VEGETATIVE FILTER STRIP: A BEST MANAGEMENT PRACTICE (BMP) FOR FEEDLOT RUNOFF POLLUTION CONTROL IN NORTH DAKOTA

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The Supervisory Committee certifies that this ***disquisition*** complies with North Dakota State University's regulations and meets the accepted standards for the degree of

DOCTOR OF PHILOSOPHY

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ABSTRACT

Runoff from animal feeding operations is a major source of water pollution. Vegetative filter strips (VFS) are effective ways to reduce nonpoint source pollution. In this study, vegetative filter strips with different designs and in climatic and management conditions of North Dakota were evaluated. Runoff samples were collected from inflow (before entering VFS) and outflow (after exiting the VFS) locations using automatic samplers. Collected samples were analyzed for solids and nutrients. It was observed that the transport reductions by VFS were ranged from very low to up to 100%. However, soluble nutrients were not as effectively removed as sediment and sediment bound nutrients. Filter with longer length was more effective in reducing transport of sediments and nutrients. Antecedent soil moisture condition had an important effect on VFS performance.

An attempt was made by varying the VFS soil pH in a broader range to investigate effect of pH on reducing transport of soluble nutrients from manure borne runoff. Soil was treated with calcium carbonate to adjust pH at different levels. Treated soil was packed into galvanized iron boxes and seeded with grasses to simulate vegetative filter strips. Runoff experiments were conducted with manure solution and inflow, outflow, and leachate samples were collected. Samples were analyzed for sediment and nutrients. It was observed that the soluble nutrients transport was influenced by the pH, and higher ortho-P transport reduction was observed in higher pH. Leaching of NO_3-N at higher pH was observed, indicating potential of groundwater pollution from the soil with higher pH. Using calcium carbonate to increase soil pH and thereby reducing transport of soluble nutrient could increase VFS performance.

To aid VFS design and evaluation, a model was developed to predict trapping efficiency of sediment, sediment bound P, and dissolved P from VFS. Two procedures were coded into

FORTRAN and added into existing VFSMOD model. The model was calibrated and validated using field data. Due to limited data points and difficulties in measuring runoff volume, the model appeared to be under or over predicting. In future, model predictability can be improved by accurately measuring runoff volume and carefully selecting input parameters.

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CHAPTER 1. GENERAL INTRODUCTION

Animal agriculture contributes large portions of surface water pollution through runoff. For example, open animal feeding operations and land application of manure are major sources of pollutant runoff which accounts for non-point source (NPS) pollution. Primary pollutants associated with runoff from concentrated animal feeding operations and surface application of manure include sediment, nutrients (e.g., nitrogen, phosphorous), and microbes (Westerman et al., 1980; McLeod and Hegg, 1984; Edward et al., 1983). After large rainfall events and snow melt, runoff from upslope areas moves into nearby water bodies and causes pollution. Nutrient-laden water that enters into surface water is responsible for eutrophication, a condition that decreases dissolved oxygen and kills aquatic animals. In addition, increased bacterial population, change in water color, and development of odor might cause loss of recreational value of surface water.

As the pollution concern increases, animal production is facing various federal and state regulations to minimize pollution. However, effluent discharge and pollution elimination guidelines required by the state and federal regulations are sometimes not cost effective (Young et al., 1980). For example, containment structures are recommended for pollution control, but they are expensive to construct and may contaminate groundwater. Therefore, cost effective pollution mitigation methods and management practices are of primary concern to the producers. The concentrated animal feeding operations (CAFO) regulation rules, effective from 15 April 2003, have made a provision to use alternative runoff mitigation measures other than the containment structures provided that the alternative runoff control measure discharges pollutants equivalent to or less than that of containment structures (Federal Register, 2003). Under this regulation, producers are becoming interested in using vegetative filter strip (VFS) as a means of controlling runoff pollutants from feedlot.

Vegetative filter strips (VFS), also known as vegetative buffer strips (VBS), are a band of planted or naturally grown vegetation at the down slope end of a non-point pollution source to reduce pollutants from effluent runoff which passes through the strips (Dillaha et al., 1988, Chaubey et al., 1995). It is an alternative to many of the pollution attenuating techniques where control of sediment transport, removal of organic matter, and wastewater treatment are all possible at the same time (Dickey and Vanderholm, 1981; Dillaha et al., 1989; Munoz-Carpena et al., 1992; USDA-NRCS, 1989). Vegetative buffers provide an environment to filter nutrients, sediment, and other pollutants from agricultural runoff by reducing sediment carrier energy (Webber et al., 2010). Specific pollution attenuating mechanisms include infiltration, sedimentation, sorption, volatilization, precipitation, dilution, microbial decomposition, chemical changes, and plant uptake. However, among them, infiltration and sedimentation are the predominant pollutant attenuating mechanisms. These mechanisms are influenced by VFS characteristics, hydrologic, and pollutant properties. Therefore, this pollutant reduction efficiency of vegetative buffers depends on: (i) buffer physical properties (length, width, slope, vegetation type and cover, and soil type), (ii) the properties of the pollutant (soluble or particle borne), and (iii) buffer location in terms of the pollutant source (Zhang et al., 2010).

