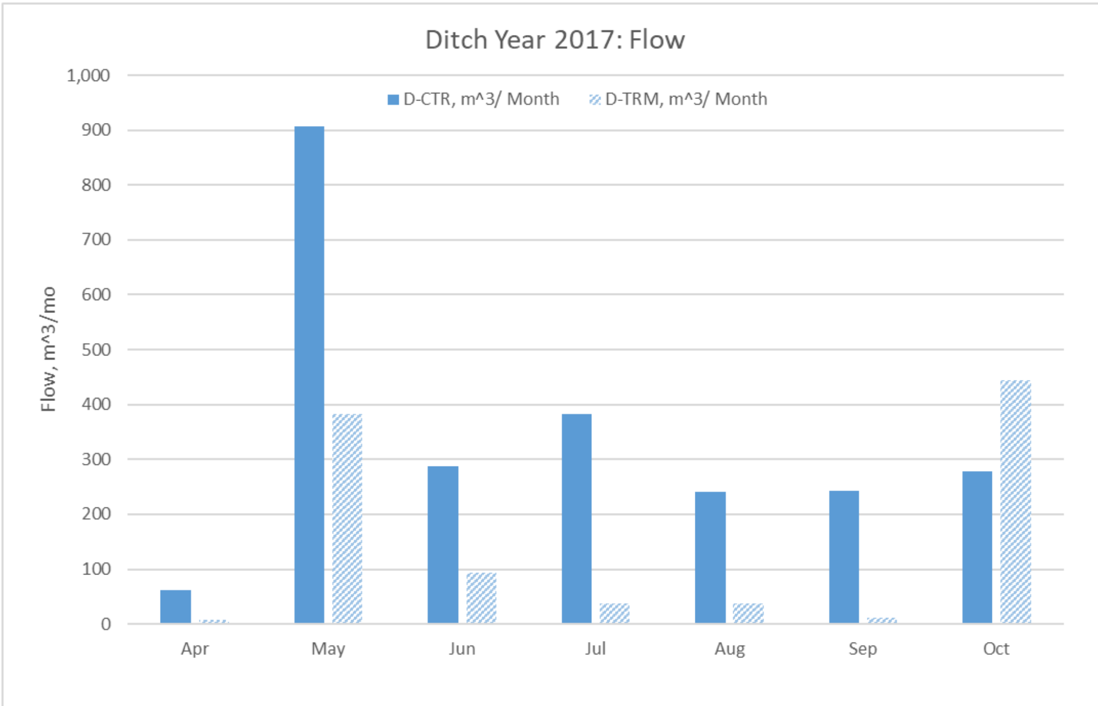

Figure S1. Monthly ditch discharge from control and treatment ditches during 2017 near Lamberton, MN.

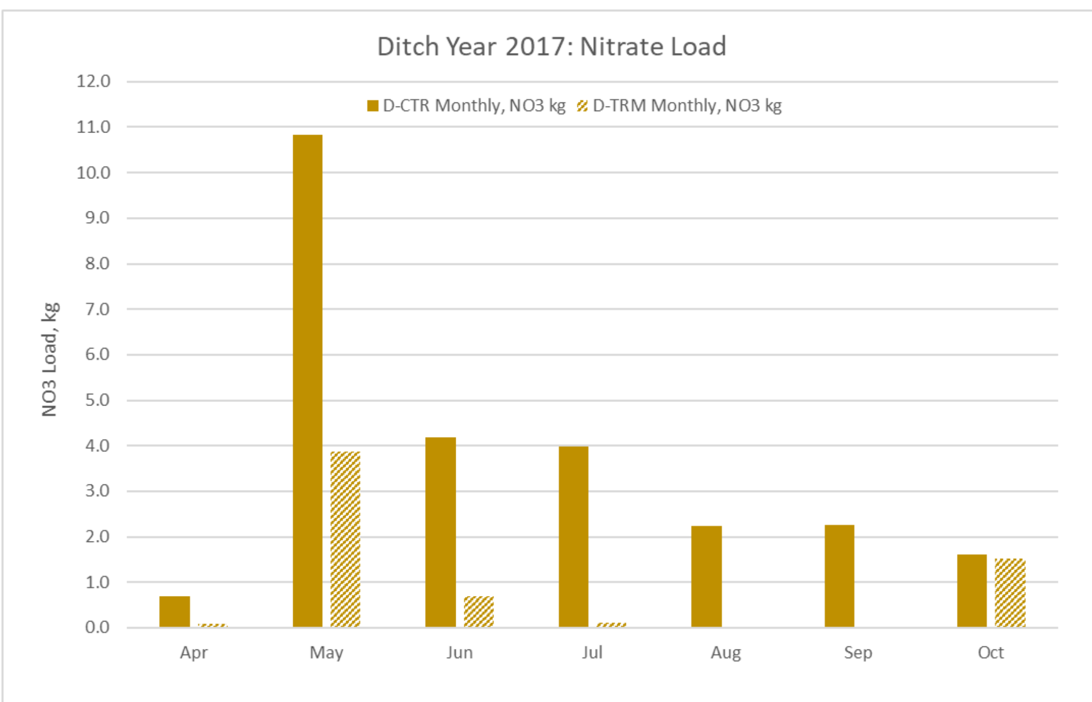

Figure S2. Monthly Nitrate-N load from control and treatment ditches during 2017 near Lamberton, MN.

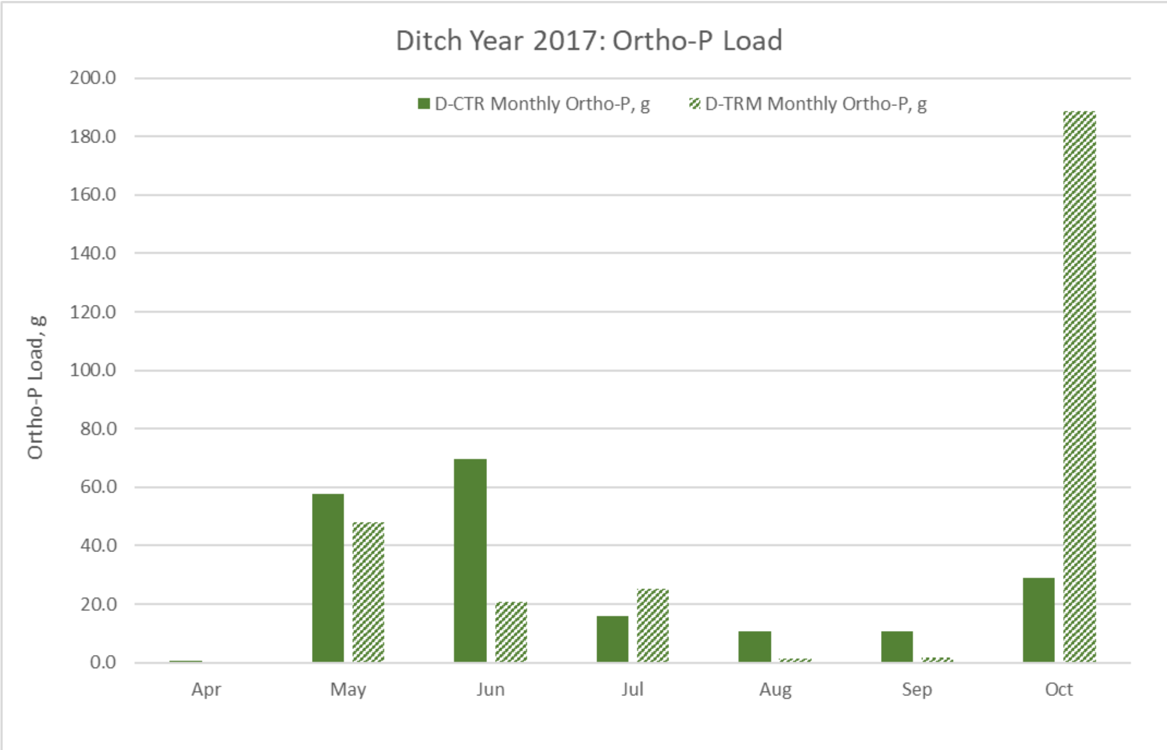

Figure S3. Monthly dissolved reactive phosphorus load from control and treatment ditches during 2017 near Lamberton, MN.

Year 2018

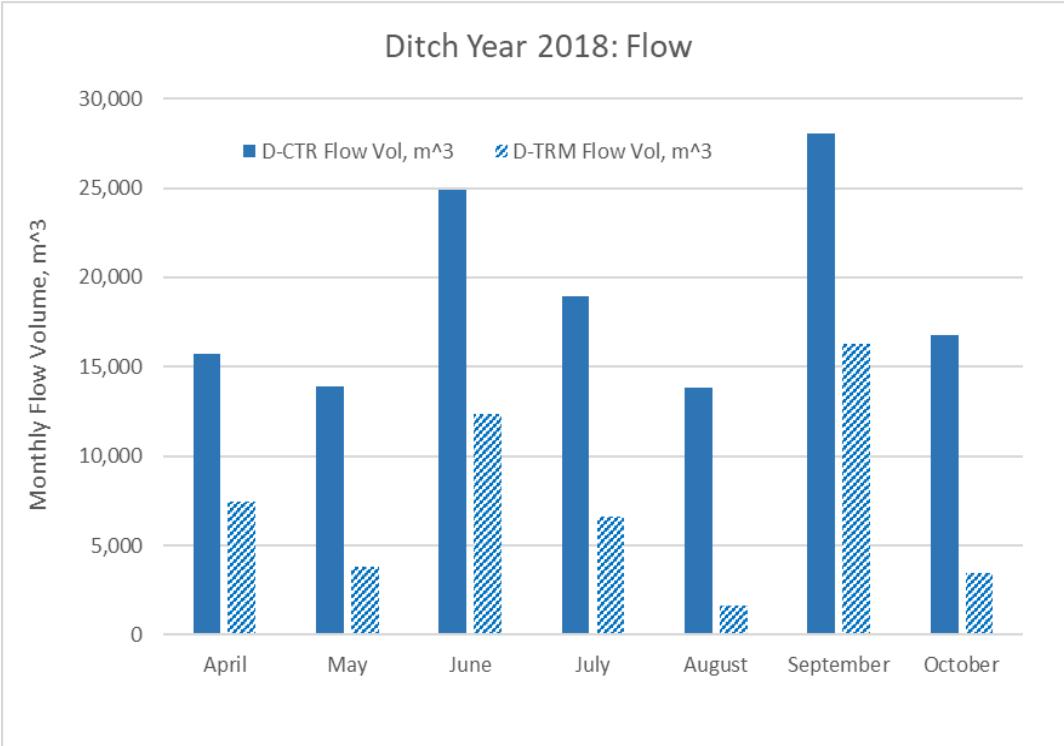

Figure S4. Monthly ditch discharge from control and treatment ditches during 2018 near Lamberton, MN.

