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Application of Drainage Water Management and Saturated Buffers for Conservation Drainage in South Dakota

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APPLICATION OF DRAINAGE WATER MANAGEMENT AND SATURATED

BUFFERS FOR CONSERVATION DRAINAGE IN SOUTH DAKOTA

BY

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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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LIST OF ABBREVIATIONS

Available soil water capacity(mm H2O (mm soil)$^{-1}$) .... SOL_AWC

Best Management Practice .... BMP

Centers for Disease Control and Prevention.. CDC

Conservation Innovation Grants .... CIG

Curve number for mositure condition II .... CN

Dissolved Reactive Phosphorus.. DRP

Drain tile lag time (hrs.) .... T_LAG

Drainage water management .... DWM

European Environmental Agency .... EEA

Evapotranspiration .... ET

Farm Service Agency .... FSA

Leaf Area Index .... LAI

Melt factor for snow on December 21 (mm H2O (˚C-day)$^{-1}$) .... MELTMN

Melt factor for snow on June 21 (mm H2O (˚C-day)$^{-1}$) .... MELTMX

Mississippi River Basin .... MRB

Plant uptake compensation factor .... EPCO

Root Mean Square Error .... RMSE

Snow melt base temperature (˚C) .... MELTTMP

Snow pack temp lag factor .... TIMP

Soil and Water Assessment Tool .... SWAT

Soil bulk density (gm cc^{-1})... BD

Soil evaporation compensation factor .... ESCO

Soil water…………………………………………………………………….…………...SW

Soil saturated hydraulic conductivity (mm hr^{-1}) .... SOL_K

South Dakota State University .... SDSU

The Louisiana Universities Marine Consortium ....LUMCOM

The United States Department of Agriculture ....USDA

Time to drain to field capacity (hrs.) .... T_FC

United States Environmental Protection Agency ....USEPA

volumetric water content .... VWC

Water Table Managment Technique 1 ....WTM1

World Health Organization ....WHO

LIST OF EQUATIONS

ABSTRACT

APPLICATION OF DRAINAGE WATER MANAGEMENT AND SATURATED BUFFERS FOR CONSERVATION DRAINAGE IN SOUTH DAKOTA.

ABHINAV SHARMA

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Edge of field practices such as drainage water management (DWM) and saturated buffers can reduce nutrient transport from croplands to surface waterbodies. DWM uses stackable weir boards in a control structure to manipulate the water table depth throughout the cropping season and reduce the amount of nutrient rich water draining out from the field. Saturated buffers, on the other hand, use a control structure to divert water draining out from the cropland to a vegetative strip via a subsurface tile installed parallel to a waterway. For the saturated buffer systems, a combination of natural denitrification, nitrogen mineralization, and plant uptake are the major causes of nitrate reduction. This study was conducted at three field sites across eastern South Dakota and the overall goal was to evaluate the effectiveness of DWM and saturated buffers in reducing nitrate loads transported from the field. A DWM site was established near Alexandria, SD in the fall of 2015 and two saturated buffer sites were established near Flandreau, SD and Baltic, SD in the summer of 2016. Water samples were collected weekly when water was flowing through control structure and analyzed for nitrate concentration in the lab. Flow records from the DWM sites were divided into two periods, the free drainage period and the managed period. Results for the Alexandria site showed that DWM reduced the total annual outflow by 8mm for 2016 during the managed period. In addition, nitrate concentrations between the two halves of the site

were compared and it was observed that the DWM half had lower nitrate concentrations as compared to conventionally drained half for most of the study. Annual nitrate loads for DWM and conventional half were calculated to be 3.3 kg ha^{-1} and 4.4 kg ha^{-1} during 2016 and 1.4 kg ha^{-1} and 2.3 kg ha^{-1} during 2017, respectively. Overall, DWM resulted in a load reduction of 26% during the managed period.

Results for the Baltic site show an overall nitrate concentration reduction of 95% for 2017, during which time 100% of water was diverted to the buffer. For Flandreau, the overall nitrate concentration reduction for 2016 was 86%, during which time 97% of water was diverted to the buffer and 65% for 2017 when 83% of water was diverted to the buffer. The lower reduction rate for 2017 was attributed to the high flow volumes that were diverted to the buffer zone throughout most of the season, resulting in inadequate nutrient uptake by the plants and insufficient time for natural denitrification.

In addition to the field study, a SWAT model was developed to assess the impact of the variability in soil properties and tile design parameters on flow volume reduction for DWM. The model was developed for the research site at Alexandria and daily measured flow data from the field study for 2016 were used for calibration, and 2017 was the validation period for the project. Model performance was evaluated using three statistics, NSE, RSR and PBIAS. The evaluation statistics ranged from 0.54 to 0.84 for NSE, -23% to 61% for PBIAS and 0.40 to 0.68 for RSR. It was concluded that SWAT simulations accurately represented the hydrological processes for the research site and that DWM resulted in an increase in ET, lateral flow and surface runoff while decreasing tile flow during relatively wet years. During dry years however, DWM resulted in an increase in tile flow. Apart from climatic conditions, DWM performance was also affected by variability in soil properties such as bulk density and available water capacity, and tile design parameters such as drain tile lag time and time to drain to field capacity.

A financial comparison between the two systems showed that DWM had a higher cost per pound of nitrate removed per acre at \$28 lb^{-1} ac^{-1} observed for Alexandria as compared to \$22 lb^{-1} ac^{-1} and \$0.6 lb^{-1} ac^{-1} for Baltic and Flandreau respectively. The lower cost for the buffer systems can be related to a higher cumulative load reduction for the study period.

Overall, both management practices were successful in reducing nitrate loads from drained croplands and expanding the model to a sub watershed or watershed scale could facilitate in decision making for agricultural water management in South Dakota.

Chapter 1. INTRODUCTION

1.1. Background

Tile drained croplands in the Midwestern US have been identified as a major contributor of nutrient loading to surface water bodies within the Mississippi River Basin (David et al., 2010; Goolsby et al., 1999). Each summer, the accumulation of excess nutrients, specifically nitrogen, results in a hypoxic zone in the Gulf of Mexico. In 2001, the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force setup an action plan to reduce the areal extent of the zone to 5000 km^2, but recent results show that the extent of the zone for 2017 was 22,720km^2, four times the goal. It was the largest zone measured in the area since monitoring began in 1984.

In South Dakota, an increase was observed in the number of tile drainage permits since 2006 (Finocchiaro, 2014). Increased streamflow in major waterbodies due to tile drainage pose a potential risk of increasing nutrient pollution downstream (Alexander et al., 2000b; Petrolia and Gowda, 2006). Conservation drainage practices like drainage water management (DWM) and saturated buffers have been developed and tested extensively in North Carolina and Iowa respectively to reduce nitrate loads delivered to surface waterbodies (Jaynes and Isenhart, 2014a; Skaggs et al., 2010). Drainage water management involves using a control structure to manipulate the water table depth to prevent water from flowing out from the field. This is achieved by stacking up boards in the structure during the growing season, forming a barrier for water outflow through the outlet. It also supplements the crop's water and nutrient requirements, thereby increasing the potential for nitrate load reduction (Evans et al., 1996; Frankenberger et al., 2004; Strock et al., 2010). Skaggs et al. (2012) compiled studies conducted across various research sites in the US and observed that the reduction rates were location specific and ranged from 18% to 85%. Similarly, Ross et al. (2016) summarized numerous studies that examined factors affecting DWM performance and concluded that tile design characteristics, climatic conditions, topographic conditions, and soil properties impacted DWM performance. The South Dakota component of the Transforming Drainage project, funded by the USDA, evaluated the performance of DWM on a plot scale setup at the South East Research Farm near Beresford, SD. The study concluded that DWM was successful in reducing the total outflow by 58% with load reductions ranging from 21% to 89%. DWM was also tested for its effect on crop yields, but the practice did not result in a yield increase (Sahani, 2017).

In contrast with DWM, saturated buffers allow water to flow from the field, but divert it to a vegetated strip using a subsurface tile running parallel to a waterway. Dosskey et al. (2002) found that subsurface buffers are more effective in reducing nitrate loads than buffers intercepting surface runoff. The N attenuation rates for a buffer is a function of soil and microbial properties exhibited by each site. Saturated buffers reduce nitrate loads by N immobilization, natural denitrification, and plant uptake (Jaynes and Isenhart, 2014a). Evaluation of saturated buffer effectiveness was part of a Conservation Innovation Grant project which spanned several sites across the Midwestern U.S. The annual report for the project showed an average nitrate reduction of 18% to 85% for the sites where more than 50% of the flow was diverted to the buffer zone (Utt et al., 2015).

Overall, both the practices have been shown to reduce N loads from croplands to receiving waters, but the selection of a Best Management Practice (BMP) is site specific. Christianson et al. (2013) concluded that DWM had an immediate effect on nitrate reduction as saturated buffers needed time to mature, but the quantitative reduction for buffers is more than that for DWM.

Utilizing BMP’s on a larger spatial scale is vital to achieve the desired N load reduction in the Gulf of Mexico. Using models can provide an accurate, economical method of evaluating the performance of BMP’s and are vital in the decision-making process for agricultural water management to achieve the desired water quality targets. SWAT is a semi-distributed model which uses a water balance equation to simulate hydrological processes at various spatial and temporal scales (Arnold et al., 1998). Since its development, SWAT has been used to determine the impact of various climatic and agronomic practices on water management in agriculture (Waidler et al., 2011). Sahu and Gu (2009) used SWAT to quantify the effect of riparian buffers in reducing nutrient loads to downstream surface water bodies in a watershed in Iowa. Other studies have also tested the use of SWAT in simulating tile drainage and its effect on watershed hydrology and water quality (Du et al., 2005; Green et al., 2006).

1.2. Problem Statement

An increase in land area used for agriculture in South Dakota combined with increasing interest in tile drainage use on croplands and changing rainfall patterns have resulted in an increase in streamflow and nutrient loads to major streams and rivers in the state (Dahlseng, 2013; Paul et al., 2017; Rajib et al., 2016). DWM and saturated buffers can potentially aid in reducing nutrient pollution from croplands. While, both the practices have been implemented successfully in other states, their use and effectiveness on the field scale in South Dakota has not been well documented. Using modeling in conjugation with field study can aid in better understanding the hydrological system and the effect of variability in climatic conditions and soil properties on DWM performance. Evaluating the performance of both the practices on field scale along with a cost comparison would be vital for agricultural water quality management for individual croplands in South Dakota.

1.3. Objectives

The objectives of this study were

- To evaluate the effectiveness of DWM and saturated buffers in reducing N loads from tile drained croplands
- To estimate the cost of systems per pound of nitrate removed
- To develop a DWM module for SWAT+, and use it to study the impact of DWM on field hydrology and crop yields

1.4. Significance of the study

The effectiveness of DWM and saturated buffers is location specific. This study focused on determining the feasibility of DWM and saturated buffer use for conservation drainage in South Dakota. The DWM module for SWAT+ will be critical in studying the effect of the practice on field hydrology under varying climatic conditions and water table management strategies. Results from the study could prove useful for agricultural producers and policy makers to improve water management across croplands in South Dakota.

Chapter 2. REVIEW OF LITERATURE

2.1. Nitrogen cycle

Nitrogen (N) is a vital nutrient for plant growth and is found in abundance in the earth's atmosphere, hydrosphere, and biosphere. However, more than 99% of this nitrogen is present in its non-reactive molecular form, N_2, with a triple bond between two the nitrogen atoms and, thus, is unavailable to more than 99% of the organisms. The energy required to break the bond can be achieved through very high temperatures or by nitrogen fixing microbes. These microbes convert N_2 to reactive forms, such as NH_4^+ and NO_3^-, which can then be utilized by plants. This continuous process of interconversion and movement of nitrogen in environment can be defined as the nitrogen cycle (Galloway et al., 2003). Ayres et al. (1994) concluded that the total denitrification processes and the microbial N fixation were equal prior to human intervention, including the application of fertilizers. These interventions have caused a shift in the balance, resulting in accumulation of reactive nitrogen in the environment at various spatial scales (Chindler et al., 1997; Galloway et al., 1995). Excess nutrient concentrations in the water impacts human as well as environmental health.

2.1.1. Excess Nitrates: Impact on human health

The maximum contaminant limit for nitrates in drinking water is 10 mg L^{-1} as per the United States Environmental Protection Agency (USEPA) regulations and 11 mg L^{-1} as per the World Health Organization guidelines (WHO, 2004). Nitrate is one of the most common chemical pollutants in groundwater aquifers around the world (Spalding and Exner, 1993). In Europe, high nitrate levels were observed mostly in private wells in rural areas (EEA, 2003). Studies elsewhere around the world in countries like China, Bostwana, Turkey, Senegal, and Mexico found elevated levels of nitrate exceeding the WHO guideline, with concentrations reaching over 68 mg L^{-1} in some cases (WHO, 2011)

Rural areas are susceptible to high nitrate concentrations in well water due to fertilizer use in agricultural areas. Gelberg et al. (1999) monitored nitrate levels in drinking water in rural New York and found that samples from shallow wells or springs were more likely to have higher concentrations of nitrates associated with them. It was also observed that wells near large farm areas were associated with higher nitrate concentrations. The most cost effective way to reduce nitrates in well water was to relocate wells away from the cropped area.

Pennino et al. (2017) observed an increase in the number of groundwater systems violations and the average duration of violation from 1994 - 2016. Nebraska and Delaware had the greatest proportion of systems under water quality violation. Ohio and California had the greatest average annual people affected by violated systems. Overall, the proportion of systems under violation increased between 1994 and 2009, from 0.28% to 0.42%, but decreased to 0.32% by 2016.

High nitrate concentrations in water cause infant methemoglobinemia, also known as blue baby syndrome, which affects infants up to six months and can prove fatal (WHO, 2011; Knobeloch et al., 2000; Gupta, 2000). Elevated nitrate concentrations have also been linked to spontaneous abortions among women (CDC, 1996) and increased mortality rates due to gastric and prostate cancer (Morales-suarez-varela et al., 1995). In addition, a study analyzing birth defects concluded that elevated levels of nitrates in drinking water led to birth defects among pregnant women. The study used data from the

National Birth Defects Program and observed the offspring of women consuming elevated levels of nitrate in drinking water to be more susceptible to birth defects such as limb deficiency, cleft palate, and cleft lip (Brender et al., 2013).

Nitrate removal from drinking water supplies can be a complex and expensive process achieved through methods such as reverse osmosis, ion exchange, and distillation. Ribaudo et al. (2011) estimated that nitrogen removal costs for a community water system can vary from $19,500 to $815,000 per year depending on the size of the water system. The cost-benefit ratio for the removal process vs prevention was difficult to accurately predict. Job (1996) estimated the cost of treatment was 30 – 40 times the cost of preventing the contaminant issue, while Heberling et al. (2015) concluded that the prevention costs were greater than the treatment costs.

2.1.2. Excess Nitrates: Impact on the Environment

In addition to human health concerns, high nitrate concentrations can adversely impact the environment. The presence of nutrients, such as nitrate and phosphorus, in water bodies supports the growth of algae, which has been cited as the primary cause for water quality impairment in many areas (Anderson et al., 2002; Goolsby et al., 1999; Turner et al., 2007). For example, algal blooms in ponds, lakes, and rivers due to excess nitrates in water reduce oxygen concentration in water and acts as a key stressor to aquatic flora and fauna (Diaz and Rosenberg, 2008).

An example of nutrient pollution, specifically nitrates affecting the coastal waters, in the United States is the hypoxic zone in the Gulf of Mexico. Nutrient loading into the gulf results in the formation of a low oxygen, or hypoxic, zone every year. The hypoxic zone has been found to be more persistent and severe in spring and summer months (Turner et al., 2005). Hypoxia occurs when a water body is deprived of adequate oxygen and has a detrimental effect on aerobic and aquatic organisms, including reduced growth, loss of reproductive capacity, increased mortality rates, and a reduction in biodiversity(Diaz and Rosenberg, 1995).

Vaquer-Sunyer and Duarte (2008) worked on an extensive literature search to evaluate oxygen thresholds for lethal and sub lethal oxygen concentrations in waterbodies. They found that there was not a fixed dissolved oxygen concentration level which was lethal for most organisms and concluded that 2 mg L^{-1}, the conventionally defined upper limit of oxygen concentrations for hypoxic zones, was under the actual level that resulted in ecological health decline. For example, Chabot and Dutil (1999) found that cod growth was reduced when oxygen levels were less than 7 mg L^{-1}, while dissolved oxygen concentrations of 2 mg L^{-1} were unfavorable for shrimp growth. Additionally, sharks and rays migrate to more favorable, higher oxygen locations once the dissolved oxygen concentration drops below 3 mg L^{-1} (Rabalais and Turner, 2001).

The hypoxic zone in the Gulf of Mexico is formed when nutrient rich water flows into the gulf and stimulates growth of the phytoplankton biomass offshore. This biomass acts as the energy source for the coastal food web and results in carbon loading at the bottom layers of the ocean. Next, bacteria decompose the carbon and consume dissolved oxygen in the water during the process. This results in low oxygen levels at the bottom of the water body and layers of different oxygen concentrations within the same water body, also called stratification. The condition worsens as the bottom layer is not resupplied with oxygen by surface water. The marine ecosystem in these zones is greatly affected as the low oxygen zones cannot support many aquatic species and ultimately result in a dead zone. A dead zone is defined as an area or region where the low oxygen conditions results in the loss of most of the existing marine life. The number of such dead zones have been increasing over the last two decades and currently there are over 550 such zones in the world (Diaz and Rosenberg, 2008; Howarth et al., 2011; Rabalais et al., 2010; Conley et al., 2011). The zone formed in the Gulf of Mexico is the second largest human caused coastal hypoxic zone worldwide. To address the issue, a task force comprising of researchers, engineers, and policy makers was formed. The Mississippi River/Gulf of Mexico Watershed Nutrient Task Force was setup in the fall of 1997. It focused on understanding the causes and effects of eutrophication in the gulf area as well as plan activities that could help reduce the size and severity of the hypoxic zone. In 2001, the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force developed an action plan to reduce the size of the zone to 5000 km^2.   The zone size for the last 5 years was consistently well over the goal setup by the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force of 5000 km^2 with an approximate average size of 15,000 km^2 from 2013 – 2017 (Figure 2.1).

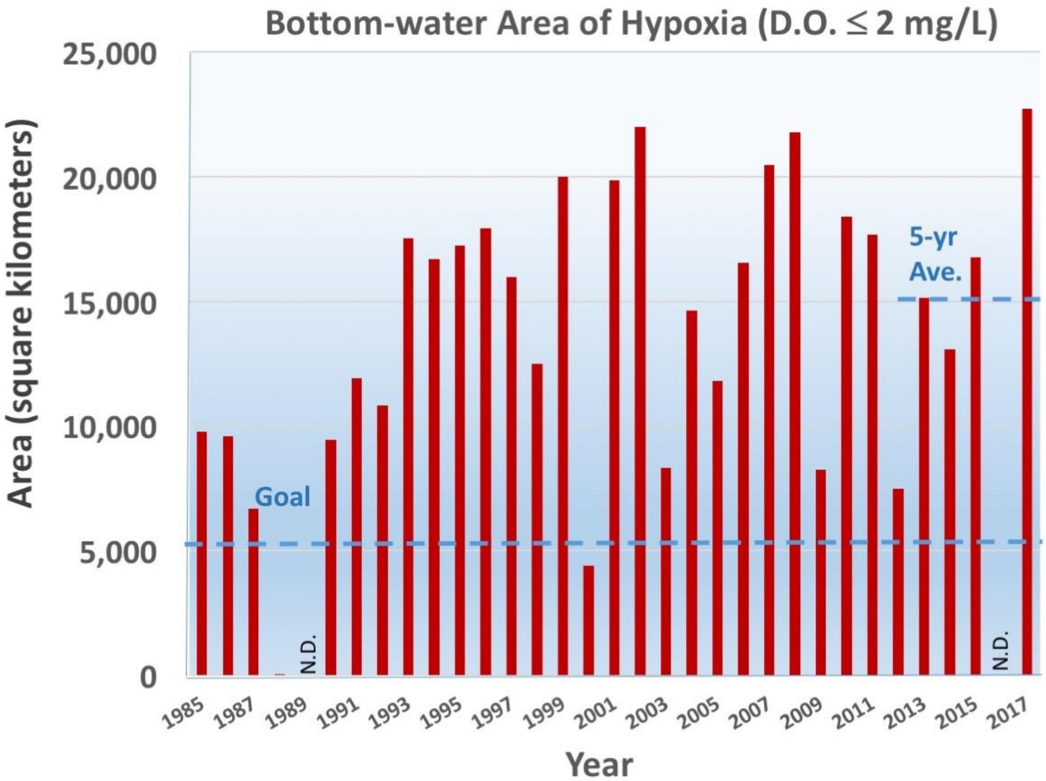


Figure 2.1 The areal extent of the hypoxic zone from 1985 – 2017 (red bars) has been consistently above the 5000km2 goal identified by the nutrient management task force (dashed blue line) since monitoring began in the 1980's (Rabalais and Turner, 2017)

The Louisiana Universities Marine Consortium (LUMCON) documents the areal extent of the zone each year. For 2017, the extent of the zone was 22,720 km^2, the largest measured to date and almost 4.5 times the goal of 5000 km^2 (Figure2.1). Also, the maximum nutrient loading occurred in May and June, similar to the findings for 2015.

While LUMCON records and documents the hypoxic zone each year in Gulf of Mexico, the monitoring and management efforts for water quality in South Dakota waters are headed by South Dakota's Department of Environment and Natural Resources (DENR). A total of seven water development districts were setup to promote conservation, development, and judicious use of water resources in the state. Each water development district runs various projects and activities including groundwater studies, development and implementation of best management practices (BMPs) affecting water quality and quantity, organizing water festivals to promote water education, and providing cost share assistance to the local communities for the improvement of water resources (Jarrett et al., 2017). For example, the East Dakota Water Development District runs a nitrate monitoring program for the Big Sioux River and its tributaries. The monitoring started in 2015 and included 28 sites, but was reduced to 25 in 2017. Water samples were taken from April through November. Results for all the three years showed an increase in the streamflow as compared to long term averages, which can be attributed to an increase in area under agriculture and a subsequent increase in subsurface drainage on the cropland. Most samples tested for nitrate were within the acceptable range (< 10 mg L^{-1}), owing to the dilution in the stream; however there were some instances when monitoring sites located near Skunk Creek, Bachelor Creek, and the Big Sioux River near Watertown, South Dakota had nitrates higher than the rest of the observation points. Also, additional points downstream and outside the development district, which were analyzed in 2016 and 2017, were found that to have nitrate concentrations consistently above 10 mg L^{-1} (Marine et al., 1990), Apart from nutrient pollution in the rivers and streams, Dakota, (2018) observed that there were 43 lakes in South Dakota experiencing hypereutrophic conditions. Hyper eutrophic zones in water bodies have been characterized with high nutrient content, frequent algal blooms and poor visibility in the water body. 75 lakes, covering 76999 acres were classified under eutrophic status which is characterized by comparatively lesser nutrients than the hypereutrophic classification but, the nutrients may result in an algal bloom and then fish kills (Meyer-Reil and Köster, 2000).

2.2. Agriculture in the Midwest: Contribution to nutrient pollution

Agricultural drainage from the Midwestern corn belt is a major contributor to nutrient pollution, including to the Gulf of Mexico (Burkart and James, 1999). This region experienced extensive hydrological modifications with an increase in the area under subsurface tile drainage and the channelization of streams, which lower the water tables on agricultural fields and provide an easy path for water draining from the field to major river systems (Baker, 2008). Tile drainage also reduces the flow path for nutrients to the riverine systems (Baker and Johnson, 1981; David et al., 2003; David et al., 1997; Dinnes et al., 2002; Gentry et al., 2007).

Figure 2.2 Major rivers in the Mississippi river basin (MRB) draining into the Gulf of Mexico (Goolsby et al., 1999).

M. B. David et al. (2010) studied N yield from each county in the Mississippi River Basin (MRB) and concluded that corn fields in conjugation with tile drainage are the major source of riverine N yields in the upper MRB. These results are similar to the ones discussed by Broussard and Turner, (2009) who reported corn fields to be the dominant source of N pollution across the U.S.

In South Dakota, a significant shift from grasslands to croplands was observed from 2006 to 2012 (Table 2.1) (Reitsma et al., 2014). Johnston (2013) identified the expansion in corn-soybean acreage to be the major factor affecting the decrease in wetland area in South Dakota. A recent increase in tile drainage permits from 2006 to 2013 demonstrates the expansion in tile drainage observed in the state which can alter nutrient transport (Finocchiaro, 2014).

Table 2.1 Change in area under agriculture in South Dakota from 2006-2012 (Reitsma et al., 2014).

Land use (acres)	2006	2012	Change	95% CI*
Cropland	15,546,600	16,986,100	1,439,500	15,600
Grassland	28,327,300	26,490,300	-1,837,100	21,100
Non – ag	1,590,300	1,617,700	27,400	110
Habitat	2,834,400	2,961,300	126,800	690
Water	1,055,600	1,299,000	243,300	860

*95% confidence interval

2.3. Agricultural Drainage

Agricultural drainage is the removal of excess water from poorly drained soils. The water is removed from the surface through drainage ditches or the subsurface through artificial tiles. Evans et al. (1996) stated that land drainage has been used in North America since the 1600's, but it was during the late 1800's that European settlers started using drainage ditches to channel water from their farms to nearby streams and rivers (Busman and Sands, 2002). Consequently, subsurface tile drains have been used in the U.S. for over 150 years to improve crop yields (Blann et al., 2009). It is particularly useful in areas with poorly drained but productive soils.

2.3.1. Agricultural Drainage: Soil and water

Subsurface tile drainage has been extensively used to enhance water transport through the soil (Kalita et al., 2006); improve the timeliness of various field operations such as tilling, planting etc. (Baker et al., 2004); increase the infiltration rate of water in the soil profile (Skaggs et al., 1994); and decrease the surface runoff (Kladivko et al., 2001).

The difference between a soil's saturation point and field capacity is the water available for removal by subsurface drainage. Water in the soil pores is held by two forces, the weaker capillary force acting between two pores and the stronger adsorptive force acting as a film surrounding individual pore. When the soil is at saturation, capillary forces are not strong enough to hold the water making it easy to drain water from the soil, however, when the soil reaches field capacity, there can be no drainage. The drainage characteristics of a particular soil also depend on the composition of soil solids. Larger soil particle sizes increase the water holding capacity and drainage capability of the soil. For example, sandy soils have better water holding capacity and drainage properties than clayey soils. Water table depth can also be related to the soil particle size. Soils comprising of small particle size drain water less efficiently and have shallow water table.

Subsurface drainage impacts the soil water balance (Sands, 2001). The water balance is represented as the mass balance of water in and out of the system.

$$P = R + ET + DP + S + D \qquad \text{(Equation 2.1)}$$

Where P is the combination of precipitation, snow melt, and irrigation (mm); ET is evapotranspiration (mm); DP is deep percolation and seepage (mm); R is runoff (mm); S is soil storage (mm); and D is drainage flow (mm).

Here, P is the major input to the soil in the form of precipitation, the melting of snow, or irrigation. Some of the water added to the system (P) is lost in the form of surface runoff, crop evapotranspiration, deep percolation, and drainage flow. In drained areas, drainage flow is the major component of water loss from the system (Sands, 2001).

A subsurface drainage system consists of a network of drain pipes typically 1- 2 m below the soil surface at a suitable grade to remove excess water from the fields. This zone is vital for crop root development and excess water present can inhibit plant growth (Sloan et al., 2016). The water flows into the perforated tiles by gravity and is routed to the outlet.

Though beneficial, tile drainage is also a major pathway for water soluble agro chemicals including nitrate to surface water bodies (Kladivko et al., 1991; Kladivko et al., 2001; Randall and Iragavarapu, 1995; Buhler et al., 1993; Kalita et al., 1998; Kalita and Kanwar, 1993; Logan, 1993). Due to increased subsurface flow from the field to outlet, nitrogen ions in subsurface soil water do not reduce to simpler, inert forms such as N_2 and are instead transported as NO_3^-, NO_2, and NH_4^+ to surface waters.

2.3.2. Agricultural Drainage: Nitrogen in soil and water

Nitrogen plays a vital role in plant nutrition. Atmospheric nitrogen is present in abundance but is biologically unavailable to the plants. It must be converted to another form for plants to use. Nitrogen fixation and nitrification convert nitrogen into more reactive, usable forms, such as nitrate, nitrite, and ammonium (Galloway et al., 2003). In agriculture, nitrogen is the major factor limiting crop production and fertilizers rich in major nutrients, such as nitrogen, phosphorus, and potassium, have been used to meet crop requirements and obtain ever increasing yield targets. Manure and chemical fertilizer application provides the plants with reactive inorganic nitrogen in the form of nitrates or ammonium ions (Galloway et al., 2003). The increased use of fertilizers has accelerated Earth's nitrogen cycle by increasing nitrogen fixation (Hofstra and Bouwman, 2005). The presence of excess nitrate in agricultural soils has been attributed to the use of fertilizers and chemoautotrophic nitrifying bacteria that oxidize ammonium ions under aerobic conditions (Betlach and Tiedje, 1981).

