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Evaluation of Woodchip Bioreactors and Phosphorus Adsorption Media for Nutrient Removal from Subsurface Drainage Water

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EVALUATION OF WOODCHIP BIOREACTORS AND PHOSPHORUS ADSORPTION MEDIA FOR NUTRIENT REMOVAL FROM SUBSURFACE DRAINAGE WATER

BY

UTSAV THAPA

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EVALUATION OF WOODCHIP BIOREACTORS AND PHOSPHORUS ADSORPTION MEDIA FOR NUTRIENT REMOVAL FROM SUBSURFACE DRAINAGE WATER

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This thesis is approved as a creditable and independent investigation by a candidate for the Master of Science in Agricultural and Biosystems Engineering degree and is acceptable for meeting the thesis requirements for the degree. Acceptance of this thesis does not imply that the conclusion reached by the candidate are necessarily the conclusion of the major department.

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Disclaimer

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ABBREVIATIONS

AIC	Akaike’s Information Criteria
cm	Centimeter
CRP	Conservation Reserve Program
CTD	Conductivity, Temperature, and Depth
gm	Grams
ha	Hectares
HRT	Hydraulic Retention Time
IL	Illinois State
kg	Kilograms
LCR	Little Cobb River
m	Meters
mg	Milligrams
MCL	Maximum Contamination Limit
N	Nitrogen
NH_3	Anhydrous Ammonia
NSE	Nash-Sutcliffe Efficiency

P	Phosphorus
PhROG	Phosphorus Removal Online Guideline
PSMs	Phosphorus Sorption Materials
RE	Relative Error
SAS	Statistical Analysis Software
SD	South Dakota
SFIR	South Fork Iowa River
USA	United States of America
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WA	Washington State
WI	Wisconsin State

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ABSTRACT

EVALUATION OF WOODCHIP BIOREACTORS AND PHOSPHORUS ADSORPTION MEDIA FOR NUTRIENT REMOVAL FROM SUBSURFACE DRAINAGE WATER

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Increased subsurface drainage over the past few decades in eastern South Dakota contributed to agricultural water quality problems. Nutrient losses, primarily nitrate-N and dissolved P, from subsurface drainage, have been identified as major contributors to eutrophication in the Great Lakes and Gulf of Mexico. Denitrifying bioreactors and P adsorption structures are edge-of field practices that can be used to protect water quality in waters downstream of subsurface drainage systems. The objectives of this study were to evaluate the effectiveness of woodchip bioreactors and a P adsorption structure in removing nitrate-N and dissolved P from subsurface drainage water. Four woodchip bioreactors were installed and monitored between 2012 and 2016 near Arlington, Baltic, Hartford and Montrose in eastern South Dakota. One P adsorption structure was designed, installed and monitored for 2016 near Baltic, South Dakota downstream of the Baltic bioreactor. Results showed that nitrate-N reductions ranged from 7% to 100% for the four bioreactors, based on upstream concentrations of 0.79 to 60.9 mg/L during 2014-2016 study period. Nitrate-N load removal rates varied between 0.01 to 14.6 gm N/m^3/day, with upstream and downstream loads of 0.03 to 116 kg/ha/year and 0.0 to 91 kg/ha/year, respectively. The average cost of nitrate removed were estimated to be $11, $20, $13, and $61 kg/N per year for the Arlington, Baltic, Hartford, and Montrose bioreactors, respectively. Dissolved P reduction ranged from 10% to 90 % and the P removal rates varied between 2.2 to 183.7 gm/m^3/day during the study period. Average annual cost of dissolved P removed was $209 kg/P. Two regression equations were developed and tested for predicting downstream nitrate-N concentrations in eastern South Dakota. The equations performed with 70% efficiency at the bioreactor sites examined, except at the Baltic site where the equations overestimated downstream nitrate-N concentrations. Based on examination of nitrate removal data of all four bioreactors since installation, their performance appears to decrease over time. The information provided in this study would be useful to increase understanding of the effectiveness of edge-of-field practices for nitrate-N and dissolved P reduction in eastern South Dakota.

CHAPTER 1: INTRODUCTION

1.1 Background

Nitrogen (N) and phosphorus (P) are essential for crop growth. They are widely used in fertilizers to achieve maximum yields (Randall and Mulla, 2001; Ringeval et al., 2017). However, they can contribute environmental concerns if they enter, in excess, groundwater and surface water bodies through leaching and surface runoff (McDowell and McGregor, 1984; Rabalais et al., 1996b; Rabalais et al., 2001).

An increase in installation of subsurface (tile) drainage over the past few decades has led to increased levels of N and P transport from agricultural fields to downstream waters (Randall and Goss, 2008; Sharpley and Syers, 1979; Smith et al., 2015). In the Midwestern United states, subsurface drainage has been extensively used to increase crop production by removing excess water from the soil profile (Skaggs et al., 1994a; Smith et al., 2015; Zucker and Brown, 1998). These artificially drained agricultural fields are adjoining to lakes, rivers, and streams (Boesch, 2002; Rabalais et al., 1996b). Annual nitrate-N and dissolved P loads measured in subsurface drainage for Midwestern states are generally in the range of 1.8 to 68 kg/ha and 0.05 to 1.0 kg/ha, respectively (Gentry et al., 2007; Goolsby et al., 2001; Randall and Mulla, 2001). These studies highlight the need for technologies to reduce nutrient enrichment in downstream waters.

1.2 Problem Statement

Subsurface drainage is essential for agricultural production in the upper Midwestern United States (Dinnes et al., 2002; Skaggs et al., 1994a; Smith et al., 2015). Nutrient losses, primarily nitrate-N and dissolved P from subsurface drainage, have been identified as major contributors to eutrophication in the Great Lakes and hypoxia condition in the Gulf of Mexico (Daigh et al., 2015; Goolsby and Battaglin, 2000; Goolsby et al., 2001; Smith et al., 2015). The concentration of nitrate-N in agricultural drainage water often exceed 10 mg/L, the maximum contamination limit (MCL) set by United States Environmental Protection Agency (USEPA) for drinking water (EPA, 2002).

Historically nitrate-N has been the major concern associated with pollution in the receiving waters; however, research suggests that dissolved P also plays a major role in the recent two decades (Dodds, 2006; Hansen et al., 2002; Sims et al., 1998). P is generally identified as one of the primary nutrients that cause freshwater eutrophication (King et al., 2015a). Dissolved P concentrations > 0.05 mg/L are considered as problematic for freshwater aquatic systems (Dunne and Leopold, 1978) . While improved management of fertilizer and animal manure is one important method for reducing nitrate-N and dissolved P losses, it is often not enough; therefore, water quality goals for nitrate-N and dissolved P require additional edge-of-field practices to reduce nitrate-N and dissolved P loads from subsurface drainage water (Dinnes et al., 2002; Penn et al., 2007b; Power, 1998). Therefore, there is a critical need for drainage water management strategies that minimize nitrate-N and dissolved P loads from subsurface drainage. Denitrifying woodchip bioreactors and P adsorption media are edge-of field practices that can be used to conserve water quality in subsurface drainage systems (Addy et al., 2016; Christianson et al., 2012a; Hoover et al., 2016; Penn et al., 2012).

1.3 Objectives

The goal for this study is to demonstrate and evaluate practices placed at the edge of fields to reduce nitrate-N and dissolved P loads from subsurface drainage systems in eastern South Dakota. The specific objectives were:

1. Evaluate the effectiveness of at least two-year old woodchip bioreactors, and assess bioreactor performance over time since installation.
2. Develop relationships between bioreactor performance and catchment characteristics;
3. Design, install, and evaluate the effectiveness of phosphorus adsorption structure.

1.4 Significance of the Study

This study evaluated the performance of four existing woodchip bioreactors installed near Arlington, Baltic, Hartford, and Montrose, and one P adsorption structure installed near Baltic. The information will inform farmers and other stakeholders about relevant and field-tested edge-of field practices to conserve water quality. This study adds to existing conservation drainage efforts and testing in eastern South Dakota.

1.5 Thesis Organization

This thesis is organized in five chapters. Chapter 1 provides the background, problem statement, and objectives of the study. Chapter 2 provides a literature review related to subsurface drainage and nutrient transport. This chapter also includes information on drainage conservation practices. Chapters 3 and 4 discuss, respectively, field evaluation of four woodchip bioreactors and phosphorus adsorption structure installed in eastern South Dakota as well as the cost of nutrient removed. Chapter 3 led to a manuscript that will be submitted for publication. Chapter 5 presents the conclusions of the study and future work.

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CHAPTER 2: LITERATURE REVIEW

2.1 Water Management with Subsurface Drainage in the Midwest

In the Midwest United States, much of the cropland would not be productive without subsurface (tile) drainage (Skaggs et al., 1994a). The Midwest region’s cool climate, water table variation, and precipitation frequently above crop demand in wet areas (King et al., 2015a) require subsurface drainage system to remove surplus soil water (Figure 2.1) (Pavelis, 1987) and promote optimum crop growth (Madramootoo et al., 1992; Skaggs et al., 1994a). Subsurface drainage started in the USA in late 1800s with the first drainage system installed in 1838 by a farmer named John Johnston in Seneca County, New York (King et al., 2015b; Pavelis, 1987). Most subsurface tile drainage systems were installed between 1870 to 1920 and between 1945 to 1960 (Zucker and Brown, 1998). By 1987, more than 20 million hectares in Illinois, Indiana, Iowa, Ohio, Minnesota, Michigan, Missouri, and Wisconsin were artificially drained using subsurface drainage systems (Zucker and Brown, 1998).

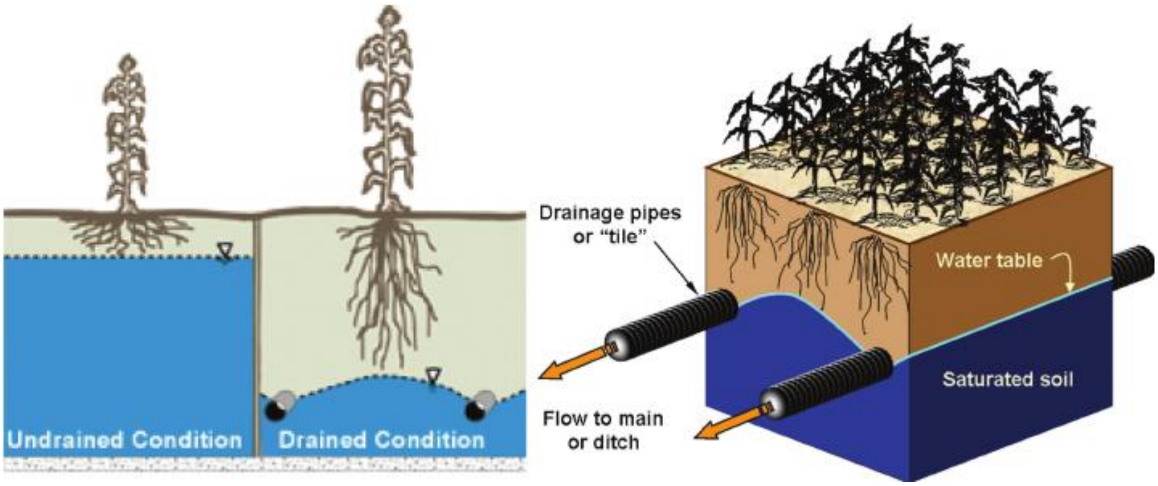

Figure 2.1 Representation of a conventional subsurface drainage system in the field (Blann et al., 2009)

Although subsurface drainage remove excess water from the soil profile, allows greater plant uptake of nutrients, and increases crop growth and productivity (Hatfield et al., 1998; King et al., 2015b), the practice also results in land cover change and, loss of agricultural nutrients to surface and ground water (Blann et al., 2009). Nutrients carried with the drainage water from agricultural drained fields to the nearby lakes, rivers, and streams are considered key factors contributing to downstream water quality problems (Gentry et al., 2007; Randall and Mulla, 2001). According to Dodds et al. (2008), lakes, rivers, and reservoirs in the US receiving excess nutrients from agricultural lands and human activities will likely have water quality problems. The hypoxic condition in the Gulf of Mexico is one of the major water quality concerns in the nation and an example of excess nitrogen (N) and phosphorus (P) discharge into receiving rivers and lakes (Figure 2.1) (Dodds, 2006; Mitsch et al., 2001; Rabalais et al., 2001). The discharge of excess nitrate-N and P into water bodies increases consumption of oxygen by accelerated algae blooms, leading to eutrophication (Carpenter et al., 1998; Crites, 1977; Litke, 1999; Mueller et al., 1996). Eutrophication refers to a nutrient enrichment of a water body and is a condition where growth of algae and aquatic weeds interferes with use of water for aquatic life, recreation, agriculture and drinking water purposes (Carpenter et al., 1998; Howarth and Marino, 2006; Litke, 1999).