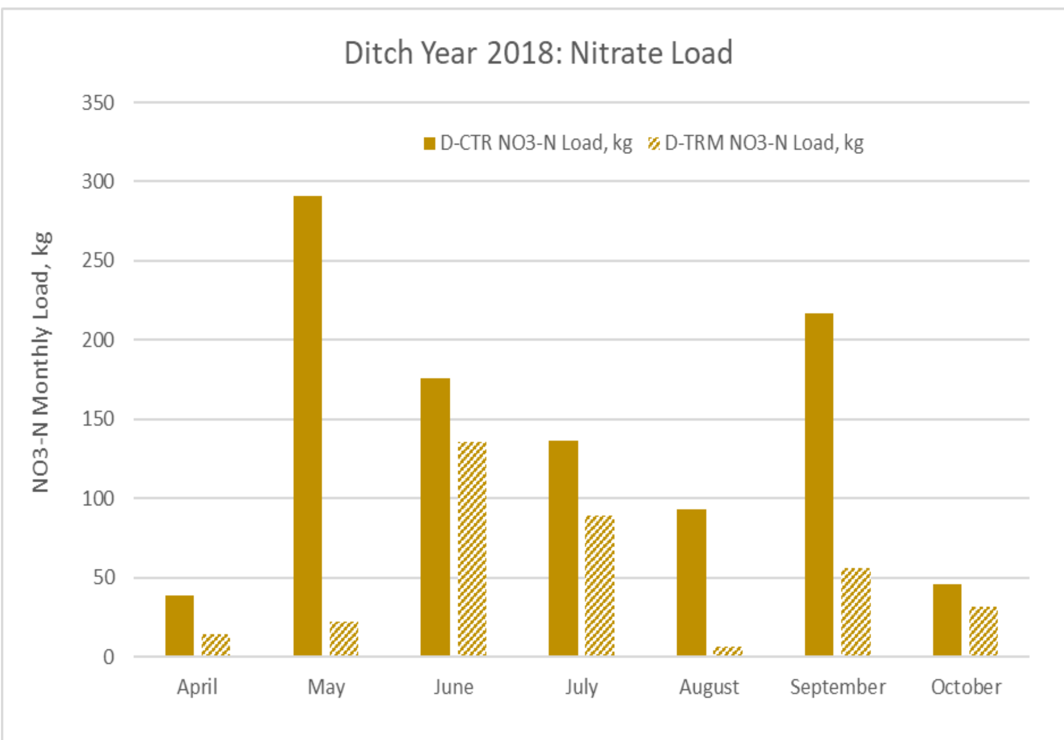

Figure S5. Monthly Nitrate-N load from control and treatment ditches during 2018 near Lamberton, MN.

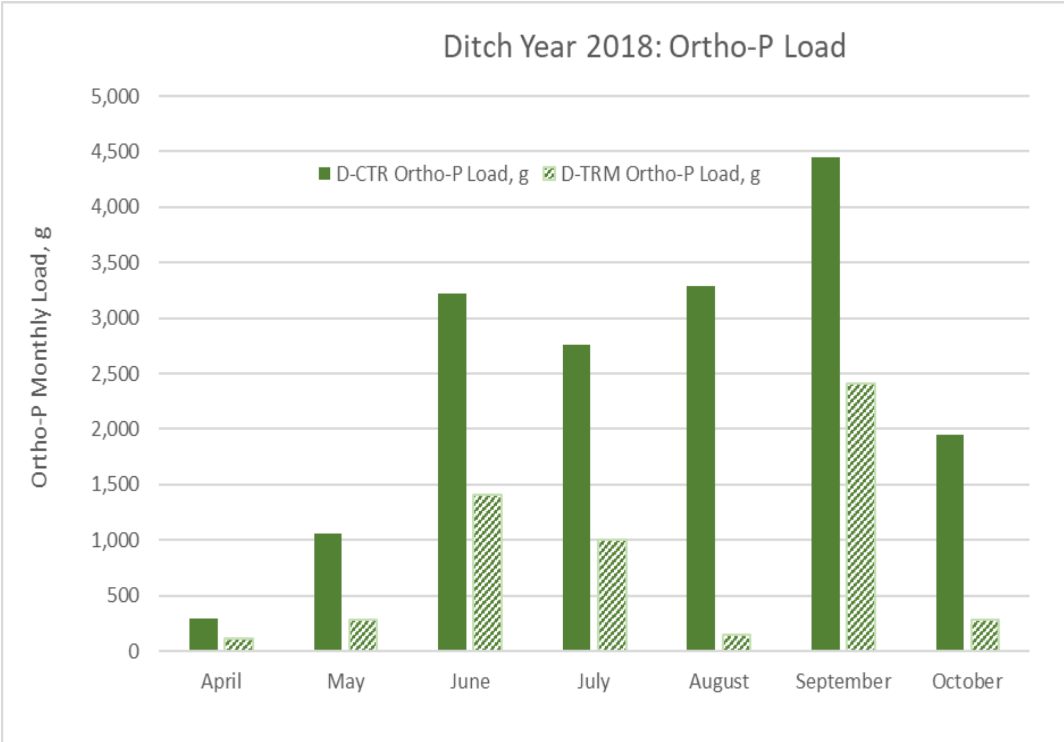

Figure S6. Monthly dissolved reactive phosphorus load from control and treatment ditches during 2018 near Lamberton, MN.

Year 2019

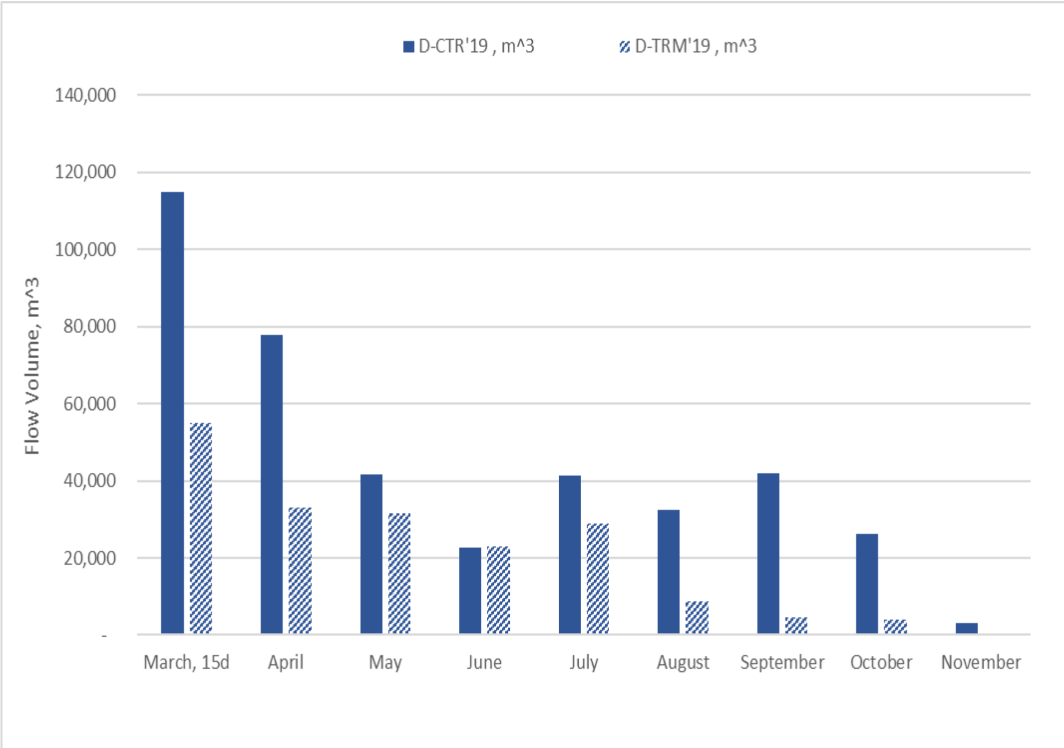

Figure S7. Monthly ditch discharge from control and treatment ditches during 2019 near Lamberton, MN.

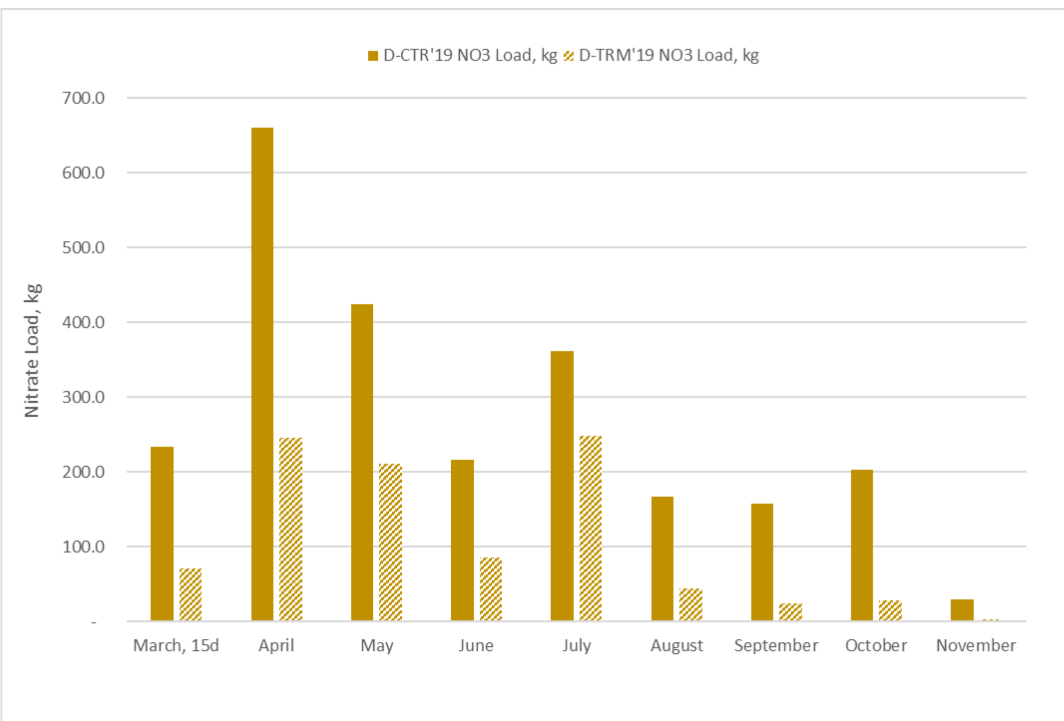

Figure S8. Monthly Nitrate-N load from control and treatment ditches during 2019 near Lamberton, MN.

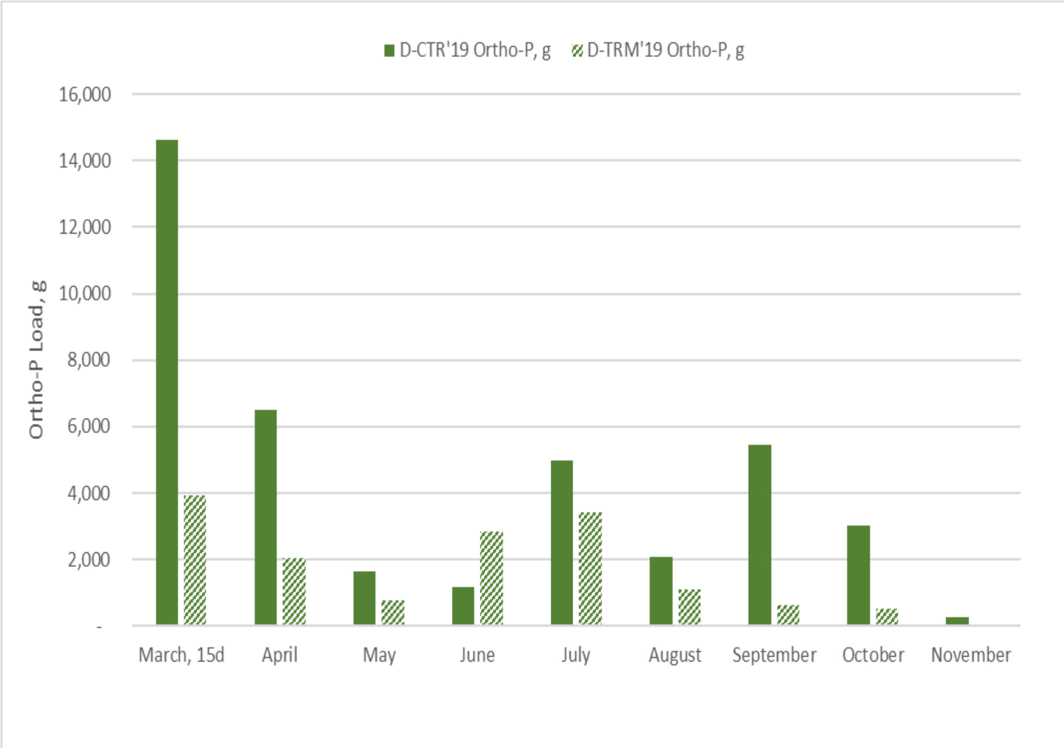

Figure S9. Monthly dissolved reactive phosphorus load from control and treatment ditches during 2019 near Lamberton, MN.