Nitrate and ammonium forms are readily taken up by plants due to their solubility in water. The residual nitrogen remains in soil water and moves out of the root zone with natural subsurface water movement or artificially by subsurface drainage (Rosen and Horgan, 2013). The rapid movement of water through an artificial subsurface drainage system, reduces the retention time in the soil, thus reducing the time necessary for denitrification (Kellman, 2005).

To reduce nitrogen outflow from croplands, many in-field and edge of field practices have been studied. Drainage water management and saturated buffers are two such edge of field best management practices which employ different mechanisms to reduce nitrate loads flowing from croplands into receiving waters (Dinnes et al., 2002).

2.4. Drainage water management

Drainage water management, also called controlled drainage, is a water management practice to reduce water outflow from tiled croplands. It is practiced on the edge of a field with a slope of < 1% (Strock et al., 2010). It uses a control structure to adjust the water table depth of the field during the growing season and prohibit water outflow from the field. Raising the water table also supplements the water and nutrient requirements of the crop (Frankenberger et al., 2004; Skaggs et al., 2012b; Strock et al., 2010).

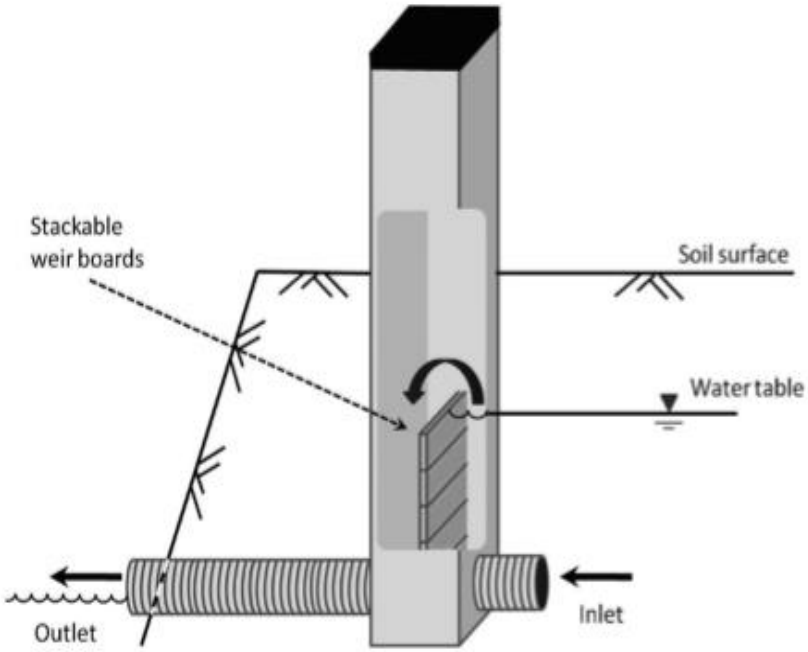


Figure 2.3 A water control structure consists of three components: an inlet, an outlet, and stop logs or stackable weir boards. The inlet receives water draining from the field via a subsurface tile. The outlet drains out water to the receiving water body. Stop logs or stackable weir boards are used to raise the table depth during the growing season (Strock et al., 2010).

2.4.1. <u>Drainage water management: Effect on hydrology and water quality</u>

DWM was developed to reduce subsurface flow from a hydrological system. To reduce drainage, either one or multiple other components of the water balance (ET, R, DP, and/or S) will also be affected by DWM.

Skaggs et al. (2010) used DRAINMOD to complement a previous field scale study by Gilliam et al. (1979) involving the use of DWM to reduce subsurface water outflow and nitrate loads. He concluded that DWM was successful in decreasing drainage flow and that the net decrease was a result of an increase in ET, seepage, and runoff. The magnitude of DWM impact on the components of the water balance depended on soil properties, site conditions, subsurface drainage intensity, climatic factors, and on-field management. The effect of DWM on N losses is impacted by the net reduction in subsurface water outflow and site-specific processes that govern N attenuation rates in the soil. Skaggs et al. (2012) compiled a list of studies examining the effectiveness of DWM in reducing drainage volumes and nitrate loads. For all the sites studied, DWM resulted in annual drainage volume decrease ranging from 18% to over 89% (Cooke and Verma, 2012; Drury et al., 2009; Fausey, 2005; Gilliam et al., 1979; Jaynes, 2012).

Numerous studies have been conducted to evaluate the factors affecting DWM performance. Fang et al., (2012) used a modeling study in conjugation with field study to evaluate the effect of N application rates and weather conditions on DWM performance. It was concluded that variability in precipitation patterns, from 600mm - 1100mm resulted in a much larger variation in Nitrate load reduction results, which ranged from 20% to 50%. The N application rates, on the other hand which ranged from (0-250) kgNha^{-1} resulted in 38% to 40 % nitrate load reduction.

DWM performance can also impacted by tile design, inherent soil properties and water table management strategies. Four studies in Ontario, Canada had reductions from 16% to 80% in water volume. All were conducted on a similar scale with areal extents ranging from 0.1 ha to 2.2 ha. Out of the four studies, three had the same soil type, Brookston Clay Loam, and resulted in similar reduction rates, 16% - 29% reduction in drainage water volume (Gaynor et al., 2002; Drury et al., 2009; Tan et al., 1998). The fourth study had a different soil type along with a deeper control depth and exhibited higher reduction in volume (80%) than the other three.

Lalonde et al. (1996) observed an effect of control depth on DWM performance. The sites for the study were setup using two different control depths 0.75 and 0.5m. The reduction in flow volumes for DWM were 49% and 80% for 0.75m and 0.5 m control depth, respectively for the two years under study. A larger flow reduction for smaller control depth was a result of more storage under the 0.5m control depth. The study did not evaluate the effect on deep percolation but in general, excess water storage can result in wet water stress for a crop and has been studied to reduce crop yields for a DWM system(Ale et al., 2009).

In addition to varying control depths, water outflow can be controlled by the water table management strategies used on the field. Jacinthe et al., (1999) used a soil column study to study the impact of different water management techniques on nitrate load reductions. He tested two techniques, WTM1 and WTM2 based on the total reduction in nitrate loads. WTM1 involved a static water table maintained at 0.5m below the soil surface for 92 days. It was then raised to 0.1m for 18 days. WTM2 involved a dynamic water table control throughout the study period. The water table was held at 0.5m for 7 days, raised to 0.1m for the next 4 days. It was then lowered to 0.7m for the next 4 days and then held at 0.5m for the next 43 days. Finally, it was raised to 0.1m and held there for 18 days. The study concluded that the rate of nitrate removal increased when water table was perched near the soil surface. The reduction rates ranged from 9 - 14% for WTM1 as compared to 24 - 42% for WTM2.

Ross et al. (2016) compiled various field scale, plot scale, and modeling studies comparing DWM with conventional drainage and concluded that around 90% of the studies focused on already established literature on tile drainage such as tile depth, tile spacing, soil type, etc. They identified a need for future work on predictor variables such as drain diameter, fertilizer application method, and fertilizer timing on DWM performance.

For DWM, reduction in flow volumes from cropland was the major factor contributing to nitrate load reduction in agricultural lands (Gunn et al., 2015; Jaynes, 2012). Although, raising the water table supports the development of anaerobic conditions in the soil which can result in the denitrification of reactive nitrogen, such as nitrate to a much more stable form N_2. Denitrification in soil requires a carbon source along with the anaerobic conditions, which is affected by various soil properties such as organic matter, bulk density, organic carbon, soil pH, and temperature (Bremner and Shaw, 1958).

Overall, DWM has successfully demonstrated a reduction in nitrate loads, but no difference was observed for nitrate concentrations (Skaggs et al., 2012b; M. D. Sunohara et al., 2014) except for (Frey et al., 2013, 2016). Reduction in nitrate loads as a result of reduced outflow ranged from 18% to 85% in studies conducted in various places such as Ontario, Sweden, Illinois, North Carolina, Iowa, Indiana and Ohio(Cooke and Verma, 2012; Drury et al., 2009; Fausey, 2005; Gilliam et al., 1979; Helmers et al., 2012; Lalonde et al., 1996; Tan et al., 1998; Wesström and Messing, 2007). The trends in nitrate concentration reduction were similar to the ones in flow reductions for each study. In addition to nitrate concentration, DWM was tested successfully in reducing ammonium, total phosphorus, dissolved reactive phosphorus, E. coli, and Enterococci (Sunohara et al., 2016).

Jaynes et al., (2010) studied the potential impact of DWM application for the entire Midwest in reducing nitrate loadings to the Gulf of Mexico. He concluded that it could result in about 6% of the required 45% reduction goal to fix the hypoxic zone issue in Gulf of Mexico if implemented effectively.

2.4.2. Drainage water management: Impact on crop yield

Apart from the studies related to water quality, there were also some that evaluated the impact of DWM on crop yields (e.g. Skaggs et al., 2012a). DWM resulted in an increase in crop yields by 1% to 19 % due to higher water tables that supplement the crop's water and nutrient requirements (Delbecq et al., 2012; Ghane et al., 2012; Helmers et al., 2012; Poole et al., 2011; Wesström and Messing, 2007). However, not all studies showed favorable impacts on crop yields; some observed no impact from DWM on crop yields which was attributed to the loss of soil aeration from the raised water table (Cooke and Verma, 2012; C. F. Drury et al., 2009; Fausey, 2005; Tan et al., 1998). In one instance, negative yield impacts were observed (Helmers et al., 2012). The relatively brief period of observation in many of the studies may have influenced the ability to detect effects of DWM on crop yields.

The limited number of studies combined with the short study durations and variable results indicates the need for more work on determining the impact of DWM on crop yields for different soils, weather conditions, drainage designs, and management strategies.

2.4.3. DWM in South Dakota

Limited information exists on DWM performance in South Dakota. Sahani, (2017) evaluated DWM on a plot scale as part of the Transforming Drainage project at sites setup near Beresford, South Dakota. DWM resulted in a 58% reduction in flow, but the nitrate concentrations between the DWM and conventional plots were not statistically different. A paired field approach was used for evaluation of DWM performance on a plot scale. A plot scale study limits the effect of variability in soil and topography, but there would be an uncertainty in results due to lateral flow between the plots unless a physical barrier was used to separate them. This uncertainty in lateral flow should be taken into account when evaluating DWM performance(Skaggs et al., 2010).

2.5. Modeling DWM

Modeling presents a possibility to comprehend the spatial and temporal variability of management practices, including DWM. Ross (2003) suggested that future research on DWM should be expanded to a watershed or larger scale using modeling techniques which are scientifically sound and less expensive than implementing these systems in fields across a watershed.

2.5.1. Hydrological modeling: Introduction

The purpose of a model is to represent an actual system. Hydrologic models have been developed to assess the impacts of various environmental parameters on the hydrology of a system.

Hydrologic models can be classified into two types: physical models and empirical models. Physical models use a process to describe and study a system whereas, Empirical models represent a system using a mathematical relationship between variables, for example using regression models and Artificial Neural Networks. Another way of classifying models is the way they treat randomness in the variables. A deterministic is a model that does not account for randomness in the system. A given input value will always result in the same output value, provided all other variables are constant. Such a model is useful when makings forecasts. A stochastic model, on the other hand incorporates a degree of randomness in its variables. Stochastic models are useful when making predictions. It uses a statistical distribution to study a system (Refsgaard, 1990).

Computer-based hydrological models can also be classified as lumped, semi-distributed, and distributed models (Singh, 1988), on the basis of spatial correlation. A lumped model is a deterministic model which accounts for the spatial average of the system and does not account for randomness in the system. Some examples include the Stanford watershed model (Crawford and Linsley, 1966) and the HBV model (Bergstrom, 1976). A semi-distributed model incorporates homogeneity in some variables and also defines some variables as a function of spatial dimensions. Examples include SWAT (Arnold et al., 1998) and TOPMODEL (Beven and Kirkby, 1979). A distributed hydrological model will divide a basin into elementary units each with spatial correlation with one another. Examples are the Institute of Hydrology Distributed Model (Beven et al., 1987) and SHE (Abbott et al., 1986).

2.5.2. Hydrological modeling: SWAT.

The Soil Water Assessment Tool (SWAT) is a semi-distributed hydrological model widely used for studying hydrological processes at basin, watershed, and sub-watershed scales (Arnold et al., 2012). SWAT was developed to predict the impact of management on water, sediment, and chemical yields (Arnold et al., 1998). The earliest version divided the watershed into sub-basins, each of which had an impact on the hydrology, but had its own dominant land use and soil type. All sub-basins in the watershed were spatially referenced to one another. Each sub-basin comprised of lumped land units called hydrologic response units, each having a unique land cover, soil type, and land slope.

Each SWAT run simulates the hydrological cycle for the system and the processes involved are divided into two modeling phases: the land phase, which emphasizes the movement of water, nutrients, sediment, and pesticides in the main channel; and the routing phase, which emphasizes the transport of water, sediments, nutrients, and pesticides through the channel network to the watershed outlet. The hydrological processes are based on the water balance equation (Equation 2.2) (Neitsch et al., 2011).

$$SW_t = SW_o + \sum_{n=i}^{t}(P - Q_{surf} - ET - W_{seep} - Q_{gw}) \qquad \text{(Equation 2.2)}$$

Where, SW_t - SW_o is the change in soil water storage, P is the daily precipitation, ET is the evapotranspiration, Q_{surf} is the surface runoff flow, Q_{gw} the groundwater flow, and W_{seep} is the deep aquifer recharge.

Since its inception in the 1990's, SWAT has undergone significant changes. The latest release, SWAT+, has some major alterations compared to previous versions such as the introduction of routing units which replace the sub basin division used in previous versions, and addition of land surface units.

2.5.3. Model evaluation

Comparison of modeled values with measured values is a vital step to study model performance. Various statistical indices were developed and are used to study model output parameters such as streamflow, sediments, and nitrates. Some of the most widely used indices are Nash-Sutcliffe Efficiency (NSE), PBIAS (percent bias) and RSR (Moriasi et al., 2007).

Mathematically,

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right)^2}{\sum_{i=1}^{n}\left(Y_i^{obs} - Y^{mean}\right)^2}\right] \quad \text{(Equation 2.3)}$$

$$PBIAS = \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right) * (100)}{\sum_{i=1}^{n}\left(Y_i^{obs}\right)}\right] \quad \text{(Equation 2.4)}$$

$$RSR = \frac{RMSE}{STDEV_{obs}} = \left[\frac{\sqrt{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right)^2}}{\sqrt{\sum_{i=1}^{n}\left(Y_i^{obs} - Y^{mean}\right)^2}}\right] \quad \text{(Equation 2.5)}$$

NSE represents the ratio of residual variance to the variance of the measured values. It ranges between -∞ to 1, with 1 being the optimal value (Nash and Sutcliffe, 1970). NSE is widely used for modeled streamflow evaluations and is recommended as one of the most accurate indices to study hydrographs (Servat and Dezetter, 1991).

PBIAS measures the over and under prediction of the modeled value in contrast to the measured value. 0% is the optimal value showing no bias in prediction, a negative value represents under prediction by the model and vice versa (Gupta et al., 1999).

RSR is the ratio of RMSE to the standard deviation of the observation. It compares the residual variance with the variance of the measured value. Zero is the optimal value and one indicates equal variation for the residual and the observation (Moriasi et al., 2007).

2.5.4. SWAT: Modeling tile drainage and BMPs

Numerous studies have simulated tile drainage in SWAT. Some have discussed the various tile drainage routines available for different versions of SWAT (Guo et al., 2018), whereas some have studied its impact on water quality in different watersheds (Boles et al., 2015; Du et al., 2005; Green et al., 2006). Tile drainage has been identified to be a source of nutrient pollution downstream, Lu et al., (2016) used SWAT to study dissolved reactive phosphorus (DRP) transport in tile drained croplands in Denmark. They developed an extension DrainP for SWAT2012, which successfully simulated P leaching throughout soils with Langmuir isotherm and its subsequent transfer to rivers on a monthly scale. Tile drainage was identified to contribute 46% of the total DRP transport for the study area. Ikenberry et al., (2017) used SWAT to study the flow pathways and soil nitrogen dynamics for tiled croplands and accurately simulated monthly water yield and NO3-N loads for the study watersheds.

For South Dakota, SWAT model was used to study the impact of climate and land use change on water quality on downstream water quality for the Big Sioux watershed. It was concluded that shifting the land use to hay/pasture resulted in a 3-14% decrease in surface runoff, sediment, phosphorus and nitrate loads for all the three climate scenarios used (Rajib et al., 2016).

In addition, SWAT has been used to study the potential effect of various BMP's on water quality. Kalcic et al., (2015) studied the effect of six management practices including no till, cereal rye, cover crops, filter strips, grassed waterways, created wetlands and restored prairie habitats on water quality in two watersheds in Indiana. It was concluded that the use of BMP's could potentially lead to a 60% reduction in the total pollutant loads. Sahu and Gu, (2009) used SWAT to model effects of buffer strips on stream water quality. It was concluded that buffer strips could be helpful in removing 55-90 % of nitrates from the sub basin. The study included running the model for different precipitation patterns, and different sizes of buffer strips. The results from the study were suggested to support the decision-making process for selecting the best management practices for nutrient management on a watershed scale.

2.6. Saturated Buffers

A saturated buffer is a vegetated strip that is fed with nutrient rich water diverted from the field through tile drainage to promote N attenuation. Like DWM, a saturated buffer is also employed along the edge of the field and can only be used on land with gentle slopes (less than 1%). The buffer width, tile line depth, plant variety, and other design characteristics depend on location, climate, and soil properties for the site.

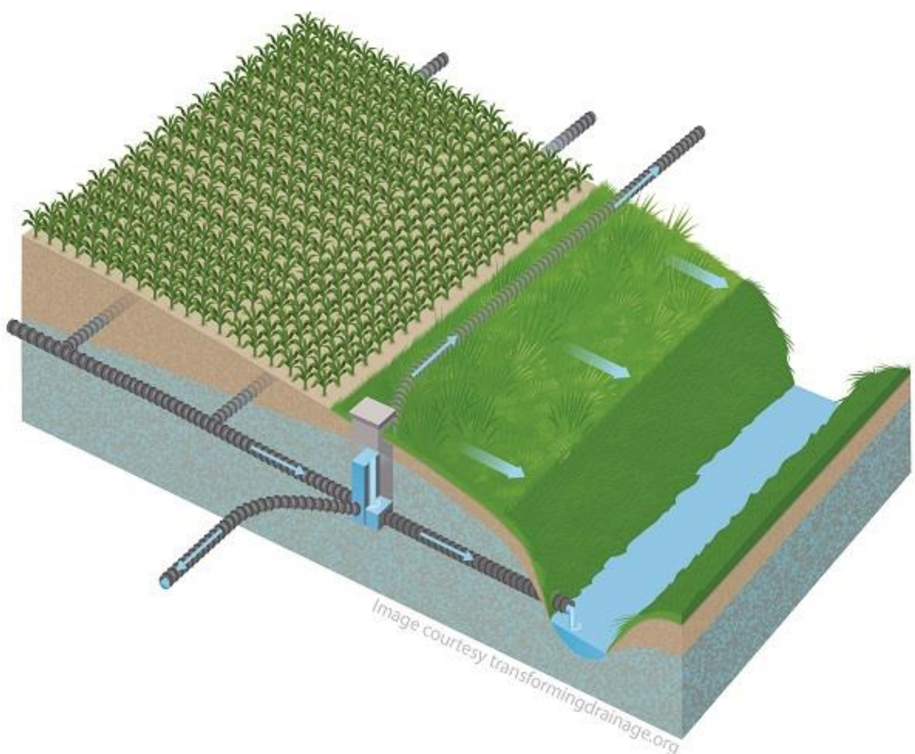


Figure 2.4 Depiction of a saturated buffer site. The control structure takes in water from the cropland, then diverts it to a vegetative strip placed parallel to a waterway. Plant uptake and denitrification result in the net reduction in nitrate load (Reinhart et al., 2016).

2.6.1. Nitrate attenuation processes

Reactive forms of nitrogen in the soil can be reduced by N immobilization, natural denitrification, and plant uptake. Immobilization refers to the conversion of ammonium to glutamate by microbes and other organisms. A combination of denitrification and plant uptake has been identified as the major source of nutrient reduction for buffers (Jacobs and Gilliam, 1985; Schipper et al., 1993; Haycock and Burt, 1993; Vought et al., 1995; Pinay et al., 1994; Lowrance et al., 1984; Fail et al., 1986). Denitrification refers to the reduction of nitrate to gaseous dinitrogen. Burford and Bremner, (1975) suggested two conditions that are necessary for nitrate removal via denitrification in a soil: the first is the presence of sufficient soil organic carbon content that would serve as an energy source for bacterial action, causing denitrification; and the second is the presence of anaerobic conditions in the soil.

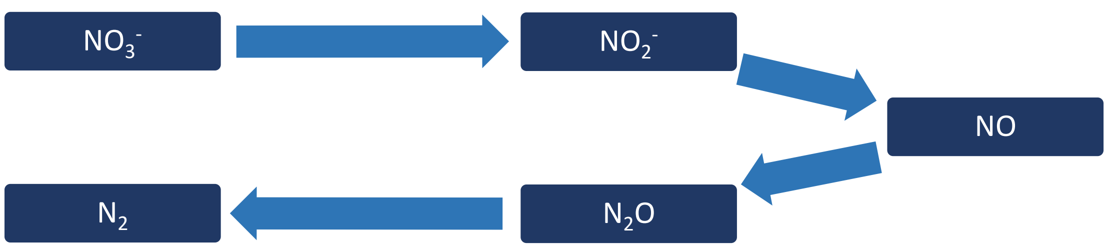


Figure 2.5 Conversion of $NO3^-$ to N_2 during the denitrification phase (Adapted from Hosftra et al., 2005)

Soils with at least 2% organic carbon can easily support denitrification and a threshold of 1% at a depth of 2.5 ft. was suggested as the minimum amount required for a setting up saturated buffer site (Utt et al., 2015). The anaerobic conditions needed for denitrification can be accomplished by raising the water table in the buffer to submerge the high carbon soil layer and restricting oxygen diffusion.

A site with either a historically shallow water table encompassing the high soil carbon layer, or the presence of a restricting layer in the buffer soil that would raise the water table by re-directing tile drainage into the buffer would eventually lead to anaerobic conditions in the buffer zone. A sandy or gravel dominated soil with absence of a restricting layer or a historically deep water table would not be conducive for denitrification, limiting the overall nitrate removal performance of the buffer (Bremner and Shaw, 1958; Burford and Bremner, 1975; McGarity, 1961).

2.6.2. Saturated Buffers: Factors affecting performance of the practice

Effectiveness of a saturated buffer system is dependent on the efficiency of the N attenuation processes taking place inside the buffer zone. Factors such input flow volumes, soil properties, and topography have been tested in different studies that can affect buffer performance.

A lower inflow volume has been studied to increase the residence time in the buffer zone, eventually increasing the efficiency of the system Lowrance et al., (2000). Similarly, concentrated flow in buffers was responsible for reduced buffer efficiency but this did not affect the sites where natural denitrification was the dominant source of nitrate removal by the buffer (Dosskey et al., 2002).

Furthermore, denitrification potential for a site has been studied to be dependent on soil properties such as organic matter, pH, temperature, and texture (Bremner and Shaw, 1958; Burford and Bremner, 1975). A lower organic matter limits water infiltration affecting water level inside a buffer zone and, it also depicts low soil organic C (Van Bemmelen, 1890). Soil carbon acts as an electron donor for microorganisms during the denitrification reaction, whereas a lower pH results in a decrease in denitrification potential.

2.6.3. Saturated Buffers: Previous work

Numerous studies have used different methods to evaluate buffer performance in reducing nutrients. Groffman et al., (1992) quantified denitrification by performing a microbial study for research sites in Rhode Island, US. A measure of denitrification enzyme activity and microbial biomass C was used to study the mode of nutrient removal from the buffer zone.

Another research conducted by Simmons et al., (1992) for the same sites used groundwater sampling of NO_3 from the buffer zone to quantify the nitrate reduction for the site. Both the studies found elevated reduction rates for the wetland area on the site.

Jaynes and Isenhart (2014) studied the impact of riparian buffers on nitrate removal when connected through tile drainage in Iowa. Here, 55% of the total flow was diverted to a 20 m wide buffer strip. It was observed that the buffer resulted in a 30-40 cm increase in water table depth and was successful in removing 228 kg of nitrate from the diverted tile water, amounting to a 100% nitrate removal rate.

A comprehensive study involving evaluation of saturated buffer sites across the U.S. states of Minnesota, Iowa, Illinois, and Indiana collected data from 2013 to 2015. Buffer performance was determined based on the pounds of nitrogen removed from incoming water. It was observed that the practice removed substantial nitrate in 17 of the 27 field years. While concentrated flow in the buffer zone can cause reduced nitrate removal capacity (Dosskey et al., 2002), the sites that performed poorly were identified to have unfavorable soil conditions for N attenuation. Some locations had coarser material above the carbon layer and some had an insufficient amount of organic carbon in the buffer zone. Lack of data from some sites also limited the evaluation of this practice. Overall, the implementation of saturated buffers was considered to be successful with nitrate reduction rates ranging from 18% to 85% when considering the sites that diverted at least 50% of the tile flow to the buffer. A continuation of the project by the Farm Service Agency examined the use of buffers within a farm operation along with the economics associated with nitrate removal. On average, the cost of nitrate removal was $2.4 per pound of nitrate removed. The producers did not record any effect on crop yields while the practice was in use. It was suggested that the practice could also be used like DWM, running on a management schedule to further improve the efficiency of N load reductions (Utt et al., 2015).

Saturated buffers have been tested to be successful in reducing nitrate transport from tiled croplands to surface water bodies but, more research needs to be done on the nutrient transport in and from the buffer zone and the fate of contaminants like dissolved reactive phosphorus in the buffer zone.

Chapter 3. DRAINAGE WATER MANAGEMENT: APPLICATION, EVALUATION, AND SIMULATION ON A FIELD SCALE SETUP IN SOUTH DAKOTA.

ABSTRACT

Drainage water management is an edge of field management practice used to reduce water outflow from tiled croplands by manipulating the water table depth throughout the growing season. The reduction in flow reduces nitrate loading from tile drained croplands, which is a major cause of water quality impairment. For this study, a field site was installed near Alexandria, SD during the fall of 2015. A paired field approach was used to compare conventional drainage and drainage water management. Tile drainage water was sampled weekly during flow conditions and analyzed for nitrate concentrations. Meanwhile, daily tile flow records were divided into free drainage and managed periods. The duration when the weir boards in the control structure were removed for both the halves comprised the free drainage period. During the managed period, boards were put into the control structure for the eastern half of the field to raise the water table. The differences in cumulative flows per acre drained during the managed periods were used to compare the two halves. It was observed that drainage water management resulted in an 8mm decrease in total outflow during 2016; however, no flow was observed for the managed period during 2017, so no data were available for comparison. Overall, drainage water management resulted in a load reduction of 26% during the 2016 managed period and cost $28 per pound of nitrate removed per acre drained. To study the impact of DWM on field hydrology, two Soil and Water Assessment Tool (SWAT+) projects were developed, one for each half of the field, and run from 2000 to 2017. The model was calibrated for 2016 (NSE = 0.81 and 0.54, PBIAS

=18% and 61%, RSR =0.43 and 0.68) and validated for 2017 (NSE= 0.70 and 0.84, PBIAS = -24% and -11%, RSR =0.55 and 0.40) using the daily tile flow (mm) measured at the site. The flow reductions due to DWM throughout the simulation period ranged from 5% to 92% during the managed period. In addition, there was an increase in runoff and ET due to DWM during most of the study period. Overall, drainage water management was successful in reducing flow volumes from tiled croplands, but the performance was dependent upon seasonal variability in precipitation, soil properties such as bulk density and available water capacity, and tile drainage parameters such as tile lag time, which were studied using hydrological modeling.

3.1 Introduction

Tile drainage is used to drain excess water from sub surface soil to optimize crop growth, however, it also increases nutrient transport from croplands to surface waterbodies (Blann et al., 2009). Accumulation of nutrients, such as nitrates, in rivers and lakes leads to algal blooms which pose a threat to aquatic flora and fauna (Diaz and Rosenberg, 2008). drainage water management (DWM) is an edge of field management practice initially developed and tested extensively in North Carolina to reduce nitrate loads delivered from croplands to receiving waterbodies (Gilliam and Skaggs, 1987).

DWM uses a control structure to manipulate water table depth during the growing season and prohibit water outflow from the field. Stackable boards are put in the control structure during the growing season, forming a barrier to prevent water from flow through the outlet, thus raising the water table. The water and nutrients retained in the field also supplements the crop’s water and nutrient requirements, thereby increasing the potential for nitrate load reduction (Evans et al., 1996; Frankenberger et al., 2004; Gilliam and Skaggs, 1987; Strock et al., 2010).

Skaggs et al. (2012) compiled numerous studies examining DWM performance and observed that the nitrate load reduction rates ranged from 18% to 85%. Tile characteristics, climatic conditions, topographic conditions, and soil properties are the dominant factors that cause this high variation in DWM performance (Ross et al., 2016). In South Dakota, plot-scale implementation of DWM has been successful in reducing the total water outflow by 58% and associated nitrate loads by 21% to 89% (Sahani, 2017). Though DWM results in more water and nutrient availability throughout the season, changes in yield due to the practice have been inconsistent (Skaggs et al., 2012b). The limited number of studies makes it difficult to reach to a general agreement.