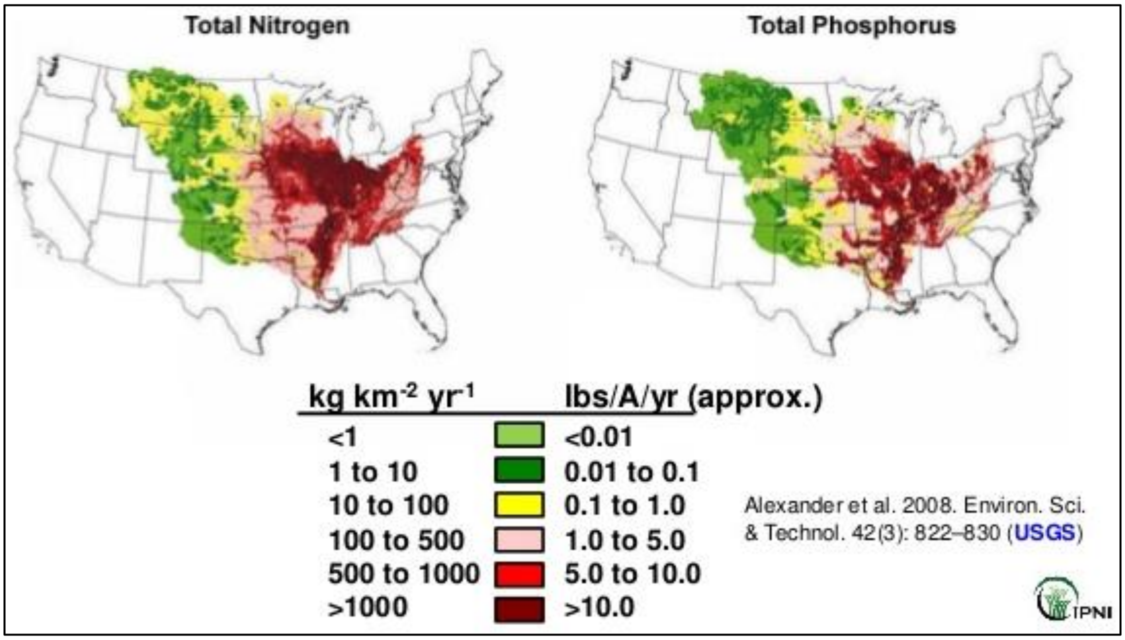

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Figure 2.2 Total nitrogen and phosphorus loads delivered to the Gulf of Mexico from 1992 to 2002 (Alexander et al., 2007)

2.2 Subsurface drainage impacts on water quality

Subsurface drainage influences soil water dynamics and downstream water yield (Schilling and Helmers, 2008; Skaggs et al., 1994b). Research showed that losses of N and P to streams and lakes are major causes of pollution in receiving waters (Algoazany et al., 2007; Baker et al., 1975a; Carpenter et al., 1998). Baker and Johnson (1981) found that average nitrate-N loss was 30 kg N/ha/year for N fertilizer application of 112 kg N/ha/year from corn and oat rotation subsurface drained fields. A study conducted in central Iowa reported that nitrate-N concentration leaving agricultural subsurface drainage system were often greater than 10 mg/L during April through July (Jaynes et al., 1999). In two Midwestern agricultural watersheds, the Little Cobb River (LCR) in Southern Minnesota and South Fork Iowa River (SFIR) in northern Iowa, researchers reported nitrate-N loads of 21.3 and 31.2 kg/ha/year from subsurface drainage systems. This study also reported that 60% of nitrate-N leaving the field occurred between April and June (Kalkhoff et al., 2016). In in Ontario, Canada, Tan and Zhang (2016) measured subsurface drainage nitrate-N losses of 87.9 and 56.6 kg/ha for 4.2 m and 7.5 m tile spacing. Daigh et al. (2015) reported 6 to 18.5 mg/L of nitrate-N concentration in subsurface drainage during four years of study in Ames, Iowa.

Due to high fixation of P in soils compared to N, P losses through subsurface drainage was generally disregarded (Baker et al., 1975a). As a result, most studies focused on reducing soil erosion in agricultural runoff and on N transport from the subsurface drainage field (King et al., 2015a; Sharpley et al., 2006). Recent studies indicated that subsurface drainage is an important contributor to dissolved P losses (Beauchemin et al., 1998; Culley et al., 1983; Gaynor and Findlay, 1995). According to Kladivko et al. (1991) P loss through subsurface drainage in the Midwest ranged from 0.01 to 0.11 kg of soluble P/ha in Indiana and 0.05 to 1 kg soluble P/ha and 0.2 to 1.3 kg total P/ha in Illinois. Kalkhoff et al. (2016) showed that two subsurface-drained watersheds located in southern Minnesota and northern Iowa yielded 0.51 and 1.13 kg total P/ha/year, with more than 50% of the total P losses occurring in April through June. In the Big Walnut Creek watershed in Ohio, subsurface drainage contributed to 47 % of total discharge in which 48% is dissolved P and 40% was total P (King et al., 2015a). From 1996-2002, 70% of total P was exported to Lake Erie through subsurface drainage systems (Dolan and McGunagle, 2005). Ruark et al. (2012) also found that 17 to 41% of cumulative total P transport, of which 16 to 58% of the total load, was from subsurface drainage systems in eastern Wisconsin. In the St. Joseph River watershed in northeast Indiana, subsurface drainage contributed 49% of soluble P and 48% of total P load (Smith et al., 2015). King et al. (2015a) studied 319 ha of subsurface drainage area in the Upper Big Walnut Creek watershed in central Ohio for eight years and found that more than 90% of all measured P concentrations coming from subsurface drainage systems exceeded the recommended MCL of 0.03 mg/L (Canada, 2004).

2.3 Eutrophication in freshwater and salt water systems

Nitrogen is the common limiting nutrient linked to eutrophication in marine water systems whereas P is the limiting nutrient for freshwater systems (Daniel et al., 1998; Hecky and Kilham, 1988). P-driven eutrophication in freshwater is typically due to excessive concentrations of P (Howarth and Marino, 2006). A laboratory experiment conducted at the University of Rhode Island tested the concept of limiting nutrient with 0, 5 10 and 25 ppt levels of salinity. This study indicated that P was limiting at 0, 5 and 10 while N was limiting at 25 ppt (Doering et al., 1995). Nitrogen fixation by planktonic cyanobacteria differs between freshwaters and estuaries due to short residence times, turbulence, salinity or limitation by iron, molybdenum, or P in freshwater (Schindler et al., 2008). High sulfate concentrations found in saline waters can delay the growth rate of cyanobacteria relative to those found in freshwaters (Conley et al., 2009). The presence of N-fixing cyanobacteria also depends on the molar ratio between N and P. High N:P ratio decreases the number of cyanobacteria while low N:P ratio increases the number of cyanobacteria in the freshwaters (Schindler et al., 2008). Paerl et al. (2001) reported that waters having N:P <15 are most likely to experience cyanobacterial dominance whereas waters having N:P >20 are more likely to be dominated by non N_2 fixing eukaryotic algae (Smith, 1983). Since P is the limiting nutrient in freshwater and N limits marine waters, best management practices to reduce both N and P inputs in waters are needed to protect aquatic ecosystems.

2.4 Nitrogen control measures

There are different strategies to reduce nonpoint source N pollutant loss to aquatic ecosystems. In-field and edge of field practices (see Figure 2.3) are some of the best management practices to reduce nitrogen loads (Dinnes et al., 2002; Power, 1998). In-field practices include improved nitrogen management, cover crops, crop rotation, and improved efficiency fertilizers (such as nitrification inhibitors and controlled release fertilizers). Managing timing and application rate of fertilizers based on requirements in the field increases the efficiency of fertilizer applied (Randall and Iragavarapu, 1995; Randall and Mulla, 2001). Research showed that applying N fertilizers in spring is more appropriate than fall application because spring application reduce N losses from the field by reducing the time between application and increased plant uptake (Dinnes et al., 2002). A study conducted in southern Minnesota showed that 36% extra nitrate-N loss was found in fall applied fertilizer compared to spring (Randall et al., 1997; Randall et al., 1992). The use of conservation or reduced tillage is also a control measure to maintain crop residues on the soil surface, increase infiltration, improve soil water storage, and reduce soil erosion. An 11-year study in Minnesota found the nitrate-N content in the residual soil depth of 0-1.5m were significantly higher with conventional tillage than no-till (Randall and Iragavarapu, 1995). Crop rotations or changing from continuous monoculture to different types of crop is a proven practice to reduce nitrate-N leaching in agricultural fields (Randall et al., 1997). Rotation of legume and nonlegume crops can also decrease nitrate-N losses in some cases. A study in Minnesota reported that crop rotation from alfalfa fields and Conservation Reserve Program (CRP) lands to mixture of alfalfa and perennial grass has less nitrate-N concentration compared to converted corn and soybean fields (Randall et al., 1997). Likewise, cover crops are usually planted to manage soil erosion, soil fertility, and improve water holding capacity; thus increasing the efficiency of N fertilizers applied in the field (Lu et al., 2000). Meisinger et al. (1991) in a review found that cover crops can reduce both load of N leached and nitrate-N concentration 20 to 80% compared with no cover crop practices in the field. Non-legume cover crops are efficient at reducing nitrate-N leaching in agricultural fields (Ranells and Wagger, 1996; Sainju et al., 2002).

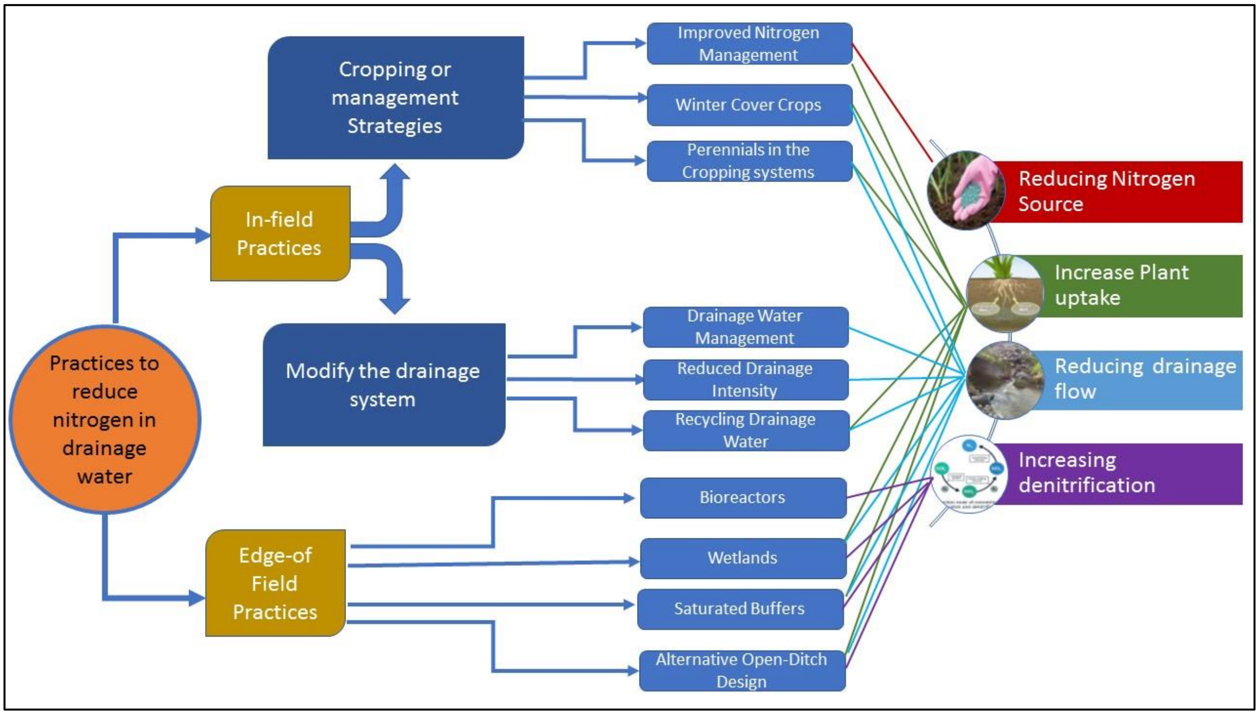

Figure 2.3 In-field and edge-of field practices to control nitrate-N in subsurface drainage water (Christianson, 2016).

Other in-field control measures include nitrification inhibitors and slow release fertilizers. Nitrification inhibitors are used to slow the conversion process from ammonium (NH_4) to nitrite. By slowing the conversion, the positively charged ammonia is retained by negatively charged soil particles. Hence, nitrogen inhibitors helps hold N in the soil longer and reduce N loss before the peak N demand by the crops (Stehouwer and Johnson, 1990). Slow release fertilizers are coated with less water-soluble materials to retard fast dissolution of fertilizers. Sulfur-coated urea is commonly used in agriculture as slow-N release product (Follett, 2008). A study in Ohio found that addition of a nitrification inhibitors (nitrapyrin) with spring applied N had no effect on grain yield but increases yield with the fall N application (Stehouwer and Johnson, 1990). Similarly, a study in Minnesota reported that the use of N fertilizer with the addition of nitrapyrin in the fall increases the efficiency of N by 16-26% and in spring application increases to 42-48% (Randall et al., 1992). In Colorado, Shoji et al. (2001) found that N losses from controlled release of N fertilizer was only 1.9% while 10 % of N losses were observed with urea application. Animal manure is also an excellent source of plant nutrients. Several studies suggested that manure application successfully reduce nitrate-N leaching, when proper rate, method and timing of manure application is taken into consideration (Ferguson et al., 2005; Van Es et al., 2006; Wu and Powell, 2007).

Nitrogen loss from agricultural drained fields can also be reduced by a number of edge-of field practices (Figure 1.6). Denitrifying bioreactors, wetlands, and saturated buffers are the most commonly used conservation practices to remove nitrogen loading from the subsurface drained water (Karpuzcu, 2012; Parkyn, 2004; Robertson and Merkley, 2009). All these practices are related to the denitrification process. A bioreactor is a trench filled with a carbon substrate material, such as woodchips, that acts as an electron donor for the anaerobic bacteria to convert nitrate into dinitrogen gas. It is a simple denitrifying process that routes the tile drain water through the woodchip trench. Christianson et al. (2012a) evaluated the performance of four bioreactors installed in Iowa and found that the percent reduction of nitrate ranges from 12-74% from the bioreactors. Similarly, a recent study done by Bell et al. (2015) in Urbana Champaign, Illinois reported 20-98 % nitrate reduction and an average nitrate removal rate of 11.6 gm N/m^3/day. A wetland study in Illinois showed nitrate-N removal rates of 12 to 63 mg/m^2/h at a temperature range of 11 to 27˚C (Xue et al., 1999).

Saturated buffers are vegetative strips along stream banks or ditches, where the tile water is diverted into shallow laterals to raise the water table and slow outflow, allowing increased nutrient uptake by the buffer plants. Hence, the saturated buffer allows drainage water to flow as shallow groundwater through the buffer soil, allowing vegetation present in the buffer to take up more nitrate-N and provide conditions for enhanced denitrification (Mayer et al., 2007; Vidon and Hill, 2004). In the Contentnea Creek Drainage Basin in North Carolina, (Spruill, 2000) found that nitrate loss was 95% lower in buffer areas compared with non-buffer areas. The study also reported that 65 to 70 % reduction was due to denitrification.