Chapter 3

Nitrogen and Phosphorus removal from Agricultural Drainage Water by a Modular Bioreactor

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Abstract

A modular bioreactor for removing nitrate-nitrogen (NO_3-N) and dissolved reactive phosphorus (DRP) in subsurface drainage flow was designed and tested. The field experiment consisted of three replications of three experimental treatments of steel slag, crushed recycled concrete and fragmented limestone for DRP removal and woodchips plus corn cobs for NO3-N removal from agricultural subsurface drainage. Potassium acetate (KCH_3CO_2) was used as an external carbon source to enhance denitrification. The experimental period lasted between May and December 2016 and March and July 2017. Mean HRT was 3.7 h during 2016 and 3.4 h in 2017. Mean P removal rates (PRR) among the three P-treatment materials was 112 mg P m^3 d^{-1} in 2016 and -0.52 mg P m^3 d^{-1} in 2017. During 2017, the negative PRR signified that the bioreactors acted as a source of P rather than a sink of P. Dissolved reactive phosphorus load reduction ranged between 10% and -6% during 2016 and 2017, respectively. The mean N removal rates (NRR) among the three P-treatment materials was 24.5 g N m^3 d^{-1} in 2016 and 9.6 g N m^3 d^{-1} in 2017. During 2016 and 2017, when acetate was added to the bioreactors, NO_3-N load reduction averaged 35% and 14%, respectively. This design for bioreactors is meant to be installed directly under a drainage outlet if the ditch geometry allows. Based on the results, these bioreactors have the potential to be used as an alternative design to classical denitrifying bioreactor beds for subsurface drainage water treatment under cold conditions.

3.1 Introduction

Nitrogen (N) and phosphorus (P) contamination of surface waters from non-point source agricultural pollution are a serious global problem, as well as in the Midwestern US (Murphy et al., 2013; Smith et al., 2015). Hypoxia in the Gulf of Mexico has been recognized since the mid 1980's and is largely attributed to nutrient enrichment of marine waters by N and P entering the Gulf from the Mississippi River (USEPA, 2013). A federal task force recommended a 45% reduction in N and P loads entering the Gulf of Mexico from the Mississippi River in order to reduce the long-term average area of the Gulf hypoxic zone to 5,000 km^2 or less (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2008; Greene et al, 2009). There has been increased interest in developing Best Management Practices (BMPs) for mitigating the effects of subsurface drainage on nutrient losses to surface waters. Ideally, a successful BMP would mitigate the negative impact of subsurface drainage on both N and P losses, while limiting its negative consequences on crop production practices and crops (Nangia et al., 2010; Strock et al., 2010).

Crop production in northern latitudes is subject to short growing seasons and in some regions, periods of frozen soil. Despite these conditions, crop productivity regularly exceeds 10.1 Mg ha^{-1} for corn, 3.1 Mg ha^{-1} for soybean and 3.0 Mg ha^{-1} for wheat (USDA NASS, 2018). Conservation practices for mitigating water quality impacts from agriculture in northern latitudes is challenging. Strock et al. (2004) reported that cover cropping reduced nitrate-nitrogen (NO_3-N) leaching in Minnesota but could be limited by late autumn planting, low temperatures in autumn and spring and excessive moisture in the autumn which may result in poor rye establishment, low dry matter yield and N uptake. Research on bioreactor performance demonstrated that the performance of denitrifying bioreactors was highly variable, with low NO_3-N removal efficiencies occurring when temperatures were low, or flow was high (Christianson et al., 2012; Hoover et al., 2016; David et al., 2016).

One promising approach for treatment of agricultural subsurface drainage water is through a bioreactor which can provided solutions to cold temperature operation, long hydraulic residence time (HRT) and carbon limitation (Addy et al., 2016). Contemporary agricultural bioreactors use denitrification to reduce NO_3-N to dinitrogen gas or calcareous materials to sorb phosphorus. Key factors influencing denitrification include: NO_3-N and P supply, dissolved organic carbon, oxidation-reduction potential, temperature, and HRT (Golterman, 2004; Piña-Ochoa and Ãvarez-Cobelas, 2006). Biological processes such as denitrification are subject to temperature effects: warm temperatures increase metabolic processes whereas colder temperatures reduce metabolic processes. In denitrifying bioreactors, colder temperatures reduce nitrate removal rates (NRR) on the order of 2 to 3 times per 10°C temperature change (Addy et al., 2016; Feyereisen et al., 2016; Hoover, 2016). Previous research has shown that application of P sorbing materials containing fragmented limestone (Mateus et al., 2012), steel slag (Penn et al., 2011) and crushed concrete (Egemose et al., 2012) have the capacity to bind dissolved P. In laboratory experiments under warm conditions (i.e., water temperatures ≥17°C), Christianson et al. (2017) and Goodwin et al. (2015) placed P-sorbing media upstream and downstream of a wood chip denitrifying chamber. Christianson et al. (2017), referencing Sibrell et al. (2009), envisioned replacement and recharge of the P-sorbing media. Egemose et al (2012) demonstrated in a laboratory experiment that crushed concrete could be an effective tool to remove P in urban and agricultural runoff.

There have been several design variations of denitrifying bioreactors including in-field denitrification walls (Jaynes et al., 2008), edge-of-field bioreactors (Woli et al., 2010) and stream bed bioreactors (Robertson and Merkley, 2009). A typical edge-of-field denitrifying bioreactor consists of a trench 3 to 8 m wide by 1 to 1.5 m deep by 10's of meters long. To date, most bioreactor trenches have been filled with woodchips as the source of carbon, although other media have been researched in the laboratory (Soares and Abeliovich, 1998; Greenan et al., 2006; Cameron and Schipper, 2010; Warneke et al., 2011b; Healy et al., 2012; Feyereisen et al., 2016).

In addition to temperature, C availability has been noted as a limiting factor in nitrate removal (Warneke et al., 2011a). Warneke et al. (2011b) and Feyereisen et al. (2016) reported that agriculturally derived media, corn cobs in particular, provided more C to denitrifying bacteria than did woodchips and therefore effected higher NRRs. Furthermore, Roser et al. (2018) found that dosing woodchips with acetate dramatically increased NO_3-N load reductions at low temperatures (5°C).

Thus, combining the best of the findings identified above – greater NRR and P removal at higher flow rates and colder temperatures with accessibility for media replacement and recharge – would represent an advancement for bioreactors that are effective at removing both N and P. We proposed a bioreactor design that uses a P-sorbing material in series with a bed of corn cobs followed by woodchips and dosed with acetate to improve N and P removal from agricultural tile drainage. We designed a novel modular bioreactor capable of removing both N and P that would be accessible for maintenance. This design was the first known vertical flow denitrifying bioreactor system conceptualized as a modular system, accessible for maintenance and media replacement, specifically designed to remove N and P from agricultural drainage water at short HRTs under cold conditions.

One of the goals of this project was a field evaluation of the N and P reduction capacity of the described bioreactor system under contrasting temperature conditions (i.e., cool versus warm). It was expected that cold water and air temperatures during the early part of the season, between March and May, and late season, between October and November (e.g. <10°C), would reduce the rate of denitrification in the bioreactor and thus affect the efficiency of nitrate removal from the subsurface drainage water. In contrast, it was expected that relatively warm conditions during the middle to late part of the growing season would have little to no effect on denitrification. It was expected that P remediation would be less affected by temperature than N.

3.2 Material and Methods

3.2.1 Site Location

The field experiment was conducted during 2016 and 2017 at the Southwest Research and Outreach Center (SWROC) near Lamberton, MN (44°14’36” N, 95°18’17” W). The climate is interior continental with cold winters and moderately hot summers with occasional cool periods. Mean annual temperature is 7˚C, with monthly extremes ranging from 21˚C in July to -9˚C in January. Total annual precipitation of 670 mm is adequate for row crop production. Subsurface drainage from approximately 73 ha discharges into a channel adjacent to the bioreactors. Subsurface drainage discharge is seasonal, with higher flows from April through June when spring snowmelt combines with spring rainfall. The contributing watershed area comprises 74% cropland (row crops), 20% pasture, and 6% farmstead. The soils of the watershed are of the Canisteo−Ves association. Canisteo soils are poorly drained and are found on the broad lowland glacial till plain. The Canisteo soils and other poorly drained soils in this association require artificial drainage to make them suitable for crop production. Ves soils are well drained and occupy convex knolls above the lowland till plain. Erosion is a concern in management of this soil. These soils are used mainly for row−crop production.

3.2.2 Bioreactor Design

The prototype design consists of a reinforced tank, porous lava rock, a sheet of non-woven, post-consumer recycled fibrous plastic (BioHaven; Midwest Floating Island, St. Paul, MN), a honeycomb shaped geotextile cellular containment material (EnviroGrid; Geo Products, LLC, Houston, TX), a layer of wood chips and a layer of corn cobs (Figure 1). The N removing layers of woodchips and corn cobs were based on specifications developed through extensive laboratory testing (Roser et al., 2018). Three materials were selected for the P-sorbing media, including sieved steel slag, sieved crushed recycled concrete and fragmented limestone. The choice of P-sorbing materials for this experiment was based on products that were readily available, had high P sorption capacity and high hydraulic conductivity. The system was designed for installation adjacent to individual tile outlets along a drainage ditch in order to remediate nitrogen-N and dissolved phosphorus. The number of modules installed at a particular location would be in part determined by the size of the outlet pipe and the desired treatment efficiency (hydraulic residence time and/or nutrient reduction). The controlled field experiment consisted of three replications of three experimental treatments for a total of nine modular bioreactors. All three experimental treatments received the same supplemental carbon (C) source and contained the same denitrifying materials but differed in their P-sorbing material as follows: (i) woodchips + corn cobs + crushed concrete, (ii) woodchips + corn cobs + fragmented limestone, (iii) woodchips + corn cobs + steel slag. The bioreactors were built on site and installed adjacent to a tile drain outlet along a drainage ditch/waterway. Subsurface drainage system discharge was measured in a water level control that was also used to divert water to the modular bioreactors.

The outlet of the bioreactor consisted of an adjustable standpipe which was used to raise or lower the elevation of water in the bioreactor. The stand pipe was also used to divert bioreactor effluent to a flow gauge and to collect water samples for chemical characterization. Each bioreactor had its own dedicated flow gauge. Fixed-time interval samples were collected with automatic water samplers (ISCO, Lincoln, NE).

3.2.3 Experimental design

The controlled field experiment consisted of three replications of three experimental treatments for a total of nine modular bioreactors. The bioreactors were built on site and installed adjacent to a tile drain outlet along a drainage ditch/waterway. A water distribution system was constructed at the field site in order to divert water from the tile drain outlet to each of the nine bioreactors. (Figure 1).

Subsurface drainage system discharge was measured in a water level control (WLC) structure outfitted with a rectangular weir and a pressure transducer connected to a datalogger. The WLC structure was also used to divert water to the modular bioreactors. The outlet elevation of the WLC structure was managed at the highest level possible which created a column of water on the upstream side of the structure. A single 5 cm dia. PVC pipe was installed in the bottom, up-stream side of the WLC structure which then diverted subsurface drainage water through an in-line PVC debris Y-filter to the bioreactors.

The single PVC pipe from the WLC structure was split into three separate distribution pipes each fitted with a paddlewheel flowmeter which was connected to a datalogger. Each of the three distribution pipes routed water to a block of three bioreactors. Flow to a block of three bioreactors was split again into three separate distribution pipes and routed to individual bioreactors in which flow was controlled with a small, PVC ball valve. Flow into the top of the bioreactor was delivered through a section of plastic gutter with holes drilled into the bottom in order to attempt to distribute water as evenly as possible across the surface of the bioreactor.