Use of modeling is an economical way of evaluating BMP performance over longer periods. It can be used at various spatial and temporal scales and is an important tool in the decision-making process for agricultural water management. SWAT has been used in numerous studies that have examined the impact of various climatic and agronomic practices on water quality and BMPs, (e.g. Arnold et al., 1998; Sahu and Gu, 2009; Du et al., 2005; Green et al., 2006). Sahu and Gu (2009) used SWAT to quantify the effect of riparian buffers in reducing nutrient loads and concluded that the practice could potentially result in 55% to 90% reduction in stream nitrate concentrations.

The objectives of this study are to demonstrate and document the effectiveness of DWM in South Dakota and develop a module for simulating DWM for SWAT to be used to study the impact of DWM on field hydrology and crop yields.

3.2 Materials and Methods

3.1.1 <u>Site setup</u>

A field scale drainage water management site was installed near Alexandria, SD (43˚40’22.28” N, 97˚48’17.05” W) during the fall of 2015. Alexandria lies in Hanson County where agriculture is the dominant land use with corn and soybeans being the major crops grown during the cropping season from April - October. For the site, the area under tile drainage was estimated to be 26 ha (65 ac). The field was split into two halves, a conventionally drained half situated on the western side and the other utilizing DWM situated on the eastern side (Figure 3.1). Tile depth and spacing for both halves were similar at 0.9 m (3 ft.) and 18 m (60 ft.), respectively, but the western half, had a larger main tile diameter (38 cm) and drained less area (12 ha) than the eastern half (dia. = 25 cm, area = 14 ha) (Table 3.1).

Table 3.1 Tile design details for the DWM site at Alexandria, SD. DWM and conventional drainage had similar Tile Depth and Tile Spacing but different Tile Size and Drained area.

	DWM	Conventional Drainage
Drained Area (ha)	14	12
Tile Size (cm)	12.7	12.7
Tile Depth (cm)	91	91
Tile Spacing (m)	18	18

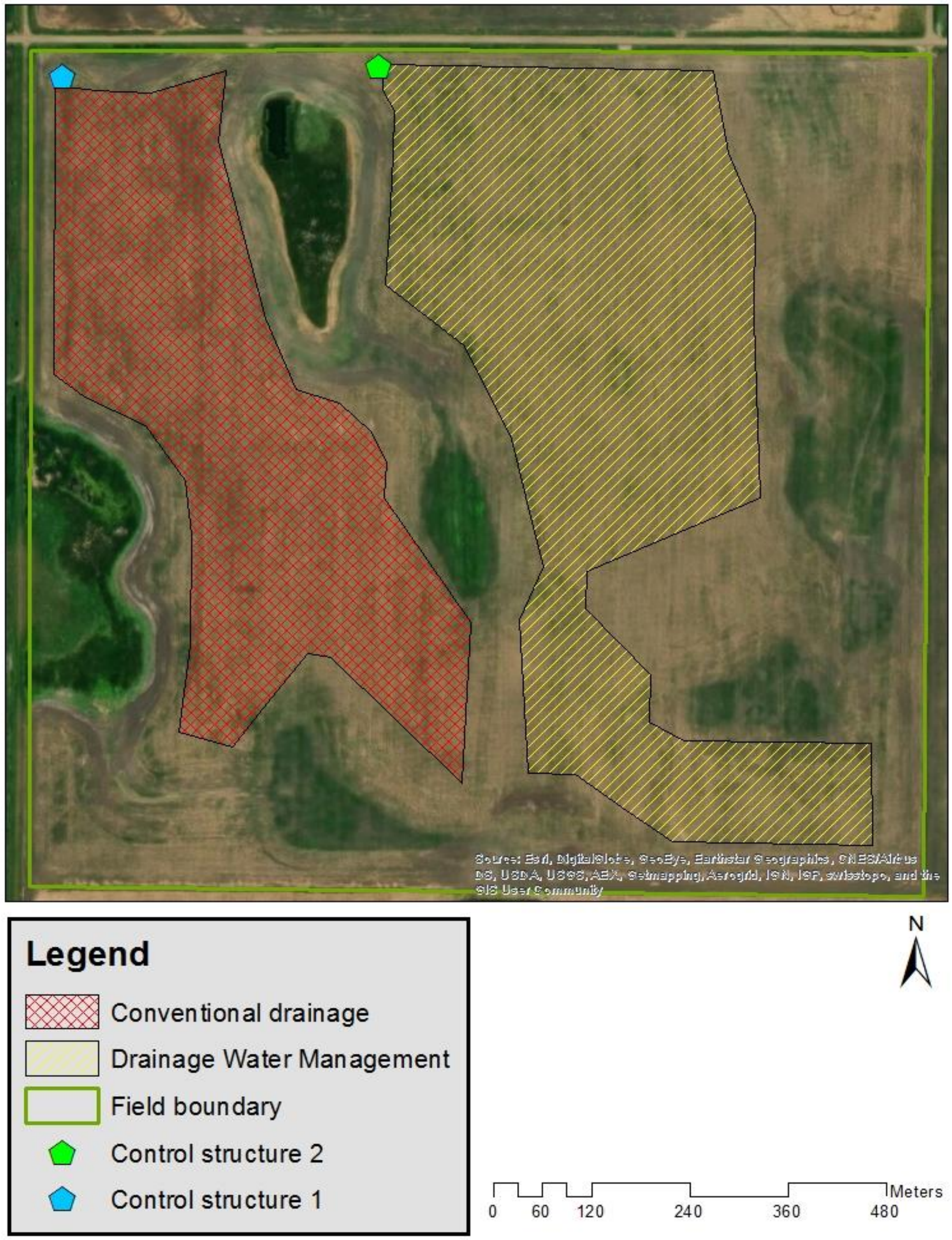


Figure 3.1 The DWM site at Alexandria, SD was split into two halves, a conventional drainage half (red crossed pattern) and another utilizing DWM (yellow striped pattern) Control structure 2 drained from the DWM half whereas, Control structure 1 drained from the conventionally drained half.

3.1.1 Instrumentation

Various sensors were deployed at the research site to record data throughout the study period from June 2016 to December 2017. The parameters measured by the sensors included water level in the control structure, air temperature, water temperature, electrical conductivity of the water, relative humidity, precipitation, and barometric pressure (Table 3.2). Each half had a control structure on the northern edge with a CTD-10 sensor (Decagon devices) which recorded data every 15 minutes. Other climate variables such as temperature and precipitation were recorded using Decagon VP-4 and Decagon ECRN-100, respectively, which were located at the control structure situated near the western half of the study site. All the sensors were connected to a Decagon EM-50G data logger, which logged data to the manufacturer's webserver where it was retrieved for use.

Table 3.2 Instrumentation deployed at the research site along with the number and parameters recorded during the study period. All the sensors like CTD-10, ECRN-100 and VP-4 were connected to the data logger EM 50G and the data was recorded every 15 minutes.

Instrument	Parameters Measured	Number of sensors
Decagon EM 50 G	Logging data	2
Decagon CTD -10	Water depth, Temperature, Electrical conductivity	2
Decagon ECRN-100	Precipitation	1
Decagon VP-4	Air Temperature, RH, Barometric Pressure	1

3.1.2 Water management

The water table was manipulated using a management schedule developed for the site. The boards were put in on June, 1 2016 and taken out on September 20, 2016. For

2017, boards were put in on June 15, 2017 and taken out on October 15, 2017. The boards were installed to a 30 cm depth for the growing season to implement DWM for the eastern half of the field. During winters, the boards were taken out to allow free drainage. It was done to prevent freezing of water stored in the soil profile and additional tile flow lag during the thawing period.

3.1.3 Field Data Collection

3.1.1.1. Water sample collection

Water samples were collected from the control structures using a grab sampler. 250 ml pre-labelled Nalgene bottles were used to store each sample upon collection. Samples were collected for two years, 2016 and 2017, during flow conditions inside the control structure. Upon collection, samples were stored in a cooler and transported to the lab at the Agricultural Engineering Department at SDSU where they were stored under freezing conditions until analyzed.

3.1.1.2 Flow rate

Water level in the control structure was measured using a Decagon CTD-10 sensor, which measures electrical conductivity, temperature, and the height of water above the sensor. Each control structure was fitted with a V-notch weir board used as the topmost board in the control structure. The dimensions of the V-notch were same for both the control structures. A flow equation (Equation 3.1) was calibrated for the same at the Agricultural and Biosystems Engineering Department at SDSU (Partheeban et al., 2014).

$$Q = 1.7406 * (H)^{1.9531} \quad \text{(Equation 3.1)}$$

Where Q is the discharge through v-notch (L min^{-1}) and H is the height of water above the bottom of v-notch (cm).

The flow above V-notch was considered as rectangular flow and calculated using the flow equations calibrated for commercially available Agri-Drain control structures by Chun and Cooke (2008).

3.1.1.3 Soil Moisture

A DeltaT ML3 probe was used for recording the volumetric soil moisture content. The measurements were made weekly and in conjugation with leaf area index (LAI) readings throughout the cropping season, from early leaf stage to senescence. The sensor uses the soil dielectric permittivity and converts it to the volumetric water content using the Topp equation (Equation 3.2) (Topp et al., 1980).

(Equation 3.2)

$$VWC = (4.3 * 10^{-6} * \varepsilon_a^3) - (5.5 * 10^{-4} * \varepsilon_a^2) + (2.92 * 10^{-2} * \varepsilon_a) - 5.3 * 10^{-2}$$

Where VWC is the volumetric water content (cm^3 cm^{-3}) and ε_a is the dielectric permittivity (dS m^{-1}).

3.1.1.4 Leaf Area Index

Leaf area index is the ratio of the aboveground leaf area to the below canopy soil area. LAI was recorded for 2016 and 2017 using the AccuPAR LP 80 Ceptometer (Decagon Devices, Inc., Pullman, WA) on a weekly basis from random locations within the field. 12 locations per half were chosen randomly to record LAI during the growing season from early leaf stage to the senescence.

3.1.1.5 Soil Analysis

Soil samples were collected (Figure 3.2) at three depths, 0 – 30 cm (0 – 12 in), 30 – 60 cm (12 – 24 in) and 60 – 90 cm (24 – 36 in) below the surface using soil augers. The samples were then put into pre-labeled plastic bags and transported to the SDSU where they were stored under freezing conditions. Samples were analyzed at the SDSU soil lab for nitrate-nitrogen (NO3-N), phosphorus (Olsen P), potassium (K), electrical conductivity (EC), organic matter (OM), pH, and particle size distribution.

In addition, soil bulk density samples were collected using AMS soil sampling kits during the summer of 2017. Samples were collected in a ring of volume $90cm^3$, which was pushed into the ground using a sliding hammer. The undisturbed samples were then transported to SDSU, where they were oven dried at 105˚C for 24 hours. Finally, the dry weight was recorded using a precise weighing balance and bulk density computed using the following formula:

$$Bulk\ Density\ = \frac{Oven\ dry\ Wt\ of\ soil\ (gms)}{Volume\ of\ soil\ sample\ (cc)} \qquad \text{(Equation 3.3)}$$

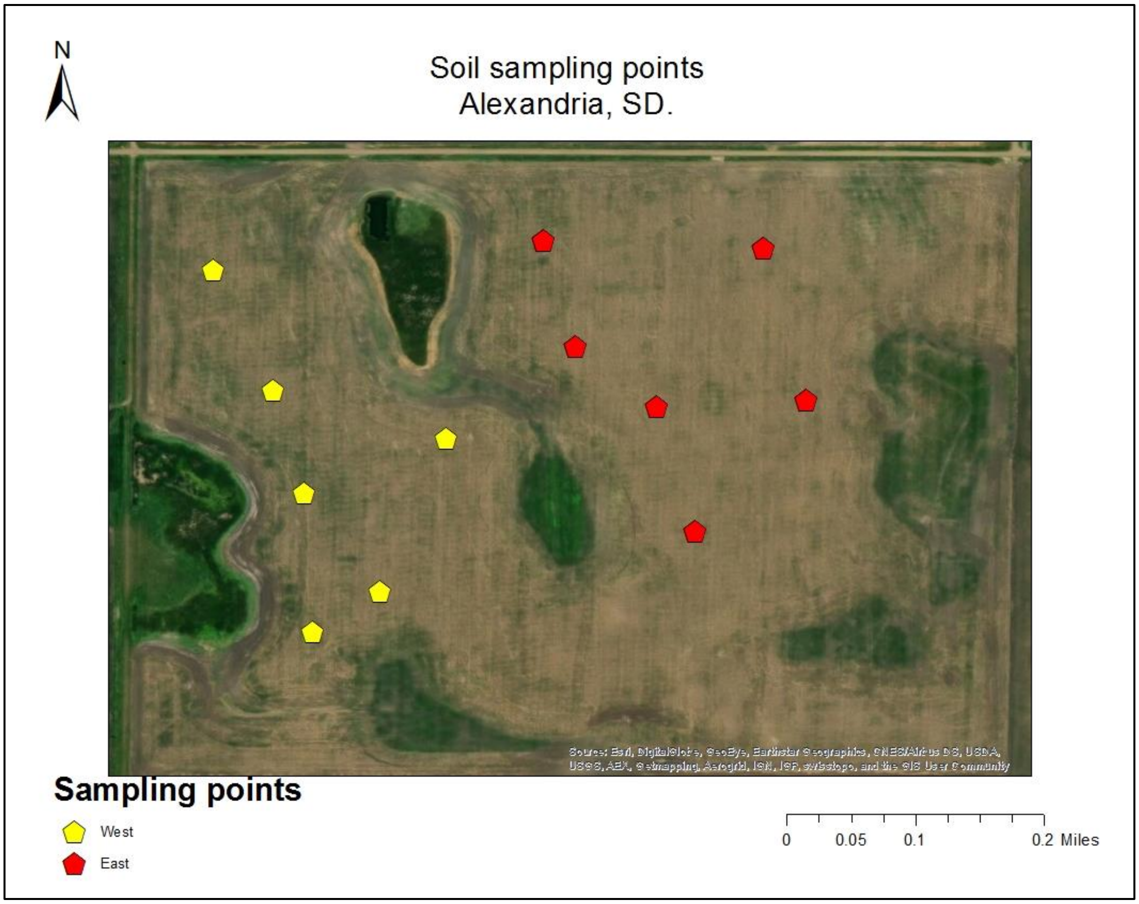


Figure 3.2 Soil sampling locations for Alexandria, SD. Three points per soil type were chosen to collect samples. There were two major soil types existing in each half. The yellow points denote sampling locations for the conventionally drained half whereas, the red points denote sampling locations for the DWM half.

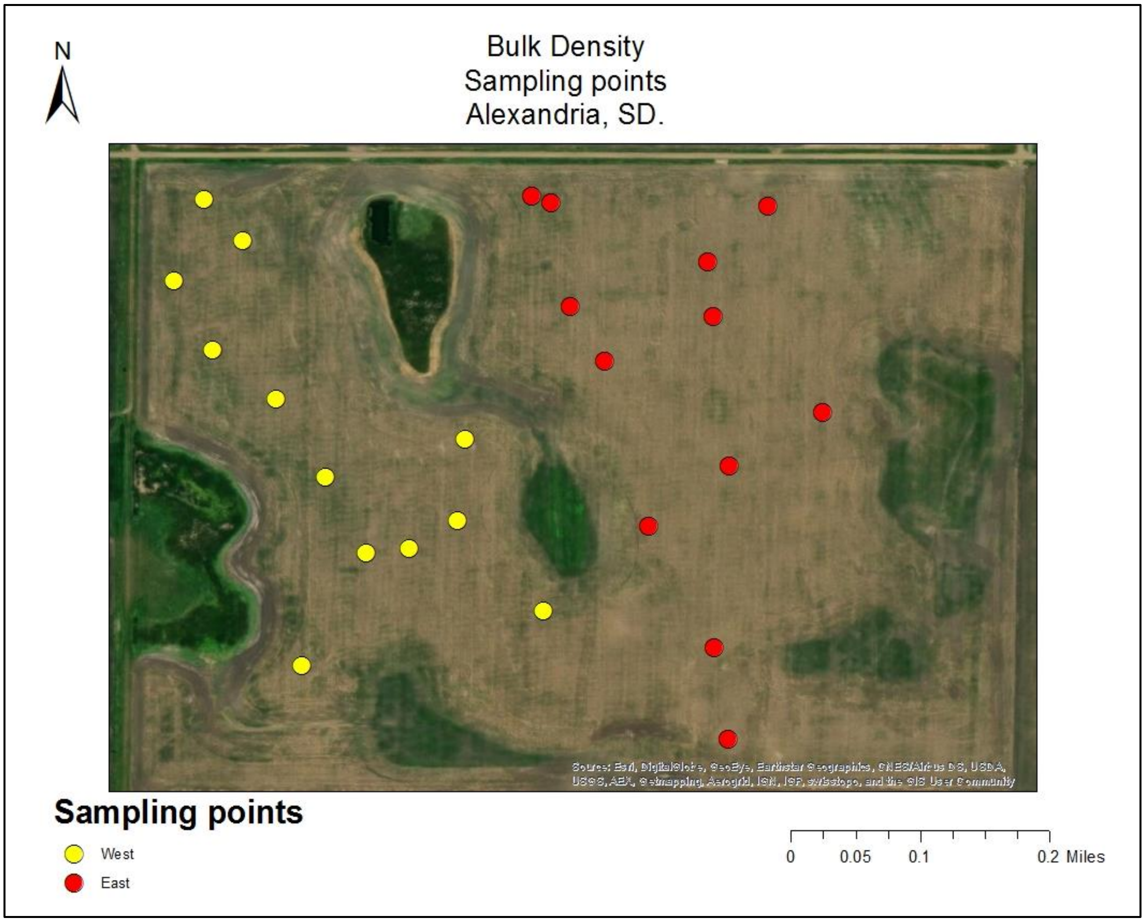


Figure 3.3 Bulk density sampling locations for Alexandria, SD. Three points per soil type were chosen to collect samples. There were two major soil types existing in each half. The yellow points denote sampling locations for the conventionally drained half whereas, the red points denote sampling locations for the DWM half. The bulk density sampling was performed in conjugation with the infiltration testing.

3.1.1.6 Infiltration rate

A single ring infiltration rate method was used to evaluate infiltration at field capacity. Three locations per soil type were chosen to perform the test each month during the cropping season (Figure 3.4).The infiltrometer ring was placed a minimum of 7.6 cm (3 in.) below the surface and supplied with 75 ml of water. The time taken for water to infiltrate was recorded and another 75 ml was added to the ring. The process was repeated until at least two stable readings were obtained. 75 ml of water corresponded to a one centimeter rise in the ring, so the time recorded was the infiltration rate for one centimeter of water.

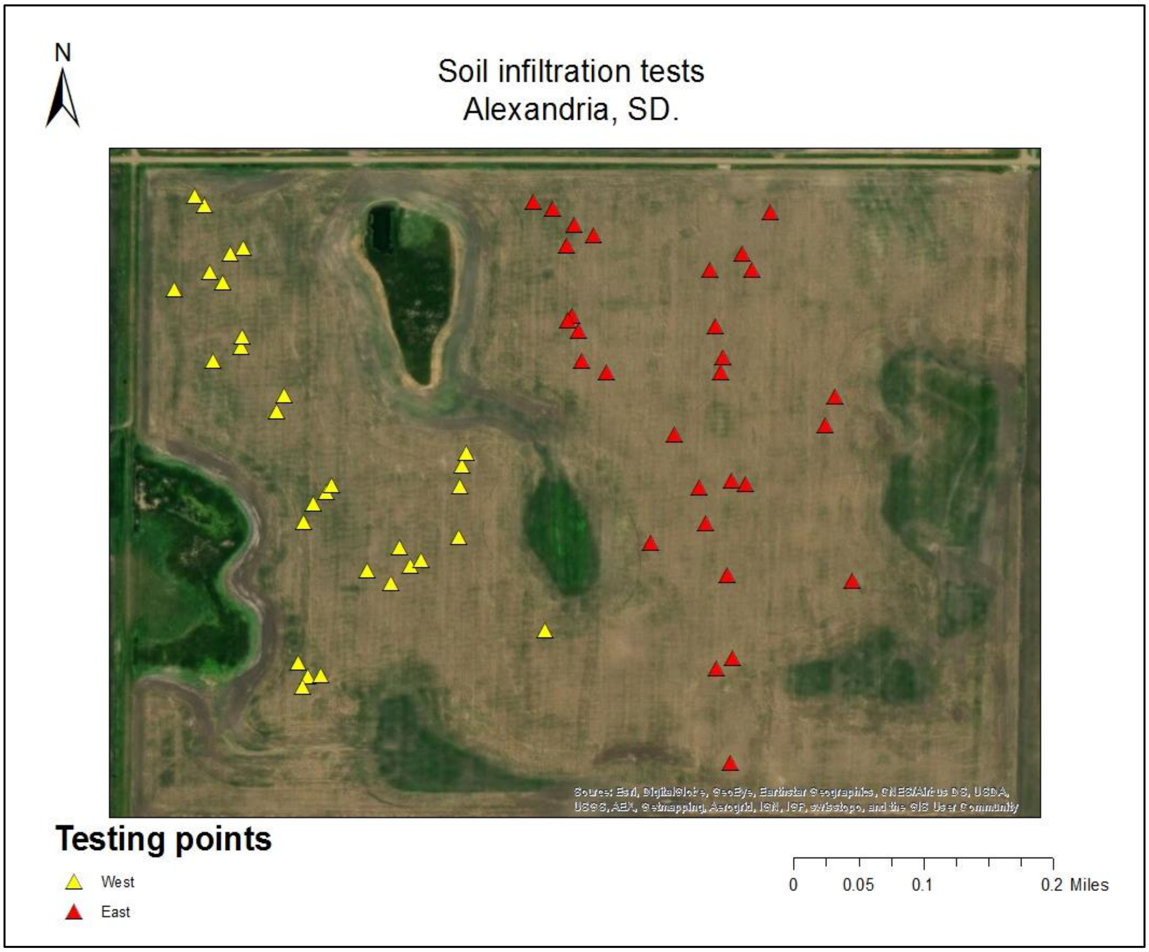


Figure 3.4 Soil infiltration testing locations at Alexandria, SD. Infiltration testing was done monthly and as a triplicate for each soil type. The red points denote the testing locations for the DWM half whereas, the yellow points denote the testing locations for the conventionally drained half.

3.1.1.7 Water Quality Analysis

Water samples were filtered using 30 ml syringes with 0.45-micron nylon membrane filters. Filtered water samples were stored in pre labelled 60 ml Nalgene bottles under freezing conditions if not analyzed immediately. Samples were analyzed for nitrate concentration using a Seal AQ2 discrete analyzer (Seal Analytical Inc., Milwaukee, WI). EPA 353.2 method was followed to calculate nitrate plus nitrite concentration in water samples. Nitrite was analyzed separately using EPA 354.1 which omits the use of the cadmium coil during the analysis. Nitrate concentration was obtained by subtracting the nitrite concentration from the nitrate-nitrite concentration (US-EPA, 1993).

3.3 Model setup

To simulate the effect of DWM on field hydrology, a SWAT+ project was setup for the research site at Alexandria, SD. Temperature and precipitation data from the closest weather station (USC00390128) were used as model inputs. The project had a six year warm up period (2000 – 2005), 2016 as the calibration period, and 2017 as the validation period.

A total of 12 parameters were included in the calibration process (Table 3.3). SWAT parameters were selected using literature pertaining to model studies around Midwestern watersheds (Rajib et al., 2016; Schilling et al., 2010) and suggestions from the development team (Neitsch et al., 2011).

Table 3.3 SWAT+ project parameterization and best simulation values for each half: 12 parameters were adjusted using the initial range. Three different types of adjustments were included to adjust the output. These were 1: multiplication by an adjustment factor (3+ given value within the range), 2: addition and 3: replacement.

Parameter	Definition	Initial range	Adjustment type	Best simulation values	
				Western half	Eastern half
CN	Curve number for moisture condition II	-10 – 10	2	-6	-6
SOL_AWC	Available soil water capacity(mm H_2O (mm soil)$^{-1}$)	-0.04 – 0.04	2	-0.02	0
BD (depth 1)*	Soil bulk density (gm cc^{-1})	1 – 1.7	3	1.19	1.25
BD (depth 2)*				1.33	1.35
BD (depth 3)*				1.6	1.62
BD (depth 4)*				1.6	1.62
SOL_K	Soil saturated hydraulic conductivity (mm hr^{-1})	-30 – 30	1	-20%	-20%
T_FC	Time to drain to field capacity (hrs.)	24 – 60	3	60	48
T_LAG	Drain tile lag time (hrs.)	0 – 200	3	90	170
EPCO	Plant uptake compensation factor	0 – 1	3	0.36	0.36
ESCO	Soil evaporation compensation factor	0 – 1	3	0.88	0.88
TIMP	Snow pack temp lag factor	0 – 1	3	0.07	0.07
MELTTMP	Snow melt base temperature (˚C)	0 – 4	3	2.0	2.0
MELTMX	Melt factor for snow on December 21 (mm H_2O (˚C-day)$^{-1}$)	1.4 – 6.9	3	6.9	6.9
MELTMN	Melt factor for snow on June 21 (mm H_2O (˚C-day)$^{-1}$)	1.4 – 6.9	3	1.4	1.4

*Soil bulk density was adjusted at four different depths beneath the soil surface, depth 1 was the topmost layer, followed by depth 2, depth 3 and then depth 4.

Daily tile flow values measured from the site were compared with simulated values. Two different models were developed to simulate the two halves of the field. The eastern half drained 14ha and used DWM was compared with the western half which drained 12ha and was under conventional drainage. Additional simulations were conducted to quantify the effect of different water table management schedules on field hydrology. Calibration was performed manually and a set of four parameters; soil bulk density, available water capacity, tile lag time, and time to drain to field capacity were chosen to have different values for each half. All other parameters such as ESCP, EPCO, CN, TIMP, MELTMX, MELTMN, and MELTTMP were adjusted, but kept the same for both models as both represented the same field.

3.4 Model Evaluation

Daily tile flow measured at the field was compared with the daily tile flow output obtained from SWAT+ simulations. Model performance for this study was evaluated using three different statistical indices; Nash-Sutcliffe coefficient of efficiency (NSE), percent bias (PBIAS), and the ratio of root mean square error to the standard deviation of the observation (RSR).

Mathematically,

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right)^2}{\sum_{i=1}^{n}\left(Y_i^{obs} - Y^{mean}\right)^2}\right] \qquad \text{(Equation 3.4)}$$

$$PBIAS = \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs}-Y_i^{sim}\right)*(100)}{\sum_{i=1}^{n}\left(Y_i^{obs}\right)}\right] \quad \text{(Equation 3.5)}$$

$$RSR = \frac{RMSE}{STDEV_{obs}} = \left[\frac{\sqrt{\sum_{i=1}^{n}\left(Y_i^{obs}-Y_i^{sim}\right)^2}}{\sqrt{\sum_{i=1}^{n}\left(Y_i^{obs}-Y^{mean}\right)^2}}\right] \quad \text{(Equation 3.6)}$$

Where, Y_i^{obs} = value of the i^th observation, Y_i^{sim}= value of the i^th simulated value and Y^{mean}= mean of the observed values.

NSE ranges from -∞ to 1, with one being the optimal value. It is a widely used evaluation statistic and has been recommended by Servat and Dezetter (1991) for evaluation of a hydrograph. The PBIAS indicates the deviation between modeled and measured values. It represents the percentage of bias in the simulation, with zero percent being the optimal value. A positive value depicts an over prediction by the model and vice versa (Gupta et al., 1999). RSR is the ratio of RMSE to the standard deviation of the observation (Equation 3.6). The standard deviation of the observed values serves as the normalization factor for RMSE which denotes the variation in the residuals. The optimal value for the statistic is zero, an RSR equal to one depicts equal variation for the residual and the observation (Singh et al., 2004). A value greater than one represents greater variation for the residual than the observation.

Moriasi et al. (2007) suggested various statistical indices for SWAT model output evaluation along with the satisfactory ranges for an accurate simulation of streamflow and other parameters (Table 3.4). The evaluation was made for a monthly time step, but the suggested ranges for NSE, PBIAS and RSR from the study are valid for daily output comparison and were used for model evaluation for this project.

Table 3.4 Satisfactory ranges for objective function for SWAT model calibration (Moriasi et al., 2007). These values are based on a monthly simulation.

Evaluation statistic	Satisfactory range
NSE	> 0.50
PBIAS	-25% to +25%
RSR	≤ 0.7

3.5 Results and Discussion

3.5.1 Field Study

3.5.1.1 Climate

Precipitation and temperature calculations were done only for the cropping season from May to October. For 2016, mean annual precipitation was 26 mm, with August being the wettest month and October the driest. For 2017, the mean precipitation was 43 mm but most of it was during September. Low precipitation during March-July period resulted in lower tile flows, the period has been studied to be a critical component for tile flow volumes in croplands (Randall and Mulla, 2001).

The temperature was the highest for June 2016 with a mean temperature of 24.1 ˚C. For 2017, July had the highest mean temperature at 25.0 ˚C. Temperature ranges were comparable with the 30-year averages. Overall, January 2017 was observed to be coldest month and the average monthly temperature was similar to the 30-year average (1981-2010) for a weather stations situated nearby the research site.

Table 3.5 Total monthly precipitation (mm) at the research site vs 30 yr. monthly average (1981-2010) from the nearby weather station (USC00390128).