2.5 Phosphorus control measures

Nonpoint source P in subsurface drainage systems can be controlled by field and edge of field practices. Field management practices such as control of rate and timing of P applied in the form of organic and inorganic fertilizers reduces P transport (Smil, 2000). Research showed that P loss in runoff increases with application of P fertilizer (Sharpley et al., 2001; Sharpley et al., 1992). Application of fertilizer in the field based on application rate and timing is an important factor affecting P loss (Smil, 2000). Hergert et al. (1981) measured P concentration from plots with 35 and 200 wet tons/ha manure application and found that mean concentration was between 0.10 and 0.44 mg/L for 200 wet tons/ha while P concentration of 0.01 to 0.02 mg/L was measured for 35 metric tons/ha. Algoazany et al. (2007) measured P concentration in subsurface drainage from four agricultural fields in east-central Illinois and found that greater application rates of fertilizers tend to increase soluble P in drainage water. The study also observed greater P concentration and loading into receiving waters when P was applied in fall compared to in spring application. Transport management controls the movement of P to the receiving waters (Sharpley et al., 2001). Phosphorus losses from subsurface drainage can be reduced by conservation tillage practices (Sharpley et al., 2006); however, majority of studies reported that conservation tilled fields contribute high levels of P to subsurface drainage (Sims et al., 1998). Gaynor and Findlay (1995) reported that total soluble P increased by 2.2 times with conservation than conventional tillage.

There is a wide range of technologies to remove P loss at the edge of fields. Phosphorus adsorption materials are the most common (Penn et al., 2011; Penn et al., 2012). Sorption of dissolved P is the combined process of adsorption and precipitation to solid forms (Penn et al., 2007a). P sorbing materials rich in iron (Fe), aluminum (Al), calcium (Ca), magnesium (Mg) or their combination showed promising results for P sorption (HiCkey and GibbS, 2009; Penn et al., 2014; Penn et al., 2007a). These materials also include iron oxide mine tailings, blast furnace slag, coconut shell–activated carbon, flue gas desulfurization gypsum, zeolite, goethite, limestone, polonite (Karczmarczyk and Bus, 2014; King et al., 2010). Penn et al. (2012) used steel slag as P sorption material in suburban watershed in Stillwater, Oklahoma and found that 25% of dissolved P was trapped by the structure for the first five months of operation. McDowell et al. (2008) also tested three ash materials and four slag byproducts and found that three ash materials i.e. fly ash, Bottom ash (0-2 mm) and Bottom ash (2-4 mm) had greater P sorption capacity than all other slag materials except basic slag. Al and Fe based salts ($Al_2(SO4)_3$ and $FeSO_4$) can also adsorb dissolved P from the drainage water (Cooke et al., 1993; Huser, 2012). Penn et al. (2011) tested six different industrial byproducts and found that materials rich in Al and Fe were more effective in P sorbing. Similarly, 50-70% of P was removed from aquaculture wastewater with iron-oxide based sorption media in Pennsylvania (Sibrell and Kehler, 2016).

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CHAPTER 3: EVALUATION OF WOODCHIP BIOREACTORS TO SUPPORT CONSERVATION DRAINAGE IN EASTERN SOUTH DAKOTA

ABSTRACT: Edge-of-field practices such as woodchip bioreactors can be used to improve water quality from agricultural subsurface drainage systems. Four woodchip bioreactors were installed and monitored for more than 20 rainfall events between 2012 and 2016 near Arlington, Baltic, Hartford and Montrose in eastern South Dakota. Results show that nitrate-N concentration reduction for the four bioreactors ranged from 7% to 100% during the study period, with removal rates that varied between 0.01 to 14.6 gm N/m^3/day. The average cost of nitrate removed was estimated to be \$11, \$20, \$13, and \$61 kg/N per year for Arlington, Baltic, Hartford and Montrose bioreactors. Two regression equations were s developed and tested to estimate the expected nitrate-N concentration downstream of a woodchip bioreactor located in eastern South Dakota when rainfall, drainage area, drainage outflow from the bioreactor, and hydraulic retention time (HRT) are known. A simple linear trend line showed that the performance of the four studied bioreactors in nitrate-N removal from tile water appeared to decrease over time during the study period. The information provided in this study is needed to increase understanding of the effectiveness of woodchip bioreactors in the United States Upper Midwest.

3.1 Introduction

Addition of plant nutrients such as N to agricultural fields is essential for improving crop growth; however, undesirable environmental consequences may arise if they are not properly managed. Nutrient loading to surface water bodies causes eutrophication in aquatic ecosystems (Alexander et al., 2007; Rabalais et al., 1996a). Eutrophication is the excessive enrichment of lakes and rivers with nutrients, causing a growth of algae (algal blooms) or other aquatic plants that depletes the oxygen level in the water when they decay (Carpenter et al., 1998; Cloern, 2001; Paerl, 2008). Eutrophication is a leading cause of impairment of waters in the Gulf of Mexico (Alexander et al., 2007; Rabalais et al., 2001), due to discharge of excess nutrients from farmlands into the Mississippi River Basin (Alexander et al., 2007; Mitsch et al., 2001). Studies suggests that as much as 20 to 40% of annual N applied as fertilizer to agricultural fields in the Midwest are lost through subsurface drainage systems (Dinnes et al., 2002; Gentry et al., 2009).

Subsurface drainage (or tile drainage) is a widely adopted water management practice utilized to increase crop production. While subsurface drainage provides many agronomic benefits (Fraser et al., 2001), the practice has been linked to downstream water quality problems (Baker et al., 1975b; Cambardella et al., 1999). A number of edge-of-field practices can be used to reduce nitrate-N exports via subsurface drainage systems. These practices include drainage water management, wetlands, saturated buffers, and denitrifying bioreactors, among others (Blattel et al., 2009; Cooke et al., 2008; Dinnes et al., 2002; Woli et al., 2010). A denitrifying bioreactor is a trench filled with a carbon source which intercepts subsurface drainage water before it reaches downstream water bodies. Denitrifying bacteria convert the nitrate-N to dinitrogen gas which is then released to the atmosphere, while the drainage water is discharged into a surface water body or conveyance system. Within the past decade, several bioreactors have been installed and monitored in United States as well as other parts of the world (e.g. Bell et al., 2015; Christianson et al., 2012a; Robertson et al., 2009; Schipper et al., 2010a). The promising performance of woodchip bioreactors for nitrate-N removal has led to increased attention in the practice (Ghane et al., 2015).

Table 3.1 Overview of field scale evaluation of denitrification bioreactors for agricultural drainage.

Reference	Site	Installation year	Carbon source	Initial concentration of nitrate-N	Percent reduction	Nitrate-N removal rate	Therotical Hydraulic retention time
Blowes et al. (1994)	Ontario, Canada	1992	Woodchip, grow-bark, sand and composted leaf	2-6 mg/L	100 *	Not reported	1-6 day
Van Driel et al. (2006)	Southern Ontario	2001	Coarse wood particle	11.8-3.9 mg/L	33 *	2.5 gN/m^3/day*	9 hours
Jaynes et al. (2008b)	Central Iowa	1999	Woodchip	<0.3-35 mg/L	55 *	0.622 gN/m^3/day*	Not reported
Schipper et al. (2010a)	Karaka, NewZealand	2006	Sawdust and woodchip	Not reported	Not Reported	5 gN/m^3/day - 10 gN/m^3/day	Not reported
Schipper et al. (2010)	Kinloch, NewZealand	2006	Sawdust and woodchip	Not reported	Not Reported	11 gN/m^3/day – 0 gN/m^3/day	Not reported
Schipper et al. (2010)	Dargaville, NewZealand	2006	Sawdust and woodchip	Not reported	Not Reported	1.4 gN/m^3/day*	Not reported
Chun et al. (2010)	Decatur, IL	2003	Woodchip	269.9 g/L*	47 *	Not reported	4.4 hours
Woli et al. (2010)	Deland, IL	2006	Woodchip and pruned limb	2.8-18.9 mg/L	33 *	6.4 gN/m^3/day*	26 min to 2.8 hours
Verma et al. (2010)	Deland, IL (East)	2006	Woodchips	3-16 mg/L	42-48	Not reported	Not reported
Verma et al. (2010)	Decatur, IL (west)	2006	Woodchip	2->9 mg/L	81-98	Not reported	Not reported
Verma et al. (2010)	Decature, IL (East)	2006	Woodchip	4-12 mg/L	54 *	Not reported	Not reported
Robertson et al. (2009)	Southern Onterio, Canada	2001	Woodchip and sawdust	4.8mg/L*	Not reported	2-16 mg N/L/day	0.41-2.2 day
Shih et al. (2011)	Southern Onterio, Canada	2006	Woodchip	0.6-4.4 mg/L	30-100	Not reported	0.8-2.8 day
Schmidt and Clark (2012)	Alachua, Florida	2009	Sawdust and sand	6.2±0.65g/m^3	77 *	5.5 gN/m^3/day (Maximum)	1.7-1.9 day
Christianson et al. (2012a)	Pekin, Iowa	2002	Woodchip	1.23-8.54 mg/L	22-74	0.38-3.78 gN/m^3/day	Not reported

Christianson et al. (2012a)	NERF, Iowa	2009	Woodchip	9.93-13.18 mg/L	12-14	0.86-1.56 gN/m^3/day	Not reported
Christianson et al. (2012a)	Greene County, Iowa	2008	Woodchip	7.7-15.2 mg/L	27-33	0.41-7.76 gN/m^3/day	Not reported
Christianson et al. (2012a)	Hamilton, Iowa	2009	Woodchip	7.74-9.59 mg/L	49-57	0.42-5.02 gN/m^3/day	Not reported
(Partheeban et al., 2014)	Montrose, South Dakota	2012	Woodchip	Not reported	51*	0.03-3.14 gN/m^3/day	4.9 hours
Bell et al. (2015)	Urbana Champaign, IL	2012	Woodchip	<0.1-17 mg/L	20-98	11.6 gN/m^3/day*	2-8 hours
David et al. (2016)	East-central, IL	2012	Woodchip	257-2127 kgN/year	3-81	0.7-116 gN/m^3/day	Not reported
(Jaynes et al., 2016)	Central Iowa		Woodchip	11.5-15.8 mg/L (2013)	38*	0.4-13.2 gN/m^3/day	Not reported
				11-16 mg/L (2014)	49*	0.04-12.6 gN/m^3/day	Not reported

* Average values

Nitrate removal efficiency of bioreactors depends on the influent nitrate-N concentration, flow rate, temperature, microbial community, and carbon substrates (Christianson et al., 2012b; Schipper et al., 2010b). The initial concentration of nitrate-N, whether low or high, affects the denitrification process (Addy et al., 2016; Schipper et al., 2005). Low influent concentration leads to N-limited conditions in the bioreactor bed, resulting in lower nitrate-N removal rates, while higher proportions of nitrate-N increase nitrate-N removal during the denitrification process (Addy et al., 2016; Schipper et al., 2010b)The HRT also affects the nitrate-N removal rates (Addy et al., 2016). A HRT less than 6 h does not typically provide enough time for nitrate removal compared to beds designed with a HRT greater than 6h (Addy et al., 2016; Christianson et al., 2012b; Robertson and Merkley, 2009). The HRT is controlled by the flow rate along with the size and design media porosity of the bioreactor. Research showed that the hydraulic retention time influences nitrate-N removal efficiency, which typically decreases with increased flow rates (Addy et al., 2016; Christianson et al., 2012b). A laboratory scale study showed a range of 10% to 40% in nitrate-N reduction with shorter retention time, while the reduction increased to 100% with a longer retention time (Chun et al., 2009). The retention time must be managed to optimize nitrate-N removal but also avoid unwanted processes to occur. If the retention time is too long and the nitrate-N concentration approaches zero within the bioreactor, other unwanted processes, such as sulfate reduction, may occur. High flow rates facilitate the transport of dissolved oxygen in the bioreactor (Greenan et al., 2009). Increased dissolved oxygen in water decreases nitrate-N removal rate in the bioreactor as the presence of oxygen competes with the presence of nitrate-N in the role of terminal electron acceptor (Healy et al., 2006; Rivett et al., 2008). An increase in water temperature typically increases bioreactor efficiency. When water temperature increases, the growth rate of denitrifying organisms also increases, leading to increased nitrate-N removal rate (Addy et al., 2016; Cameron and Schipper, 2010; Robertson et al., 2009). Saleh-Lakha et al. (2009) reported a positive correlation between the growth rate of denitrifying organisms and water temperature. The denitrification processes in a bioreactor generally occur between 5˚and 30˚C (Christianson, 2011; Robertson and Merkley, 2009), although nitrate-N removal at lower temperatures also occurs at reduced rates (Blowes et al., 1994; Van Driel et al., 2006).

Denitrifying bacteria are highly dependent on the carbon source, which acts as an electron donor to facilitate the denitrification process (Cameron and Schipper, 2010; Rivett et al., 2008). Examples of carbon source materials include woodchips, sawdust, maize cobs, wheat straw, cornstalks, and green waste, among others (Blowes et al., 1994; Greenan et al., 2006). The properties of the carbon media influence the hydraulic retention time and longevity of the bioreactor (Christianson et al., 2011). The carbon source facilitates the denitrification process by catalyzing aerobic respiration to reduce oxygen concentration in the water (Schipper et al., 2005).

Even though bioreactors have been proven as an effective and low-cost technology for nitrate-N removal (Christianson et al., 2012a; Cooke et al., 1998), there is a need for continued evaluation of their performance in additional geographic regions and climate conditions to document region-specific bioreactor performance and factors that affect this performance. The goal of this study was to evaluate four woodchip bioreactors installed in eastern South Dakota in terms of improving water quality from agricultural subsurface drained fields. The specific objectives were to (1) evaluate the effectiveness of woodchip bioreactors in removing nitrate-N and estimate cost to remove nitrate Kg N/ha two years after installation; (2) develop relationships between bioreactor performance and catchment characteristics; and (3) examine changes in bioreactor performance over time.

3.2 Materials and Methods

3.2.1 Study Sites

Four woodchip bioreactors were installed in eastern South Dakota with different dimensions and treatment areas (Table 3.2). The bioreactors are located near Arlington, Baltic, Hartford, and Montrose, South Dakota. The contributing area of the Baltic bioreactor has silty clay loam soil in a corn-soybean-wheat rotation during the study period (2013-2016). A theoretical HRT of 5.4 hours was used to design the bioreactor to handle up to 25% of the peak flow from the contributing area.