The stand pipe was also used to divert bioreactor effluent to a flow gauge and to collect water samples for chemical characterization. A tipping bucket flow gauge with approximately 4.0 L capacity per tip was used to measure bioreactor discharge volume and rate. Each bioreactor had its own dedicated tipping bucket. Fixed-time interval samples were collected with automatic water samplers (ISCO, Lincoln, NE). Tipping buckets were made from stainless steel. The volume of each tip was calibrated in the laboratory at a series of known flow rates, ranging from 1.2 to 18.9 Lpm. Tips were counted using magnetic switches located on the tipping bucket.

An acetate supply system was constructed which consisted of a 1,136 L black plastic tank, a peristaltic pump, datalogger and solar panel (Figure 1).

3.2.4 Carbon source biostimulation

Denitrification is mainly accomplished by heterotrophic bacteria and is strongly dependent on the availability of organic carbon, which serves as an energy source and electron donor of the denitrification process. Wood products have been commonly used as a C source in bioreactors used to treat agriculturally derived nitrate because of their availability, low cost, favorable hydraulic conductivity properties and relatively high C:N ratio (100-300:1). One of the advantages of a relatively high C:N ratio is that materials like wood are stable and do not require frequent replenishment unlike relatively low C:N ratio, labile carbon sources such as corn cobs which may be depleted more rapidly. One disadvantage of having a relatively high C:N ratio is that the availability of labile carbon could limit denitrification rates.

In biological treatment processes which promote denitrification, an external carbon source is frequently added as an electron donor in order to stimulate the process of denitrification. Some carbon sources which have been used to promote denitrification include: acetate ($C_2H_3O_2$), glucose (corn syrup, $C_6H_{12}O_6$), Lactate ($C_3H_6O_3$), sucrose (molasses, $C_{12}H_{22}O_{11}$), ethanol (C_2H_6O) and soybean oil ($C_{18}H_{32}O_2$). Commercially available carbon substrates have some characteristics that are similar, including some degree of degradability and solubility. They differ in the speed with which the material becomes bioavailable and is degraded, in the complexity of their composition and in their cost. A substance's physical characteristics and molecular composition determine whether they serve to provide a slow, low concentration release of soluble carbon or are immediately available at a higher concentration. In this project, potassium acetate (KCH_3CO_2) was used as the external carbon source. The concentration was based on reducing nitrate concentration at 20 mg/L for a subsurface drain flow rate of 34 Lpm in the 5 cm-PVC water mainline; the stochiometric C:N ratio was 0.82 (Lew et al., 2012). Nitrate reduction only requires mildly reducing conditions (E_h^o, -82 to -119 mV), so it is important to maintain redox potential in the optimum range to minimize electron donor scavenging for use in supporting Mn(IV), Fe(III) or sulfate reduction reactions. The process of biological P removal by phosphorus accumulating organisms (PAO) is also strongly dependent on anaerobic conditions that promote the consumption of readily biodegradeable organic carbon. According to Morling (2001), biological phosphorus removal declined when the availability of readily biodegradeable organic carbon was low in a sequencing batch reactor for wastewater treatment. A type of PAO called denitrifying PAO can combine P removal and denitrification into one process using the same organic carbon substrate (Meinhold, et al., 1999; Yuan and Oleszkiewicz, 2008; Flowers et al., 2009).

Data collection periods

The experiment was divided into two phases. The first phase lasted for two months, May and June 2016, and consisted of a bioreactor calibration phase in which no acetate was added to the subsurface drainage water entering the bioreactors. This phase allowed the constituent materials in the bioreactors to settle and equilibrate and for the formation of biofilms on bioreactor components. The second phase consisted of dosing the bioreactors with acetate which lasted 11 months beginning July 2016 through July 2017. During 2016, drain flow was relatively constant beginning in April and continuing through early December. Bioreactor monitoring began in early May and mean discharge rate was analyzed on a daily basis. Drainage system and bioreactor outflow and water analysis operations were terminated in December when air temperatures dropped to at or below freezing for an extended period of time. Eight (125 mL) water samples were collected at three hour intervals on a daily basis, preserved and refrigerated at 4° C for analysis of nitrate (NO_3-N) by cadmium reduction and dissolved reactive phosphorus (DRP) by ascorbic acid colorimetric analysis using flow injection analysis (Lachat Quikchem 8500, Hach, Loveland, CO.). Oxidation-reduction potential and dissolved oxygen levels, as indicators of reducing conditions within the bioreactors, were collected using a YSI 556 handheld multiparameter instrument (YSI Inc. Yellow Springs, OH) through the standpipe of each bioreactor. The second phase of the experiment consisted of the bioreactor treatment phase in which dissolved potassium acetate was added to the subsurface drainage water entering the bioreactors.

Statistical analysis

Statistical analysis was conducted to identify significant differences between N and P-treatments with five dependent variables: HRT, NO_3-N concentration, NO_3-N load, DRP concentration and DRP load using R statistical package (R Core Team, 2017). Three factors were introduced into the model (3-way ANOVA) with their respective paired interactions: N and P-treatment, flow group, and months. All analyses were conducted at a significance level of $P \leq 0.10$ to assess which treatments were statistically different. The interaction terms in the model allow the comparison between monthly mean values and their statistical significance using Tukey HSD test. Each model was checked for normality by examining Q-Q plots of model residuals for each dependent variable.

3.3 Results and Discussion

Precipitation

The weather conditions during 2016 would be considered abnormal. Annual precipitation at the SWROC was 41% above average, 960 mm compared to the 30-year average 670 mm. Mean long-term growing season precipitation, April through September, was 435 mm whereas during 2016 a total of 646 mm was observed. Annual precipitation for 2017 was 765 mm, which was 13% above the 30-year normal average, with low amounts during the first three months of the year. Growing season rainfall was 578 mm in 2017 (April to September), 25% higher than that of the long-term average (Figure 2).

Discharge Rate and Hydraulic Residence Time

Statistical analysis did not reveal any significant differences (p=0.359) for mean discharge rate among bioreactors during 2016. Between May and December, discharge rate ranged between 1.07 to 5.96 m^3 d^{-1}. Variability in drain flow rate from the bioreactors over time was attributed to the impact of source water availability and apparent changes in the physical features of P-treatment materials that may have affected water percolation through the bioreactors. Source water supply was affected by several factors including variation in monthly precipitation, evaporation at the soil surface, water uptake by plants and the observation that the fragmented limestone and steel slag materials began to conglomerate. For example, during mid to late summer, discharge rate decreased when crop growth and crop water uptake were high. The P-treatment materials, fragmented limestone and steel slag, appeared to have changed characteristics from small individual aggregates into cohesive masses which may have reduced bioreactor discharge rate. The cause of this conglomeration was unknown but warrants further investigation. One hypothesis was that dissolved minerals were precipitating and effectively “cementing” individual P-treatment materials together. The P-treatment materials were left in place for both 2016 and 2017. Subsurface drainage restarted in late March 2017. There were small differences in mean daily discharge rate between P-treatments and months but none of them were significant. During 2017 all P-treatment materials were gently mixed on a weekly basis in order to ensure that the materials maintained their granular structure so as not to inhibit flow through the bioreactors.

Statistical analysis of mean hydraulic residence time (HRT) of the modular bioreactors did not reveal any significant differences (p=0.203) among P-treatments during 2016 (Table 1). The target HRT for the modular bioreactors was one hour. During 2016, mean HRT (3.7 h) was always greater than the one-hour target HRT. During the growing season, the shortest HRT (1.8 h) occurred during June and the longest HRT (7.25 h) occurred during August. The longest HRT (7.63 h) of the 2016 monitoring campaign occurred during November. The data indicate that the HRT for the Crushed Concrete and Limestone P treatments were similar across all periods. In contrast, the HRT for the Steel Slag treatment was more variable and frequently had the longest HRT. During 2016, the steel slag HRT ranged from 1.1 to 1.8 times greater than the other two treatments. As noted earlier the observed changes in HRT during 2016 for the fragmented limestone and steel slag could be due in part to a conglomeration of the individual aggregates into a conglomerate with lower hydraulic conductivity. Statistical analysis for HRT resulted in a significant difference across P-treatments during July 2017 (p-value = 0.0004); post-hoc Games-Howell method was used for the test since the data was strongly non-normal, unbalanced, and unequal variance. The overall mean HRT for all treatments was 3.4 h during 2017.

Oxidation-Reduction and pH Values

Oxidation-reduction potential (Eh) and pH were measured on a monthly basis during 2016 and bi-weekly during 2017. The pH of the bioreactors was relatively stable during 2016 and 2017, remaining between 7.0 and 7.4. During 2016, Eh was +200 mV in May and early June before acetate was added to the bioreactors (Figure 3). The Eh dropped briefly to -300 mV during July after acetate was added. By September Eh had risen to approximately -50 mV and by December Eh was observed to be about +200 mV. Negative Eh values represent high electron activity and intense anaerobic conditions dominated by anaerobic microbial metabolism. Typical electron acceptors under these conditions are sulfate and carbon dioxide. During 2017, measured Eh among bioreactors fluctuated between +150 and +200 mV from the end of March through the first week of June. During the second week of June, Eh dropped to approximately +50 mV and then gradually rose to about +125 mV through July. There was a noticeable increase in water temperature from 20°C to 27°C in early June which coincided with the decline in Eh. Positive Eh values represent low electron activity and aerobic to moderately anaerobic conditions dominated by aerobic and facultative microbial metabolism. Under the conditions observed in the field during 2017 the data suggest that the bioreactors were under moderately reduced and moderately anaerobic conditions. Typical electron acceptors under these conditions are nitrate, manganese and iron. Acetate was added continuously to the bioreactors during 2017. Measured rates of denitrification demonstrated that denitrification can occur over a wide range of Eh values ranging from +400 to -100 mV (Seo and DeLaune, 2010). In a laboratory experiment to evaluate the relative contribution of fungi and bacteria on denitrification, Seo and DeLaune (2010) showed that denitrification was dominated by fungi under moderately reducing to weakly oxidizing conditions (Eh > +250 mV), whereas it was dominated by bacteria under strongly anaerobic reducing conditions (Eh < -100 mV). Oxidation-reduction potential values below -250 mV may result in biological phosphorus release, sulfide formation and methane production which could lead to environmental pollution (Karanasios et al., 2010).

Phosphorus Removal and Reduction

During 2016, dissolved reactive P (DRP) removal rates (PRR) among the three P-treatment materials ranged between 80.0 and 133 mg P m^3 d^{-1}. Crushed concrete and steel slag had similar PRRs but fragmented limestone exhibited a smaller PRR (Table 2). Dissolved phosphorus loads followed the same trend as PRR in 2016 and ranged from 10.0 to 16.8 g P. Reinhardt et al. (2005), in an analysis of P retention from small constructed wetlands, reported P retention of 1.1 g P m^2 yr^{-1}.

Phosphorus concentrations range between 50 and 150 ug L^{-1} during 2016 and 2017 (data not shown). Statistical analysis revealed no significant differences for monthly DRP load among P treatments for 2016 ($p = 0.386$) (Table 2). No acetate was added to the bioreactors during May and June 2016. Mean source P and P-treatment P loads for 2016 were 5.4 g and 1.7 g P, respectively. Changes in mean monthly bioreactor DRP load were attributed to a combination of differences in mean bioreactor HRT, water temperature, pH and Eh. For example, a combination of low DRP in the source water, coupled with relatively long HRT, on average 6.7 h, and relatively low discharge rate, on average 1.2 m^3 d^{-1}, resulted in the lowest mean DRP load (0.75 g) across P-treatments during August 2016. It was apparent from mean DRP loads during September, October and December that some of the P-treatment bioreactors were acting as a source of DRP rather than a sink (Table 5). Husk et al. (2018) reported similar results for DRP in two out of three years for horizontal flow woodchip bioreactors (n=3) and one out of three years for a mixed media bioreactor (n=1) containing woodchips plus activated alumina and gravel in south-central Quebec, Canada. In this study, no supplemental carbon source was added to the bioreactors.