Months	Alexandria		
	Total 2016	Total 2017	(1981-2010) 30 year average.
January	-	1.4	12.8
February	-	6.2	14.5
March	-	0.2	36.5
April	-	45.0	73.2
May	-	64.0	81.9
June	29.0	1.8	102.7
July	42.4	0.8	79.0
August	55.4	17.4	69.1
September	1.6	104.0	64.9
October	1.4	68.6	51.1
November	2.0	15.0	29.6
December	2.0	0.0	12.8

Table 3.6 Average monthly temperatures (˚C) at the research site vs 30 yr. monthly averages (1981-2010).

Months	Alexandria		
	Average 2016	Average 2017	(1981-2010)
January	-	-6.6	-7.0
February	-	-0.4	-4.3
March	-	2.0	2.0
April	-	9.1	9.4
May	-	14.8	15.8
June	24.1	22.3	21.1
July	24.1	25.0	23.9
August	22.8	19.7	22.8
September	18.0	17.5	17.8
October	11.1	9.5	10.4
November	5.0	1.4	1.7
December	-6.9	-6.2	-5.8

3.5.1.2 Soil Analysis

Soil Bulk Density

Soil bulk density ranged from 1.16 gm cm^{-3} to 1.47 gm cm^{-3} for the DWM side of the field and 1.14 gm cm^{-3} to 1.55 gm cm^{-3} for the conventionally drained side of the field, however, the mean soil bulk density was higher for the DWM half as compared to the conventionally drained half. Compaction due to heavy machinery and slight variation in soil texture is a possible reason for this difference (Horn et al., 1995; Richard et al., 1999; Wolkowski, 1990).

Soil texture and nutrients

In general, the DWM side had lower nutrient concentrations for all the measured parameters with a slight variation in magnitude for different sampling depths. Soil OM, NO3-N, Olsen P, K, and pH for the entire site were observed to decrease with an increase in depth. The gradual decrease in OM with depth may be related to the mineralization resulting in the release of oxide solids through decomposition of organic matter. Higher OM at surface can be related to the process of enrichment due to mixing of crop residue at the surface soil. Higher soil OM also relates to the differences in bulk density and observed during the field study. OM ranged from 1.2% to 3.3% for the site. Lower values for OM limit the denitrification potential for the entire site, similar to the discussion in Burford and Bremner, (1975).

Furthermore, lower EC values were observed at the top layer indicating reduced soil nutrient movement. This was conclusive from the higher nutrient values for NO3-N, Olsen P, and K at upper depths. The pH was neutral at the surface but increased with depth in all the four sites. This indicates the leaching of basic cations from the surface to the deeper layers due to the occurrence of rainfall events, which is supported by the increase in salt concentration with depth. The texture of the studied soil profiles was largely clay loam.

Table 3.7 Soil analysis for DWM and conventionally drained halves. Mean concentrations for soil properties including organic matter (OM) (%), soil nitrate (NO3-N) (ppm), soil phosphorus (Olsen P) (ppm), soil K (ppm), soil pH, Electrical conductivity and texture properties (percent sand, silt, and clay) were analyzed at three sampling depths, 0-30cm, 30-60cm, 60-90cm.

Sampling depth	(0-30) cm		(30-60) cm		(60-90) cm	
	Conventional	DWM	Conventional	DWM	Conventional	DWM
OM	3.3	3.0	2.2	2.2	1.2	1.2
NO3-N	9.5	6.9	7.6	4.1	7.6	5.3
Olsen P	8.7	8.3	5.2	4.2	2.8	2.3
K	216.8	185.2	127.8	118.3	100.2	88.5
pH	7.2	7.0	7.8	7.3	8.1	7.9
EC	0.62	0.63	0.60	0.57	1.38	1.00
Sand	36.8	37.2	37.5	40.8	39.2	42.2
Silt	33.3	29.8	30.0	27.8	29.7	28.3
Clay	29.7	33.0	32.3	31.3	31.0	29.5

3.5.1.3 Soil Infiltration rate

The monthly infiltration rate for both the halves was variable and no trends were observed for the infiltration rates throughout the cropping season. The infiltration rates ranged from 6.2 mm hr^{-1} to 150.0 mm hr^{-1} for the entire site. Infiltration rates were lower for the DWM half during the managed period as compared to the free drainage period. This may be related to the greater amount of water stored by DWM during the managed period, limiting the air-filled pore volume during that time. Throughout the study period, infiltration rates were fairly constant for the western half, but dynamic for the eastern half likely due to changing soil water storage for the managed half.

Table 3.8 Descriptive statistics for infiltration rates observed (mm hr^{-1}) between the two halves at Alexandria, SD.

	Conventional	DWM
Minimum (mm hr^{-1})	7.9	6.2
Mean (mm hr^{-1})	37.2	34.4
Maximum (mm hr^{-1})	130.0	150.0

3.5.1.4 Soil moisture

Weekly soil moisture readings ranged from 5.7% to 39.7% by volume for corn raised during 2016 and 4.7% to 36.3% by volume for soybeans planted during 2017. There were no trends observed as the season progressed as the moisture was easily influenced by rain events. The mean soil moisture for the DWM was slightly greater for 2017 indicating greater storage during the managed period, similar to the results found for monthly infiltration rates during the same period. For 2016 however, it was similar for both the halves which might be due to greater rainfall during June, July and August as compared to 2017, where most of the water was stored due to early summer rainfall.

Table 3.9 Descriptive statistics for the soil moisture content (%VWC) observed during 2016 and 2017 for both the halves.

	Minimum (% VWC)		Mean (% VWC)		Maximum (% VWC)	
	Conventional	DWM	Conventional	DWM	Conventional	DWM
2016	5.8	5.7	16.2	16.2	35.6	39.7
2017	4.7	6.3	18.7	19.6	36.3	34.2

3.5.1.5 LAI

As expected, LAI readings increased as the season progressed owing to crop growth but decreased after the senescence stage during both the years of study. For 2016, corn LAI values ranged from 0.5 during the beginning of the season to a max value of 4.1 around August and then decreased to 0.05 nearing the end of September, similar to as observed in Nguy-Robertson et al., (2012). However, soybean LAI reached a higher value of 5.1, rose to 6.0 around 10 August nearing the maturity stage and then decreased after 30 August to 4.5 around 14 September. LAI values were higher for soybeans because of the dense cover as compared to corn.

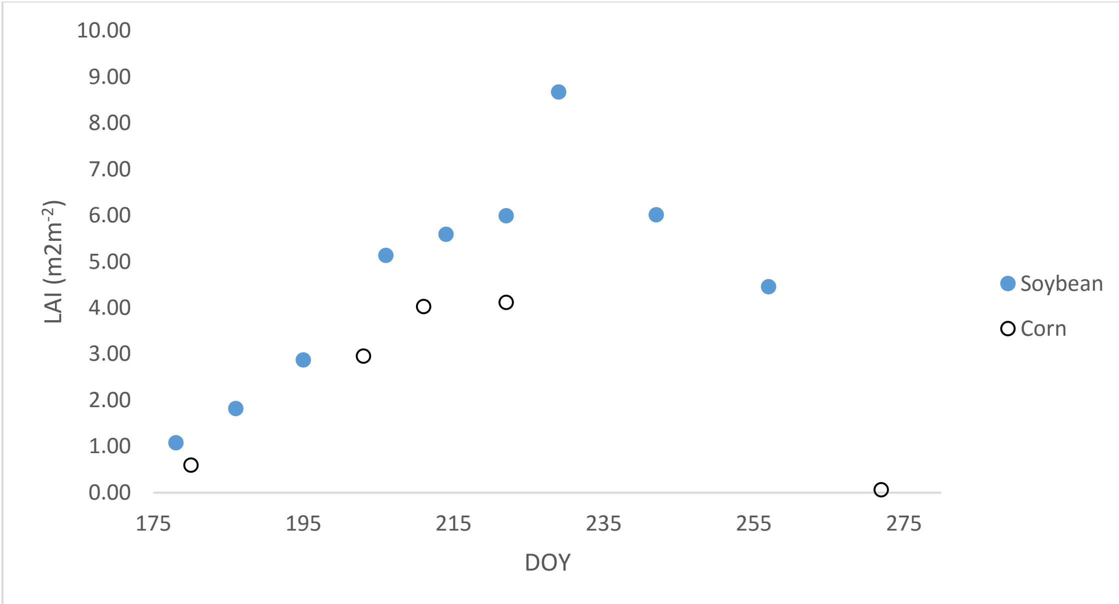


Figure 3.5 LAI values for corn and soybean throughout the cropping season.

3.5.1.6 Water depth at plot outlet

The daily water level recorded in the control structure for the DWM half was higher than the conventionally drained half during the managed period for both the years of the study. The mean water level in the control structure during 2016 was 233 mm and 81 mm for the DWM and conventionally drained halves, respectively (Figure 3.6). For 2017, the mean water level was 132 mm and 57 mm for the DWM and conventionally drained halves, respectively. A rise in the water table was observed for the DWM half, after the boards were put in the structure, indicating water storage in the soil profile during the managed period. The level gradually declined, supplementing the crop water requirements. Similar observations were made by Randall and Mulla (2001), where the majority of the flow occurred during early summer months and declined as the year progressed, probably due to increasing ET losses as the growing season progressed.

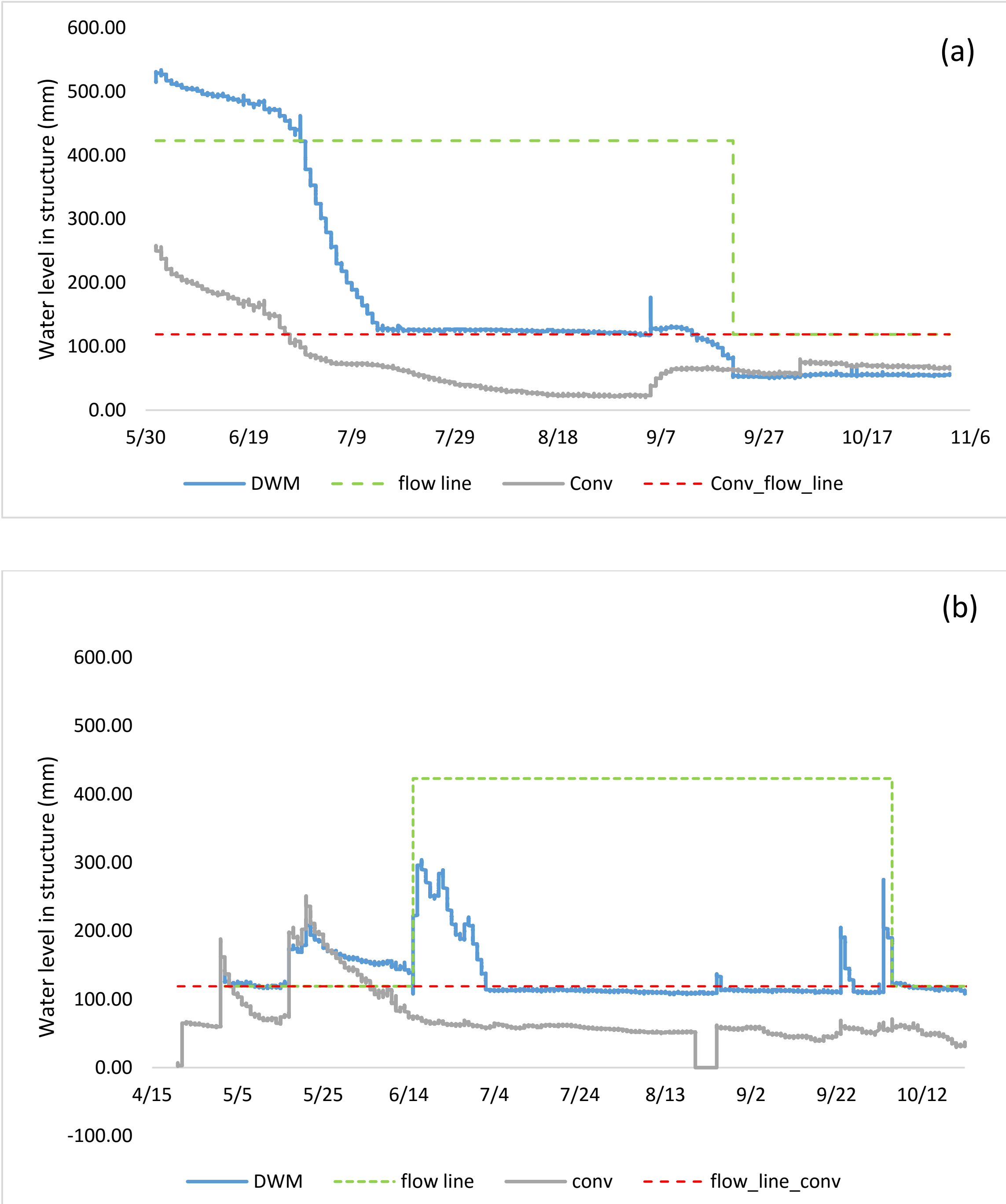


Figure 3.6 Water level in control structures for the DWM and conventionally drained half (conv) during 2016(a) and 2017(b). Dotted lines represent the height of the boards in the control structure while the solid lines show water level in the control structure.

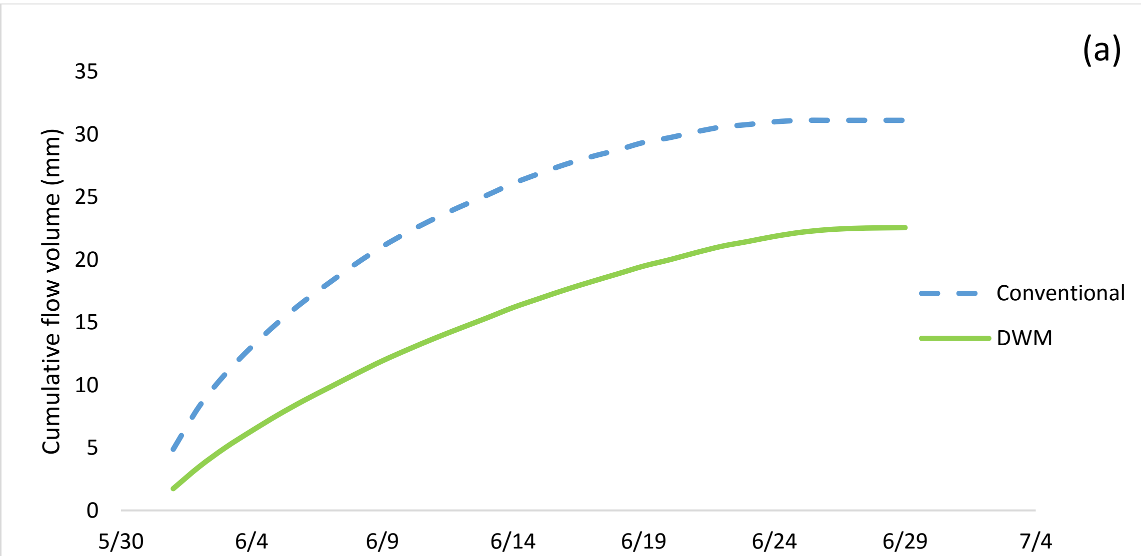
3.5.1.7 Tile drain outflow

CTD-10 sensors were installed and recorded flow during the growing season (June - October 2016, April – October 2017). Measured flow records were split into two periods, the free drainage period and the managed period. During the free drainage period, both the halves drained freely, but during the managed period, the eastern half utilized DWM and the western half drained freely. Flow occurred for both the halves during the managed period for 2016 and was 8mm less for DWM as compared to conventional drainage (Figure 3.7 (a)). For 2017, no flow occurred during the managed period and the annual drain flow for the DWM half was 6mm less than the conventional half (Figure 3.7 (b)).

The peak flow rate was also lower for the DWM half as compared to the conventional half. The peak discharge for the DWM half were 1.7 mm day^{-1} and 1.3 mm day^{-1} for 2016 and 2017, respectively, whereas, they were 3.4 mm day^{-1} and 3.2 mm day^{-1} for the conventionally drained half during 2016 and 2017, respectively. A possible reason to support this observation was lower tile lag time for the conventional drainage half. The impact of the tile lag time, water management strategies, and tile design on field hydrology was further studied during the modeling study.

Tile drainage volumes were observed to be minimal from July to October for both years, indicating high evapotranspiration needs (Randall and Mulla, 2001). Overall, tile drainage comprised of 5.2% and 3% of the total precipitation received during 2016 and 2017 respectively at the conventional half. For the DWM half, it was 3.7% and 1.8% for 2016 and 2017 respectively. The difference within each year was due to seasonal variability in precipitation patterns especially during the months of March, April and

May. The cumulative rainfall for the three months amounted to 280 mm for 2016 which was substantially higher than 164 mm for the same period during 2017. In addition, the difference between the two halves for each year can be due to the variability in soil properties and tile design parameters. Variability in soil is supported by the results from the soil analysis for OM, infiltration rate testing and soil bulk density tests. This was used for setting up each SWAT+ model.



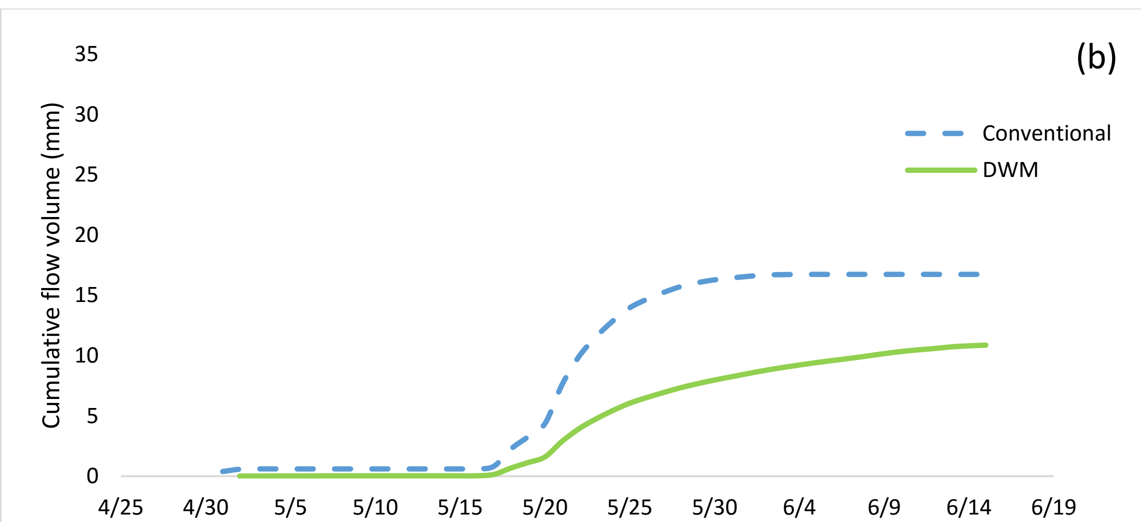


Figure 3.7 Comparison of cumulative flow volume (mm) between DWM (solid line) and conventional drainage (dashed line) for Alexandria during 2016(a) and 2017(b).

3.5.1.8 Water Quality

Nitrate concentration

Nitrate concentrations were lower for the DWM half than the conventionally drained half during the 2016 managed period, indicating increased nutrient uptake by the crop. Similar observations were made by Frey et al. (2013) and Frey et al. (2016). However, for the free drainage period during 2017, the nitrate concentrations were maximum during peak flow events. These results are similar to those in numerous studies (Bakhsh et al., 2002; Drury et al., 1993; Randall and Iragavarapu, 1995; Randall and Mulla, 2001). For 2017, there was a difference between the concentrations observed at some instances, which can be related to the difference in flowrate between the two halves. The average concentrations for both the years was above 10 mg L^{-1} for both the halves, indicating additional requirement for load reduction (Figure 3.8).

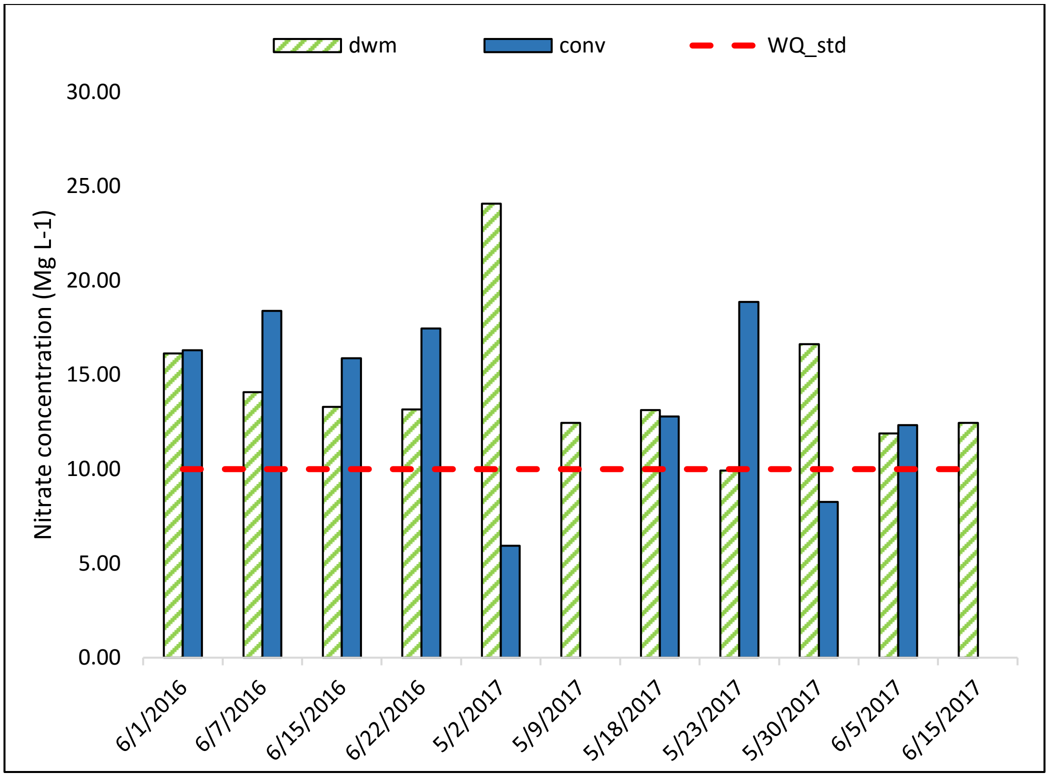


Figure 3.8 Comparison of nitrate concentrations observed in water samples from DWM (striped green bars) and conventional drainage (solid blue bars) along with the EPA drinking water quality standard (dashed red line) (10 mg L^{-1}).

3.5.1.9 Evaluation of performance

The annual nitrate loads ranged from 1.4 kg ha^{-1} (1.2 lbs. ac^{-1}) to 3.3 kg ha^{-1} (2.9 lbs. ac^{-1}) for DWM and 2.3 kg ha^{-1} (2.1 lbs. ac^{-1}) to 4.4 kg ha^{-1} (3.9 lbs. ac^{-1}) for conventional drainage during 2017 and 2016 respectively. The percent reduction by DWM amounted at 40% and 25% for 2017 and 2016, respectively. This was within the range observed in literature (Skaggs et al., 2012b). The difference in total loads for the entire site for the two years studied can be related to the difference in flow volumes caused by lower precipitation during spring 2017 as compared to 2016.

The total input cost for the DWM system was $52 per acre (Table 3.10). Dividing the total input cost by the pounds of nitrate removed (Table 3.11) equaled $28 per pound of nitrate removed. This was higher than the cost observed for DWM, \$1.2 per pound of nitrate removed per year by Jaynes et al. (2010). The critical factor affecting the pound removal rate was the cumulative pounds removed and duration of study. For the site at Alexandria, dry years resulted in less tile flow which affected the nitrate load reduction and the cost per nitrate removed. Considering a 20-year lifespan for the control structure and the average nitrate load reduction per year during the field study, the cost per pound of nitrate removed could be \$2.8 for the management practice for a 20 year implementation period.

Table 3.10 Input costs for the DWM half on the field.

Input	Cost (USD)
Cost of control structure	\$1321.7
Installation costs	\$500
Total cost per acre	\$52.1 acre^{-1}

Table 3.11 load removed by DWM for the entire duration of study

	Load per acre (lbs. ac^{-1})
DWM	4.2
Conventional drainage	5.9
Load reduction per acre due to DWM	1.8

3.5.2 Modeling Study

3.5.2.1 Evaluation of model performance

To determine the efficiency of the model in accurately predicting the hydrological response of the system to DWM, daily tile flow hydrographs from the calibrated setup were compared with the measured values from the field study (Figure 3.9 (a) and (b)). The entire flow for the eastern half (Fig 3.9 (a)) was under managed period. The hydrograph represents flow under the recession limb. For Eastern half, tile flow was observed to be maximum at 1.7 mm for June 1, 2016 and decreased to zero around June 30, 2016. The simulated values followed a similar decreasing pattern and had maximum value at 2.4 mm on June 1, 2016 while eventually becoming 0.04 mm on June 30, 2016.

For the western half, tile flow was observed maximum at 3.9 mm on June 1, 2016 and decreased to zero around 26 June 2016. The simulated values followed the decreasing trend but were mostly lower than the measured values. The maximum tile flow was simulated at 2.8 mm for June 1, 2016 and it decreased to 0.006 mm around June 26, 2016.

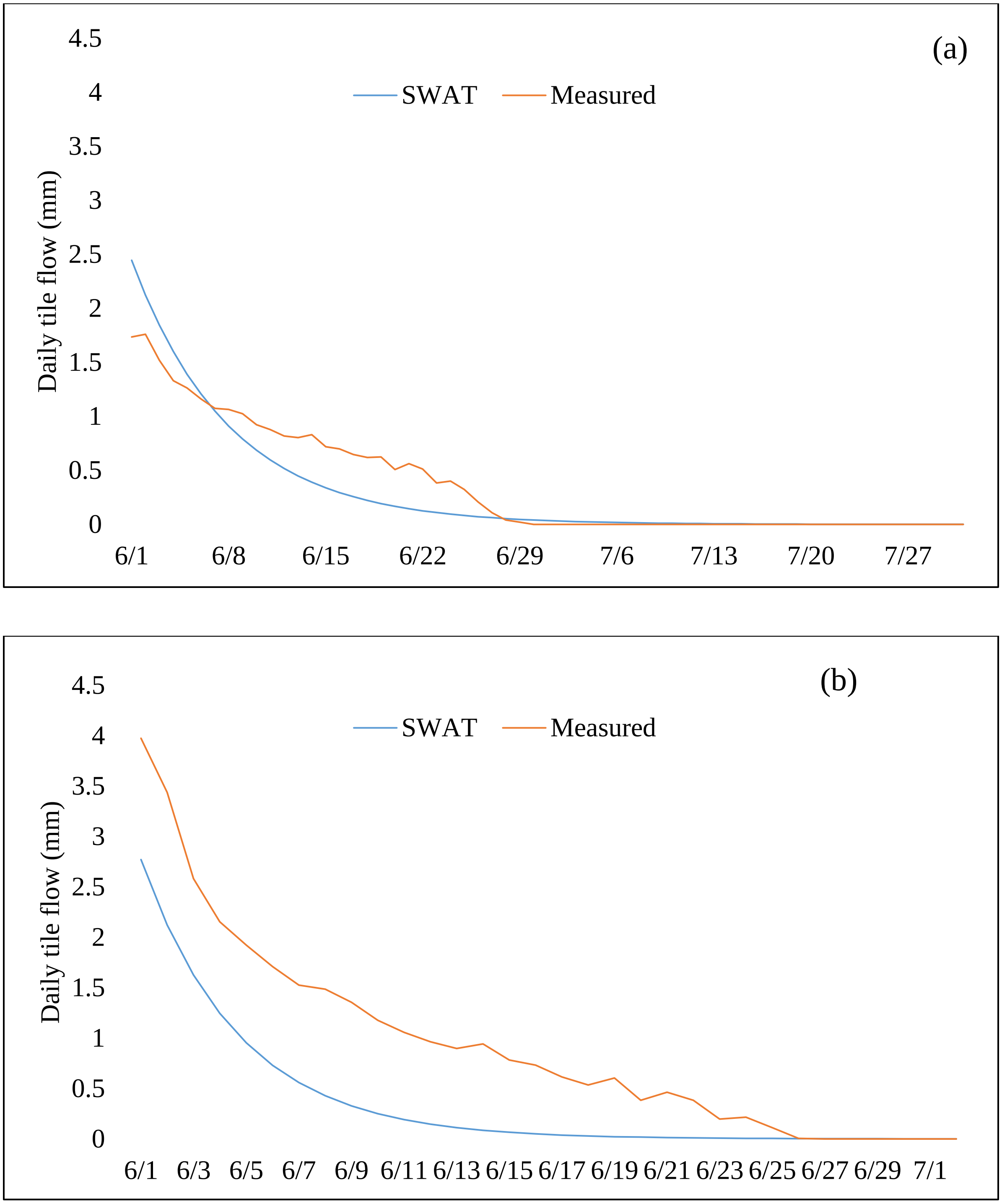


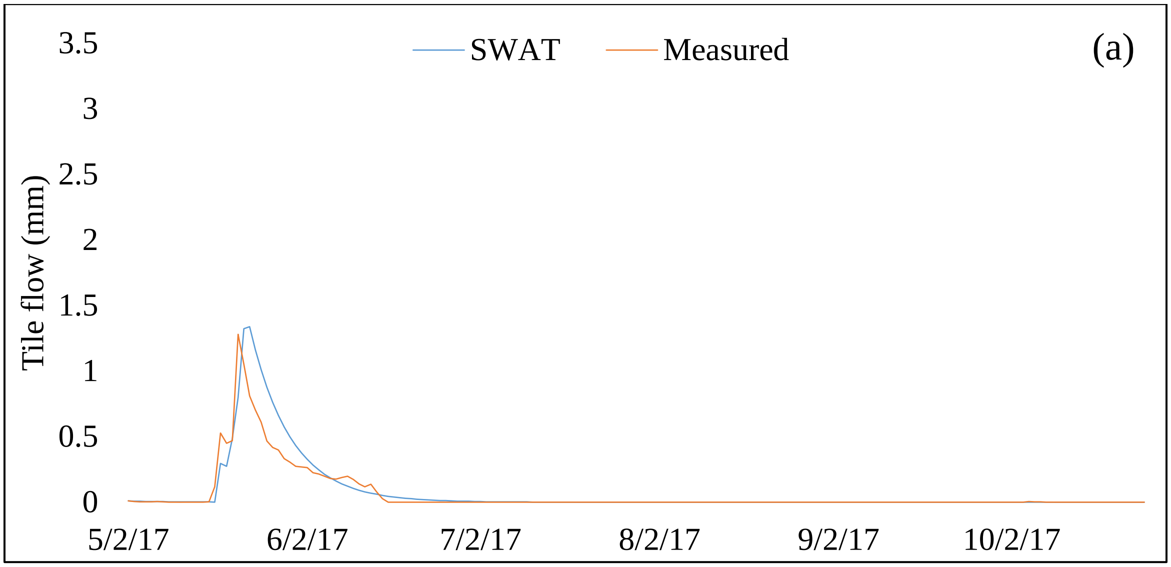
Figure 3.9 Graphical comparison between SWAT and measured daily tile flows for the eastern half (a) and western half (b) during the calibration period.