The Montrose bioreactor receives drainage from approximately 15.4 ha of silty clay loam soil in a corn-soybean rotation during four years of study period (2013-2016).

This bioreactor was designed with an estimated HRT of 4.9 hours and designed to handle up to 25% of peak flow from a 20 cm main drain line.

The drainage area for the Arlington bioreactor is approximately 6.9 ha of silty clay loam soil. This site was in a corn-soybean rotation during the study period (2014-2016), and the bioreactor was designed to handle 18% of peak flow from a 15 cm main drain line with 6.3 hours HRT.

The Hartford bioreactor was designed with 6.3 hours HRT to handle 18% of peak flow from a 15 cm main drain line that drains approximately 8.1 ha. This site was in a corn-soybean rotation during the study period (2015-2016).

Common for all sites was the installation of 3-chamber or 4-chamber control structures on the main subsurface drainage line upstream of the bioreactor to divert water through the bioreactor. During high flow conditions, excess water is directed through a bypass and directly into the outlet of the drainage system. A 2-chamber control structure was installed downstream of all bioreactors to measure the outflow before the open ditch, except at the Arlington site, which has a 4-chamber control structure. The retention time is controlled by adjusting the elevation difference between the inlet and outlet of the bioreactor. The particle size of the woodchips ranged from 0.6 to 5.1 cm.

Table 3.2 Descriptive information for the four woodchip bioreactors in Eastern South Dakota.

Location	Installation Date	Dimension (m)	Catchment Name	Area Drained (ha)
Arlington (Brookings County)	July 2013	38.1 L x 3.6 W x 1.52 D	Lake Sinai	6.9
Baltic (Minnehaha County)	July 2012	35 L x 5.4 W x 1.52 D	Silver Creek	16.2
Hartford (Minnehaha County)	November 2014	38.1 L x 3 W x 1.52 D	Wall Lake	8.09
Montrose (McCook County)	December 2012	39.6 L x 6.4 W x 1.52 D	Skunk Creek	15.4

L = Length, W = Width, and D = Depth

3.2.2 Data Collection

Decagon CTD-10 sensors (Decagon Devices, Pullman, WA) were used to measure water depth, temperature, and electrical conductivity in the control structures. The Baltic site used Campbell Scientific sensors (Campbell Scientific, Inc., Logan, UT) while the other three sites were outfitted with Decagon sensors. These sensors utilize pressure transducers to obtain water level measurements. Tipping bucket rain gauges were also installed near upstream of the bioreactors to measure precipitation every 10 minutes. Campbell Scientific TE525 and Decagon ECRN-100 double spoon gauges were installed, respectively, at the Baltic and Arlington sites, and Montrose and Hartford sites.

Each transducer was suspended between 0.63 to 1.27 cm above the bottom of the control structure, and flow depth readings were relative to the pressure transducer location. A standard 45˚ V-notch weir (AgriDrain Corp, Adair, Iowa) was installed in the control structures and the drainage flow rate was estimated based on the thickness (i.e. depth) of the sheet of water flowing over the weir. All the sensors, including pressure transducers and rain gauges were connected to data loggers that record and store the data every 10 minutes.

During drain flow events, grab water samples were collected directly within the upstream and downstream control structures. Samples were collected using a water bottle attached to a steel rod placed in each control structure. Collected water samples were immediately stored in a cooler with packed ice until transported to the analysis laboratory. The samples were manually filtered using a 30 mL high-density polyethylene (HDPE) syringe and a 0.45 µm nylon filter membrane into a pre-labeled 60 mL Nalgene bottle. The samples were kept at 0˚C until analyzed for nitrate-N plus nitrite-N (usually negligible) using the Sulfanilamide and N-(1-naphthyl)-ethylenediamine dihydrocholoride reduction by cadmium method (EPA Method 353.2) with Seal AQ2 Discrete Analyzer (Mequon, WI).

3.2.3 Performance Evaluation of Woodchip Bioreactors

Nitrate-N concentration was used to evaluate the performance of the bioreactors in removing nitrate-N by calculating concentration-based percent nitrate-N removal as:

$$\%\text{Removal} = \frac{C_{up} - C_{down}}{C_{up}} * 100 \qquad (3.1)$$

where C_{up} is the upstream nitrate-N concentration (mg/L) and C_{down} is the downstream nitrate-N concentration (mg/L). The depth of water flowing over the weir in the control structures was converted into flow rate (L/min) by using the following equations (Chun and Cooke, 2008; Partheeban, 2014):

$$Q = 1.7406 H^{1.9531} \qquad (3.2)$$

where Q is the discharge (L/min) and H is the depth of water flowing through the V-notch, and

$$Q = 0.027 * L * H^{1.2} \qquad (3.3)$$

where Q is the discharge (L/sec), L is the width of spillway, and H is the depth of water flowing through the spillway. Equation 3 was used under high-flow conditions for water depths exceeding the V-notch cut away

Nitrate-N removal rate (gram N/m^3/min) was calculated for each sampling event as:

$$R_N = \frac{(C_{up} - C_{down})Q}{V_{sub}} \tag{3.4}$$

where R_N is the nitrate-N removal rate (mg/m^3/min), C_{up} is the upstream nitrate-N concentration (mg/L), C_{down} is the downstream nitrate-N concentration (mg/L), Q is the flow rate (L/min), and V_{sub} is the volume of carbon material (i.e. woodchip) used for the treatment, i.e. the wetted volume in the bioreactor (m^3). The bypass flow volume was considered while calculating nitrate-N loading. Nutrient load computation accounted for bypass flow for sampling days with bypass flows, except the Montrose and Hartford bioreactor sites which did not have any bypass events.

3.2.4 Cost Estimation of Nutrient Removed

The cost to remove nitrate from tile drain water was estimated per hectare per year for each bioreactor. The total installation cost for the bioreactor installation was categorized for different cost components such as excavation and backfilling, woodchips, plastic liner, control structure, and others (e.g. personnel transport, labor, etc.). The total installation cost for each bioreactor is approximately \$7914, \$9014, \$9714, and \$10,414, for Arlington, Baltic, Hartford, and Montrose, respectively. Based on design, the life expectancy of the bioreactors is 20 years. The total installation cost per year of the bioreactors were \$894, \$976, \$1022 and \$1075 for Arlington, Baltic, Hartford, and Montrose, respectively. These costs were obtained from Partheeban (2014).

Total nutrient load removed per year (kg /year) was estimated with following equation:

$$\text{Nitrate-N load removed (NLR)} = \left(C_{up} - C_{down}\right) * Q \tag{3.5}$$

where C_{up} is upstream nutrient concentration (mg/L) and C_{down} is downstream nutrient concentration (mg/L) and Q is the discharge (L/min).

Total nutrient load removed per year was used to determine the cost to remove a unit mass of nitrate-N as:

$$C = \frac{Total\ \ Installation\ \ Cost/Year}{NLR} \tag{3.6}$$

where *NLR* is, the nutrient load removed per year (kg/year) and the total Installation Cost/ Year is the total amount spent for the installation of the practice divided by the life expectancy of the practice (20 years for a bioreactor).

3.2.5 Relation Between Bioreactor Performance and Site Characteristics

The relation between bioreactor performance and contributing area characteristics was developed by identifying dominant factors that control performance of the bioreactor through literature review and through the development of linear models. These factors include upstream nitrate-N concentration flowing into the bioreactor, HRT, calculated flow rates, temperature, microbial community and carbon materials (Christianson et al., 2012b; Schipper et al., 2010a). In this study, the dominant factors were first selected from the literature review based on their availability at the study sites. Data from the Arlington, Montrose, and Hartford sites were used to develop performance equations, while data from the Baltic site were used to test the equations (Table 3.3). The data from Baltic were not included in model development because this site is used as a demonstration site for various agricultural activities and products, including field days; thus, the Baltic data may not be representative of typical field operations in eastern South Dakota. The equations developed were further tested with one year data from Montrose (Table 3.3). Nitrate-N data used were based on grab samples collected 2-3 times per week. Flow rates were calculated using the aforementioned equations (3.2) and (3.3). The HRTs used for the analysis are the theoretical HTRs obtained from the bioreactor design tool (Christianson et al., 2011). It should be noted that the design tool requires drain size, drain grade, hydraulic conductivity and porosity of wood media, and dimensions of the bioreactor as input to estimate the theoretical HRT.

The equations were developed using multiple linear regression analysis. Prior to equation development, the distribution of the data was checked for normality and a logarithmic transformation was applied to all variables. A correlation test was also performed to further tune the variable selection process, ensuring that the predictor variables are not statistically strongly correlated with each other. The final group of variables was used as predictor variables (Table 3.4) to develop regression equations for estimating downstream nitrate-N concentration from the bioreactors. The models were developed using the Statistical Analysis System, version 9.3 (SAS Institute Inc., 2011). The SAS multiple linear regression model procedure (PROC REG) with the “Best 10” option was used to select the models that best describe variation in nitrate-N downstream of the bioreactor due to variation in upstream factors (i.e. predictor variables; see Table 3.4) based on adjusted coefficient of determination (R^2) and Akaike’s Information Criteria (AIC) values.

Two models were developed to estimate downstream nitrate-N concentration of the bioreactors. One equation included HRT, while the second one excluded HRT. While HRT is an important factor in bioreactor performance, (Robertson and Merkley, 2009), HRT is not a readily available parameter to obtain. In addition, these equations were intended to be simple tools for estimating bioreactor performance of a given site in South Dakota, leading development of the equation without HRT. The equations developed for downstream nitrate-N concentration estimation were:

$$\log(DConc) = \log\beta_0 + \beta_1 \log(x_1) + \beta_2 \log(x_2) + ...\beta_n \log(x_n) \tag{5}$$

where DCONC is the downstream concentration (mg/L); β_0 is the regression constant; β_1, β_2...... β_n are regression coefficients; and x_1, x_2.......... x_n are the predictor variables. Since the equations were developed with logarithm transformed variables, the final equations are expressed as:

$$DConc = \beta_0 x_1^{\beta_1} x_2^{\beta_2} x_3^{\beta_3} .....x_n^{\beta n} \tag{6}$$

Model performance was evaluated on the basis of R^2, Nash-Sutcliffe Efficiency (NSE) coefficient, and relative error (RE). The mathematical expressions of R^2, NSE, and RE can be found in the literature (Moriasi et al., 2007; Nagelkerke, 1991; Nash and Sutcliffe, 1970).

The Baltic and Montrose bioreactors have four years (2013-2016) of data records. The Arlington bioreactor has three years (2014-2016) and Hartford has two years (2015-2016) of data. The data lengths used for model development and testing are presented in Table 2. The models were further tested with one year data (2013) from Montrose bioreactor (Table 3.3).

Table 3.3 Data length used for model development and testing to predict outflow nitrate-N concentration from the bioreactors examined.

Study Site	Development Period	Testing Period
Arlington	2014 - 2016	-
Baltic	-	2013 - 2016
Hartford	2015 - 2016	-
Montrose	2014 - 2016	2013

Table 3.4 Catchment characteristics used as independent variables for model development to predict outflow nitrate-N concentration from the bioreactors examined.

Variable	Notation	Unit
Upstream Concentration	UpConc	mg/L
Precipitation	PRECIP	mm
Downstream Flow Rate	FLOW	L/min
Hydraulic Retention Time	HRT	hours
Drainage Area	DA	ha

3.2.6 Examination of Bioreactor Performance Over Time

A simple linear regression approach was used to examine trend or change in bioreactor performance over time. Calculated recent nitrate removals since the installation of the bioreactor (see Table 3.2) were plotted versus time and a trend line was fitted through the data to determine an increasing or decreasing pattern. Microsoft Excel was used for the analysis. The procedure was repeated for each bioreactor (Table 3.2) to assess whether their performance changes over time.

3.3 Result and Discussion

3.3.1 Nitrate-N Concentration and Reduction

Upstream nitrate-N concentration was generally highest at the Baltic bioreactor for the three year study period (2014-2016) compared to the other three sites (Table 3.5) Table 3.5). Baltic upstream nitrate-N concentrations ranged from 16.1 to 61.0 mg/L, with an average of 34.9 mg/L, while downstream concentrations ranged from 0.0 to 40.2 mg/L, with an average of 14.9 mg/L (Table 3.5). Most downstream concentrations from the Baltic site exceeded the 10 mg/L maximum contaminant level (MCL) prescribed by USEPA (2002) for drinking water. Elevated levels of upstream nitrate-N concentration may be due to upland activities and field management at the Baltic site. Upstream concentrations at the other three sites ranged from 0.8 to 30.7 mg/L with an average of 12.8 mg/L, and downstream concentration varied between 0 and 22.8 mg/L, with an average of 4.6 mg/L (Table 3.5). Downstream concentrations at three other sites were generally less than the 10 mg/L MCL for all samples, except some collected in 2014 and 2016. The maximum rainfall events of the study period were recorded in 2014 and 2016, resulting in high flow rates carrying elevated levels of nitrate-N into the tile system. Water samples collected during June and July upstream of the bioreactors tended to have nitrate-N concentration levels of more than 20 mg/L, especially at the Arlington, Hartford and, Montrose sites; however, a majority of the samples were between 10 and 20 mg/L. In Iowa and Illinois, field studies also showed ranges of less than 0.1 to 51 mg/L and 2.8 to 18.9 mg/L of nitrate-N, respectively, in upstream tile water (Daigh et al., 2015; Jaynes et al., 2008a; Jaynes et al., 2016; Woli et al., 2010) and 0.1 to 14.5 mg/L of nitrate-N downstream of woodchip bioreactors (Christianson et al., 2012a; Moorman et al., 2010; Woli et al., 2010).