There is a distinct decline in DRP load after the addition of acetate beginning in July 2016 (Table 2). This was unexpected since the main mechanism for P removal was expected to be via adsorption and precipitation to the P-treatments materials. Research conducted to evaluate nutrient removal efficiency from wastewater treatment plants showed that biological P removal was performed by specific phosphate-accumulating organisms (Lesjean et al., 2003). Researchers demonstrated that biological P removal was possible when appropriate conditions were insured including anaerobic conditions plus supplemental carbon (acetate) (Wang et al., 2002; USEPA, 2009). Biological phosphorus removal is achieved by phosphorus accumulating organisms in a zone free of nitrates and dissolved oxygen (anaerobic zone) (USEPA, 2009). Similar conditions to these existed in the bioreactors in July 2016. During 2017, ANOVA for DRP concentration by P-treatment did not show any significant differences in mean values ($p = 0.56$) (data not shown). Statistical analysis did not show any significant differences between the mean DRP load for the P-treatment bioreactors ($p = 0.582$) (Table 2). The mean monthly DRP load from the bioreactors was lower than that of the source water for the months of March, June and July. We hypothesized that the low monthly source water P concentration in April and May was driving the desorption of P from crushed concrete, fragmented limestone, and steel slag materials which resulted in DRP load outputs higher than DRP load inputs during these months. It is also likely that P was released from the phosphate accumulating organisms present in the bioreactors under the aerobic conditions that existed during spring 2017.

On average, the highest phosphorus load reduction occurred during June with values ranging between 28 and 57%; the lowest nitrate load reduction, 25 %, occurred in March. Overall, when the P sorbing material was acting as a sink it was quite effective at removing P. Cold climate limitations were less important for P removal than for N removal.

Nitrate Removal and Reduction

During 2016, the range in N removal rates (NRR) among the three P-treatment materials ranged between 15.4 and 42.4 g N m^3 d^{-1}. Crushed concrete and fragmented limestone had similar NRRs but steel slag exhibited greater NRR (Table 2). Data from May and June 2016 were excluded from the analysis because no acetate was added during that time. The goals of the initial no-acetate period were to establish anaerobic conditions in the bioreactors and for the colonization of microorganisms within the bioreactors. In 2017, NRR (range, 9.1 to 10.0 g N m^3 d^{-1}) was similar among all P-treatments but smaller in magnitude than in 2016. These NRRs compare favorably with the range reported in a meta-analysis of bioreactor studies, 2 to 22 g N m^3 d^{-1} (Addy et al., 2016), and second year NRR typically drops from the first year (David et al., 2016).

Previous studies have shown that several factors, including temperature, pH, dissolved oxygen (DO) level, organic carbon species, and carbon to nitrogen (C:N) ratio can affect denitrification efficiency (Strong et al., 2011; Kraft et al., 2014). Nitrate removal efficiency increases with increasing C:N ratio (Chen et al., 2017). Consequently the higher monthly C:N ratios in 2016 would be expected to increase nitrate removal compared to the lower monthly C:N ratios in 2017. Declines in N load and NRR in 2017 were attributed to an intentional decrease in the stoichiometric C:N ratio between C (acetate) added to bioreactors and N in the drainage water feeding the bioreactors to 0.35, approximately 40% of the recommended value of 0.82. The reason for this reduction in 2017 was to minimize the production of extracellular polymeric substances (EPS) discovered in the source water supply piping periodically in 2016. Nitrate-N removal rates observed in this field experiment were like those reported by others investigating denitrification in the laboratory using woodchips over a range of HRTs (7.2 to 51 h) (Hua et al., 2016; Christianson et al., 2017). In this experiment, the mean HRT across all P-treatments and both years was 3.6 h, which was half of the shortest reported HRT by Christianson et al. (2017).

There were no significant differences ($p = 0.186$) for mean monthly NO_3-N concentration among P treatments in 2016 (data not shown). Typical N concentrations found in tile drainage waters during the experimental period ranged between 1 and 24 mg L^{-1}. In 2016, NO_3-N load reduction during the period prior to acetate addition was minimal (Table 4). The apparent lack of denitrification activity during this period could be due to the absence of or insufficient populations of denitrifying organisms. Mean bioreactor NO_3-N load was generally uniform and behaved similarly across P treatments regardless of month. The largest mean bioreactor NO_3-N loads observed occurred during June and the smallest during August (Table 4). The June data reflect the relatively large discharge rate during June, coupled with a relatively short HRT and no acetate addition (Table 1). In the case of these modular bioreactors, mean bioreactor NO_3-N load for the different months was attributed to a combination of differences in mean bioreactor NO_3-N concentration, discharge rate and HRT. During 2016, the mean NO_3-N load reduction from the bioreactors was 30% and ranged between -13 and 99%. The largest percent NO_3-N load reduction from the bioreactors occurred during August when HRT was long, on average 6.7 h, and discharge rate was low, on average 1.2 m^3 d^{-1}. When conditions turned cold, during the months of November and December, the range in NO_3-N load ranged between -13 and 42%. The NO_3-N load increased from the crushed concrete and fragmented limestone bioreactors and decreased from the steel slag bioreactors. This was an unexpected result. It was hypothesized that N may have been released from microbial cells when ambient temperatures declined but the lack of consistency among the P-treatment materials, specifically for the steel slag, remains unexplainable.

No statistical difference between P-treatments was observed for monthly nitrate concentration in 2017 ($p = 0.904$). Nitrate concentrations from all bioreactors ranged between 10.0 and 18.3 mg L^{-1} (data not shown). Statistical analysis did not show any significant difference between mean NO_3-N load by P-treatment for the bioreactors ($p = 0.643$). The mean monthly NO_3-N load from the bioreactors remained lower than that of the source water except for the fragmented limestone P-treatment for the month of May in 2017 (Table 4). During 2017, the mean NO_3-N load reduction from all bioreactors was 14% and ranged between -6 and 41%. The largest percent NO_3-N load reduction from the bioreactors occurred during July (23%) when HRT was long, on average 6.8 h, and discharge rate was low, on average 1.2 m^3 d^{-1}. An increase in water temperature of 7^{o}C contributed to a sharp drop in Eh in early June. The net effect was an increase in NO_3-N load reduction, from 10% in May, to 17% in June and 23% in July. As noted previously, the combination of Eh values conducive to denitrification coupled with declining discharge rates and longer HRT values resulted in the increasing NO_3-N load reduction between May and July for the crushed concrete and steel slag bioreactors. The fragmented limestone treatment exhibited the opposite behavior showing the highest NO_3-N load reductions in March and April follow by a gradual decline. At the beginning of 2017, the stoichiometric C:N ratio was reduced from the 0.82 target to 0.35, approximately 40% of the recommended value. We hypothesized that due to a carbon limitation in the bioreactors during 2017, the magnitude of NO_3-N load reduction declined from 2016 to 2017.

Previous field research demonstrated that under low temperatures (< 10˚C) NRR was low where cold conditions were common (David et al., 2016). Roser et al. (2018) demonstrated that a 25% reduction in NO_3-N (20 mg L^{-1} to 15 mg L^{-1}) was possible for HRT of < 2 h when supplemental C was added to a woodchip bioreactor under laboratory conditions. Based on the denitrification data collected during 2016 and 2017, under field cold conditions, a classification of HRT was produced (Table 5). Based on the current configuration of HRT and supplemental C addition, an HRT of less than 2 hours may generate very little nitrate load reduction during cold portions of the year in northern climates. In the presence of adequate C, low temperature was considered to be the main inhibitor of denitrification.

3.4 Conclusions

It is essential to develop effective approaches to reducing offsite impacts of agricultural drainage in northern latitudes which experience cold conditions. This field experiment assessed the impact of a novel two-phase bioreactor designed for removing N and P from agricultural subsurface drainage water. Modular bioreactors were constructed using mixed woodchips plus corn cobs for facilitating denitrification, plus either crushed concrete, steel slag or fragmented limestone for P sorption. Experimental modular bioreactors were installed adjacent to an existing drainage ditch/waterway. Modular bioreactors showed the ability to reduce N load from agricultural drainage water through denitrification and P load through sorption and biological P removal. It was apparent that denitrification and P removal efficiency were variable and declined under cold conditions during late winter-early spring and again during late autumn-early winter. The variability observed in N and P removal during cold periods of the year was attributed to the inhibition of microbial processes affecting biological denitrification and biological P removal.

In this experiment, HRT was on average less than 4 h which was a considerable improvement over previous bioreactor designs. Nitrate removal rate was related to HRT in the bioreactor, but could be enhanced by the addition of acetate; longer retention time resulted in a greater removal of N. The addition of acetate as a biostimulant, in order to promote denitrification, improved NRR compared to previous research. All three P sorbing materials performed adequately for removing P from drainage water during 2016. During the second year of data collection, PRR of the P-treatment materials declined. It was also observed that there was a potential adverse effect due to DRP loss as a result of desorption from P-treatment materials in late winter-early spring. Additional considerations are necessary in order to develop strategies for the management of P-treatment materials in order to minimize unintended release of P into the environment. During a portion of these experiments the P-treatment materials acted as a source of P rather than a sink for P removal. Additional research is necessary to determine the longevity of the P removal materials and to optimize bioreactor performance under cold conditions.

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Table 1. Mean hydraulic residence time (hour) for bioreactors grouped by P-treatment.

Date	P-treatment					
	Crushed concrete (n=3)		Limestone (n=3)		Steel slag (n=3)	
	2016	2017	2016	2017	2016	2017
March		2.39 a		2.90 a		3.27 a
April		2.59 a		2.45 a		2.71 a
May	2.52	2.34 a	2.21 a	1.87 a	1.66 a	1.91 a
June	1.92 a	3.08 a	2.19 a	2.48 a	1.33 a	2.84 a
July	3.80 a	8.59 a	4.04 a	5.28 b	2.92 a	6.29 b
August	7.25 a		5.54 a		7.35 a	
September	2.41 a		2.20 a		2.10 a	
October	5.76 a		5.31 a		6.66 a	
November	3.84 a		4.38 a		7.63 a	
December	1.67 a		1.68 a		2.66 a	

Means followed by the same lowercase letter within a given row, by year, are not significantly different at $\alpha = 0.10$.

Table 2. Mean bioreactor volume, duration of data collection, mean nitrate and DRP removal rates in g of nitrate-N or mg DRP removed per m^3 of woodchips and P-filter media per d during 2016 and 2017.

	P-treatment					
	Crushed concrete		Limestone		Steel slag	
	2016	2017	2016	2017	2016	2017
Mean bioreactor vol., m^3	0.838		0.825		0.796	
	Nitrate					
Duration, d	99	119	99	119	99	119
Removal rate, g N m^{-3} d^{-1}	15.6	9.1	15.4	10.0	42.4	9.8
	DRP†					
Duration, d	151	119	151	119	151	119
Removal rate, mg DRP m^{-3} d^{-1}	133	-1.1	80	-0.10	122	-0.36

†Negative removal rate denotes desorption or biological release of phosphorus.

Table 3. Mean dissolved reactive phosphorus input and output loads (g) from bioreactors grouped by P-treatment. Long-term mean monthly air temperature (C) and departure from mean air temperature for 2016 and 2017.