Simulation results for eastern half were better than the simulation results for the western half. The eastern half model setup had a higher NSE (0.81) as compared to the western half setup (0.54). The PBIAS and RSR values were 17.7% and 0.4, respectively, for the eastern half as compared to 60.8% and 0.7 for the western half. All the statistics except PBIAS for western half were within acceptable ranges (Moriasi et al. 2007), but the ranges were developed for a monthly simulation and not for a daily simulation, widening the range of acceptability for the current project. The high PBIAS values indicate that the models under predicted the tile flow values consistently for both the halves during the calibration period (Table 3.12).

During the validation period (2017), the model accurately predicted peak flows better for the eastern half than the western half, but the mid and low flow volumes simulated for the western half were closer to the measured values (Figure 3.10).

For the eastern half, trickle flow was recorded from May 2, 2017 to May 9, 2017. The flow was recorded again on May 16, 2017 and peak flow was observed on May 21, 2017. SWAT simulated peak flow around May 23, 2017, use of a higher lag time during the model setup led to this delay, which proved to be crucial in simulating the rising limb portion of the hydrograph. The flow gradually declined to zero around June 15, 2017 and was similar to the simulated flow, 0.03 mm for June 15, 2017.

For the western half, flow was first recorded on May 1, 2017 and reached zero on May 4, 2017. After precipitation events during mid-May, flow was observed again on May 16, 2017 and peaked on May 21, 2017. SWAT+ simulations were closer to the recorded values for most period except the peak flow, which was under predicted by the model. In addition, the model over predicted flow during October, 2017. It was due to the lower bulk density, higher available water capacity, and lower tile lag time values used to setup the SWAT project which increased the percentage of sub surface flow generated for a precipitation event as compared to the setup for eastern half.



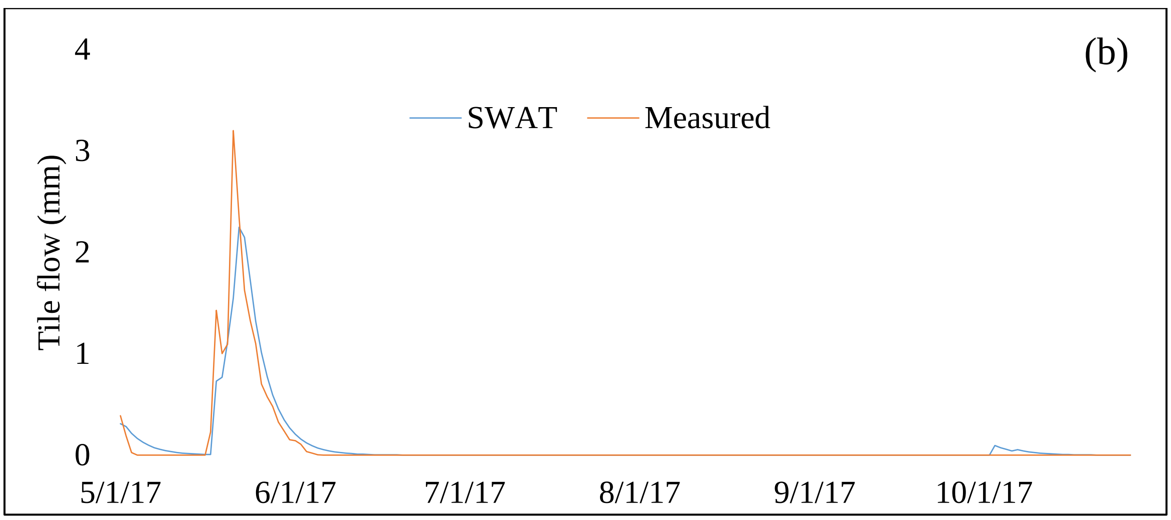


Figure 3.10 Graphical comparison between SWAT simulated and measured daily tile flows for the eastern half (a) and western half (b) during the validation period.

The western half had a better NSE (0.84) as compared to the eastern half (0.69) during the validation period. The RSR values for both the halves were similar and in the acceptable range. The PBIAS was greater for the eastern half at -23.6%, which denotes that the model over-predicted the daily tile flow values during the validation period, specifically after May 23, 2017 but it was -7.1% for the western half (Table 3.12).

Table 3.12 Evaluation statistics observed during the calibration and validation period.

Objective function	DWM (East)		Conventional (West)	
	Calibration	Validation	Calibration	Validation
NSE	0.81	0.70	0.54	0.84
PBIAS (%)	17.68	-23.56	60.83	-7.11
RSR	0.43	0.55	0.68	0.40

3.5.2.2 Comparison of different management schedules

To study the impact of different water table management schedules on field hydrology, the SWAT model was systematically adjusted. Three different management scenarios were run for each half of the field. First, the model was run under conventional drainage system (Conv). Second, the model was run under DWM with the boards taken out during the winter (MG1) and third run involved DWM with boards put in the structure until the land preparation period, taken out until planting and then put in again after planting. Finally, the boards were taken out close to the harvesting period and then put back in after harvesting to store water during the winter (MG2).

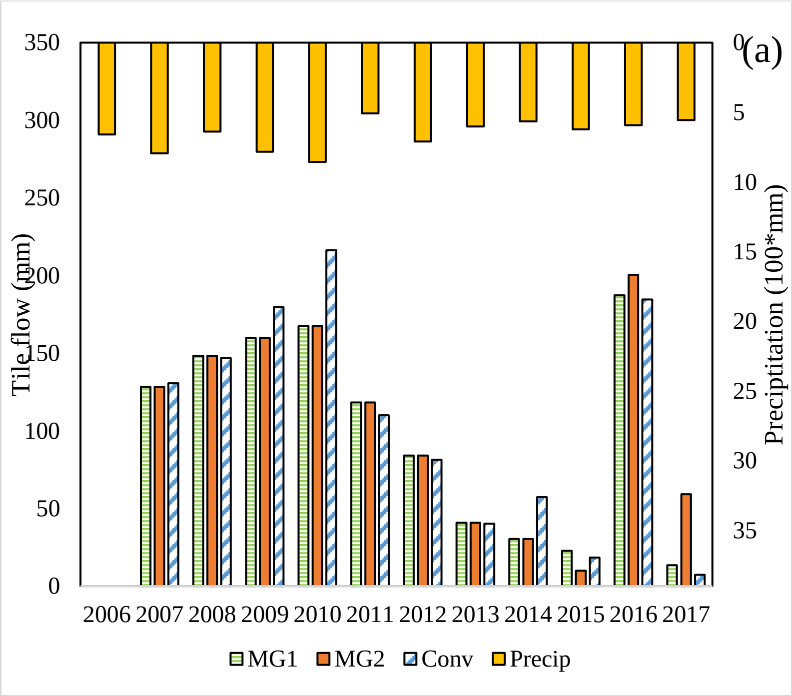
The study period included both dry and wet years. For wetter years such as 2007, 2009, 2010, 2011, 2012, and 2015; DWM resulted in greater surface runoff, lower tile flow volume, and higher ET as compared to conventional drainage similar to the results discussed in Skaggs et al. (2010).

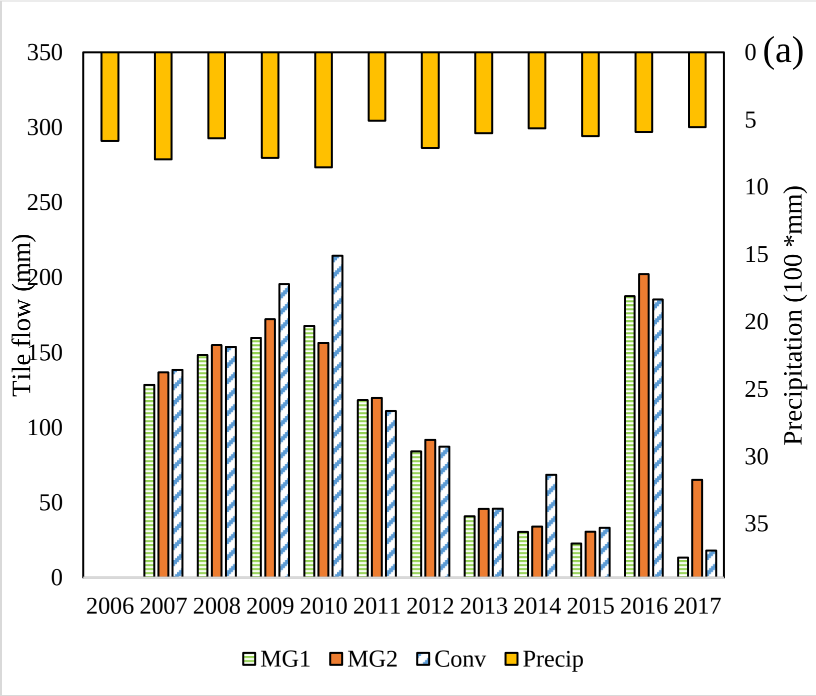
Raising the water table resulted in lowering the total outflow during the growing season. Similar results were also observed for the western half, setup with a lower bulk density, higher available water capacity, lower drain tile lag time, and higher time to drain to field capacity as compared to the eastern half. The variability in these properties affected the magnitude, but not the overall trends.

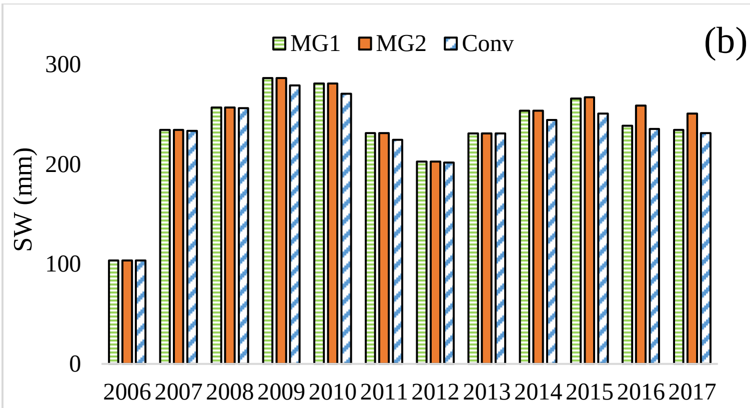
For considerably dry years, it was observed that MG1 and MG2 resulted in an overall increase in total tile flow during the growing season except for 2015 which was affected by heavy precipitation during July leading to flow in the Conv scenario and storage in MG1 and MG2 scenarios. Greater soil water content (SW) for the MG1 and MG2 scenarios contributed to the tile flow during the growing season. Freezing of soil water stored during the winter and then subsequent thawing resulted in tile flow during spring free drainage period.

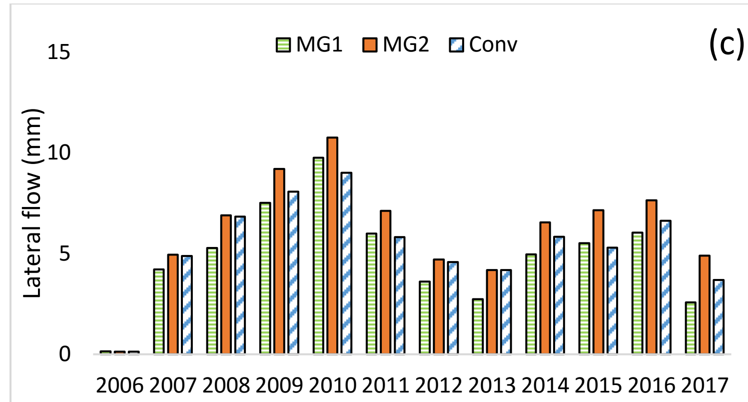
Furthermore, lateral flow for MG2 was more than MG1. Greater water present underneath the soil resulted in grater lateral flow. The eastern half had lower lateral flow as compared to the western half owing to the different soil parameters used to setup the different scenarios. In addition, higher SW stored for the western half resulted in an increase in lateral flow.

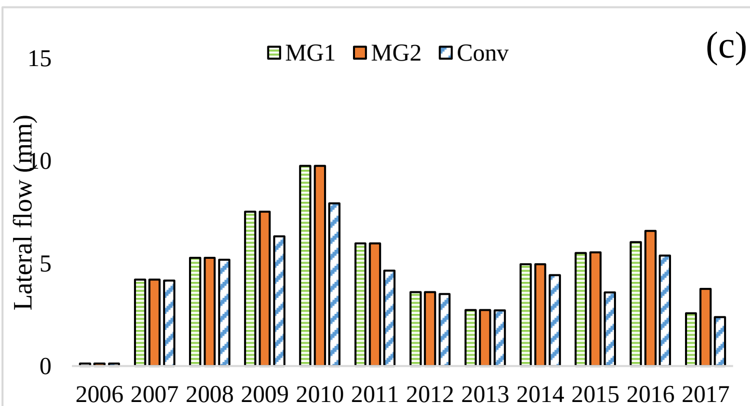
In general, MG1 and MG2 reduced the flow volume during the managed period, but resulted in higher flow during the free drainage period throughout the study period. Overall, tile flow comprised of 1.3% to 33.8% of the annual precipitation across the three scenarios for the eastern half and ranged from 2.4% to 34.1% of the annual precipitation across the three scenarios for the western half. The amount of tile flow was dependent on the timing rather than the intensity of precipitation. From the daily tile flow charts, it was observed that precipitation during June and July resulted in flow through the system and reduction in outflow for the MG1 and MG2 scenarios. Continuous precipitation events, such as one around June 1, 2010 and June 10, 2010 had a greater impact on DWM performance as compared to a higher intensity but isolated event, such as one on June 5, 2008. Interestingly, a difference in daily tile flows was observed between MG2 and MG1 for the eastern half during 2015 (Figure 3.13). For the majority of the study, MG2 and MG1 acted similarly, but precipitation events during the free drainage period for MG1 during November 2015 resulted in a higher flow as compared to MG2. The following year, MG2 resulted in a greater flow as compared to MG1 as a direct consequence of storing more water during the winter of 2015. A similar trend was observed for the western half with a slight variation in magnitude and occurrence of tile flow for 2014 and 2015. Overall, the net increase of flow during fall of the year was equal to the decrease in tile flow for the following year.


(a)
Tile flow (mm)
Precipitation (100*mm)
2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016 2017
MG1 MG2 Conv Precip


(a)
Tile flow (mm)
Precipitation (100 *mm)
2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016 2017
MG1 MG2 Conv Precip


MG1 MG2 Conv
(b)
SW (mm)
300
200
100
0
2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016 2017


MG1 MG2 Conv
(c)
Lateral flow (mm)
15
10
5
0
2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016 2017


MG1 MG2 Conv
(c)
Lateral flow (mm)
15
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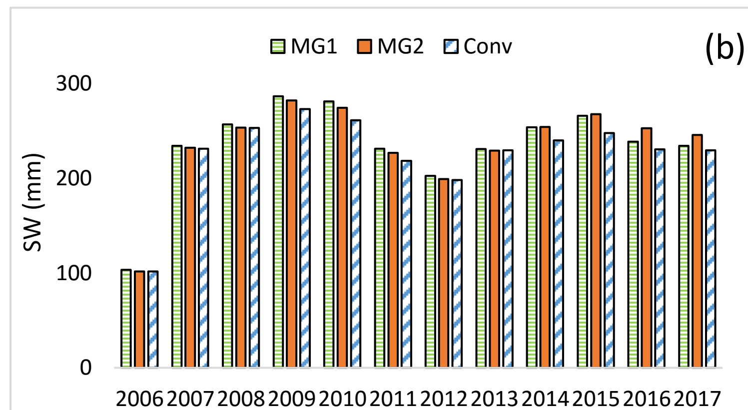

MG1 MG2 Conv
(b)
SW (mm)
300
200
100
0
2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016 2017

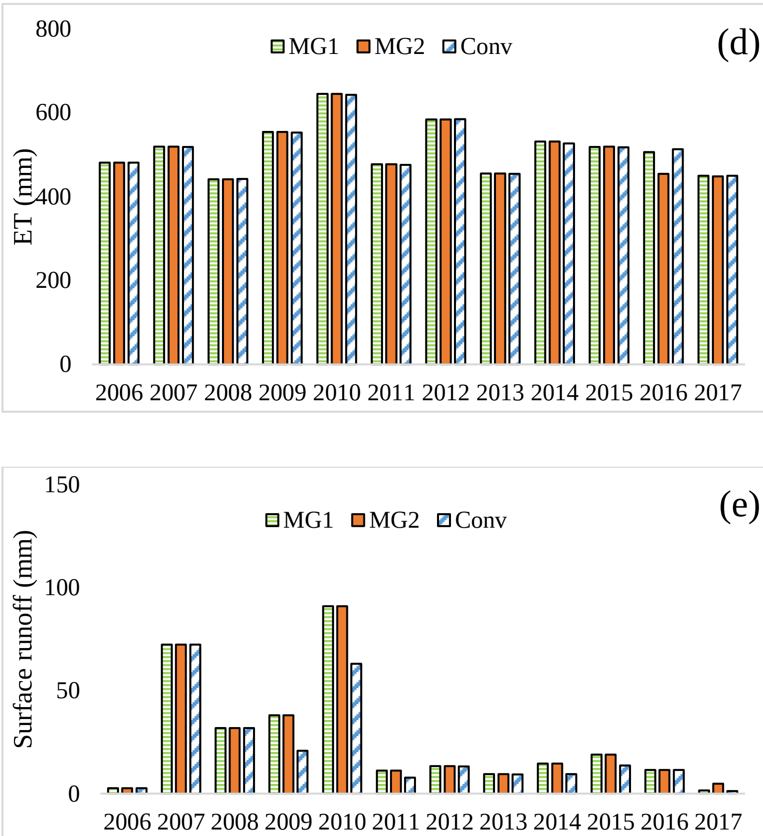


Figure 3.12 Annual tile flow (a), Soil water stored in the profile (b), sub surface lateral flow (c), ET (d) and surface runoff (e) for the eastern half in mm for three different management schedules MG1, MG2 and Conv throughout the study period.

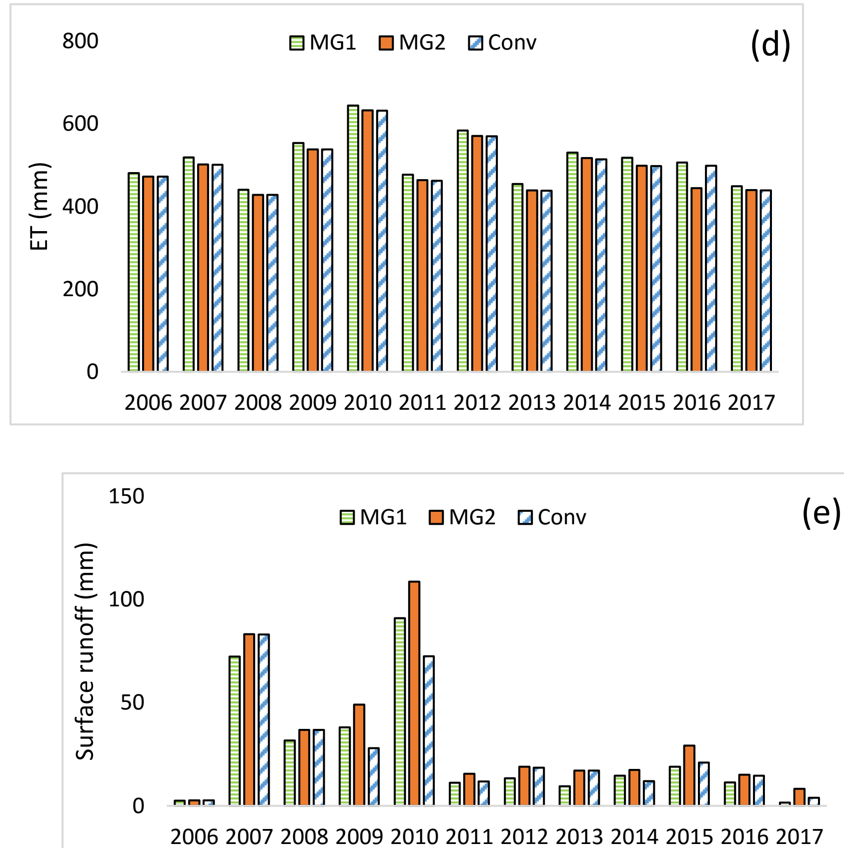


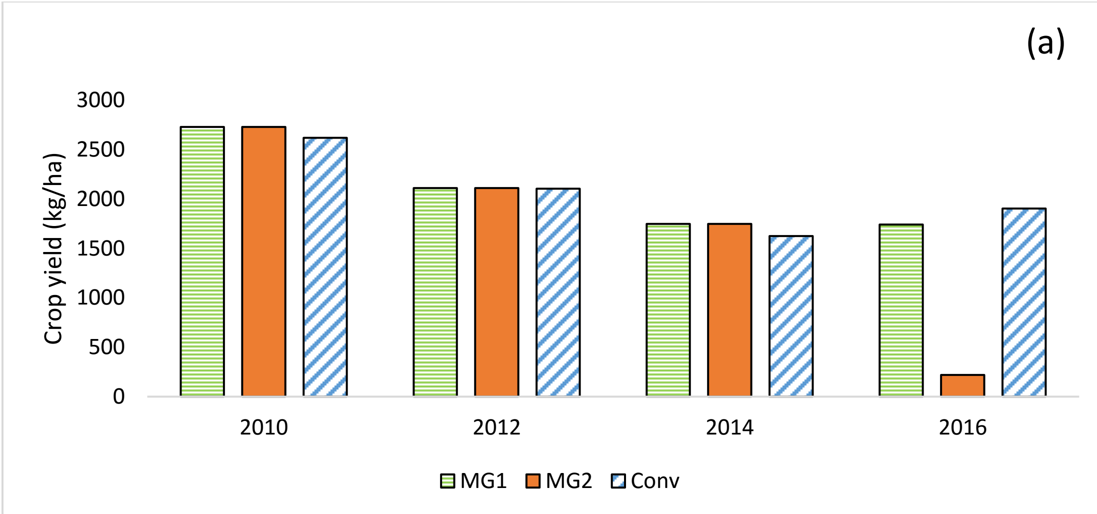
Figure 3.11 Annual tile flow (a), Soil water stored in the profile (b), sub surface lateral flow (c), ET (d) and surface runoff (e) for the western half in mm for three different management schedules MG1, MG2 and Conv throughout the study period.

Figure 3.13 Difference in daily tile flow for MG2 and MG1 management schedules throughout the study period for the eastern (a) and western half (b). Positive values represents greater flow in MG2 than MG1 and vice versa.

Daily tile flow volumes pertaining to all the three scenarios were analyzed for the reduction in outflows during the managed period. The percent flow reduction for the eastern half ranged from 5.4% to 58.4% for MG1 when compared to Conv. MG1 and MG2 had no differences in flow for most of the study. For the western half, higher flow reduction rates were observed (12.2% - 92.1%). Lower bulk density and tile lag time for the western half might allow for more drainage, thereby increasing the total outflow during the managed period. Although, this resulted in higher peak flow rates flowing out through the site, similar to observations during field study.

3.5.2.3 Impact on crop yield

Increased nutrient rich water availability during the managed period might result in increased yield as it supplements the crop's water and nutrient requirements. The annual crop yield was highest for MG1 for most of the study. Both Mg1 and MG2 had similar crop yields for both soil type and tile configurations as compared to Conv for dry years, however, for wetter years, Conv showed a slightly greater yields as compared to MG1 and MG2. Excess water on the field prevents proper aeration beneath the soil surface and may lead to crop stress and eventually decrease yield. Ale et al. (2009) studied the impact of drainage on crop yields as affected by excessively dry and wet conditions. During the 2016 growing season, MG2 had the maximum SW and led to potential crop failure or extremely low yields due to excess water during the germination stage. Overall, there were no trends observed for crop yields based on the on-field water table management strategies (Figure 3.14 and Figure3.15).



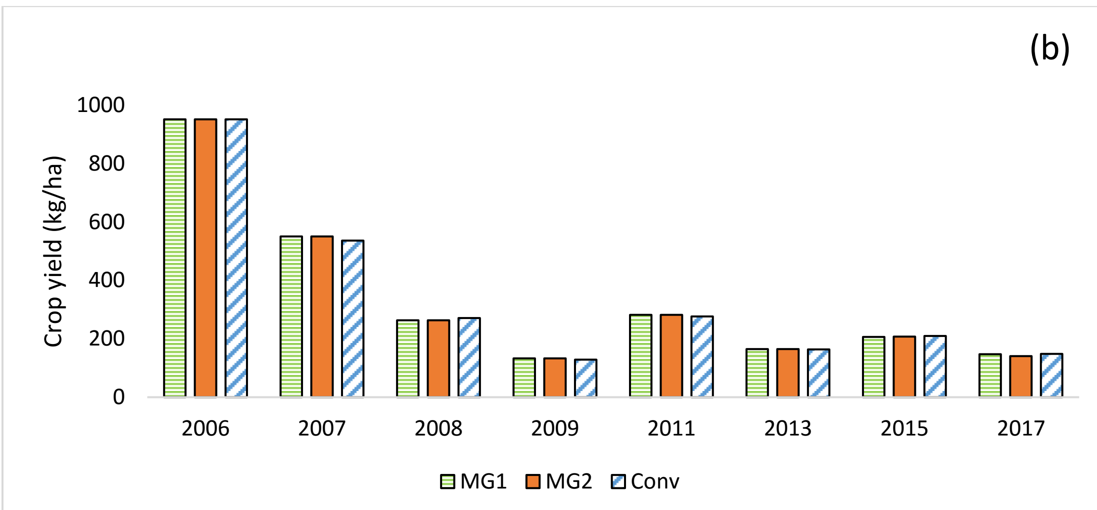


Figure 3.14 (a) Corn and (b) soybean yields (kg ha^{-1}) for three different management schedules MG1, MG2, Conv throughout the study period for the eastern half.