Nitrate-N concentration reduction for all four bioreactors ranged from 7% to 100% (Table 4). These values are consistent with the range of 12% to 100% reported by Bell et al. (2015) and Woli, et al. (2010) for similar field-scale bioreactors in Illinois, suggesting the four bioreactors examined in eastern South Dakota have similar performance capacity. Christianson et al. (2012) also found a nitrate-N reduction range of 14% to 75% for four woodchip bioreactors in Iowa. The Arlington bioreactor had the highest nitrate-N reduction with an average of 81% for the three years while other bioreactors had average nitrate-N reductions of 57%, 56%, and 59% (Baltic, Hartford and Montrose, respectively). Generally, low nitrate-N reductions were observed during high flow events and during events where inflow nitrate-N concentration was high for all the four bioreactors. High nitrate-N reduction tended to occur during late spring and early fall. This is likely due to the fact that during these times of the year, water flow through the bioreactor is low and temperature is warmer. Warmer temperatures increase controls on transport of dissolved oxygen in the bioreactor by increasing denitrifying bacterial communities and activities, resulting in increased nitrate-N reduction (Christianson et al., 2013; Elgood et al., 2010; Pluer et al., 2016).

Nitrate-N load removal rates ranged from 0.01 to 14.6 gm N/m^3/day for the four bioreactors, based on 0.01 to 0.8 kg N/ha/day upstream loads and 0 to 0.7 kg N/ha/day downstream loads (Table 4). These values translate to 0.03 to 116 kg/ha/year for upstream loads and 0 to 91 kg/ha/year for downstream loads (Table 4). Researchers in the Midwest region also reported a similar range of 0.38 to 13.2 gm N/m^3/day for similar field-scale bioreactors (Christianson et al. (2012a); (Jaynes et al., 2016); Schipper et al. (2010a). Bell et al. (2015) reported an average nitrate-N removal rate of 11.6 gm N/m^3/day for a woodchip bioreactor monitored over two years in Illinois. Nitrate-N load removal rates for all the bioreactors increased with greater concentration reduction, with the exception of bypass flow events where the bypass flow had no concentration reduction. Removal rates also increased when drainage water temperature increased. Increased temperature in influent water contributes to increased bed temperature and anaerobic microbial activity, leading to greater nitrate-N removal rates (Christianson et al., 2012a; Christianson et al., 2013; David et al., 2016). Conversely, removal rates decreased during low flow conditions and low nitrate-N concentrations entering the bioreactor. Low inflow concentrations often create nitrate-N-limiting conditions in the bioreactor bed, which results in lower nitrate-N removal rates compared to a non-nitrate-N limited bed (Addy et al., 2016; Pluer et al., 2016; Woli et al., 2010).

Nitrate-N concentration reductions of 7% to 100% suggest that woodchip bioreactors can be an effective technology for nitrate-N reduction from subsurface drainage water. For the three study sites with representative cropping conditions in eastern South Dakota (i.e. Arlington, Hartford, and Montrose), nitrate-N concentrations downstream of the bioreactors were less than 10 mg/L across all sampling events. During the study period, nitrate-N load removal was highly influenced by drain flow and influent concentration similar to results reported for similar bioreactor field monitoring studies (Christianson et al., 2012a; David et al., 2016).

Table 3.5 Annual range of upstream and downstream nitrate-N concentrations, loads, and removal for four bioreactors in eastern South Dakota.

Site	Nitrate-N Concentration			Nitrate-N Load			Rainfall
	Upstream	Downstream	Reduction	Upstream Load	Downstream Load	Removal Rate	
	(mg/L)		(%)	(kg N/ha/year)		(gm N/m3/day)	(mm/yr)
Arlington							
2014	6.8-25.0	0.3-1.2	87-97	0.03-6.9	0.0-0.7	0.01-2.6	310
2015	0.8-20.1	0.0-5.1	55-100	0.6-14.6	0.0-3.7	0.2-5.1	545
2016	8.9-23.8	0.0-11.8	31-100	9.0-116.1	0.0-91.0	2.2-14.6	422
Baltic							
2014	17.9-40.7	0.2-33.6	17-99	0.0-20.8	0.0-20.2	0.01-9.3	483

2015	26.0-61.0	2.7-40.2	27-92	0.3-16.7	0.1-7.6	0.1-11.3	475
2016	16.1-52.7	0.0-30.8	7-100	8.2-41.5	0.3-21.2	0.5-8.8	656
Hartford							
2015	6.1-15.7	0.2-7.6	34-98	0.5-4.8	0.0-0.8	0.1-1.9	291
2016	5.9-30.7	2.3-22.8	8-61	4.6-35.1	1.8-22.2	0.6-3.7	307
Montrose							
2014	5.7-20.1	0.1-16.1	7-99	0.1-65.1	0.0-60.0	0.01-5.1	313
2015	6.9-20.4	0.0-5.2	42-100	1.8-12.6	0.0-6.2	0.6-4.3	117
2016	7.9-23.1	6.7-11.2	15-57	3.4-25.8	2.0-17.3	0.3-6.9	530

3.3.2 Cost of Nitrate-N Removed

Total average nitrate loads removed per year for the bioreactors were 87.5, 52.1, 67.3 and 16.6 kg N per year for Arlington, Baltic, Hartford, and Montrose sites, respectively. Using the nitrate load removed (*NLR*) per year and total installation cost per year, the average cost of nitrate removed was estimated to be \$11, \$20, \$13 and \$61 kg/N removed per year for the Arlington, Baltic, Hartford and Montrose bioreactors. Annual estimated cost for the Arlington and Baltic bioreactors was comparable to 2.39 to \$15.17 kg/N published by Christianson et al. (2012a) and Schipper et al. (2010b) for field-scale bioreactors in the region. The installation cost for the bioreactors in Hartford and Montrose sites was comparatively higher than the cost for bioreactors installed in Arlington and Baltic sites.

3.3.3 Prediction of Bioreactor Downstream Nitrate-N Concentration

The regression equations developed for estimating downstream nitrate-N concentrations are herein referred to as model A for the model with HRT and model B for the model without HRT (Table 3.6). As mentioned earlier, HRT is an important factor that influences bioreactor performance but not readily available; thus, a second model was developed to allow estimating of bioreactor performance with relatively readily available information such as rainfall, drainage area, and drain flow in the bioreactor. While it is not common to directly measure drainage flow rates, it would be relatively easy to get a point measurement of drain flow through manual measurement of the water depth of water behind the board(s) in the control structure.

Table 3.6 Regression equations for estimating nitrate-N concentration downstream of bioreactors

Model Description	Equation
Model A	$\log(DConc) = -24.046 + 1.6543 \log(UpConc) - 0.0899 \log(PRECIP) + 0.436 \log(FLOW) + 6.8463 \log(DA) + 20.287 \log(HRT)$ $DConc = 8.989*10^{-25}\ U_pConc^{1.654}\ PRECIP^{-0.0899}\ FLOW^{0.436}\ DA^{6.846}\ HRT^{20.287}$
Model B	$\log(DConc) = -2.397 + 2.030 \log(UpConc) - 0.167 \log(PRECIP) + 0.341 \log(FLOW) - 0.011 \log(DA)$ $DConc = 0.0040\ U_pConc^{2.030}\ PRECIP^{-0.167}\ FLOW^{0.341} DA^{-0.011}$

DConc = Downstream Concentration (mg/L); UpConc = Upstream Concentration (mg/L); PRECIP = Rainfall (mm); FLOW = Downstream Flow (L/min); DA = Drained Area (ha), HRT = Hydraulic Retention Time (hrs).

The performance of the regression equations is shown in Figure 3.1. The simulated downstream concentration for the models A and B ranged from 0.03 to 18.8 mg/L and from 0.01 to 14.2 mg/L, respectively, while the observed values ranged from 0.0 to 16.2 mg/L, suggesting that the predicted and observed ranges of bioreactor downstream nitrate-N concentration were similar. Model A generally performed better than model B, as indicated by the performance statistics shown in Figure 3.1. HRT is an important factor in bioreactor performance (e.g. Christianson et al., 2011b; Robertson and Merkley, 2009), and adding it to the model improved prediction performance. Bell et al. (2015) also developed two models to predict nitrate-N removal and load reduction for three woodchip bioreactors in Illinois using temperature and HRT as predictor variables. The authors found that the equation with HRT improved predictions.

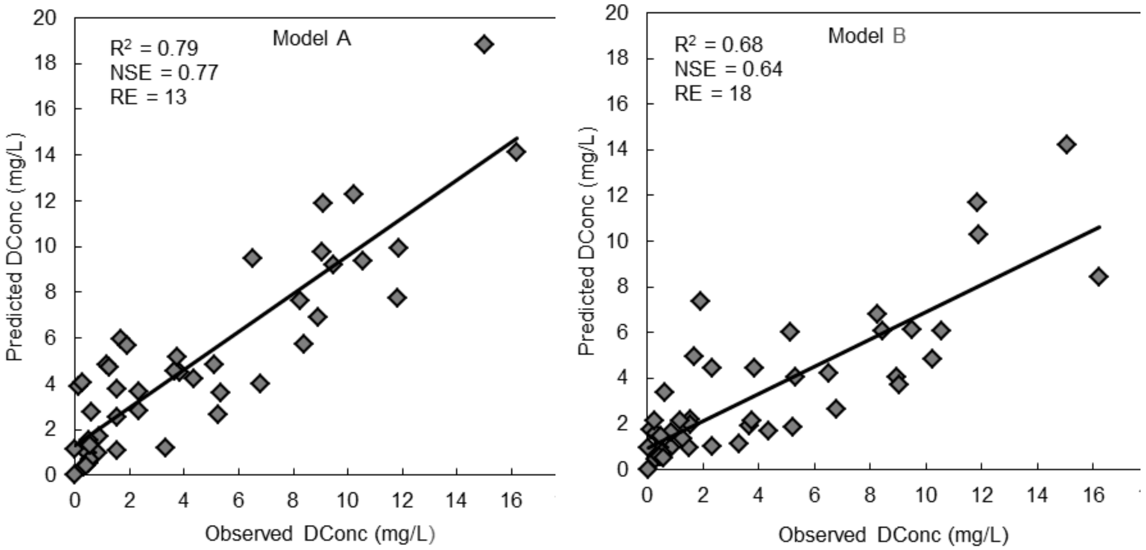

Figure 3.1 Predicted versus observed downstream nitrate-N concentration for bioreactors in eastern South Dakota

The equations developed are able to reproduce the downstream nitrate-N concentration data of the sites used for model development (FiguresFigure 3.2Figure 3.3,Figure 3.4). The equations were then tested with three years of data (2013-2016) from the Baltic site and one year of data (2013) from the Montrose bioreactor (Table 3.3). The performance of the equations at the Montrose site resulted in NSE and R^2 values greater than 0.5 (Figure 3.6), which is good in terms of model performance as discussed by Moriasi et al. (2007). RE values for both equations are 4% and 14%, respectively for the Montrose site (Figure 3.6). When applied the Baltic site, the performance of the models was unsatisfactory, as indicated by the three model performance evaluation statistics (Figure 3.5). R^2 values are low for both models while NSE values indicate poor performance, especially for model A (Figure 3.5). As a demonstration site, Baltic has field management conditions that may not be accurately be captured in the models

(Figure 3.5). This likely leads to large deviations between the predicted and observed downstream concentration at the Baltic bioreactor site.

Overall, the models appear to predict satisfactorily nitrate-N concentration downstream of the bioreactor based on Montrose data (Figure 3.6), with Model A performing better than Model B across the sites examined. Although the models performed well, some of the simulated values were considerably lower and higher than the observed values (Figures Figure 3.5Figure 3.6). This could be the result of differences in inflow nitrate-N concentrations and flow events during specific sampling events.

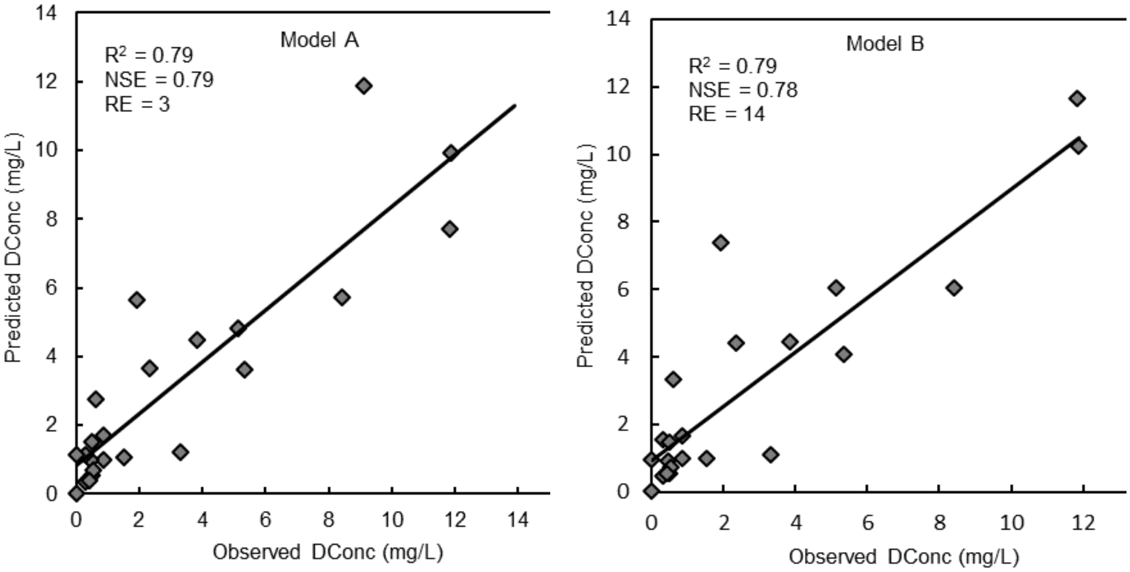

Figure 3.2 Predicted versus observed nitrate-N concentration downstream of Arlington bioreactor during 2014-2016