				P-treatment Output							
				Input source		Crushed concrete (n=3)		Limestone (n=3)		Steel slag (n=3)	
	30-yr mean	2016	2017	2016	2017	2016	2017	2016	2017	2016	2017
March	-0.7	4.4	0.4		1.01 a		0.53 a		0.52 a		0.75 a
April	6.9	1.7	1.5		1.91 a		4.16 a		3.31 a		2.99 a
May	14.3	0.4	-0.7	7.14 a	2.49 a	4.16 a	5.64 a	6.57 a	5.86 a	(8.67) a	4.17 a
June	20.2	1.1	0.5	15.69 a	5.35 a	10.63 a	2.27 a	10.70 a	3.84 a	14.28 a	3.48 a
July	22.2	0.0	0.2	10.15 a	2.43 a	2.47 a	1.07 a	2.57 a	1.72 a	3.88 a	1.59 a
August	20.6	0.8	-1.8	1.87 a		0.72 a		0.88 a		0.66 a	
September	16.4	1.3	1.3	2.99 a		2.89 a		(3.44) a		(3.48) a	
October	8.9	1.1	0.5	2.22 a		(2.53) a		(2.79) a		1.76 a	
November	0.4	5.0	-0.6	1.93 a		1.92 a		1.72 a		0.99 a	
December	-7.1	-1.1	0.0	1.00 a		(1.53) a		(3.46) a		0.76 a	

Means followed by the same lowercase letter within a given row are not significantly different at $\alpha = 0.10$. Means enclosed in parenthesis indicate desorption or biological release of P.

Table 4. Mean NO_3-N input and output loads (kg) from bioreactors grouped by P-treatment. Long-term mean monthly air temperature (C) and departure from mean air temperature for 2016 and 2017.

						P-treatment Output					
				Input source		Crushed concrete (n=3)		Limestone (n=3)		Steel slag (n=3)	
	30-yr mean	2016	2017	2016	2017	2016	2017	2016	2017	2016	2017
March	-0.7	4.4	0.4		0.252		0.237 a		0.198 a		0.247 a
April	6.9	1.7	1.5		1.671		1.485 a		1.358 a		1.387 a
May	14.3	0.4	-0.7	1.912	2.185	1.440 a	1.932 a	1.754 a	2.309 a	1.768 a	2.076 a
June	20.2	1.1	0.5	3.311	1.576	2.873 a	1.241 a	2.374 a	1.493 a	2.821 a	1.194 a
July	22.2	0.0	0.2	1.202	0.638	0.587 a	0.377 a	0.592 a	0.629 a	0.477 a	0.463 a
August	20.6	0.8	-1.8	0.098		0.001 a		0.003 a		0.001 a	
September	16.4	1.3	1.3	0.744		0.582 a		0.572 a		0.572 a	
October	8.9	1.1	0.5	0.520		0.420 a		0.452 a		0.315 a	
November	0.4	5.0	-0.6	0.494		0.557 a		0.514 a		0.283 b	
December	-7.1	-1.1	0.0	0.328		0.371 a		0.367 a		0.247 a	

Means followed by the same lowercase letter within a given row, for a given year, are not significantly different at α = 0.10.

Table 5. Class of hydraulic residence time (HRT), percent nitrate load reduction, and percent nitrate load reduction range per category of hydraulic residence time from all P-treatment bioreactors during 2016 and 2017

HRT, hour	Nitrate load reduction, %	Range Nitrate Load Reduction, %
< 2	7.5	0 - 15
2 - 3	19	3 - 60
3 - 4	24	9 - 51
> 4	51	13 - 99

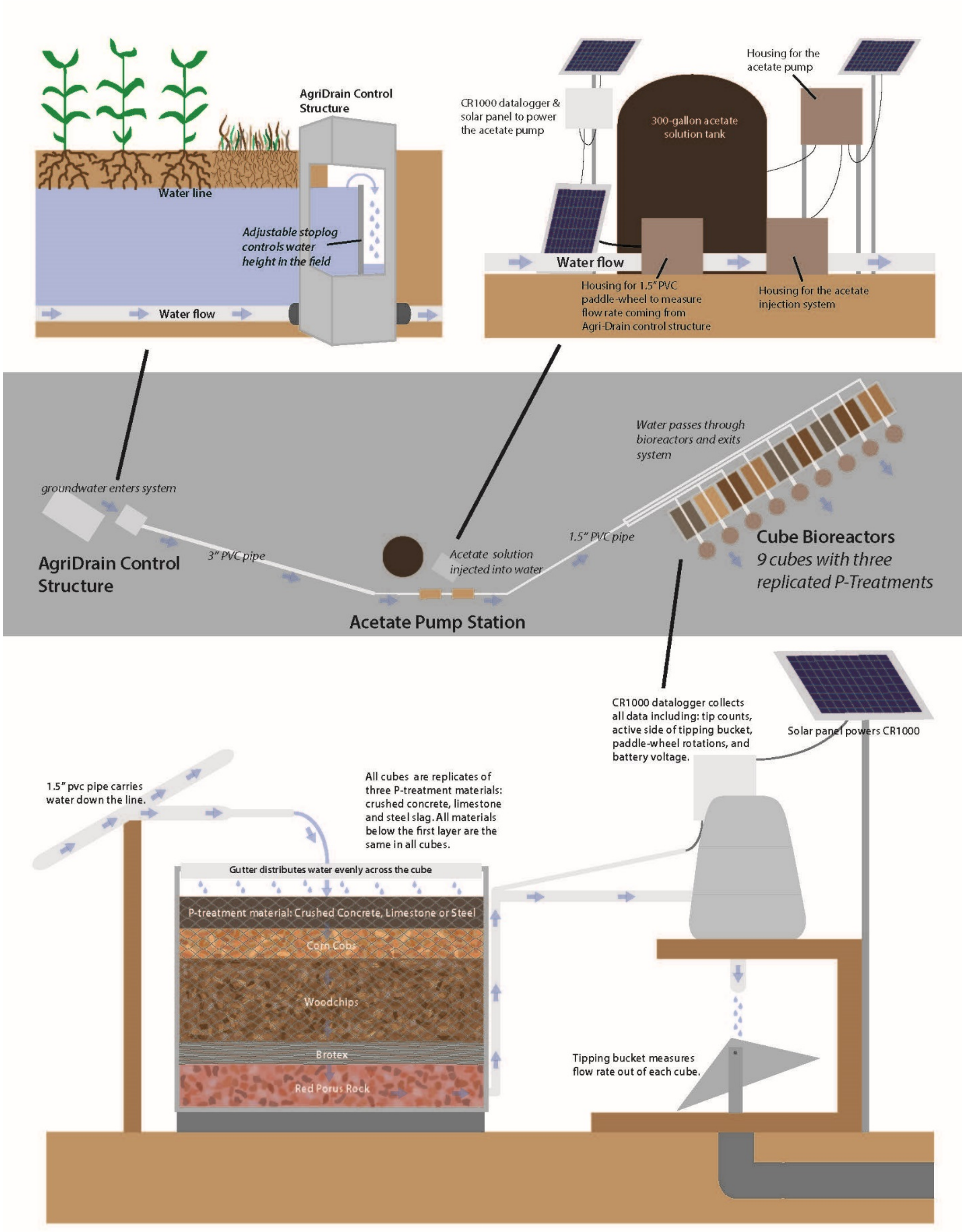

Figure 1. Experimental design and components of novel N and P removal modular bioreactors.

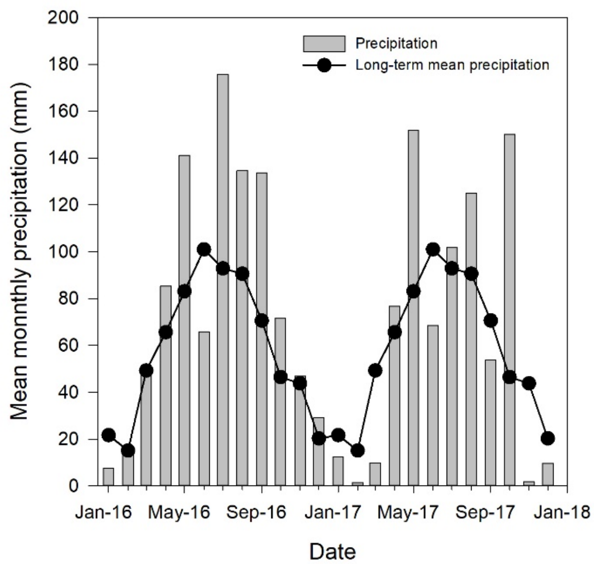

Figure 2. Mean monthly precipitation near Lamberton, MN during 2016 and 2017.

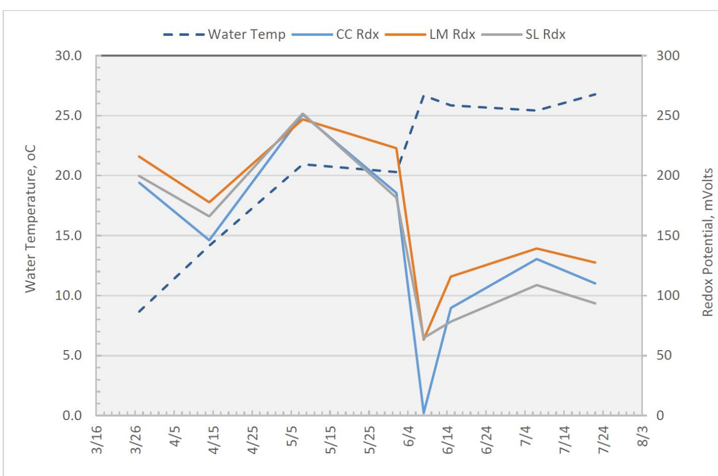

Figure 3. Oxidation-reduction potential and average water temperature in the bioreactors for 2017.

Chapter 4

Simulating individual farms in the Cottonwood River Basin for evaluating water quality benefits from the Minnesota Agricultural Water Quality Certification Program (MAWQCP).

4.1 Introduction

The Minnesota Department of Agriculture's Minnesota Agricultural Water Quality Certification Program (hereafter referred to as the MAWQCP or the Certification Program) aims to improve Minnesota's water quality by providing farmers with incentives and technical support to adopt improved management practices on their farms. More specifically, farmers who enroll in the program gain some freedom from potential legislative uncertainty; they are deemed to be in compliance with environmental laws for a period of 10 years (even if environmental laws change during that time). Farmers in the Certification Program also get priority for technical assistance and cost sharing programs to help achieve improved management practices on their farms. At the time of writing, the program has over 850 producers in Minnesota enrolled.

While there have been over 580,000 acres of farmland successfully enrolled in the Certification Program, it is difficult to quantify the water quality benefits of these changes in farm management because the program does not include an on-site monitoring component. The diffuse nature of nonpoint source pollution and Minnesota's weather variability make it difficult to detect water quality benefits that may arise from changes made to a single farm. That does not necessarily mean that these conservation practices do not work, however, and it is important to develop methods to estimate the environmental benefits that may accumulate from the actions of individual farmers. In order to quantify the water quality benefits of the MAWQCP, we employed a modeling approach to estimate water quality outcomes for a suite of Certified Farms located in the Cottonwood River Basin.