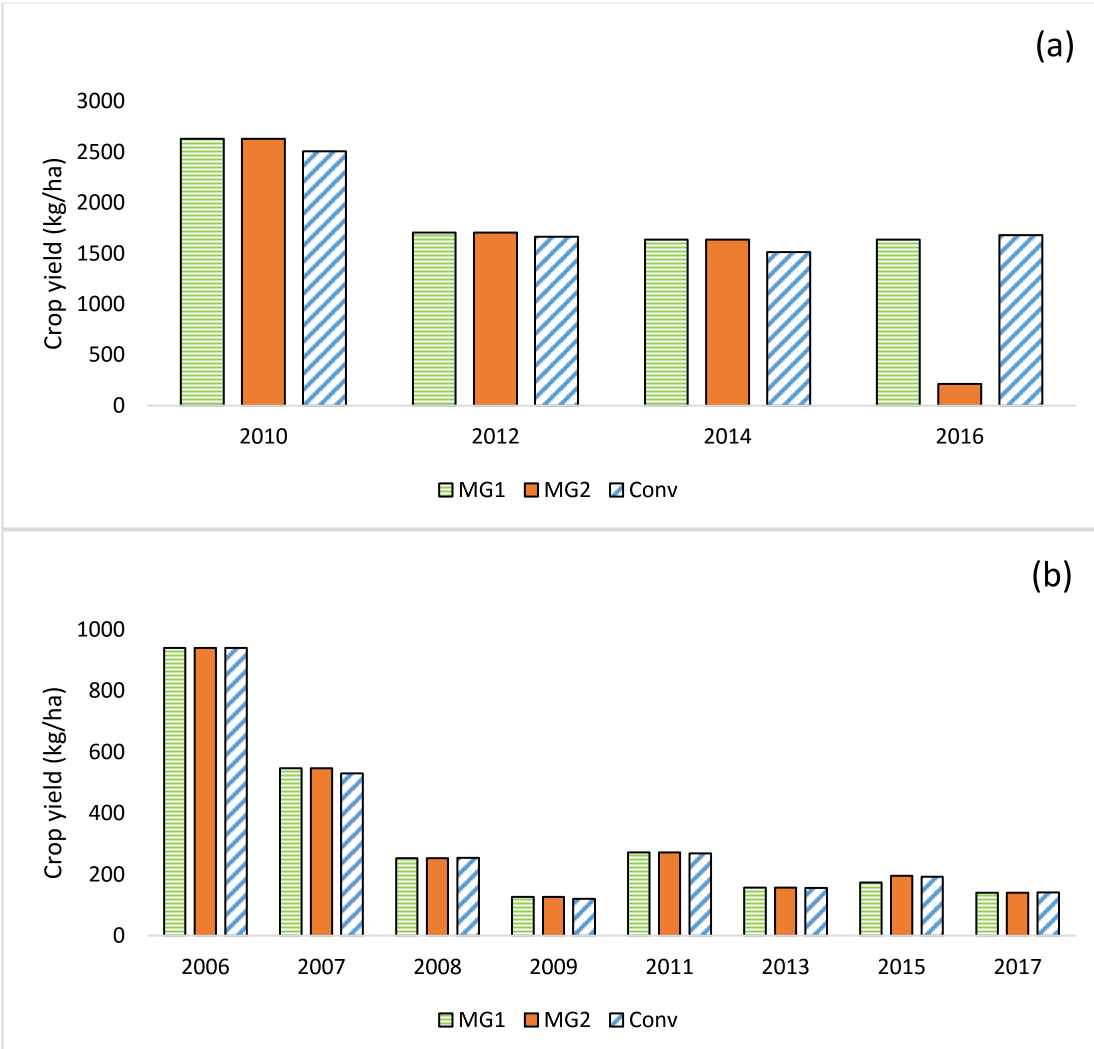


Figure 3.15 (a) Corn and (b) soybean yields (kg ha^{-1}) for three different management schedules, MG1, MG2, Conv, throughout the study period for the western half.

3.6 Conclusions

This study demonstrated and assessed the impacts of drainage water management on hydrology and water quality at a field in eastern South Dakota. DWM was successful in reducing the total outflow of water from the field by 8 mm during 2016. There was also a difference in the nitrate concentrations during the managed period which supports plant nutrient uptake as the soil analysis enabled us to discard the possibility of denitrification. Overall, there was a 25% nitrate load reduction during the managed period, which was within the range reported in previous literature. Total cumulative load reduction by the practice amounted to 1.83 lbs. ac^{-1} (2.05 kg ha^{-1}). The cost of removing a pound of nitrate per acre amounted to approximately $28, which is higher than mentioned in Jaynes et al. (2010) due to lower cumulative nitrate loads resulting from lower tile flows during dry years 2016 and 2017.

Further, A SWAT+ project was setup to study impact of DWM on field hydrology and it was observed that seasonal variability in precipitation played a major role in DWM performance under different water management strategies. An overall reduction in outflow was observed for the DWM scenarios during wet years. For relatively drier years, soil water storage for both DWM scenarios resulted in increased flow during the free drainage period. MG1 also had higher crop yields as compared to MG2. Both MG1 and MG2 resulted in greater yields during dry years because of reduced water stress for the managed area, but during relatively wetter years, crop yield was less for the managed scenarios as compared to conventionally drained scenario because of excess water due to reduced drainage from DWM.

Overall, it was concluded that DWM is a useful management practice to not only achieve water quality targets, but also reach productions goals from a producer's perspective.

3.7 Limitations and recommendation for future work

The study focused on the effect of drainage water management on field hydrology, but the duration of the study limited testing the impact of the practice on crop yield. In addition, to facilitate the decision making process for agricultural water management on a large scale, the modeling setup needs to be expanded to a watershed scale.

Chapter 4. APPLICATION OF SATURATED BUFFERS ON A FIELD SCALE SETUP TO IMPROVE WATER QUALITY IN SOUTH DAKOTA.

ABSTRACT

Saturated buffers are an edge of field practice developed to reduce nutrient transport from croplands to surface waters. This study involves the evaluation of the practice at two locations in South Dakota. Two field scale buffer sites were installed in 2016 near Flandreau and Baltic, SD. Water from tiled croplands was diverted to the buffer zone using a control structure with a tile running from the mid chamber of the structure through the entire length of the buffer parallel to a waterway. To study the reduction in nitrate concentrations as a result of the practice, a set of well transects were installed and sampled under flow conditions. Results for Flandreau showed an average nitrate removal rate of 86% and 65% for 2016 and 2017, respectively. The lower reduction rate for 2017 was associated with high flow volumes fed to the buffer zone resulting in inadequate nutrient uptake by the plant and hindering reduction through denitrification. For Baltic, the average reduction rate was 95% for 2017 when 99% of the drainage water was diverted to the buffer. Both the saturated buffers were successful in removing nitrate from tile drainage water, but the efficiency was dependent on input flow volumes fed to the buffer throughout the study period. The cost of removing a pound of nitrate per acre drained were $22 for one year under observation at Baltic and $0.6 for two years under observation at Flandreau.

4.1.Introduction

Accumulation of excess nitrogen in water bodies leads to algal blooms, which are harmful for aquatic ecosystems. Tile drainage has been identified as a major pathway for nutrient transport from croplands to surface water bodies (Alexander et al., 2000a; Petrolia and Gowda, 2006). Conservation drainage practices, like saturated buffers, were developed and tested in Iowa for reducing nitrate loads from agricultural fields to surface waterbodies (Jaynes and Isenhart, 2014a).

Saturated buffers use a control structure to divert flow to a vegetative strip via a subsurface tile installed parallel to a waterway. Nutrient content is reduced from the subsurface water due to a combination of N microbial immobilization, plant uptake, and natural denitrification (Jaynes and Isenhart, 2014b). Dosskey et al. (2002) found that subsurface buffers were more effective in reducing nitrate loads than buffers intercepting surface runoff.

A joint program across the Midwest evaluated the efficiency of saturated buffers in reducing nutrient transport from croplands from 2014-2015. Buffers were able to remove a substantial amount of nitrate from tile drainage water. The average percent nitrate concentration reduction ranged from 18% to 85% from the sites which diverted at least 50% of the water to the buffer zone(Utt et al., 2015).

The goal of this study was to evaluate the effectiveness of saturated buffers in reducing nitrate transport from tiled croplands in South Dakota. It was achieved by studying the nitrate reduction performance of a saturated buffer system on a field scale and the factors that impacted its performance. The results from this would be helpful to the producer looking to implement the practice to improve agricultural water management for a cropland. In addition, policy makers across the state could use the results in encouraging producers and organizations to implement the practice and improve the overall stream water quality across South Dakota.

4.2.Materials and Methods

4.2.1.Site Setup

Two saturated buffer sites were installed near Baltic (43˚ 43’ 52.54” N, 96˚ 40’ 54.20” W) and Flandreau (43˚ 57’ 39.98” N, 96˚ 29’ 55.60” W), SD during the summer and fall of 2016, respectively. Water was diverted through a control structure using a subsurface perforated tile line running parallel to a stream. The buffer zone near Flandreau was split into two sections separated by a non-perforated tile section to minimize lateral flow between the distribution tile and the stream (Figure 4.1).

The drainage system near Baltic drained 6 ha, whereas the system at the Flandreau site drained 35 ha to the buffer zone. To facilitate computation of nitrate reduction rates along the length of the buffer, two pairs of well transects were installed at Baltic (Figure 4.1) and labelled as B1A and B1B near the control structure and, B2A and B2B at the farther end. Further, B1A and B2A were closer to the distribution tile and, B1B and B2B were farther from the tile and closer to the outlet stream. The buffer zone near Flandreau drained a larger area and so was longer in length as compared to the buffer near Baltic; it was established with three pairs of monitoring wells. Similar to the labelling pattern followed for Baltic, the monitoring wells were labelled as F1A, F1B, F2A, F2B, F3A and F3B (Table 4.1).

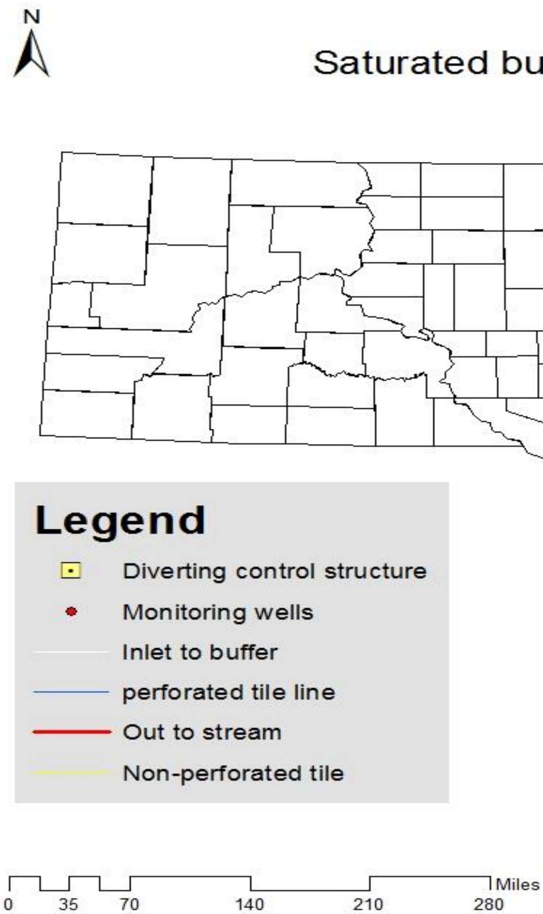


Figure 4.1 Two saturated buffer sites were setup near Baltic and Flandreau, SD. Each site was setup with a pair of well transects (point feature) to observe nitrate reduction throughout the length of the buffer. The distribution tile consisted of perforated (solid blue line) and non-perforated (soil yellow line) sections diverting water from a control structure (yellow box), which drained water from tiled croplands.

Table 4.1 Site specifications for saturated buffer sites. Flandreau had a longer buffer length and area drained as compared to the Baltic buffer site. There were three pairs of monitoring wells setup at Flandreau as compared to two at Baltic.

	Baltic	Flandreau
Drained area	6 ha (15 ac)	35 ha (87 ac)
Tile size	0.15 m (6 in)	0.13 m (5 in)
Buffer length	61 m (200 ft)	101 m (330 ft)
Monitoring wells	4	6
Depth of tile in the buffer zone	0.91 m (3 ft)	0.86 m – 0.96 m (34 in – 38 in)

4.2.2. Instrumentation

Numerous sensors were used on each site to record multiple weather and flow parameters. The sensors were deployed during the cropping season each year and recorded data every 15 minutes. The recorded parameters included water level in the control structure (Decagon CTD – 10), precipitation (Decagon ECRN100), air temperature (Decagon VP-4), and water table depth in each monitoring well (HOBO water level data logger U20). With the exception of the Hobo water level loggers used to monitor the wells, all the sensors were setup next to the control structure at each site. These were connected to an EM-50g data logger which recorded and logged in the data. The data was viewed and downloaded by using the manufacturers’ web service address http://www.ech2o.com/.

4.2.3. Field data collection

4.1.1.1 *Water sampling*

During flow conditions, weekly water samples were collected from the control structure and monitoring wells. For Flandreau, samples were collected for two cropping seasons, 2016 and 2017 but for Baltic sampling was done only for 2017. Pre-labelled 250 ml Nalgene bottles were used for transporting and storing samples. Upon collection, the samples were kept in a cooler and transported to the laboratory where they were stored under freezing conditions until further processing.

4.1.1.2 *Flow rate*

Decagon CTD 10 sensors, were used at Baltic and Flandreau to record the water level (mm) in the control structures. The CTD 10 sensor is a pressure transducer that records the water conductivity, water temperature, and water depth. The water level readings were then used to calculate the flowrate in the structure. Each control structure was fitted with a V-notch and a flow equation (Equation 4.1) was developed for the boards at the Agricultural and Biosystems Engineering Department, SDSU (Partheeban et al., 2014). The flow above the V-notch was calculated using calibrated flow equations for different sized commercially available Agri-drain control structures as computed by Chun and Cooke (2008)

$$Q = 1.7406 * (H)^{1.9531} \qquad \text{(Equation 4.1)}$$

Where, Q is the discharge through v-notch ($L min^{-1}$) and H is the height of the water in the v-notch (cm).

4.1.1.3 *Soil analysis*

Soil samples were collected for each buffer site near each monitoring well. Each sample had 5 replications to account for variance in the analysis. Soil samples were collected at three depths, 0 – 12 inches (0 – 30 cm), 12 – 24 inches (30 – 60 cm), and 24 – 36 inches (60 – 90 cm) below the surface using soil augers. The samples were then placed into pre-labeled plastic bags and transported to the laboratory. Upon arrival at the laboratory, samples were stored under freezing conditions and sent for further analysis to the soil lab at SDSU for soil nitrate-nitrogen, Olsen phosphorus, Potassium, electrical conductivity, organic matter, and pH.

4.1.1.4 *Water Quality Analysis*

Frozen water samples were thawed and then filtered using 30ml syringes and 0.45µm nylon membrane filters prior to nitrate analysis. Filtered water samples were labelled and stored in 60 ml Nalgene bottles under freezing conditions until analysis. Samples were analyzed using a Seal AQ2 discrete analyzer (Seal Analytical Inc., Milwaukee, WI). The EPA 353.2 method was followed to calculate the nitrate plus nitrite concentration in water samples. Nitrite was analyzed separately using EPA 354.1 method, which omits the use of cadmium coil during the analysis. The nitrate concentration was calculated by subtracting the nitrite from the nitrate-nitrite concentration (US-EPA, 1993).

4.3.Results and discussion

4.3.1.Baltic

4.3.1.1.Climate

The monthly temperatures recorded at the site were higher than the 30 year averages indicating a warmer year. The research site received maximum rainfall during May, 2017 which was higher than the 30 year average for the month. The months before that were drier and received a total of 82.6 mm approximating to a third of the rainfall for May 2017. The period after July 2017 was also substantially wet and resulted in rapid but short duration flow events being diverted to the buffer zone.

Table 4.2 Monthly precipitation and temperatures observed at the site and compared with the long term averages (1981-2010) at the nearest weather station (USC00391851).

Month	Precipitation (mm)		Temperature (˚C)	
	Total 2017	(1981-2010) 30 year average	Average 2017	(1981-2010) 30 year average
January	1.4	14.2	-7.0	-8.3
February	8.6	15.3	-1.1	-5.5
March	2.2	44.6	0.6	0.8
April	71.4	76.5	8.4	8.3
May	258.4	86.3	13.4	14.7
June	67.2	99.6	20.7	20.2
July	34.0	78.4	23.4	23.1
August	157.6	77.4	18.9	21.7
September	43.8	70.5	17.3	16.5
October	109.2	55.0	10.1	9.1
November	-	34.6	1.5	0.6
December	-	17.6	-7.3	-6.8

4.3.1.2.Soil Analysis

Soil organic matter

The soil organic matter (OM) ranged from 3.9% to 5.0% and was maximum for the 12 in. – 24 in. soil layer at the wells on the northern end of the buffer. OM is an estimation of the soil organic carbon (SOC) present in the soil, which is a major factor controlling the denitrification process (Weier et al., 1984). Measured OM was converted to a percent SOC using the Van Bemmelen factor which yielded a mean value of 2.5% for the site (Van Bemmelen, 1890). It supports the threshold SOC value of 2% for setting up a buffer site that would easily support denitrification, as discussed by Utt et al., (2015). Higher SOC values also have a direct relationship with the denitrification potential of soil (Bremner and Shaw, 1958; McGarity, 1961).

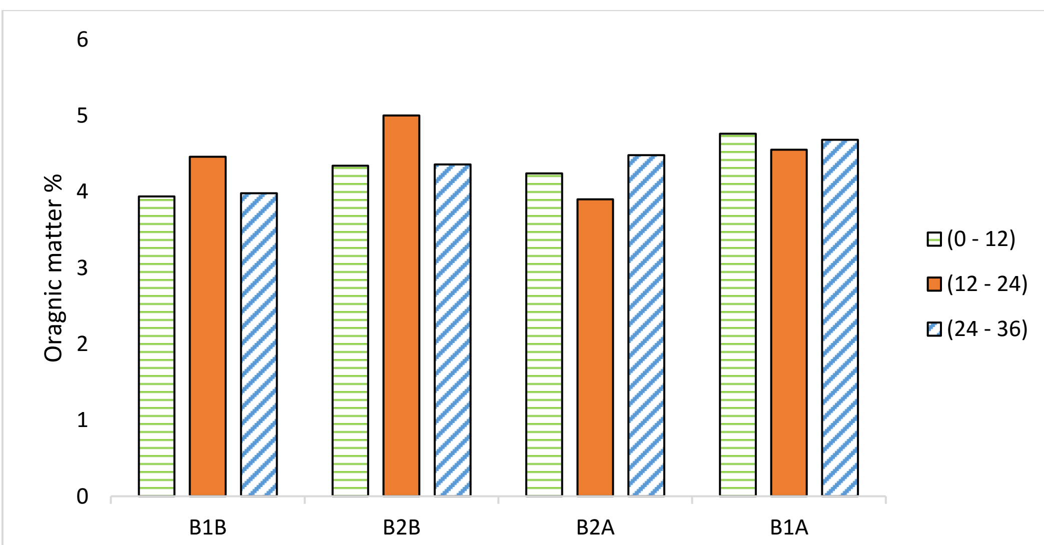


Figure 4.2 Organic matter (%) observed at each sampling point near the monitoring wells B1B, B2B, B2A and B1A at three depths 0-12in., 12-24in., and 24-36in.

Soil NO_3-N

Higher NO_3-N concentrations were observed at B2B and B2A wells suggesting N mineralization at lower depths. Overall, the NO_3-N concentrations ranged from 3.9ppm to 9.0ppm. The concentration was lower for the uppermost layer, as compared to the middle layer which can be related to high organic matter and plant cover (Jaynes and Colvin, 2001).

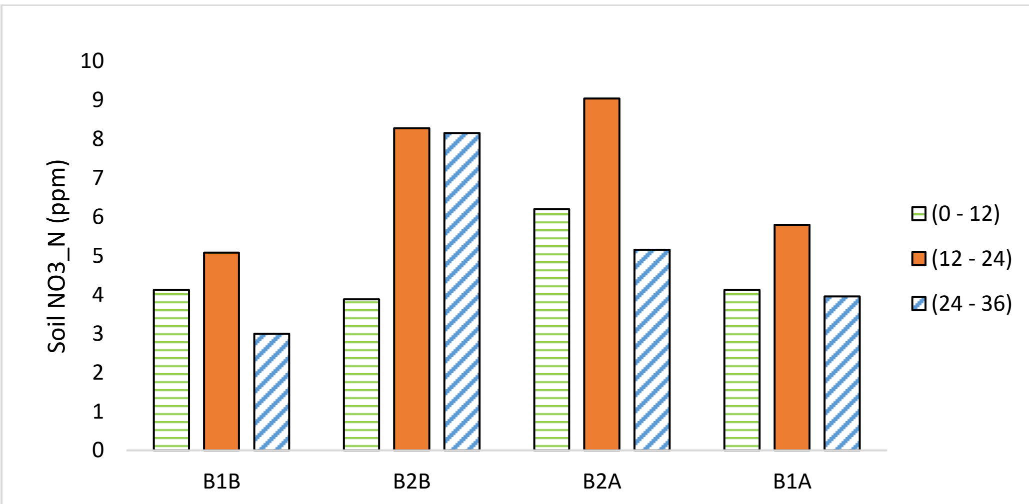


Figure 4.3 Soil NO3-N content (ppm) observed at each sampling point near the monitoring wells B1B, B2B, B2A and B1A at three depths 0-12in., 12-24in., and 24-36in.

Soil pH

Soil pH ranged from 7.6 to 7.9 and decreased with increasing depth. pH values throughout the buffer zone represented the presence of anoxic conditions beneath the soil surface (Sallade, Y E; Sims, 1997; Valero et al., 2007); for the buffer it was due to shallow water table with relatively high nitrate concentrations.

Soil pH also exerts a strong impact on the denitrification potential as it controls the carbon availability for the denitrifying bacteria (Koskinen and Keeney, 1982). A neutral or slightly alkaline pH has been studied to support denitrification enzyme activity in the soil (ŠImek and Cooper, 2002). Values measured for the site support existence of lower mole fractions for N_2O, indicative of a rapid conversion of N_2O to N_2(Koskinen and Keeney, 1982).

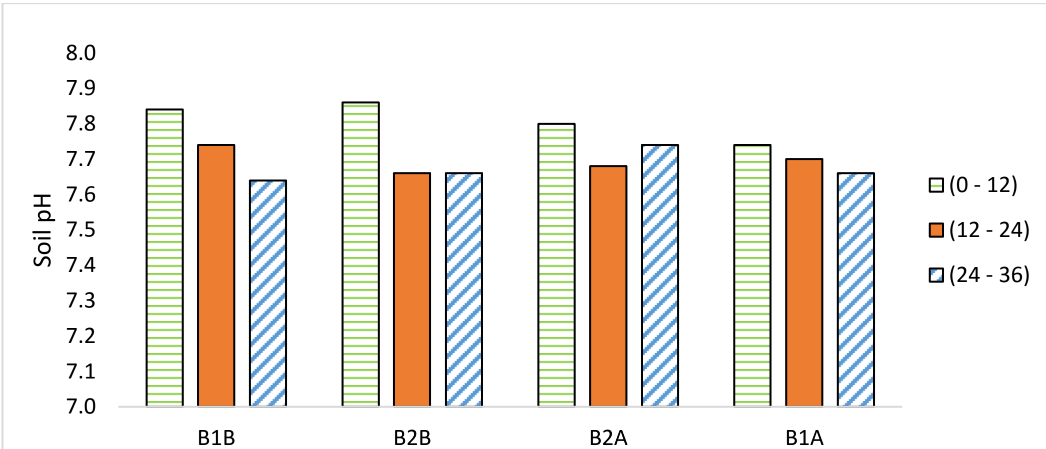


Figure 4.4 Soil pH observed at each sampling point near the monitoring wells B1B, B2B, B2A and B1A at three depths 0-12in., 12-24in., and 24-36in.

Soil P

Soil P concentration ranged from 30.8 to 51.9 ppm for the topmost layer, 22.1 to 69.8 ppm for the mid layer and 18.3 to 36.2 ppm for the bottom most layer. Higher soil P concentrations were measured at the B2B well indicating leaching and possible mineralization of the nutrient in the buffer zone.

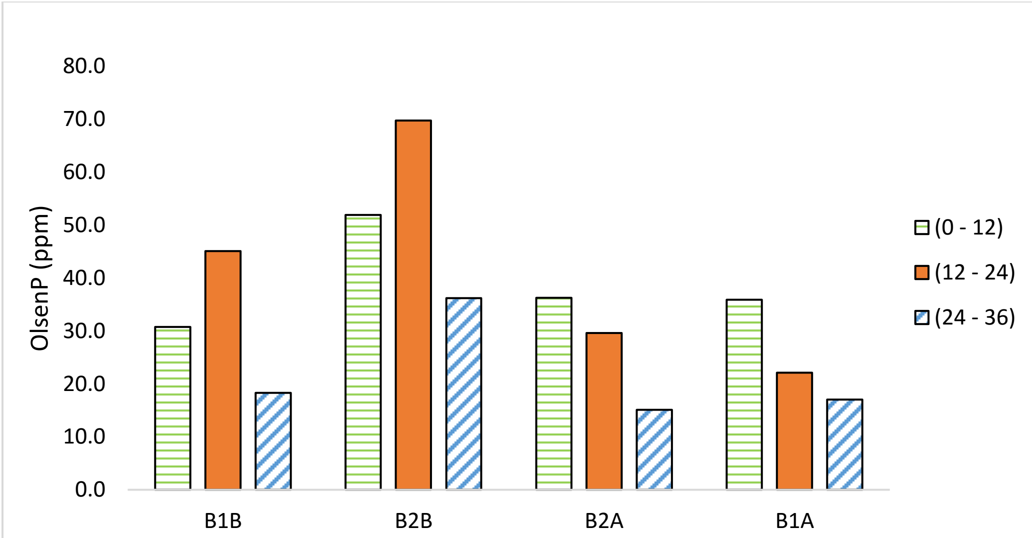


Figure 4.5 Soil P (ppm) observed at each sampling point near the monitoring wells B1B, B2B, B2A and B1A at three depths 0-12in., 12-24in., and 24-36in.

4.3.1.3.Conditions inside the buffer zone

Shallow Groundwater Table

The mean water table depth ranged from 0.39 m to 1.37 m for 2017 inside the buffer zone (Figure 4.6). The water table declined from May to August apart from occasional storm events which temporarily increased the water table depth in the buffer zone. Most of the samples were collected around peak flows which corresponded to a sharp rise in water table.

Buffers have been tested to perform considerably well in areas with a high water table because it leads to anaerobic conditions in the upper layers of soil rich in SOC and promotes denitrification (Burford and Bremner, 1975). For the buffer zone near Baltic, shallow water table observed during the study and sufficient OM supports the development of anaerobic conditions leading to denitrification.

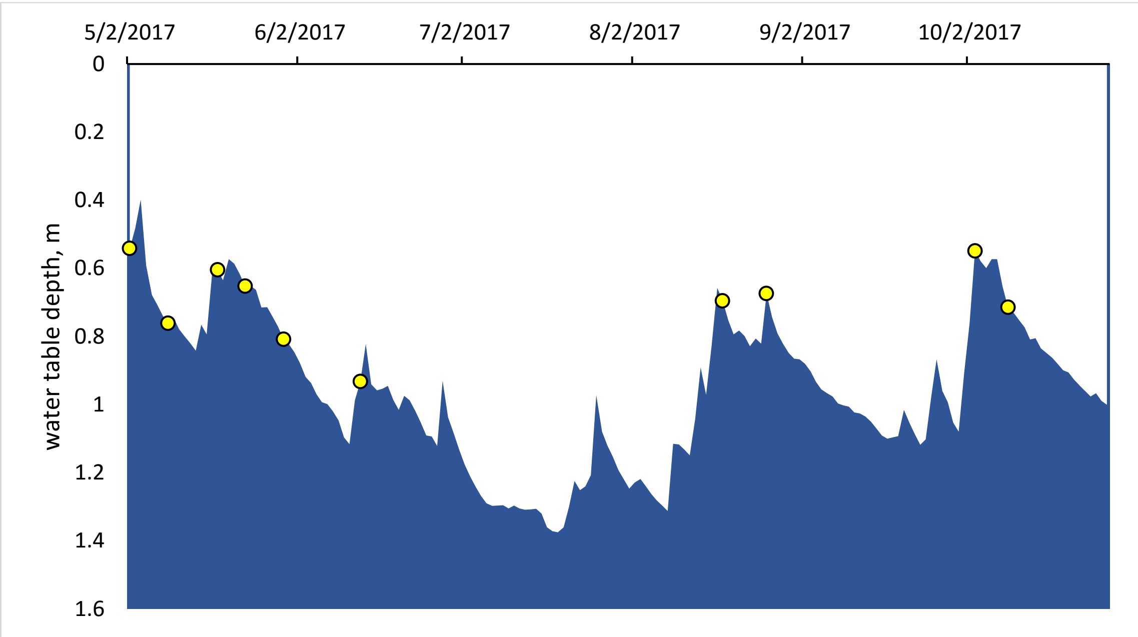


Figure 4.6 Daily water table depth (m) below the soil surface and days of water sampling from the monitoring wells during 2017.

Hydraulic gradient between wells

A positive gradient was observed between the wells for most of the season. A slight variation was observed around July 26, 2017 where, due to receding flow conditions, a negative gradient was observed between the eastern pair of wells. This might have been due to rapid flow from the B1A to B1B, but a relatively slow movement from B1B to the stream (Figure 4.7).

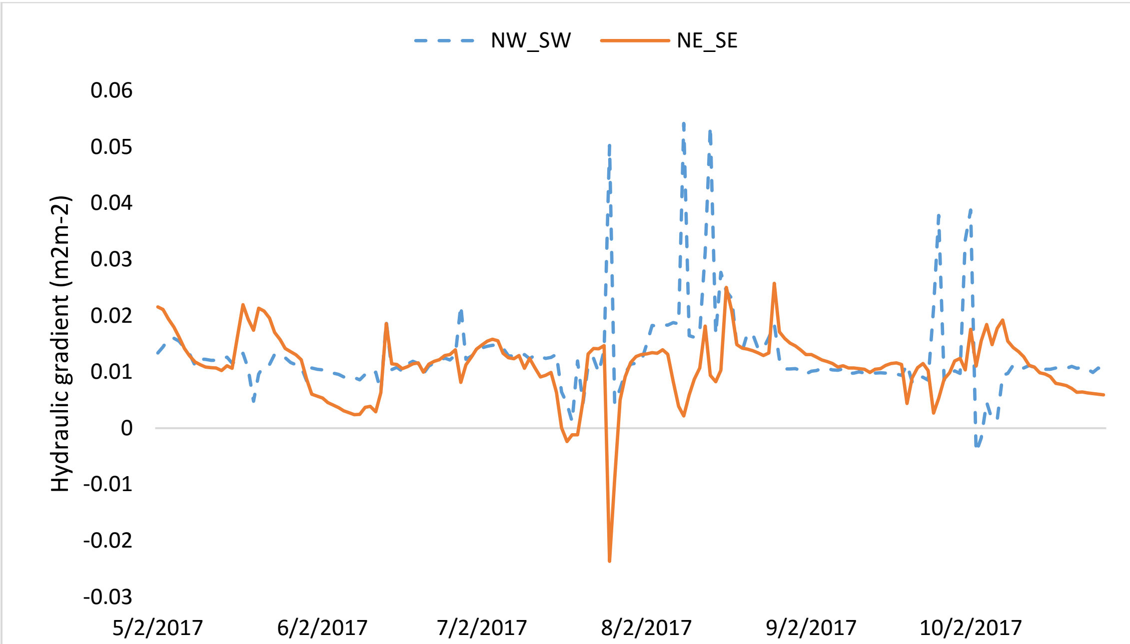


Figure 4.7 Hydraulic gradient (m^2m^{-2}) between wells B2B and B2A, and B1A and B1B during 2017 at the Baltic buffer site.