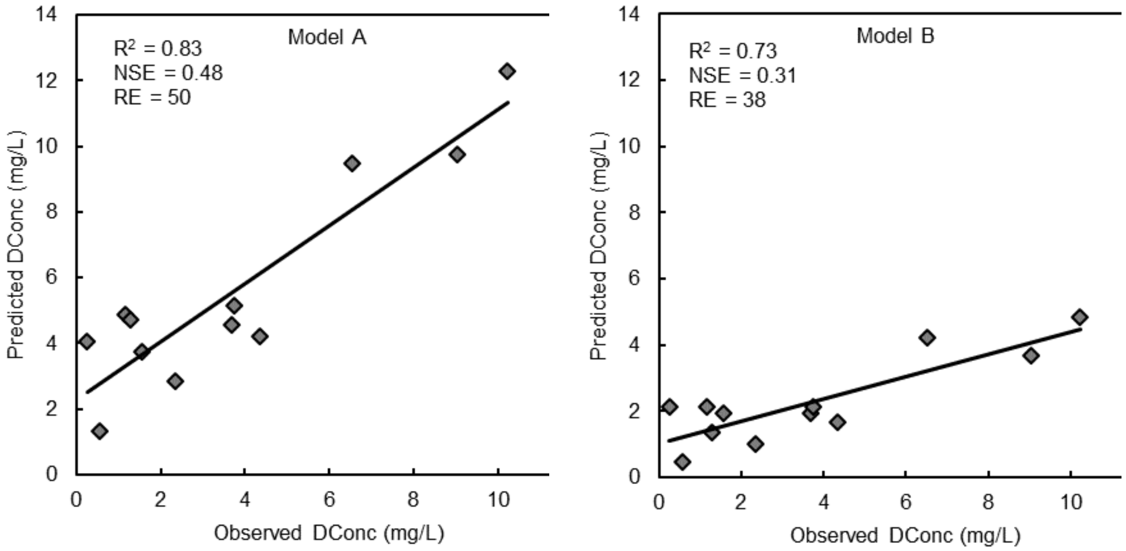

Figure 3.3 Predicted versus observed nitrate-N concentration downstream Montrose bioreactor during 2014-2016

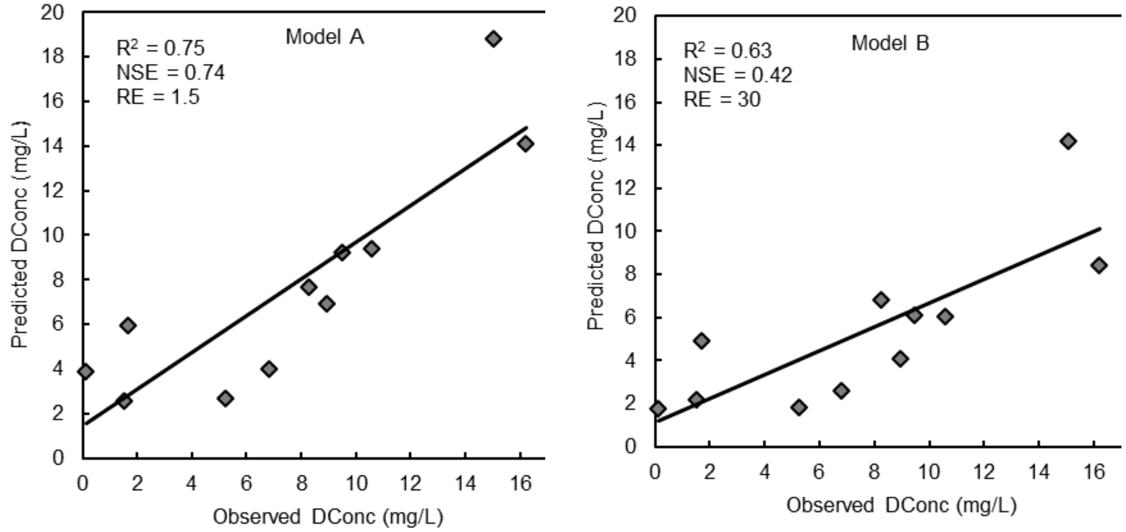

Figure 3.4 Predicted versus observed nitrate-N concentration downstream Hartford bioreactor during 2015-2016

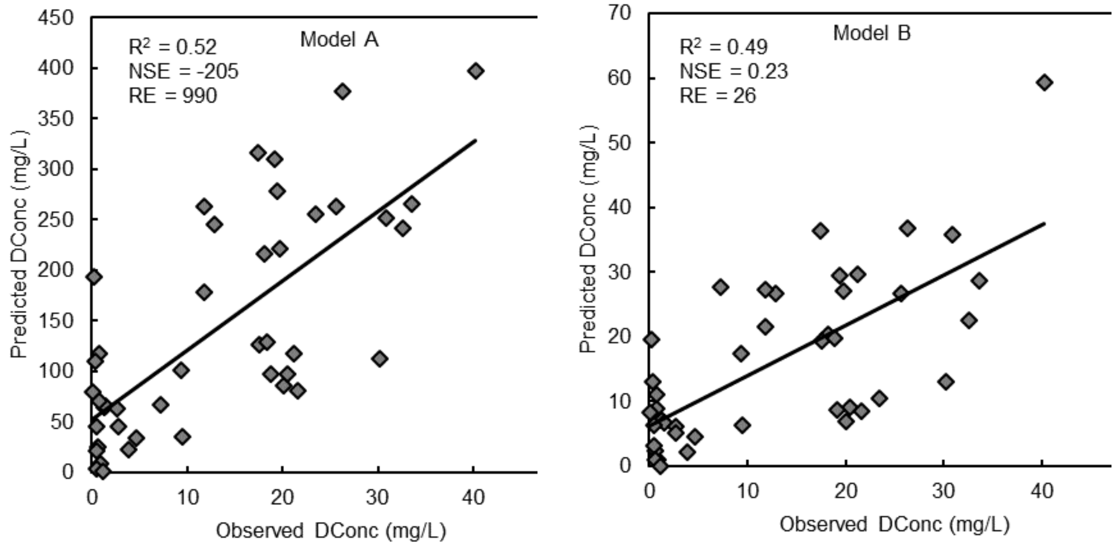

Figure 3.5 Predicted versus observed nitrate-N concentration downstream of Baltic bioreactor during 2013-2016 testing period.

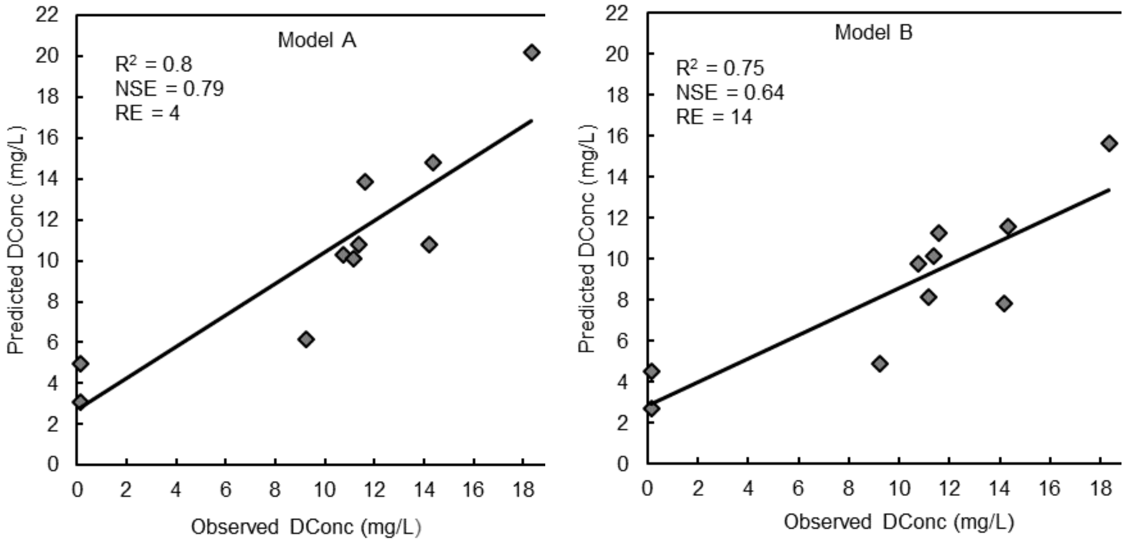

Figure 3.6 Predicted versus observed nitrate-N concentration downstream of Montrose bioreactor during 2013 testing period.

3.3.3 Bioreactor Performance Over Time

Without paying attention to differences in flow events and incident precipitation events, the trend line representing nitrate percent reduction consistently shows a decrease in performance of the four bioreactors over time (Figure 3.7). Although nitrate reduction records are not the same for all four sites (Tables Table 3.2 and Table

3.5), the data suggest that the effectiveness of the bioreactors decreases gradually over the study periods. This decreasing pattern in performance is consistent across all the study sites, except at Montrose which shows a relatively flat trend line (Figure 3.7). On average, percent nitrate reduction decreased from 94% to 61% for Arlington, 81% to 61% for Baltic, 80% to 36% for Hartford, and 51% to 31% for Montrose during the two- to four-year year study period. In the year 2015, nitrate reduction in Montrose bioreactor was higher than other years (Table 3.5). At the Montrose site in 2015, variation in rainfall from May to June resulted in low flow events, which likely increased nitrate reduction in the bioreactor during that year compared to other years. Nitrate removal in bioreactor highly depends on flow rate which fluctuates the HRT, which in turn affects the denitrification reactions (Chun et al., 2009; Robertson and Merkley, 2009). During 2015 majority of samples were collected in July at Montrose, and almost all events had 100% nitrate reduction. Factors such as July temperature and low flow rate resulted in increased nitrate reduction (e.g. Addy et al., 2016; Christianson et al., 2013; David et al., 2016). The least monitored site (i.e. Hartford; Tables 2 and 3) also shows decreases in nitrate reduction between 2015 and 2016. For this site, there is almost 50% reduction in performance between 2015 and 2016. Although this could be due to differences in flow events between the two consecutive years, it is likely that the performance of the practices monitored decreased over time . Christianson et al. (2012a) also reported decreasing performance from 55% to 31% for a bioreactor in Iowa based on six years of monitoring data. While the overall bioreactor performance may decrease over an extended period of time, the year to year performance may vary due to factors such as initial nitrate concentration, temperature and flow rates that influence bioreactor effectiveness (Hoover et al., 2016; Christianson et al., 2013).

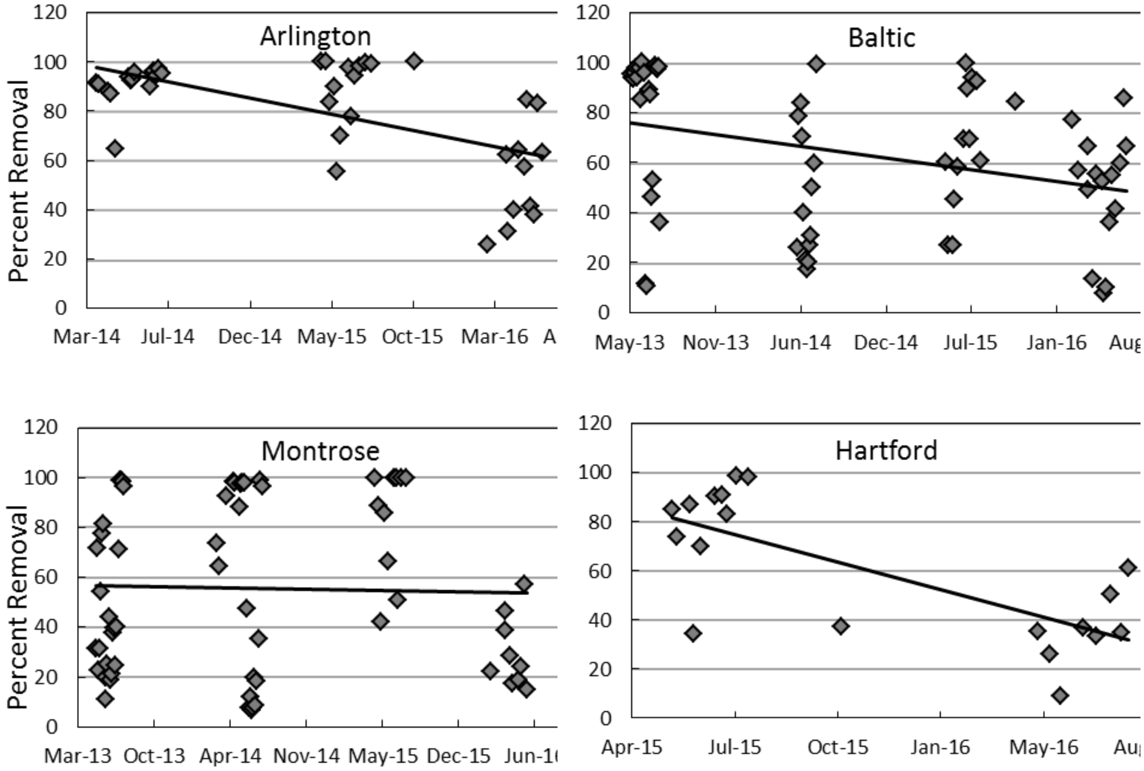

Figure 3.7 Nitrate removal by four woodchip bioreactors over two to four years monitoring periods in eastern South Dakota.

3.4 Conclusions

This study evaluated four woodchip bioreactors located near Arlington, Baltic, Hartford, and Montrose in eastern South Dakota for nitrate-N reduction for a 2014-2016 study period. Nitrate-N concentration reductions ranged from 7% to 100% for the four bioreactors, based on upstream concentration of 0.79 to 60.9 mg/L. Nitrate-N load removal rates for the four practices varied between 0.01 to 14.6 gm N/m^3/day, with upstream and downstream loads of to 0.03 to 116 kg/ha/year and 0.0 to 91 kg/ha/year, respectively. The average cost of nitrate removed was estimated to be $11, $20, $13, and $61 kg/N per year for Arlington, Baltic, Hartford and Montrose bioreactors. Two regression equations were developed and successfully tested for predicting downstream nitrate-N concentration in eastern South Dakota. The equations generally performed reasonably well at the bioreactor sites examined, except at one site (Baltic) where the equations appeared to overestimate downstream concentration. Based on simple linear trend lines, the performance of the four studied bioreactors in removal nitrate from tile water appeared to decrease over time during the study period. This study provides information on field-scale denitrification bioreactors that can be used to inform agricultural water management in South Dakota and similar regions.

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CHAPTER 4: EVALUATION OF DISSOLVED PHOSPHORUS REMOVAL FROM SUBSURFACE DRAINAGE WATER USING STEEL SHAVINGS

ABSTRACT: Phosphorus sorption materials can be used at the edge-of –fields to bind dissolved P present in agricultural subsurface drained waters. This study designed, installed, and evaluated for one year (2016) a P adsorption structure near Baltic in eastern South Dakota. The cost of dissolved P removed was also estimated. Results showed that dissolved P reduction ranged from 10% to 90% and the P removal rates varied between 2.2 to 183.7 gm/m^3/day. Average annual cost of unit mass of dissolved P removed was $209 per kg. The information provided in this study will be useful to inform producers and policy makers about conservation drainage in the state.