4.2 Material and Methods

4.2.1 Water Quality Modeling

A watershed scale model was constructed for the Cottonwood River Basin in order to track water, sediment, and nutrient export under varying conditions of weather and farm management. The SWAT model (Soil and Water Assessment Tool) operates on a daily time-step and relies on a mixture of empirical and process-based approaches to determine farm management impacts on water quality under varying environmental conditions (Neitsch et al., 2011; Arnold et al. 1998; Arnold and Fohrer 2005; Gassman et al. 2010). The key spatial inputs for the SWAT model are soils, land cover, and topography (slope). Spatial datasets were intersected using GIS software (ArcGIS 10.4.1) to generate the basic computational unit of the model: the Hydrologic Response Unit (HRU). Land cover data are based on the 2011 National Land Cover Dataset (NLCD; Homer et al., 2015) which includes data for broad land classes including cultivated cropland. In the Cottonwood River Basin, cultivated land is assumed to be in a two-year rotation of corn and soybeans, the dominant practice across much of the agricultural Upper Midwest. Slope data are derived from a digital elevation model (pixel size = 30m) from the National Elevation Dataset (U.S. Geological Survey 2016). County-level soils data (SSURGO) were downloaded from the Web Soil Survey (Soil Survey Staff). In order to prepare the watershed scale model for this study, additional steps were taken to represent the fields in the watershed that are enrolled in the MAWQCP. This was accomplished by obtaining GIS data containing the polygons of field boundaries from the Minnesota Department of Agriculture (MDA) and intersecting them with the base land cover dataset. An arbitrary identifying number was assigned to each MAWQCP farm so that model inputs and outputs could be tracked separate from the broader set of modeled data. Farm-specific information about management practices were obtained by MDA personnel. This included information about (1) what practices were in place prior to Certification Program enrollment, and (2) what new practices were initiated as a result of enrollment in the Certification Program. This approach allowed us to evaluate how Certification Program farmers compared against a more general set of baseline management for the watershed as well as quantify additional changes to environmental outcomes as a result of enrollment in the Certification Program. The final model of the Cottonwood River Basin contained 2,028 HRUs including 291 that collectively represent the farm fields of nine different landowners enrolled in the Certification Program in the Cottonwood River Basin.

The SWAT model also requires daily weather inputs of precipitation, temperature, relative humidity, wind speed, and solar radiation. Weather input data were gathered and applied to the model following methods described in Antolini et al., (2019) and Dalzell and Mulla, (2018). Briefly, NEXRAD data were used for precipitation and the Climate Forecast System Reanalysis dataset (CFSR) (Fuka, et al. 2014) was used for remaining weather data. The spatial weather data (NEXRAD and CSFR) are represented in the model as a synthetic network arranged on a grid (NEXRAD: 4km spacing; CSFR: roughly 30km spacing). SWAT operates at a daily timestep and key model outputs are stream flow as well as sediment and nutrient export. For this study of the MAWQCP, we focused on farm-scale outputs of sediment, phosphorus and nitrogen (as NO3-).

We began with a baseline scenario to simulate current conventional agricultural practices, in which corn and soybean crops are grown in a two-year rotation typical for the Upper Midwest. Tillage, fertilizer application, and planting/harvest dates are based on farmer surveys (Minnesota Department of Agriculture 2007) and feedback from local stakeholders and commodity groups and from prior modeling work in the Minnesota River Basin (Dalzell et al. 2012; Pennington et al. 2017; Dalzell and Mulla 2018). Following model calibration and validation, we applied alternative management practices based on information about farm-specific practices before and after Certification Program enrollment. Evaluation of alternative management scenarios to achieve water quality goals are typically based on relative differences in nutrient export compared to the baseline management scenario. In the case of this study, we have the benefit of additional farm-specific management that was in-place both before and after enrollment in the Certification Program. As such, differences in SWAT outputs across farms result from both the nature of Certification Program practices as well as the physical differences in HRU characteristics, namely: soils, land cover, and topography. While the agricultural landscape of the Cottonwood River Basin seems homogenous, minor differences in soils and topography can influence the initial environmental footprint of a field as well as the relative changes in water quality outputs that are achieved through enrollment in the Certification Program. Results presented here reflect average annual values based on 10 years' worth of model outputs from 1, Jan 2004 through 31, Dec 2013.

4.2.2 Simulated Conservation Practices

Water and Sediment Containment Basins (WASCOBs)

Water and Sediment Containment Basins were simulated in the SWAT model by following the approach employed by Her (2016) and Waidler (2011). This approach relies on simulating terraces within SWAT as a means to represent the use of earthen embankments to retain water and sediment. More specifically, curve numbers were decreased by 9% to reflect tillage along a contour and the USLE P factor was adjusted to a value of 0.5 based on the assumption that WASCOBs were placed in the middle of field watercourses.

Grassed Waterways

Grassed waterways were simulated with the conservation operations scheduler within the SWAT model. The width of the waterways was assumed to be 10m while the slope and length were based on the dimensions and topographic characteristics of each individual HRU (the model default setting).

Reduced Tillage

Conservation Tillage and No-Till were simulated in the SWAT model by replacing the baseline tillage practice (chisel till) with generic model database practices for Conservation Tillage (less efficient soil mixing) and No-Till (no soil mixing) that resulted in greater levels of crop residue on the soil surface.

Cover Crops

Winter rye was simulated to be planted immediately following fall harvest of corn or soybeans. Cover crop growth occurred through the winter as allowed by environmental conditions and the rye was terminated in the spring just prior to planting corn or soybeans.

Fertilizer Management

Rates of applied fertilizer were decreased from the baseline scenario by 25% and changes in nutrient export were expressed as percent decrease relative to the baseline scenario.

Blind Tile Inlets for Subsurface Tile Drainage.

While the SWAT model does have the ability to simulate subsurface tile drainage, it is not able to simulate the environmental impacts of open surface inlets. Some practices supported as part of the MAWQCP program include the replacement of surface tile intakes with blind inlets. In order to include those water quality benefits in this analysis, we rely on a field-based study by Feyereisen et al. (2015) in order to predict water quality differences between open and blind tile inlets. In that work, the authors showed that replacing surface inlets with buried blind inlets was an effective management practice for reducing sediment and phosphorus loss from agricultural fields. We selected more conservative values of sediment and phosphorus reduction observed by Feyereisen et al. (2015) study in order to estimate the performance of that management practice in MAWQCP farms. We estimate that replacement of surface intakes with blind inlets for tile drainage systems will reduce the loss of sediment and total phosphorus by 59 and 57%, respectively (c.f. Feyereisen et al., 2015). Replacement of surface inlets with blind inlets is assumed to not have any impact on NO3- losses from agricultural fields.

4.3 Results and Discussion

Caveats about interpreting model results
Results presented here reflect estimated annual average export of sediment, phosphorus, and nitrogen from selected farm fields in the Cottonwood River Basin for the 10-year period from 1 Jan 2004 through 31, Dec 2013. There are also important non-field sources of sediment and phosphorus that originate from failing streambanks and eroding river bluffs. While these non-field sediment and phosphorus sources can be linked to field-scale processes because of farm management impacts on water yield, those non-field sources of sediment and phosphorus are not tracked in this study of export from MAWQCP farms. Simulated values are dependent upon accurate input data about soils, land use, daily weather, and farm management. Model-simulated sediment and nutrient export values fall within in the range of what has been observed in plot- and field-scale studies and these results are helpful for characterizing farm-level responses to changes in management. While farm-level results from this study reflect a greater level of specificity about farm-level management than is typical for most watershed scale studies, they are not an absolute measure of actual export from farms in the Cottonwood River Basin. Nevertheless, results presented here provide insight into the expected water quality outcomes from varying farm management practices and they also provide a more direct measure of water quality changes that result from enrollment in the MAWQCP. For purposes of representing results from Certification Program farms in this study, farms were assigned arbitrary ID numbers to keep data organized but no additional personal identifying information is included here.

4.3.1 Scenarios
For each MAWQCP farm in the Cottonwood River Basin, results from three different scenarios were collected for comparison:

1. a general model baseline scenario based on typical management in South-Central Minnesota,
2. farm-specific management based on practices in place prior to enrollment in the Certification Program, and
3. farm-specific management based on new practices that were implemented in response to enrollment in the MAWQCP.

For each farm, we compare estimated Certification Program water quality outcomes against both baseline conditions and conditions prior to enrollment in the MAWQCP. This allows us to quantify direct water quality benefits of the Certification Program as well as provide a more general comparison of participating farmers against more typical watershed conditions. These results are summarized in Figures 1-3 for sediment, phosphorus, and NO_3^-, respectively.

The farmers who enrolled in the MAWQCP were already employing a variety of conservation practices on their farms before certification. These included: fertilizer management, reduced tillage, grassed waterways, edge of field filter strips and cover crops. In comparison to the general baseline management scenario, MAWQCP farmers in the Cottonwood River Basin were exporting less sediment (27.5%), less phosphorus (16.1%) and less NO_3^- (19.9%) from their fields.

Following changes made for enrollment into or technical assistance that resulted from the Certification Program, enrolled farmers were still exporting less sediment (78.7%), less phosphorus (71.8%) and less NO_3^- (9.2%) from their fields in comparison to the baseline scenario (Figure 1). Additional changes in water quality outcomes resulting from enrollment in the Certification Program were mixed. Model results indicate additional increases in water quality benefits from reductions in sediment (51.2%) and phosphorus (55.0%), but a slight increase in export of NO3- (8.5% increase) relative to the pre-certification conditions (Figures 1-3).

These water quality outcomes reflect the nature of the practices adopted by farmers in this study and they highlight the challenges of improving water quality in agricultural landscapes. In general, the practices added were effective at reducing overland flow, which was most effective in reducing sediment and phosphorus loss. However, decreased overland flow often translated into increases in tile drainage (and, to a lesser extent, lateral soil water flow) which is responsible for small increases in NO_3^- loss from farm fields. Fertilizer management was effective for reducing P and NO_3^- loss from farm fields. But overall NO_3^- export increased slightly as fertilizer management was often implemented in coordination with WASCOBs which increased infiltration of surface water (resulting in increased tile drainage).

The farms that showed the greatest improvements in NO_3^- (relative to the baseline) were those who had adopted cover crops – which were mostly in place prior to enrollment to the Certification Program. These farms with cover crops in place were exporting an average of 69% less NO_3^- from their farm fields (as well as 55% less sediment and 41% less phosphorus) relative to the baseline management scenario.