4.3.1.4.Input flow rate and volume

The mean flowrate to buffer zone tile for 2017 was $9.05\text{x}10^{-5}$ m^3sec^{-1} (0.0032cfs). Overall, 99.99% of the total flow from the control structure was diverted to the buffer zone, but no flow was observed in the control structure from mid-June to early July. High crop water requirements resulted in no tile drainage during this period, but after reaching maturity, frequent and intense precipitation events, especially during late September and early October, resulted in flow in the control structure (Figure 4.8). The peak discharge was found to be 0.0026 m^3sec^{-1} and occurred around October 3, 2017 following heavy precipitation. Overall, a total of 1506 m^3 (53,180ft^3) of water was diverted to the buffer zone during 2017. During the entire duration of study, there existed frequent dry and wet spells of water outflow to the buffer zone. Such fluctuating spells of increasing and decreasing soil moisture impacts denitrification dynamics within the soil profile (Aulakh and Rennie, 1987). Irrespective of the wet and dry spells, flow volumes fed to the buffer were relatively low, which lead to a longer residence time in the buffer zone and supports higher nitrate reductions rates (Lowrance et al., 2000b).

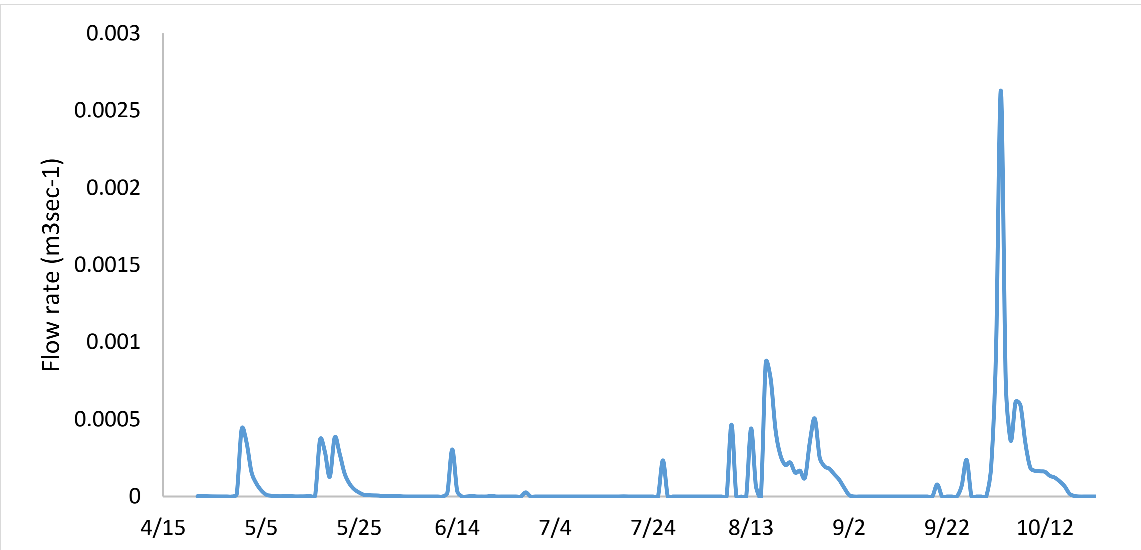


Figure 4.8 Flow rate through the control structure at Baltic buffer site for 2017 (m^3sec^{-1}).

4.3.1.5.Effect on water quality and performance evaluation

The mean reduction rate observed in the Baltic buffer was 96%. In addition, nitrate reduction was calculated along the length of the buffer and was similar for three out of four wells, B1B, B2A, and B2B. For B1A, a single monitoring event was responsible for lowering the overall reduction rate for the study period (Table 4.3). Overall, the reduction rates observed at each monitoring event were well within the annual reduction rate ranging from 48% to 100%, found in Dinnes et al., (2002).

In addition to the reduction rates, nitrate concentrations observed in the control structure were well above the EPA safe drinking water limit of 10 mg L^{-1}. However, none of the concentrations in the wells exceeded the safety limit.

The nitrate concentration data were tested for normality using the Shapiro-Wilk test ($p<0.05$) and observed to be non-normal. So, A non-parametric paired Wilcoxon test was used for the statistical analysis ($p<0.05$) to compare the inlet concentrations with the concentrations observed in wells. The mean inlet concentration was 48.64 mg L^{-1} and compared to 1.44 mg L^{-1} for all the wells. The nitrate concentrations observed at each well were significantly lower than the inlet concentrations (Figure 4.9).

Table 4.3 Average nitrate reduction rate observed at each of the 4 wells B1B, B2B, B2A and B1A.

Wells	B1B	B2B	B1A	B2A
Average percent nitrate reduction	98.5	96.8	88.5	97.8

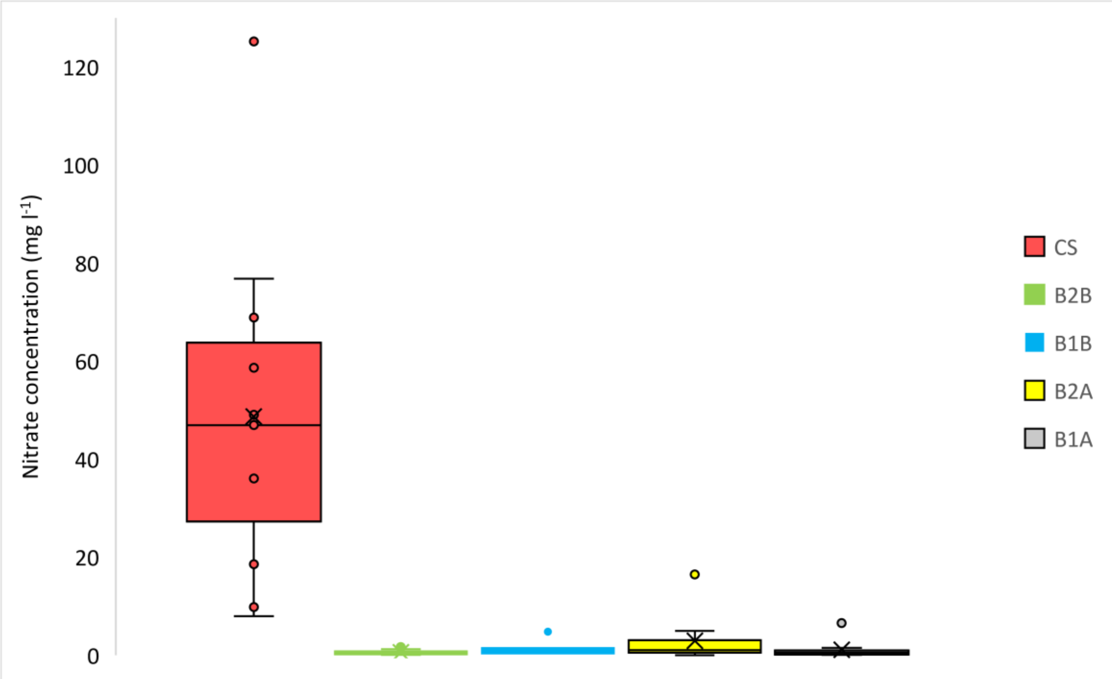


Figure 4.9 Box plots depicting nitrate concentrations observed throughout the year 2017 at the inlet control structure and 4 monitoring wells B1B, B2B, B1A and B2A.

A cumulative nitrate load of 44 kg (97 lbs.) of NO_3^- was fed to the buffer zone for 2017. Out of this, a total of 42 kg (92.7 lbs.), was removed by the buffer leaving 2 kg (4.3 lbs.) as a residual in the buffer zone. The buffer zone drained 6 ha (15 acres), resulting in a removal rate of 7 kg ha^{-1} (6.18 lbs. ac^{-1}).

Cost of system per pound removed

For Baltic, the total site installation cost was $2041.18 or $136 acre^{-1} (Table 4.4). Using the input cost per acre and the pounds per acre removed for the entire duration of study, it cost $22 to remove one pound of NO3-N using the buffer zone, which received nutrient rich water from an acre of cropped area. The working cost for the buffer is higher as compared to the results from previous studies which range from $ 0.5 to $ 4.6 per pound of nitrate removed (Utt et al., 2015). The major reason for a higher cost is the shorter monitoring period but, since the maintenance cost for the entire system would be negligible during future years, the operational cost would reduce after each year of implementation. Considering a 20-year implementation period, the cost per pound of nitrate could reduce to $1.1 per year assuming the same removal rate as observed for the site for 2017.

Table 4.4 Input costs for the buffer zone at Baltic, SD

Item	Cost (USD)
Cost of control structure	$1191
Installation costs	$500
Tile and fittings	$350
Total cost per acre	$136 acre^{-1}

4.3.2.Flandreau

4.3.2.1.Climate

Precipitation during fall 2016 for the site was greater compared to fall 2017. For 2017, May had the maximum precipitation, amounting to 105mm. In addition, fall 2016 was hotter than fall 2017 and with July having the highest mean temperature during the year at 22.3˚C. Overall, the annual rainfall for 2017 was less than the 30-year average rainfall. For 2016, the rainfall observed was comparable to the 30-year averages.

Table 4.5 Precipitation and temperatures recorded at the research site at Flandreau as compared to long term averages (1981-2010) for the closest USGS weather station (USC00392984).

Months	Precipitation (mm)			Temperature (˚C)		
	2016	2017	(1981-2010) 30 year average.	2016	2017	(1981-2010) 30 year average
January	-	1.2	12.1	-	-7.7	-10.1
February	-	11.0	13.9	-	-1.4	-7.4
March	-	1.6	36.2	-	0.0	-1.0
April	-	38.0	66.7	-	8.2	7.0
May	-	105.4	82.0	-	13.2	13.7
June	-	60.0	107.8	-	20.1	19.1
July	-	62.2	84.6	-	22.3	21.7
August	70.0	88.8	85.8	19.7	18.1	20.3
September	63.8	9.0	80.6	16.7	16.4	15.3
October	53.4	7.6	55.3	9.4	7.8	8.1
November	28.6	-	28.7	3.5	-0.9	-0.4
December	15.6	-	15.9	-8.3	-	-8.2

4.3.2.2.Soil Analysis

Soil organic matter

The mean organic matter observed for the site was 5.4%. The organic matter content for each well was greatest at the topmost layer and decreased with increasing in depth. This can be related to the enrichment or melanization process (mixing of decomposed plant residue with soil) at the soil surface. Wells F3A and F3B had the lowest organic matter in the 12 – 24 in and 24 – 36 in horizon as compared to other wells (Figure 4.10). The SOC levels for the three soil layers were computed as 4.3%, 2.8% and 2.2% using the Van Bemmelen factor (Van Bemmelen, 1890) which was higher than the threshold value of 2% discussed in Burford and Bremner (1975). The relationship between SOC and denitrification potential was developed in the 1960's and higher SOC favors greater denitrification potential (Bremner and Shaw, 1958; McGarity, 1961).

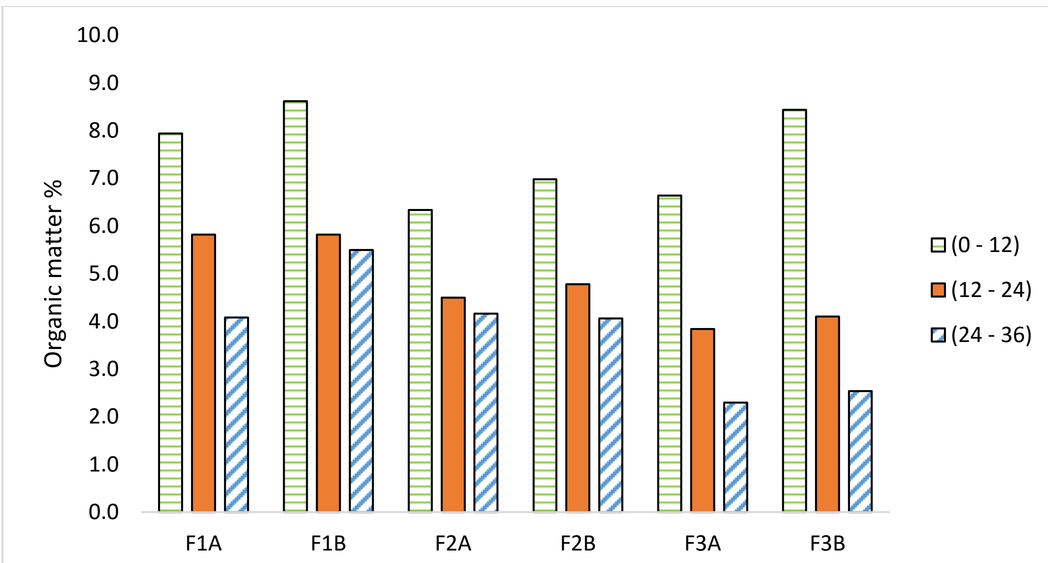


Figure 4.10 Organic matter (%) distribution at three depths 0-12in, 12-24in and 24-36in for sampling points near each monitoring well namely F1A, F1B, F2A, F2B, F3A and F3B located at the buffer zone at Flandreau, SD.

Soil NO3-N

The mean soil NO3-N at the site was 7.1 ppm. The highest concentrations were observed for the 0 – 12 in horizon for wells F1A and F1B. The concentration decreased with increasing in depth in the majority of sites with a slight variation in magnitude. For wells F2A and F2B, NO3-N concentrations were lower than the other two well pairs. The analysis also indicated higher mineralization in the upper layers around wells F1A and F1B as compared to the rest of the wells (Figure 4.11).

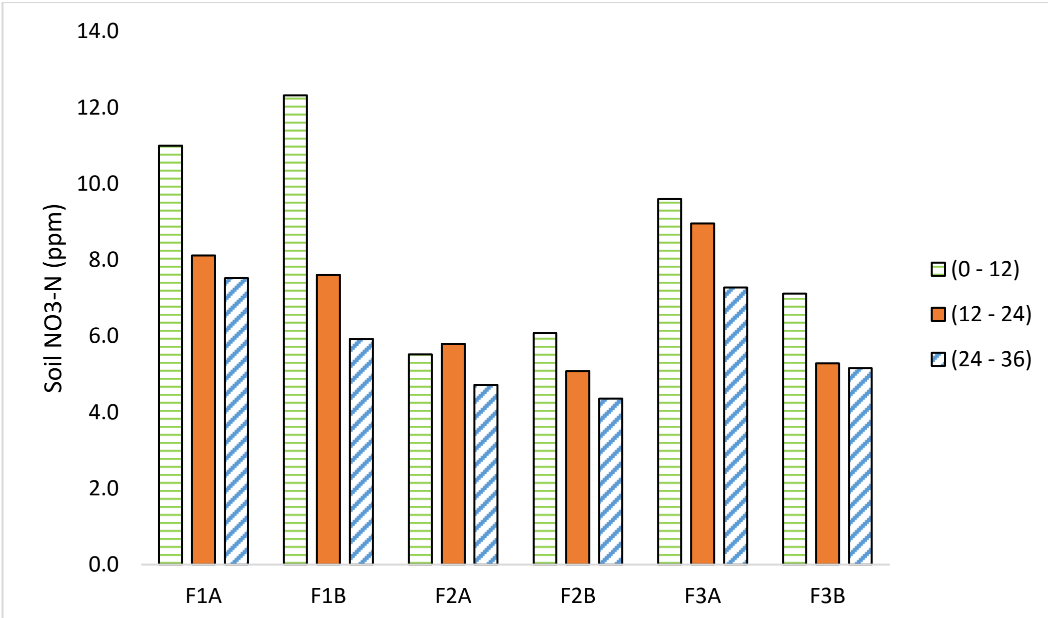


Figure 4.11 Soil NO3-N (ppm) distribution at three depths 0-12in, 12-24in and 24-36in for sampling points near each monitoring well namely F1A, F1B, F2A, F2B, F3A and F3B located at the buffer zone at Flandreau, SD.

Soil P

Soil P was higher for wells F1A and F1B and mostly decreased with increasing depth except well F1B (Figure 4.12). Mean Soil P for each well location was computed to be 16.3, 43.9, 3.2, 4.16, 2.76 and 5 ppm. Phosphorus solubility in soils increases with an increase in pH over 7.5 (Olsen, 1954). The lower values for wells F2A, F2B, F3A and F3B in comparison with F1A and F1B suggest increased solubility of phosphorus and subsequent vertical or horizontal movement around these wells.

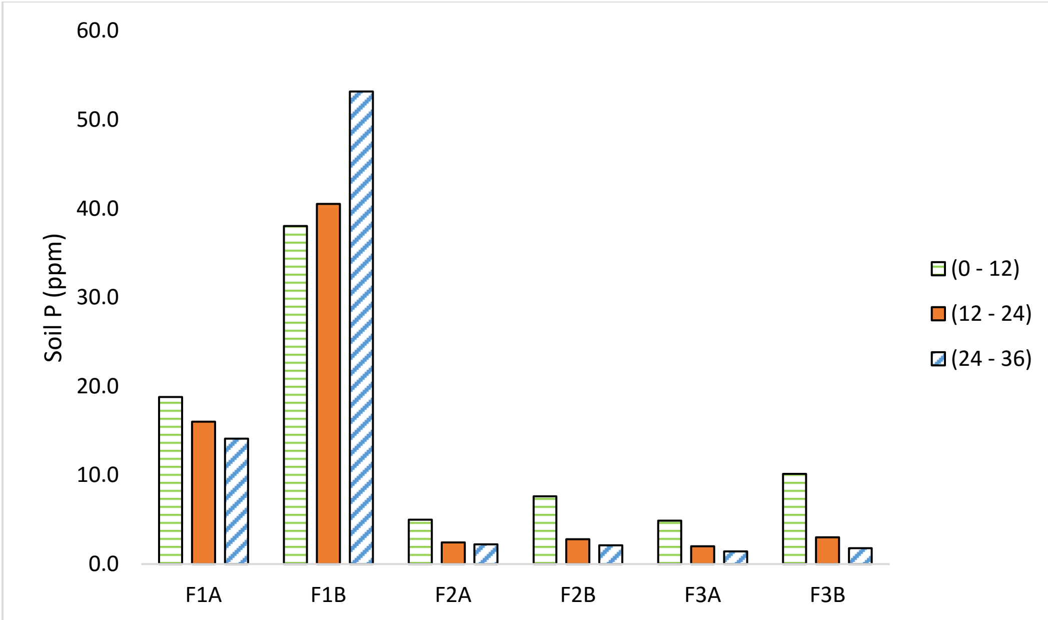


Figure 4.12 Soil P (ppm) distribution at three depths 0-12in, 12-24in and 24-36in for sampling points near each monitoring well namely F1A, F1B, F2A, F2B, F3A and F3B located at the buffer zone at Flandreau, SD.

Soil pH

Soil pH values were observed to be above 7 for all the samples at various depths for the site. The mean pH for the entire site was 7.9. Alkaline soils have also been found to optimize the growth of denitrifying bacteria which affects availability of carbon for denitrification (Bremner and Shaw, 1958) and supports the rapid reduction of N_2O to N_2 during the denitrification reaction (Koskinen and Keeney, 1982). In general, there were no trends observed for soil pH values with depth. Wells F3A and F3B had an increase in pH at lower depths which supports leaching of basic cations from the surface to lower layers.

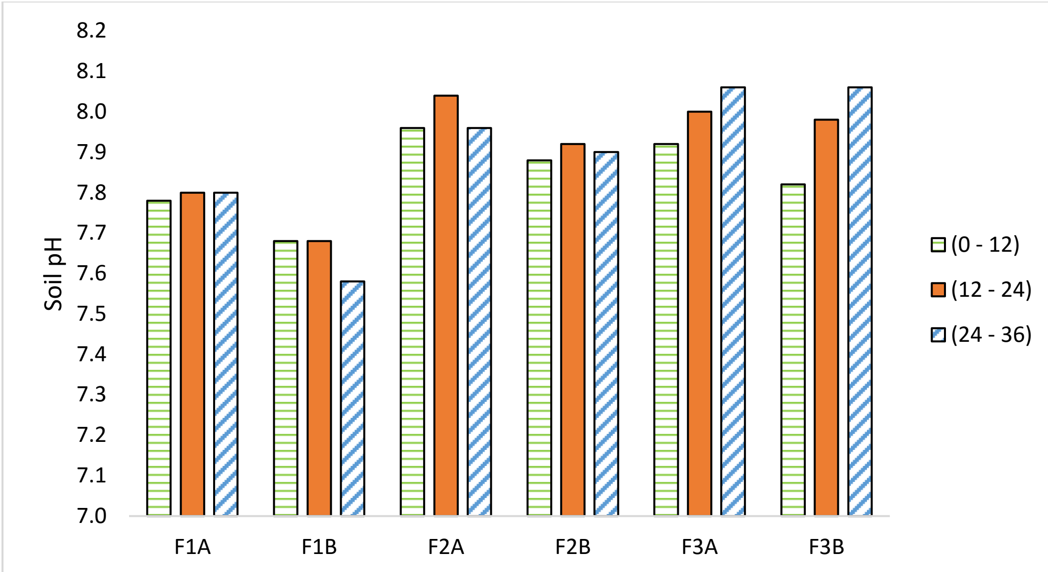


Figure 4.13 Soil pH distribution at three depths 0-12in, 12-24in and 24-36in for sampling points near each monitoring well namely F1A, F1B, F2A, F2B, F3A and F3B located at the buffer zone at Flandreau, SD.

4.3.2.3.Conditions inside the buffer zone

Shallow Groundwater Table

The mean water table depth inside the buffer zone ranged from 0.43 m to 1.11 m for 2016. The site had a deep water table just after the installation period (late August, 2016), but rose as a more water was diverted from the field to the buffer zone.

For 2017, the mean water level ranged from 0.38 m to 1.33 m. The water level dropped during drier periods, such as July and August in 2017, as more water was used by the crop, but the levels rose again around September and continued to rise until late October (Figure 4.14 (a) and Figure 4.14 (b)). Fluctuations in the water table were consistently observed before and after storm events during the study period. The site exhibited a shallow water table which is important development of anaerobic conditions which further support denitrification (Bremner and Shaw, 1958; Hofstra and Bouwman, 2005).

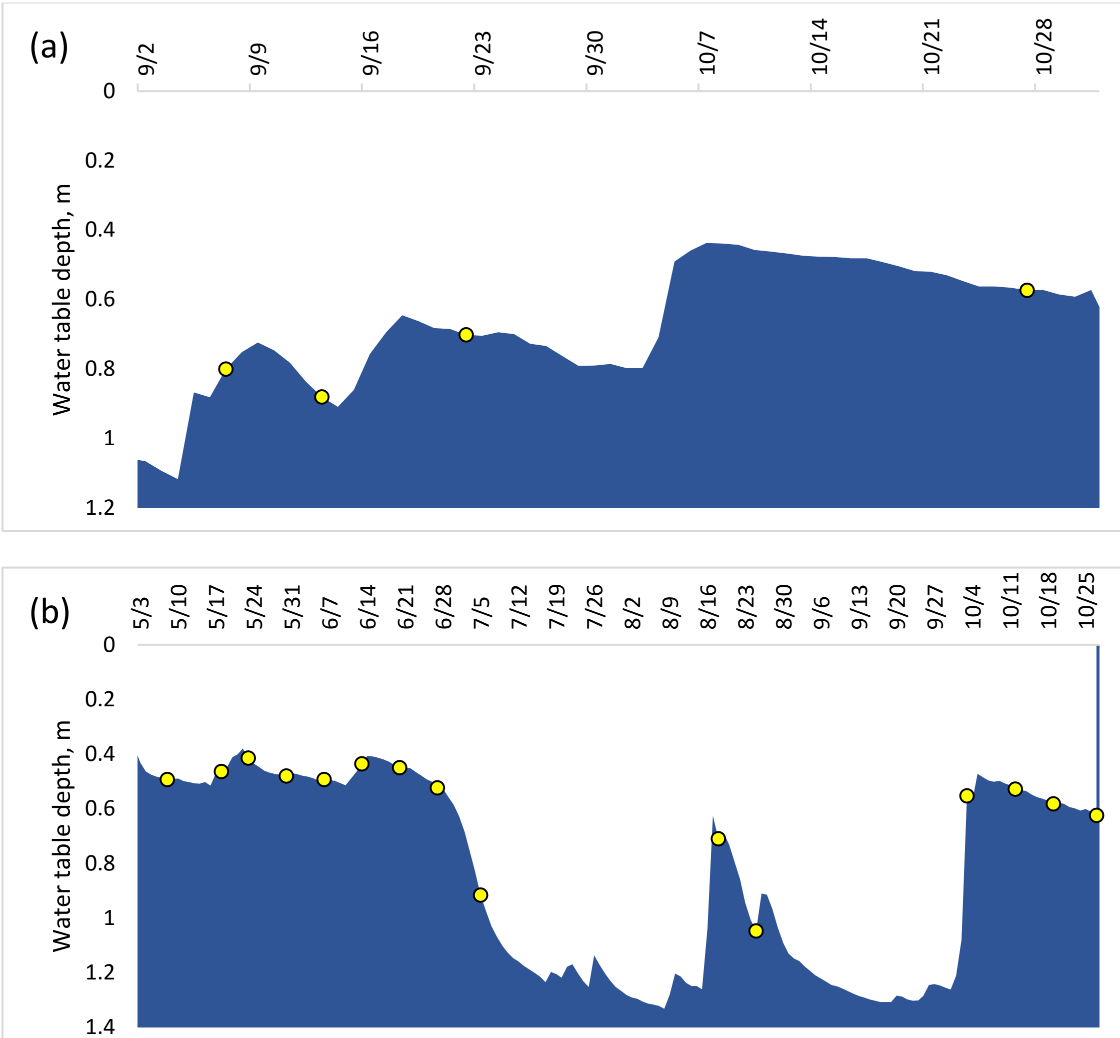


Figure 4.14 Water table depth (m) below the soil surface and days of water sampling from the monitoring wells at Flandreau during 2016 (a) and 2017 (b).

Hydraulic gradient between the well transects

To study flow direction between each well transect, the daily hydraulic gradient between the well transects was computed. For drier conditions inside the buffer, a smaller gradient existed between all the three pairs. However, this changed when the input flow volume to the buffer zone increased, resulting in a greater gradient between each well pair (Figure 4.14). 2017 started with a higher gradient between the wells, but rapidly decreased to a lower value and increased again during the high flow events (Figure 4.15). Overall, wells F3A and F3B had a higher gradient as compared to the other well pairs. This may be due to a greater head difference between the two wells.