4.1 Introduction

Reactive phosphorus (P) leaving agricultural fields is a primary source of eutrophication in freshwater bodies Phosphorus concentration > 0.05 mg/L is considered problematic for freshwater streams (Dunne and Leopold, 1978). In agricultural landscapes, P transport to freshwater bodies through surface runoff and subsurface flow contribute to water quality problems (Baker et al., 1975a; Jamieson et al., 2003). Subsurface drainage removes excess water, leading to runoff reduction from agricultural fields (Hatfield et al., 1998; King et al., 2015b). Previously, subsurface drainage was not regarded as a substantial P transport pathway because of reduction in volume of surface runoff (Smith et al., 2015). Recent studies showed that subsurface drainage is a potential source of P in agricultural watersheds (Sims et al., 1998; Smith et al., 2015). Research also reported that subsurface drainage exports the same amount of P as the amount of P transported in surface runoff (Sims et al., 1998).

Leaching of P to subsurface drainage systems can be enhanced by reduction in soil phosphorus adsorption capacity and development of preferential flow in the field (Algoazany et al., 2007; Kladivko et al., 1991). A study conducted in Wood County, Ohio showed that dissolved reactive phosphorus (DRP) in subsurface drainage ranged from 0.03 and 0.3 mg/L, which increased in magnitude with rainfall events (King et al., 2017). In southcentral Minnesota, a three-year study also reported 0.2 to 0.9 mg/L for DRP concentration at tile outlets (Ginting et al., 2000). Similarly, research conducted in Ontario, Canada for four year (1999-2003) showed a range of 0.05 to 0.33 mg/L/year DRP in subsurface drainage water (Zhang et al., 2015). Hence, it is necessary to implement both in-field and edge-of-field practices to control P losses from subsurface drainage for aquatic ecosystem protection.

Phosphorus removal technologies like flow-through filters, cartridges, and adsorption media installed in-line at the drainage outlet using minerals (limestone, gypsum, zeolite, goethite, dolomite sand, and others) and industrial byproducts (fly ash, steel slag, steel turnings, steel wool, acid mine drainage residue, and others) are proven technologies for P removal from drain flow (Christianson et al., 2017; Hua et al., 2016; Morse et al., 1998; Penn et al., 2007a). These phosphorus sorbing (PSMs) materials are generally rich in calcium (Ca), iron (Fe), Magnesium (Mg) and aluminum (Al), which promote P sorption and removal (Penn et al., 2016; Penn et al., 2011; Penn et al., 2014).

The sorption materials provide metal cations which bind with dissolved P to form insoluble compounds (Penn and Bryant, 2006). Christianson et al. (2017) conducted a laboratory test to determine P removal efficiency of acid mine drainage treatment residuals and steel slag and found a removal range of 56 to 98%, and 23 to 89%, respectively, for different hydraulic retention times. A laboratory study in Urbana, IL found that a mixture of 60% fly ash and 40 % clay can result in up to 80% P reduction in 30 second and almost 100% in 10 minutes (Li et al., 2017). (Penn et al., 2011) also tested six different industrial byproducts as PSMs and found that Fe and Al-rich PSMs and Ca and Mg-rich PSMs are the two most effective groups to precipitate dissolved P available in the water column. The objectives of this study were to design, install, and evaluate the effectiveness of a P adsorption structure and to estimate the cost per kilogram (kg) of dissolved P removed.

4.2 Materials and Methods

4.2.1 Study Area

A P adsorption structure was installed near Baltic, SD downstream of an existing bioreactor (Figure 4.1). Detailed information about the study site is shown in Table 4.1.

Figure 4.1 Location of P adsorption structure installed near Baltic, South Dakota (Red dot).

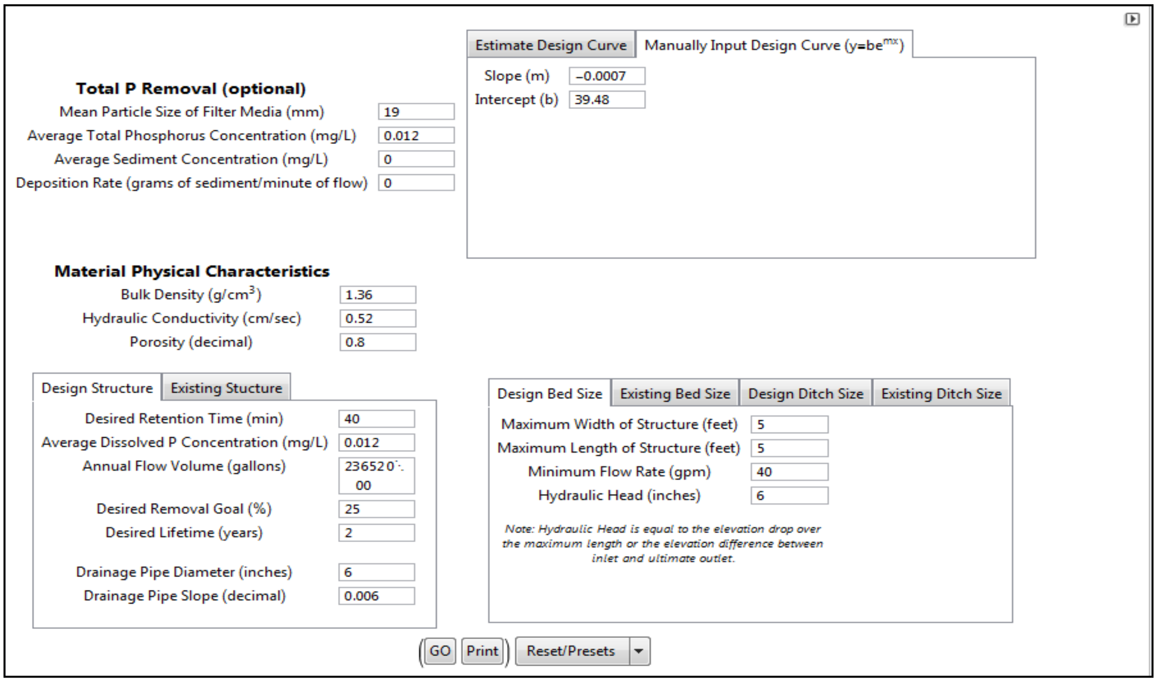
Table 4.1 Descriptive information on the P adsorption media

Location	Installation Date	Dimension (m)	Catchment name	Area Drained (ha)
Baltic (Minnehaha County)	October 8, 2015	2.5 L x 0.45 diameter	Silver Creek	16.2

4.2.2 Design of P Adsorption Structure

The P adsorption bed was designed with a mixture of small steel chips, medium steel chips, and large steel turnings based on findings from lab scale experiments (Sellner, 2016) (Figure 4.2). Using the PhROG software developed at Oklahoma State University (Penn et al., 2016), 2.5 m long and 0.45 m diameter barrel was determined for construction of the phosphorus adsorption structure based on the mass of the steel shavings. The barrel was connected to a 6-inch tile drain pipe and has a grade of 0.6%. Based on the design tool (PhROG software), the retention time was estimated to be 40 minutes and the mass of steel chips used was approximately 1.2 tons (Figure 4.3).

Figure 4.2 Steel shavings used as P adsorption media installed near Baltic, South Dakota



Slope (m) –0.0007
Intercept (b) 39.48

Total P Removal (optional)

Mean Particle Size of Filter Media (mm) 19
Average Total Phosphorus Concentration (mg/L) 0.012
Average Sediment Concentration (mg/L) 0
Deposition Rate (grams of sediment/minute of flow) 0

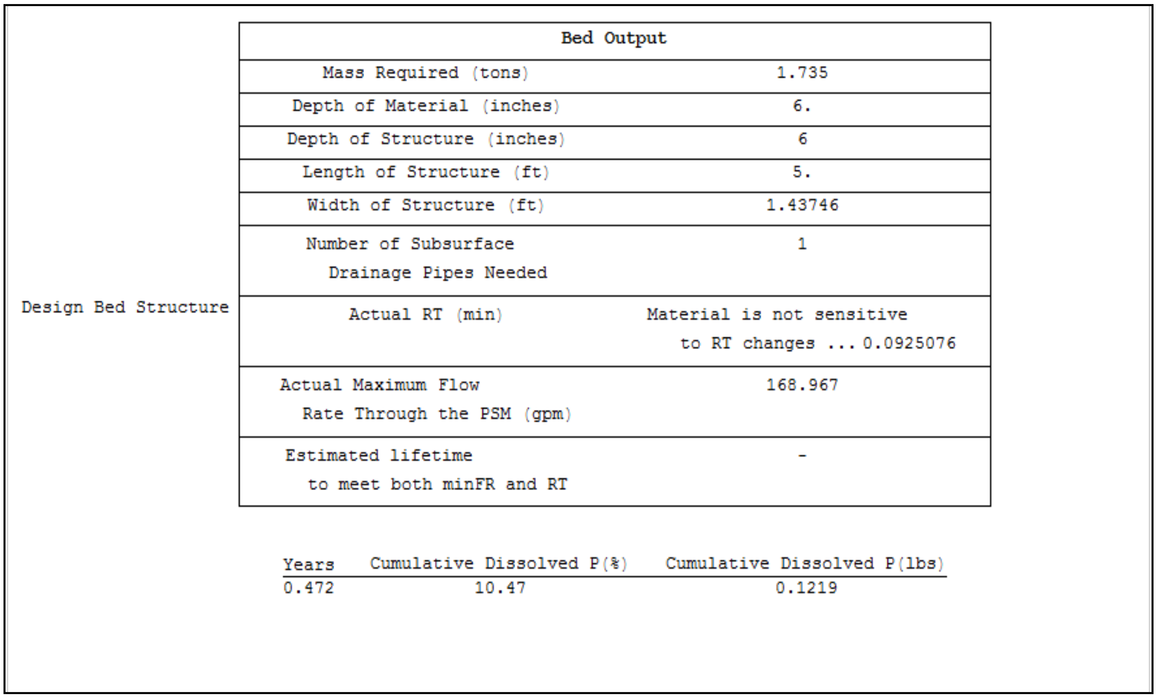
Material Physical Characteristics

Bulk Density (g/cm^3) 1.36
Hydraulic Conductivity (cm/sec) 0.52
Porosity (decimal) 0.8

Desired Retention Time (min) 40
Average Dissolved P Concentration (mg/L) 0.012
Annual Flow Volume (gallons) 2365200.00
Desired Removal Goal (%) 25
Desired Lifetime (years) 2
Drainage Pipe Diameter (inches) 6
Drainage Pipe Slope (decimal) 0.006

Maximum Width of Structure (feet) 5
Maximum Length of Structure (feet) 5
Minimum Flow Rate (gpm) 40
Hydraulic Head (inches) 6

Note: Hydraulic Head is equal to the elevation drop over the maximum length or the elevation difference between inlet and ultimate outlet.



Design Bed Structure

Bed Output	
Mass Required (tons)	1.735
Depth of Material (inches)	6.
Depth of Structure (inches)	6
Length of Structure (ft)	5.
Width of Structure (ft)	1.43746
Number of Subsurface Drainage Pipes Needed	1
Actual RT (min)	Material is not sensitive to RT changes ... 0.0925076
Actual Maximum Flow Rate Through the PSM (gpm)	168.967
Estimated lifetime to meet both minFR and RT	-

Years	Cumulative Dissolved P(%)	Cumulative Dissolved P(lbs)
0.472	10.47	0.1219

Figure 4.3 Design sheet for P adsorption media from the PhROG software interface.

4.2.3 *Installation of P Adsorption Structure*

A trench was excavated downstream of an existing bioreactor near Baltic, SD based on the dimensions of the P adsorption media mentioned above. The adsorption media was placed in the trench connected to the downstream tile drain pipe of the existing bioreactor. A drainage control structure was installed in both upstream and downstream of the P adsorption media to measure tile flow from the media (Figure 4.4). The 2-chamber control structure was installed in both upstream and downstream of the structure. As mention earlier, the P adsorption structure was installed downstream of the bioreactor has a three chamber control structure to divert excess water through a bypass around the bioreactor and P adsorption structure.

Figure 4.4 Installation of the P adsorption bed control structure near Baltic, South Dakota

4.2.4 *Evaluation of P Adsorption structure*

Similar, to the bioreactor monitoring, the control structures installed upstream and downstream of the phosphorus adsorption structure contain sensors and pressure transducers to measure water depth, electric conductivity and temperature of the tile water (see section 3.2.2 from Chapter 3 for details about monitoring equipment).

Tile water grab samples were collected from the control structures upstream and downstream of the phosphorus adsorption structure. Samples were collected using a water bottle attached to a steel rod placed in each control structure. The collected water samples were immediately stored in a cooler and packed with ice until transported to the laboratory for analysis. The samples were manually filtered using a 30 mL HDPE syringe and a 0.45 µm nylon filter membrane into a pre-labeled 60 mL Nalgene bottle. The samples were kept at 0˚C until analyzed for phosphate using the acidic molybdate/antimony with ascorbic acid reduction (EPA Method 365.1) on a Seal AQ2 Discrete Analyzer (Mequon, WI).

Dissolved P concentration and tile flow were used to evaluate the performance of P adsorption structure. Percent removal of dissolved P was calculated as:

$$\%\text{Removal} = \frac{C_{up} - C_{down}}{C_{up}} * 100 \qquad (4.1)$$

where C_{up} is P concentration upstream (µg/L) and C_{down} is P concentration downstream (µg/L).

Tile flow depth inside the control structure was converted into flow rate (L/min) by using the flow equation mentioned in chapter 3 (equation 3.2 & 3.3).

Phosphorus removal rate per day (gm P/day) was calculated for each sample event using the following equation:

$$R_P = \frac{(C_{up} - C_{down})\, Q}{V_{sub}} \qquad (4.2)$$

where R_P is P removal rate (mg/min/m^3), C_{up} is upstream P concentration (µg/L), C_{down} is downstream P concentration (µg/L), Q is the flow rate (L/min), and V_{sub} is volume of metal chips used for the treatment (m^3).