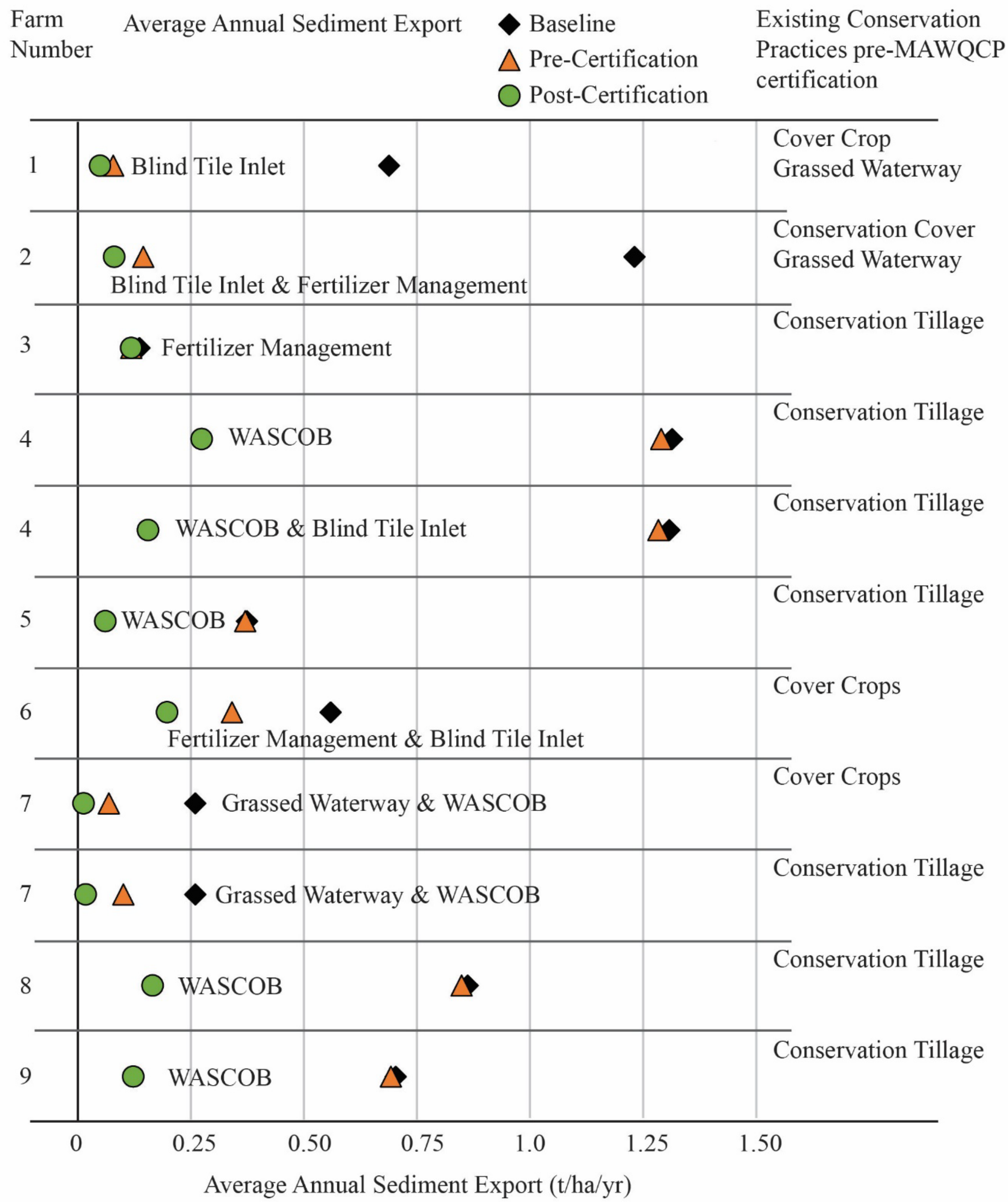

Figure 1. Average annual sediment export simulated for nine MAWQCP farms in the Cottonwood River Basin. Model results reflect the average value from 10 years of simulation outputs (2004 through 2013, inclusive). Water quality outcomes from MAWQCP farms (green circles) are compared against a general baseline management scenario (black diamonds) as well as farm management prior to Certification Program enrollment (orange triangles). Prior conservation practices are indicated along the right-hand column while new Certification Program changes are labeled near the data points.

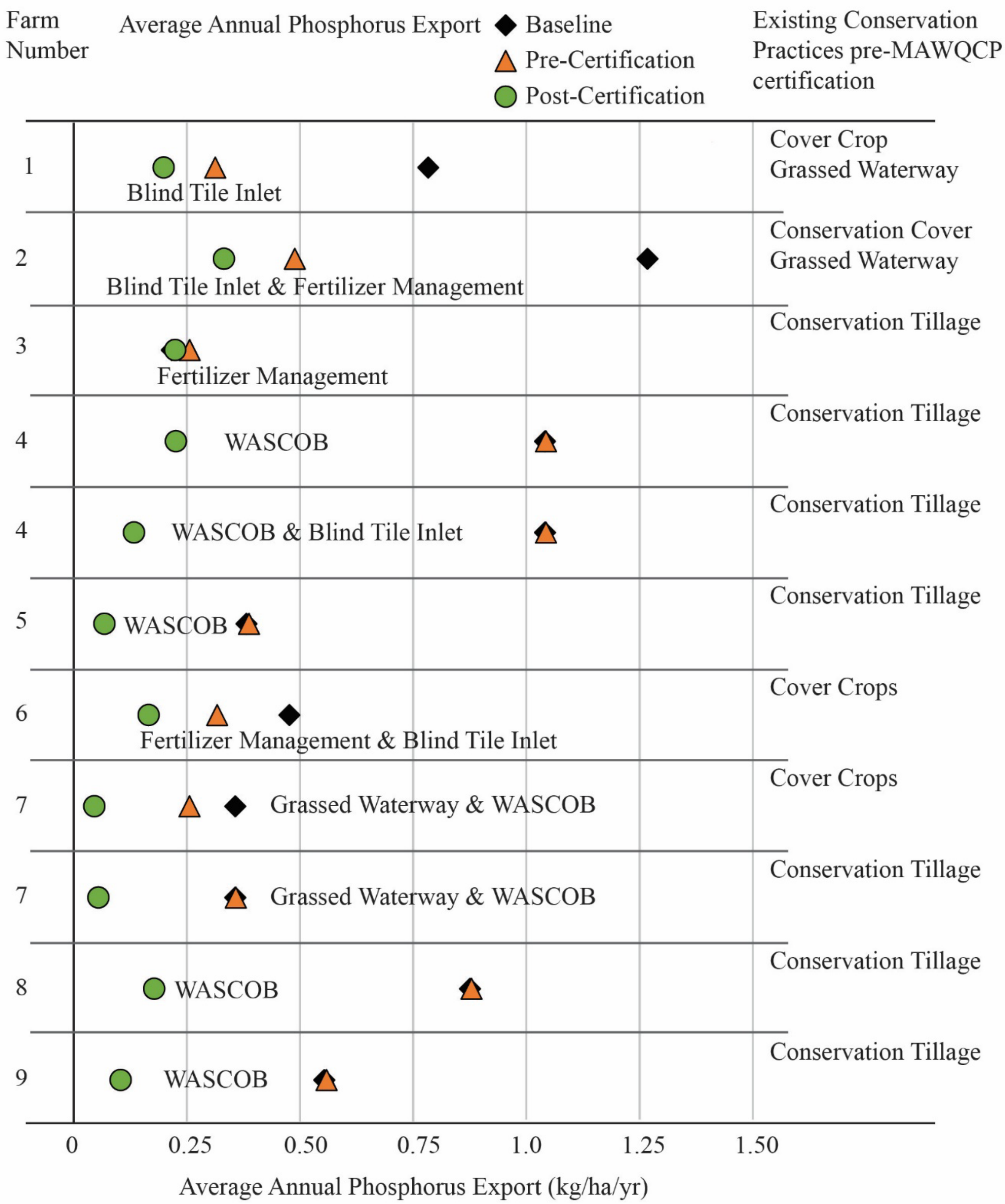

Figure 2. Average annual phosphorus export simulated for nine MAWQCP farms in the Cottonwood River Basin. Model results reflect the average value from 10 years of simulation outputs (2004 through 2013, inclusive). Water quality outcomes from MAWQCP farms (green circles) are compared against a general baseline management scenario (black diamonds) as well as farm management prior to Certification Program enrollment (orange triangles). Prior conservation practices are indicated along the right-hand column while new Certification Program changes are labeled near the data points.

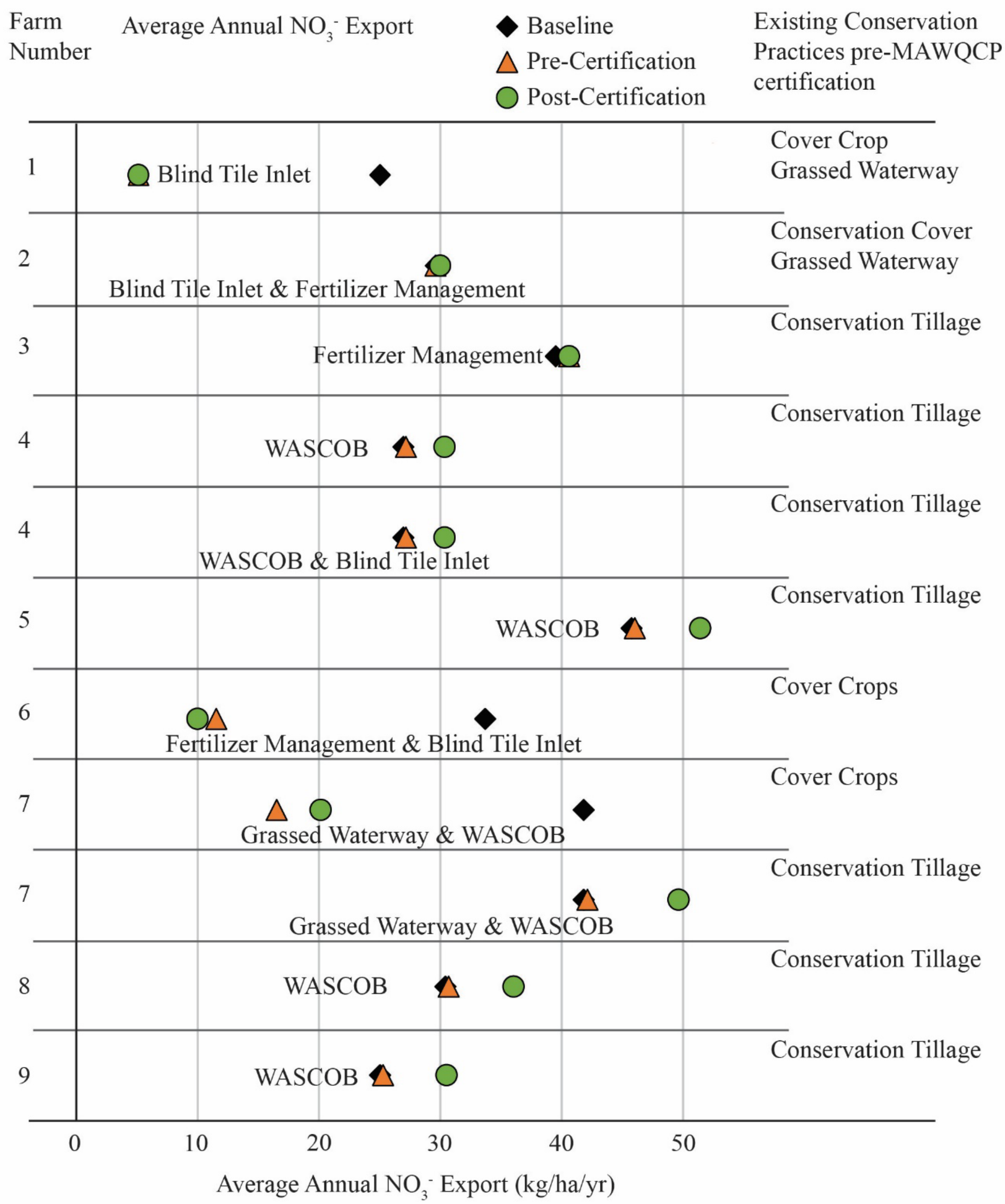

Figure 3. Average annual NO_3^- export simulated for nine MAWQCP farms in the Cottonwood River Basin. Model results reflect the average value from 10 years of simulation outputs (2004 through 2013, inclusive). Water quality outcomes from MAWQCP farms (green circles) are compared against a general baseline management scenario (black diamonds) as well as farm management prior to Certification Program enrollment (orange triangles). Prior conservation practices are indicated along the right-hand column while new Certification Program changes are labeled near the data points.

4.4 Conclusions

Based on results from this study, we make the following general observations about conservation practices and the MAWQCP:

Farmers enrolled in the Certification Program are already conservation minded and applying conservation practices on their farms prior to enrollment. Following management changes made in response to program enrollment, certified farms achieved further reductions in export of sediment (51.2%) and phosphorus (55.0%).

Overall NO_3^- export increased slightly (8.5%) as a result of simulated WASCOBs. This occurred because WASCOBs increase infiltration of water, which results in greater simulated subsurface tile drainage, the primary pathway for NO_3^- loss from these fields.

The conservation practice evaluated that achieved the greatest overall water quality benefits for sediment, phosphorus, and NO_3^- was cover crops. Fields with cover crops present exported an average of 69% less NO_3^- than the baseline management scenario. In the farms with cover crops, this practice was present prior to enrollment in the Certification Program.

For farms in the Cottonwood River Basin, the MAWQCP is improving water quality. Additional benefits would be achieved by encouraging greater adoption of winter cover crops to satisfy farm enrollment in the Certification Program.

Future work will focus on combining results from this study with spatial and economic optimization in order to help prescribe the where conservation efforts should be targeted in the Cottonwood River Basin.

4.5 References

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