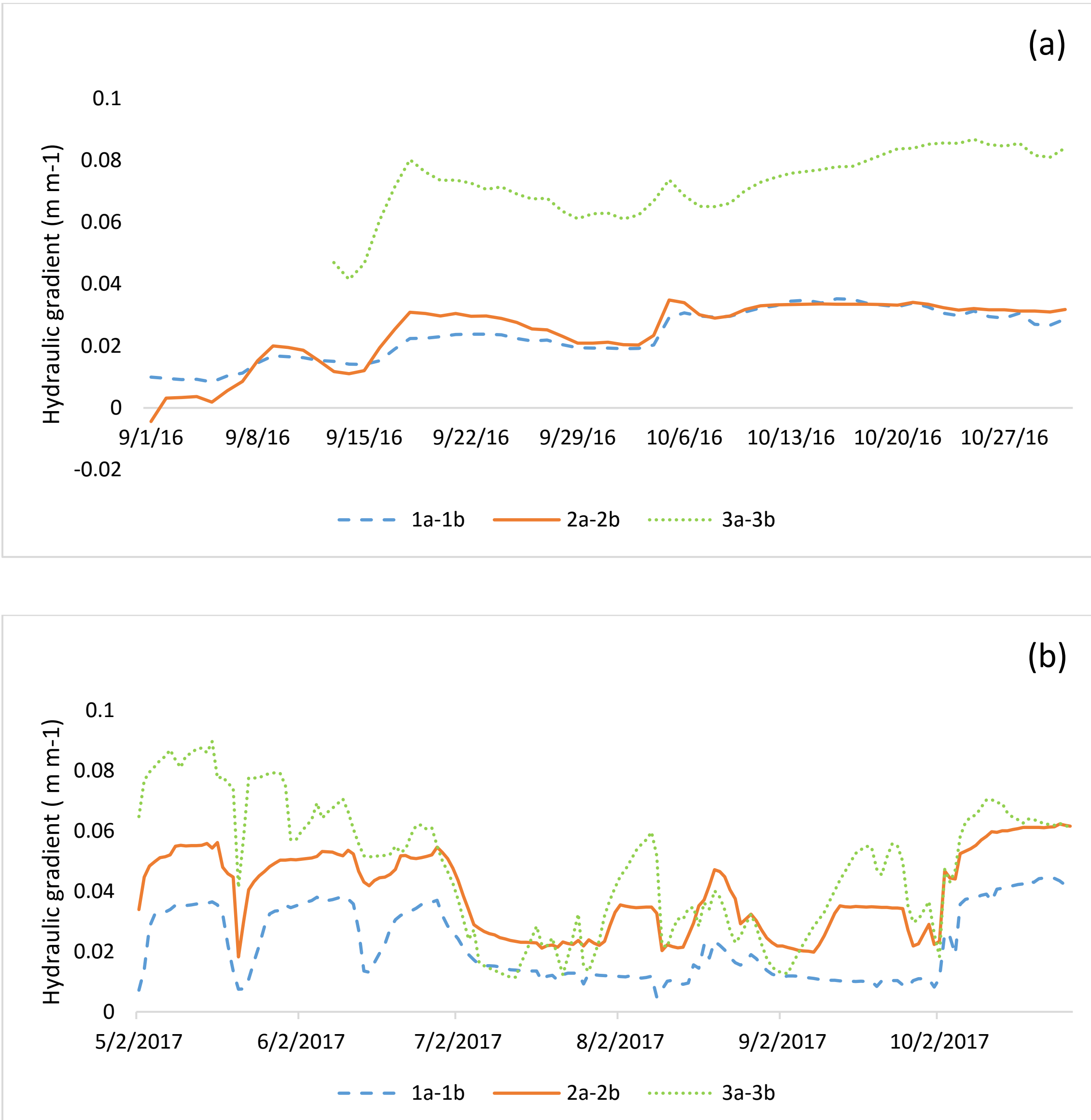
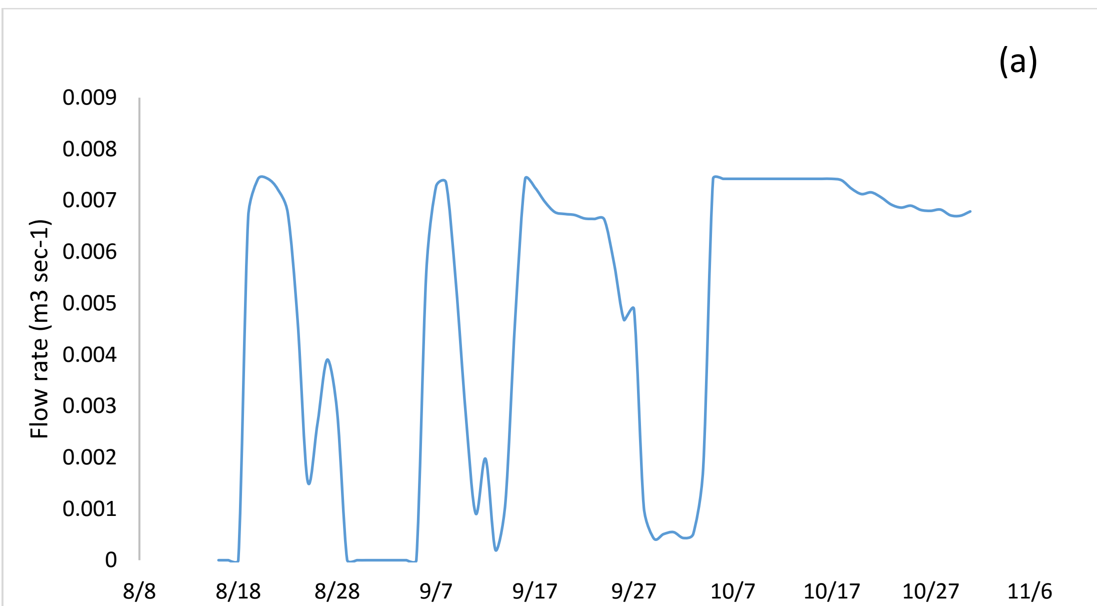


Figure 4.15 Hydraulic gradient between each well pair, F1A and F1B (dashed), F2A and F2B (solid), and F3A and F3B (dotted) for 2016 (a) and 2017 (b).

4.3.2.4.Input flow rate and volume

Decagon CTD-10 sensors recorded water level in the control structure from June 2016 to October 2016 and from May 2017 to October 2017. The mean tile flowrate was

0.0053 m^3sec^{-1} (0.19 ft^3sec^{-1}) for 2016 and 0.0062 m^3sec^{-1} (0.22 ft^3sec^{-1}) for 2017. Higher tile drainage flowrates occurred during spring and early summer. The rate of drain flow was maximum during the early summer period and then declined during crop growth. Duration and intensity of storm events during the cropping season affected the volume of water that was fed to the buffer zone. Overall, the buffer zone was fed a cumulative volume of 31,515m^3 for 2016 and 61,044 m^3 for 2017 (Figures 4.16). 97% of the flow was diverted to the buffer during 2016, but higher precipitation events during 2017 reduced the buffer volume percentage to 82.7%. The water table depth was not manipulated using the control structure as opposed to Jaynes and Isenhart (2014a), where 50% flow was diverted leading to the buffer and led to a 100% nitrate reduction rate.



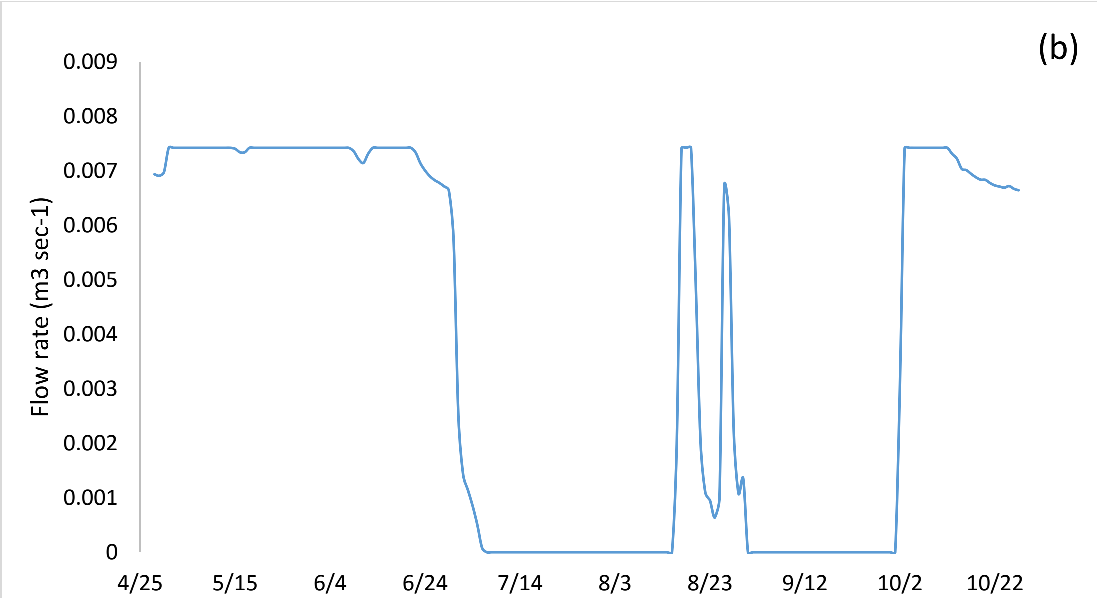


Figure 4.16 Flow rate (m^3sec^{-1}) observed at the control structure during 2016 (a) and 2017 (b).

4.3.2.5. Effect on water quality

The mean nitrate reduction rate at the buffer site was 86% for 2016 and 65% for 2017 (Table 4.6). Nitrate reduction was also computed along the length of the buffer and was lowest for the first pair of wells. The reduction rate was greatest for the second well pair, F2A and F2B. A lower reduction rate observed at F3A could be explained by lower OM content as compared to the other well transects and higher hydraulic gradient leading to greater lateral flow between F3A and F3B. So, the reduction observed here, might have been due to plant uptake. The lower reduction rate observed at wells F1A and F1B can be related to high flow volumes being diverted to the buffer zone not allowing adequate time for denitrification and plant uptake (Lowrance et al., 2000b). It is recommended to study the soil textural properties around each well location to understand the water dynamics within the buffer zone.

Table 4.6 Percent nitrate removed observed at each monitoring well (F1A, F1B, F2A, F2B, F3A and F3B) at the buffer zone Flandreau, SD throughout the study period.

Wells	F1A	F1B	F2A	F2B	F3A	F3B
Percent average nitrate reduced	55.81	53.70	82.37	68.19	54.19	77.53

The mean nitrate concentration observed at the control structure was 31.6 mg L^{-1} while the mean nitrate concentration for the wells ranged from 6.1 mg L^{-1} to 16.5 mg L^{-1}. The concentration data were analyzed for normality using the Shapiro-Wilk test ($p<0.05$) and found to be non-normal. So, the statistical analysis was done using a non-parametric paired Wilcoxon test ($p<0.05$).The nitrate concentration at each well was found to be significantly lower than the inlet nitrate concentrations.

Furthermore, the mean concentration for the wells throughout the entire study period was 12.33 mg L^{-1}, which was higher than the EPA designated safe drinking water limit of 10 mg L^{-1} (Figure 4.17). Therefore, though significant reductions in nitrate concentrations were observed due to the practice, additional treatment was required to achieve nitrate concentration levels below the EPA drinking water limit.

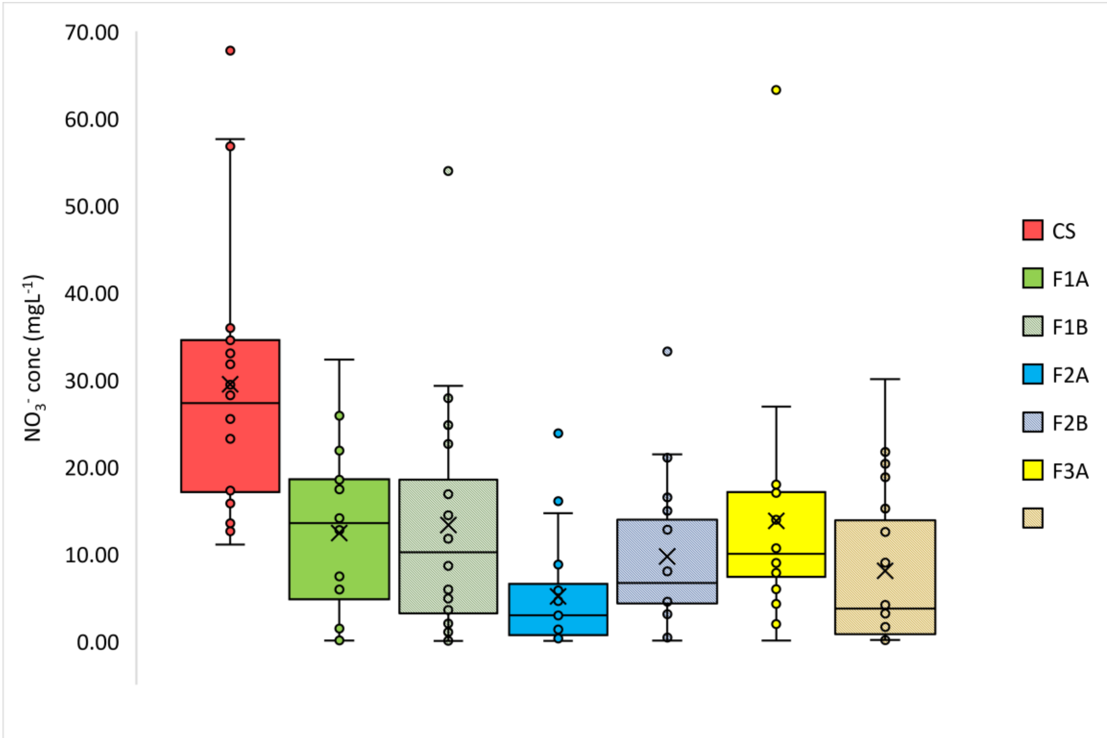


Figure 4.17 Box plots showing nitrate concentrations(mg L^{-1}) observed across the buffer site, from control structure (CS) to all the monitoring wells F1A, F1B, F2A, F2B, F3A and F3B throughout the study period The soil boxes represent the wells near the tile line and the striped boxes represent the wells near the outlet stream.

4.3.2.6.Performance evaluation

A load duration curve for the buffer zone at Flandreau showed high nitrate loads at higher flows (Figure 4.18). The loads observed during dry and mid-range conditions were near the nitrate load pertaining to the acceptable drinking water limit of 10 mg L^{-1} nitrate concentration, but during moist and high flow conditions were higher and show a reduced nitrate removal efficiency of the buffer zone at higher flow volumes. Six out of eight instances under high flow and moist conditions needed additional reduction in nitrates to reach the drinking water quality target of 10 mg L^{-1}. Higher flow volumes fed to the buffer zone did not allow adequate plant uptake and natural denitrification to facilitate N attenuation (Lowrance et al., 2000b).

Overall, the buffer zone near Flandreau drained around 35 ha (87 acres). For 2016, it was fed with a total of 635kg (1378 lbs.) of nitrate and the buffer removed 534 kg (1177 lbs.). For 2017, a total of 2105 kg (4640 lbs.) of nitrate was fed to the buffer zone and the buffer removed 1300 kg (2867 lbs.). The total removal for the two years of study amounted at 52.08 kg ha^{-1} (46.47 $lbs.ac^{-1}$).

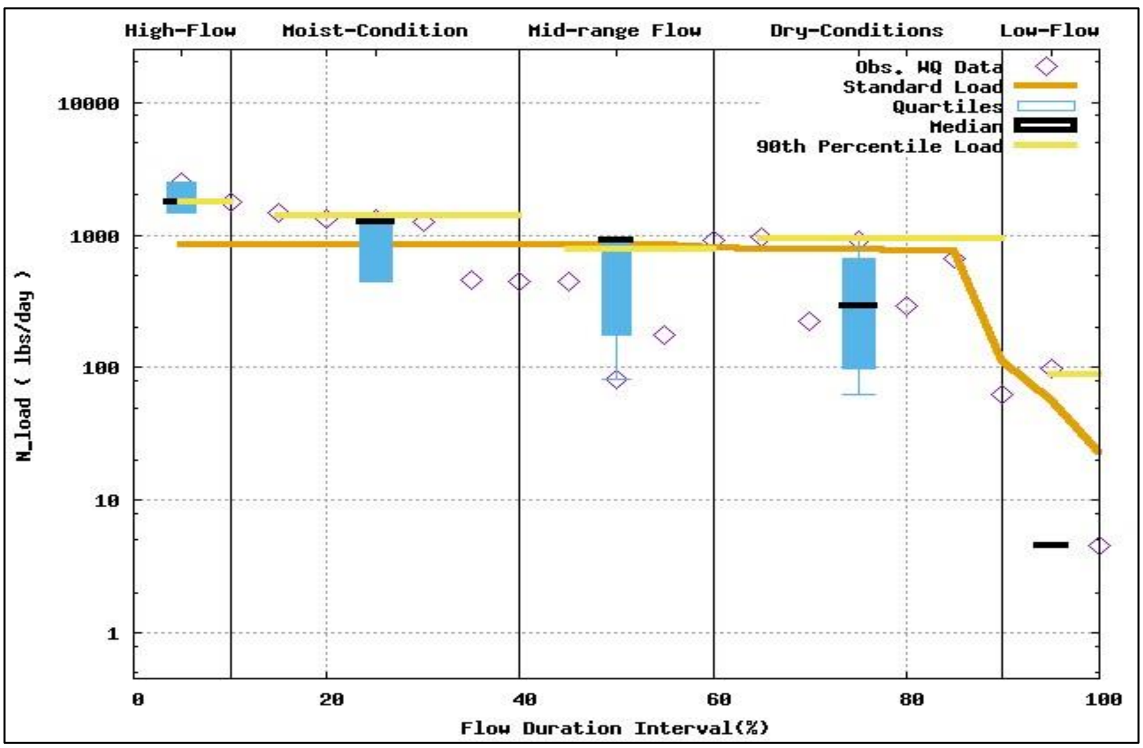


Figure 4.18 Load duration curve for Flandreau, SD. LDC compared the measured nutrient load with the standard load for different flow conditions. Higher nitrate load was observed during high flow and moist conditions as compared to the load pertaining to 10 mg L^{-1} (EPA safe drinking water limit).

<u>*Cost of systems per pound removed*</u>

The cost for setting up the site was $2500. The area drained was around 87 acres. The cost per pound of nitrate removed was computed to be $0.6 for the two years under observation. The working cost of the system was within the $ 0.55 – $ 4.64 range that was observed by Utt et al., (2015). Similar to the Baltic site, a 20-year implementation for the Flandreau buffer zone could further reduce the cost per pound of nitrate removed to $0.06 per year.

Table 4.7 Input costs for the buffer zone at Flandreau, SD

Item	Cost (USD)
Control Structure	$1,191.18
Tile & fittings	$591.60
Installation of structure & tile	$718.00
Total	$2,500.78
Total cost per acre	\$28.74 acre^{-1}

4.4.Discussion

The two major conditions required for denitrification in soil profile are presence of SOC acting as the energy source and the potential for a high-water table, enabling anaerobic conditions to readily develop and facilitate denitrification. Soil properties such as SOM, SOC, pH, NO3 have a direct or indirect impact on those conditions. Bremner and Shaw (1958) studied the impact of total, water soluble and decomposable organic matter on the denitrification potential of a soil and found significant correlation between denitrification capacity and total organic carbon. It was suggested that a measure of mineralizable carbon be used as an index for studying denitrification capacity of nitrate in soils. The OM pertaining to both the sites was within the desired range to sustain denitrification process as suggested by Utt et al., (2015).

Soil testing for both the sites was initiated and showed a mean SOC of 2.5% and 3.1% for Baltic and Flandreau respectively. In addition, soil pH for both the sites was above 7 during soil analysis; a slightly alkaline or neutral pH has been shown to favor the denitrification enzyme activity facilitating denitrification and a rapid conversion of N_2O to N_2 as compared to acidic soils(ŠImek and Cooper, 2002).

Table 4.8 Descriptive statistics involving comparison of soil properties such as N, P, K, EC and OM for the two sites under consideration.

Soil parameter	Sampling depth	Flandreau			Baltic		
		Minimum	Mean	Maximum	Minimum	Mean	Maximum
OM (%)	(0-30)	5.3	7.5	9.9	3.6	4.3	5.9
	(30-60)	3.5	4.8	6.5	3.7	4.5	5.5
	(60-90)	2.1	3.8	5.6	3.8	4.4	4.9
NO3-N (ppm)	(0-30)	4.4	8.6	15.2	2.8	4.6	7.0
	(30-60)	4.2	6.8	11.0	3.4	7.1	11.4
	(60-90)	4.0	5.8	8.8	1.8	5.1	11.2
Olsen P (ppm)	(0-30)	2.8	14.1	40.9	25.1	38.7	60.1
	(30-60)	1.7	11.1	49.7	18.6	41.7	84.6
	(60-90)	1.1	12.5	69.0	10.8	21.7	69.0
K (ppm)	(0-30)	163.0	191.6	219.0	152.0	253.3	387.0
	(30-60)	133.0	164.1	203.0	149.0	189.5	277.0
	(60-90)	89.0	157.6	200.0	124.0	163.3	247.0
pH	(0-30)	7.6	7.8	8.0	7.6	7.8	7.9
	(30-60)	7.6	7.9	8.1	7.5	7.7	7.9
	(60-90)	7.5	7.9	8.1	7.6	7.7	7.8
EC	(0-30)	0.5	0.9	1.0	0.7	0.8	1.0
	(30-60)	0.3	0.8	1.0	0.4	0.8	1.0
	(60-90)	0.3	0.8	1.0	0.3	0.8	1.0

Conditions inside the buffer were studied using the flow volumes fed to the buffer, water level in the buffer zone, and hydraulic gradient between the wells. The buffer at Baltic was had an average flow rate of 0.0032 ft^3sec^{-1} as compared to 0.2 ft^3sec^{-1} for Flandreau and can be related to the different acreage drained by both the sites.

Table 4.9 Comparison of tile flow rate (L min^{-1}) fed to the buffer zone for Baltic and Flandreau.

Flow rate ($Lmin^{-1}$)	Flandreau	Baltic
Minimum	0.00	0.0
Median	399.8	0.0
Mean	247.2	5.5
Maximum	445.5	157.8

Flandreau showed a consistent shallow water table throughout the summer months of May-June and in fall for October 2017 as compared to cycles of high and low water table observed at Baltic for the same period. Higher flow volumes diverted to the buffer zone near Flandreau did not allow for sufficient retention time and had a detrimental effect on buffer performance (Dinnes et al., 2002; Lowrance et al., 2000b).

Table 4.10 Descriptive statistics for water table depth, m for the buffer zones at Flandreau and Baltic.

Water table depth (m)	Flandreau	Baltic
Minimum	0.38	0.40
Median	0.69	0.97
Mean	0.80	0.96
Maximum	1.33	1.38

Furthermore, denitrification hysteresis caused during wet and dry cycles was also more profoundly seen at Flandreau than at Baltic. Higher retention time resulted in consistent nitrate reduction results for Baltic throughout the season. For Flandreau, higher reduction rates were observed for the rewetting period and lower during the drying period. For example, a 97% reduction was observed during the rewetting phase around October 3, 2017, whereas lower rates were observed during the sampling time around the drying period; a 70%, 51%, 69%, 58%, 52% and 47% reduction during July 5, 2017 until October 27, 2017 when conditions beneath the surface experienced subsequent drying.

In addition, the performance of each system was compared using average nitrate reduction rate, pounds of nitrate removed, and cost of system per pound of nitrate removed. The reduction rates for both the systems were 95% for Baltic for 2017 and 85% and 65% for Flandreau for 2016 and 2017, respectively. These values were well within the observed reduction rates in numerous studies which have ranged from 48% to 100% (Jaynes and Isenhart, 2014b; Utt et al., 2015). The difference in nitrate concentrations observed between control structure and at monitoring wells was significant ($p<0.05$) for both the sites. The lower reduction rate at Flandreau was also evaluated using a load duration curve and it was found that the buffer was not able to adequately reduce nitrate concentrations for high flow and moist conditions, but the cumulative load reduced per acre drained was greater for Flandreau at 46.47 $lbs.ac^{-1}$ as compared to 6.18 $lbs.ac^{-1}$ for Baltic which contributed to a lower cost of $0.6 per pound of nitrate removed for Flandreau as compared to $22 for the buffer at Baltic. Overall, the sites cannot be compared with each other in terms of cost of per pound of nitrate removed as the results from Baltic were obtained after one year of observation and that for Flandreau were obtained after two years of monitoring. For both the sites, the area designated to the buffer zone was not taken out of production, thus minimizing the input costs. In addition, there was no maintenance performed throughout the period of study. This supports for a further decrease in operational costs after each year of implementation.

4.5.Conclusions

Nitrate removal efficiency for saturated buffers was dependent upon the input flow volume fed to the buffer zone similar to previous findings (Jaynes and Isenhart, 2014a; Utt et al., 2015). Another factor that affected buffer performance was the denitrification hysteresis during the wet and dry cycles existing throughout the season. The performance for Flandreau was impacted to a larger extent as it had a profound drying and rewetting cycle occurring during 2017(Austin et al., 2004; Groffman and Tiedje, 1988).

The difference in nitrate reduction rates along length of the buffer was not significant for both the sites and an additional 42% reduction was required for the site at Flandreau to meet the water quality target of 10 mg L^{-1}. Overall, the practice was successful on both the sites in significantly reducing the nitrate content from tile outflow from the field.

However, the cost of removal of a pound of nitrate per acre drained was higher for Baltic at $22 as compared to a mere $0.6 for Flandreau. The difference in the costs can be explained by a larger area draining water into the buffer resulting in a lower input cost per acre for Flandreau. In addition, the cumulative load removed for Flandreau was computed for two years as compared to a single year for Baltic. Also, larger and continuous flow volumes were being fed to the buffer zone at Flandreau, resulting in a larger cumulative load removal and decreasing the removal cost.

4.6.Limitations and Recommendations for future work

The study only looked at impact of buffers on nitrate removal, but after soil P analysis indicating potential P transport, it is recommended to study the fate of dissolved reactive phosphorus within and outside the buffer zone.

The study also did not look upon the streambank stability during the operation period of the practice. Use of tile line close to a water body can lead to bank instability or sloughing. This is an important factor that could affect the performance of the entire system.

Further, use of modeling can help in understanding the nutrient dynamics in the buffer zone. Commercially available models such as Hydrus could be used to simulate the buffer sites and have a better understanding of the entire system on a field scale, while using a basin scale model, like SWAT, could help in analyzing the impact of saturated buffers on a bigger spatial scale.

Saturated buffers can also be used in conjugation with controlled drainage and can reduce nitrate transport via a combination of reduced outflow and subsequent nitrate removal from tile drainage water.

Chapter 5. CONCLUSIONS AND FUTURE WORK RECOMMENDATION

5.1. Conclusions

This study demonstrated the impact of drainage water management and saturated buffers on water quality at field sites across eastern South Dakota. A paired field approach was used to compare DWM and conventional drainage by considering the total reductions in water outflow and nitrate loads per acre. The difference between cumulative flows for conventional and DWM was 8 mm for 2016 and 6 mm for 2017. The mean reduction percentage in outflow was 31% for both years of the study. It was observed that 2016 and 2017 had similar annual rainfall amounting to 595 mm and 555 mm, respectively. The annual tile flow was 5.2% and 3% of the total rainfall for the conventional half for 2016 and 2017, respectively, and was lower for the DWM half with annual tile flow at 3.7% and 1.8% of the total rainfall for 2016 and 2017, respectively. Seasonal variability in rainfall had an impact on tile flows from the site for both years. In addition, soil variability had an impact between the two halves as the flow measured per acre differed during 2017 which was entirely under the free drainage period. Overall, the total nutrient reduction for the site amounted to 40% and 26% for 2016 and 2017, respectively. The average nitrate concentrations were greater than 10 mg L^{-1} for both the halves of the research site. Nitrate concentrations were lower for the DWM half as compared to conventional half suggesting plant uptake during the managed period. Soil testing for the site revealed higher bulk density for the eastern half as compared to the western half. The soil nitrate was higher for the conventional half as compared to DWM half for all the depths sampled. Texture analysis of the soil revealed that the soil on the site had little variation and was largely classified as clay loam with some pockets classified loam and sandy clay loam. The OM throughout the site was observed to be low indicating poor aggregate properties, susceptibility to erosion, and low potential for denitrification.

Furthermore, to study the impact of soil variability and management schedules on field hydrology, a SWAT+ project was developed for each half of the field (conventionally drained and DWM) and calibrated and validated using the daily tile flow values measured for the field study. The NSE ranged from 0.54 to 0.84, PBIAS from -23% to 61%, and RSR from 0.40 to 0.68. The values were largely within the satisfactory ranges.

Testing the impact of different management schedules, Conv, MG1 and MG2, on field hydrology revealed that the performance of each practice was dependent on precipitation patterns that existed over the entire year. MG1 reported lower tile flow and higher crop yields than the other management schedules for average and wet years. For dry years, MG2 had more tile flow and crop yield due to maximum soil storage as compared to the other two management schedules and, thus, had greater tile flow and crop yield.

For saturated buffers, two field scale sites were installed and monitored for 2016 and 2017. The average nitrate reduction rates observed were 95% (2017) for the Baltic site and 86% (2016) and 65% (2017) for the Flandreau site. Soil analyses from the sites show an ample amount of SOC (>2%) present for both sites to support denitrification. The mean flowrate for the two sites were substantially different with the Baltic buffer receiving 0.0032 ft^3sec^{-1} as compared to 0.2 ft^3sec^{-1} that was fed to the buffer zone at Flandreau. A lower flow rate led to an increased retention time which can explain the 95% reduction rate observed at the buffer zone for 2017. For Flandreau, a load duration curve showed that the buffer was not adequately able to reduce nitrogen for moist and high flow conditions and required additional reduction to meet the water quality target for nitrate of 10 mg L^{-1}. Overall, the reduction rates for both the sites depended less on the input nitrate concentration and more on the flow volumes fed to the buffer.

A cost comparison for a pound of nitrate removed per acre for the three sites and two management practices showed the lowest cost of $0.6 for Flandreau, followed by $22 for the buffer site at Baltic, and finally $28 for the DWM site at Alexandria, SD. There were no maintenance costs involved for all the three sites. In addition, the area adopted for buffer zones was not taken out of the main cropland which kept the initial setup costs at a lower end. For all the three sites, installation of control structures was the major investment. The lowest cost for buffer zone at Flandreau was due to the largest cumulative nitrate load removed amongst the three sites. Baltic had considerably higher cost which was a result of a shorter period of observation, one year for the site, and lower annual nitrate loads reduced by the buffer. For DWM, two dry years did not generate enough tile flow for considerable reductions in water outflow, resulting in little reduction in the pounds of nitrate removed and, therefore, the higher cost of the system per pound of nitrate removed.

5.2. Study limitations and recommendations for future work

- Effect of DWM on crop yield needs to be studied on the field scale which requires long term observations and sampling.
- CTD 10 sensors were taken out during the winter months. Some flow during spring was unaccounted for during both the years of study.
- Modeling should be expanded to a watershed scale to promote decision making for agricultural water management on a large scale.

- Buffer hydrology and Nitrate dynamics for each buffer site should be further studied to understand the inherent processes taking place in the buffer zone.

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