4.2.4 Cost Estimation of Dissolved Phosphorus Removed

The cost to remove the amount of dissolved P from drain flow was estimated per acre per year. The P adsorption structure installation cost was categorized based on excavation and backfilling, control structure, pipes and fittings, personnel transport, and labor. The total installation cost for the P adsorption structure installed near Baltic, South Dakota was $4264 with a life expectancy of two years. Total nutrient load removed per year (gm/year) was estimated with the following equation:

$$\text{Phosphorus load removed (PLR)} = \left(C_{up} - C_{down}\right) * Q \qquad (4.3)$$

The total nutrient load removed per year was used to determine the cost to remove a unit mass of P or the cost per kg as:

$$C = \frac{Total\ \ Installation\ Cost / Year}{PLR} \qquad (4.4)$$

where *PLR* is P load removed per year (kg/year), and total Installation (cost/year) is the total amount spent for installation of the practice divided by the life expectancy of the practice (two years for the P adsorption structure).

4.3 Results and Discussion

4.3.1 Dissolved Phosphorus Concentration and Reduction

Upstream flow weighted dissolved P concentration ranges from 1.1 to 71.2 µg/L with an average of 25.9 µg/L, while downstream concentration ranges from 0.3 to 29.6 with an average of 11.5 µg/L in year 2016 (Figure 4.5). During the year 2016, which is the study period, drain flow ranged from 5 L/min to 2826 L/min from mid of May to the end of October (Figure 4.6). Bypass flows were also recorded during the flow measurement period. Flow greater or equaled to 1118 L/min is bypassed from the structure. Few bypass flow events were also recorded in June-October following rainfall events.

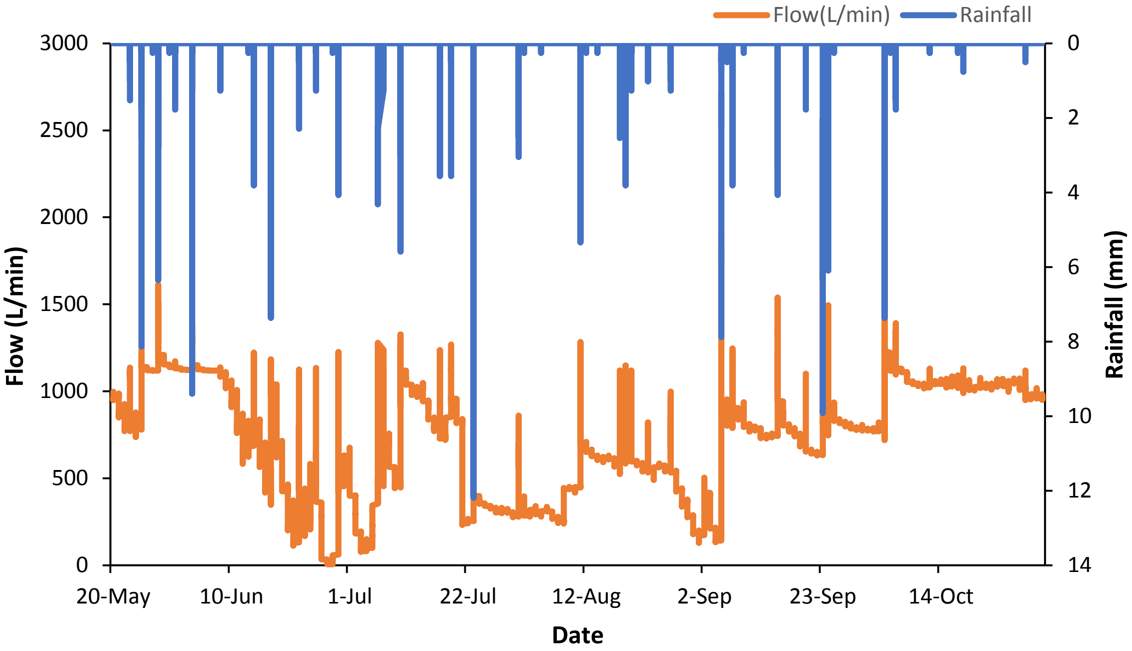


Figure 4.5 Drain flow and rainfall recorded at the phosphorus adsorption structure site near Baltic, South Dakota during the year 2016

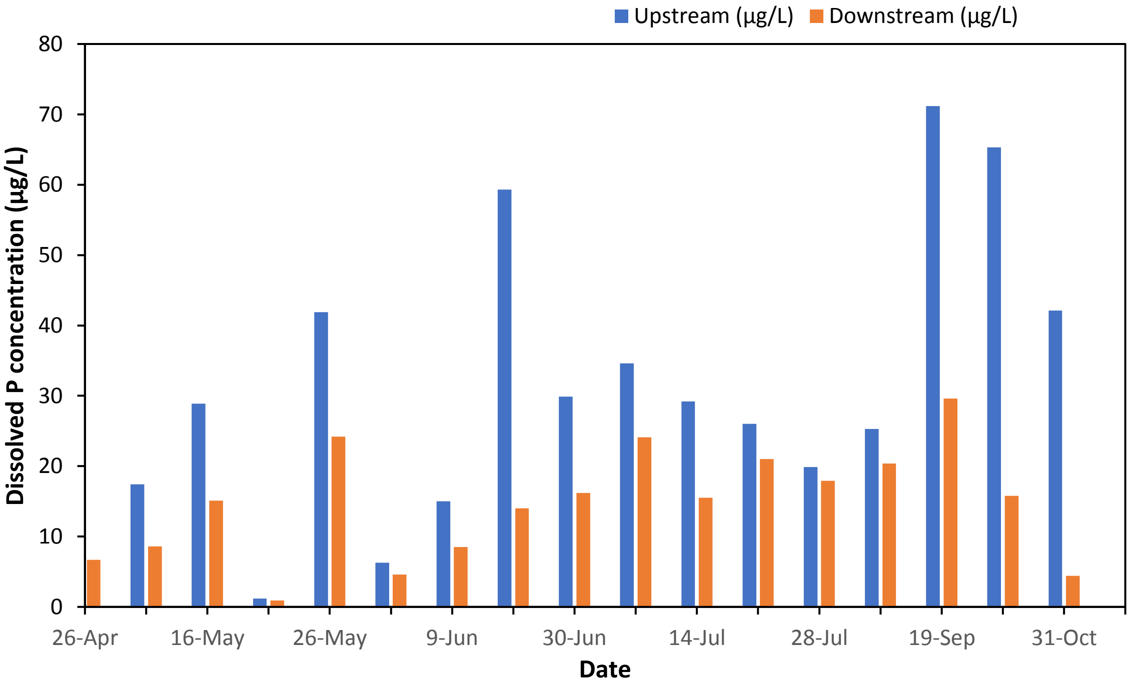


Figure 4.6 Dissolved phosphorus concentration in upstream (blue) and downstream (orange) near Baltic, South Dakota

Dissolved P reduction for the one year study period ranged from 10 to 90 % with an average of 45% (Figure 4.7). The percent of dissolved phosphorus reductions obtained for the phosphorus removal structure were comparable to P reduction of structures evaluated in a similar study conducted in Stillwater, Oklahoma (Penn et al., 2012), where the authors reported 54% of dissolved P removed. A filter designed with flue gas desulfurization (FGD) to remove dissolved P in runoff water from a 17 ha catchment area showed 65% reduction in dissolved P load during the three years monitoring period Bryant et al. (2012). However, the P removal efficiency decreased to 22% when bypass flow and base flow were considered.

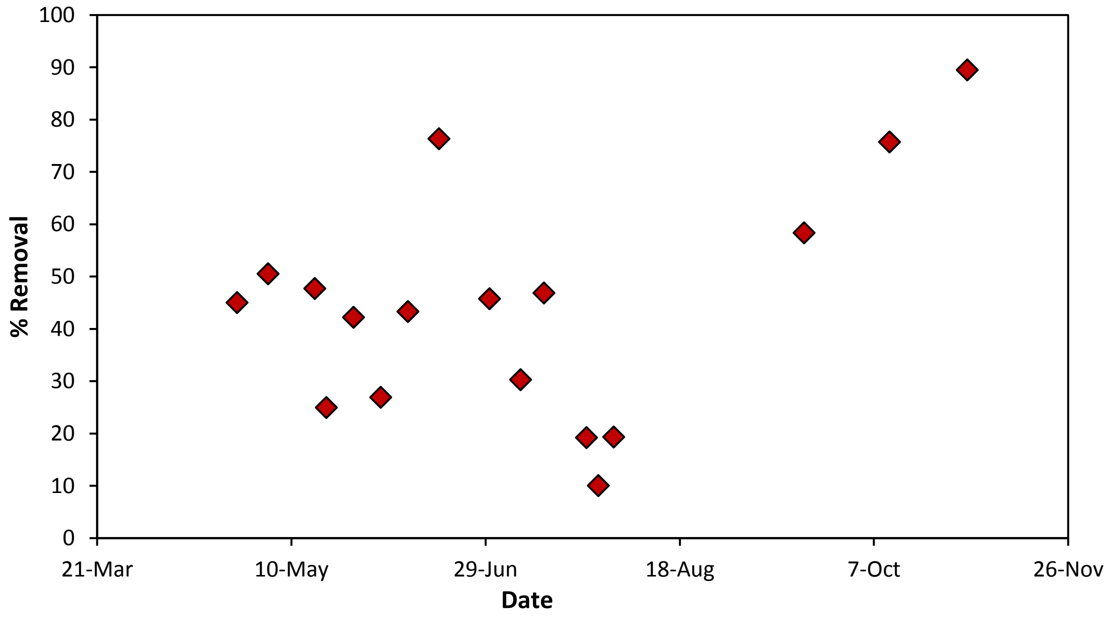


Figure 4.7 Percent dissolved P removed during the year 2016 with the phosphorus adsorption structure near Baltic, South Dakota

Dissolved P load removed by the structure ranged from 2.2 to 183.7 gm/m^3/day with an average of 49 gm/m^3/day. During the entire season, the total dissolved P removed was 4.5 kg. The removal rate of the structure varied according to the flow and initial P concentration. Upstream dissolved P load for the total sampling event ranged from 8.9 to 96.9 gm/day with an average of 36.3 gm/day including the bypass flow. Downstream dissolved P load ranged from 6.2 to 27.1 gm/day with an average of 15.3 gm/day.

4.3.2 Cost of Dissolved Phosphorus Removed

The cost to treat tile drain water for dissolved P per treatment area per year was estimated for the P adsorption structure (Table 4.2). The total installation cost per year of the phosphorus adsorption structure was $940 or $58 per year per hectare. Using the total dissolved P load removed per year, the cost of unit mass of dissolved P removed was estimated to be $209 per kg or $95 per lb.

Table 4.2 Cost details for installing and maintaining the phosphorus adsorption structure evaluated in this study

Cost category	Cost ($)	Interest	total cost	Cost/year	Replacement year
Excavation and back filling	1,000	60	1,060	530	2
Metal chips	0	0	0	0	2
Control structure and CTD sensor	1650	1353	3,003	75	40
pipes and fittings	700	294	994	49	20
Transport and media replacement	500	30	530	265	2
Travel and miscellaneous cost	400	328	728	18	40
Stop logs	14	2.52	17	2	8
Total cost/year				940	
Total treatment area				16.2	
Cost per treatment area				$58 /year/ha	
Total cost	4,264				

4.4 Conclusion

This study designed and evaluated a P adsorption structure installed downstream of an existing bioreactor near Baltic, South Dakota. Results showed that the dissolved P reduction range from 10 to 90%, based on upstream concentration of 1.1 to 71.2 µg/L. Averaged dissolved P removed from the structure was 49 gm/m^3/day, with upstream and downstream loads of 8.9 to 96.9 gm/day and 6.2 to 27.1 gm/day, respectively. Based on the cost estimates, average cost for a unit mass of dissolved P removed was $209 per kg. This study shows that the P adsorption structures is an effective edge-of field practice to remove dissolved P from drain flow but relatively expensive water quality conservation technology.

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CHAPTER 5: CONCLUSIONS

5.1 Summary

The objective of the study was to demonstrate and evaluate practices placed at the edge of fields to reduce nitrate-N and dissolved P loads from subsurface drainage systems in eastern South Dakota. The concentration based reduction of nitrate-N ranged from 7% to 100% for the four bioreactors, with upstream concentrations of 0.79 to 60.90 mg/L during 2013-2016 study period for the four bioreactors studied. Nitrate-N removal rates ranged from 0.01 to 14.6 gm N/m^3/day for the four bioreactors during the study period. Nitrate-N loads in the subsurface drainage water varied with variation in precipitation and drain flow. Upstream loads for the four bioreactors ranged from 0.01 to 15.5 kg N/ha/day and downstream loads ranged from 0 to 7.5 kg N/ha/day. The performance of the studied bioreactors appeared to decrease over the study period. The cost estimated to remove nitrate-N ranged from $11 to 61 kg/N for all four bioreactors. Two regression equations were developed and tested for predicting downstream nitrate concentration in eastern South Dakota. The equations generally performed reasonably at the bioreactor sites, except at one site (Baltic, South Dakota) where the equations seem to overestimate downstream concentrations.

Phosphorus adsorption structure was designed, installed and evaluated. Dissolved P reduction ranged from 10 to 90% with an average of 45% for the one year study period (2016). Dissolved P removal rates from the phosphorus adsorption structure ranged from 2.2 to 183.7 gm/m^3/day, with an average of 49 gm/m^3/day. The cost estimated to remove was $209 per kg of dissolved P. Cost analysis revealed that the cost for a unit mass of dissolved P removed appears expensive, indicating that the P removal structure could be an expensive practice for water quality protection.

5.2 Recommendations for Future Work

- Continuous evaluation of woodchip bioreactors over time is important to understand age effect on bioreactor.
- Understanding the microbial communities and activities will help in the future to develop ways to control the population of denitrifying bacteria.
- Measurement of additional water quality parameters such as dissolved oxygen and pH, and greenhouse gas emission will be helpful to increase understanding of bioreactor performance.
- Additional P adsorption structure with different P sorption materials is needed to understand the performance of these practices for water quality protection against P loading from subsurface drainage systems.