Vegetative filter strips are now an established method of non-point source pollution control. In the last four decades, a wide range of research has been conducted, both at field and plot scale studies, to show VFSs' effectiveness removing pollutants from runoff from feedlot (Woodbury et al., 2002, 2005; Edwards et al., 1983; Dickey and Vanderholm, 1981; Mankin and Okoren, 2003; Paterson et al., 1980; Young et al., 1980), simulated feedlot (Dillaha et al., 1988;

Robinson et al., 1996), simulated pasture (Lim et al., 1998), grazed pasture (Chaubey et al., 1994, 1995), livestock waste stockpiles (Fajardo et al., 2001), and cropland (Dillaha et al., 1989). Researchers have also investigated herbicide (Arora et al., 1996; Asmussen et al., 1977) and pesticide transport reduction (Syversen and Bechmann, 2004; Dousset et al., 2010). Vegetative filter strips were found to be effective in mass and concentration reductions of incoming sediment, nutrients, and fecal bacteria, which have been reported in various buffer studies. For instance, Coyne et al. (1998) found 98% sediment, 91% fecal coliform, and 74% fecal Streptococci concentration reductions from runoff from poultry waste-amended soil in a 9 m long buffer strip on silt loam soil. Dickey and Vanderholm (1981) found a 95% mass reduction of nutrient and 80% concentration reduction of oxygen demanding materials in runoff from beef feedlot. Dillaha et al. (1988) observed a 91% sediment removal from simulated feedlot runoff from a 9.1 m long VFS. They also noted total phosphorous (TP) and total nitrogen (TN) removals were up to 69% and 74% for applied phosphorous (P) and nitrogen (N), respectively. Chaubey et al. (1995) observed mass transport reductions were 81% for total Kjeldahl nitrogen (TKN), 98% for ammonia nitrogen (NH_3-N), 91% for TP, and 90% for ortho-phosphorous (PO_4-P). Young et al. (1980) found up to 98% total solids (TS), 98% TKN, and 98% TP reduction from a 27 m VFS. Where runoff did not exceed past the VFS area, retention rate showed up to 100%.

Nevertheless, considerable spatial variations of vegetative buffer strip effectiveness were observed in the published articles. Edward et al. (1986) observed 61% TS, 65% chemical oxygen demand (COD), 72% ammonium nitrogen (NH_4-N), and 70% ortho-P transport reduction on mass basis in a 27.5 m long strip for feedlot runoff. Schellinger and Clausen (1992) observed 33% total suspended solids (TSS), 18% TKN, 15% NH_3-N, and 6% ortho-P transport reduction from a 22.9 m long filter strip. In a five meter long (5 m) grass and grass-tree buffer vegetation species combination study, Duchemin and Hogue (2009) found 87% and 85% of TS, 53% and 54% of NH_4-N, 59% and 63% of NO_3-N, 76% and 76% of TP, 34% and 28% of ortho-P, 20% and 20% of E. coli, and 14% and 15% reduction of runoff from grass and grass-tree filter strips, respectively. Abu-Zreig et al. (2001) used 23 mm runoff and 0.9 kg m^{-2} of sediment loads in a VFS study and observed 86% and 57% of sediment and runoff reduction, respectively. Whereas, with the 7.8 kg m^{-2} sediment and 438 mm of runoff loads, VFS reduced transport of sediment, runoff, and nitrate nitrogen loads by 95%, 91%, and 97%, respectively (Blanco-Canqui et al., 2006). Basically, there are no definite design criteria for a designing vegetative buffer strip due to its variability of performance. Customarily, vegetative buffer strips are designed based on the local conditions, performance of existing buffer at the locality, and experience of the designer.

In the state of North Dakota, NRCS is supporting establishing vegetated buffer strips for controlling feedlot runoff pollution. About 1030 animal rearing facilities have been established and used for rearing animals in North Dakota (North Dakota Department of Health- Personal Communication, February, 2011). Among the 1030 animal rearing facilities, 555 are beef feedlots. Of these beef feedlots, few are using vegetative filter strips for controlling runoff pollution due to limited information on well-established design criteria, performance results (Abu-Zreig, 2001), and limited financial support. The design criteria established elsewhere may not be directly transferable in North Dakota due to different agro-climatic environments.

The climate of North Dakota is unique and is characterized by its unpredictability. It is characterized with semi-arid conditions with low annual rainfall (<250 mm) in the western half of the state; whereas, the eastern portion experiences more rainfall with an average of 560 mm. Most of the rainfall occurs during May through August. Before and after this period, rainfall is low to moderate. Summer months are hot, and winter months are very cold. The average annual temperature for the state of North Dakota ranges from 2.8 °C (37 ºF) in the northern part to 6.1 °C (43 ºF) in the south (USGS, 2013). Due to these unique climatic conditions, buffer effectiveness needs to be evaluated to optimize the buffer design under local climatic conditions.

Although removal of sediment and sediment bound nutrients by VFSs is well documented, soluble pollutants are not effectively removed by VFSs (Dorioz et al., 2006). Dillaha et al. (1988) observed 26% and 19% removal of total soluble P and soluble nitrogen, respectively, in their experiment. Lim et al. (1998) found that the VFS was ineffective in removing dissolved solids as indicated by same EC values of inflow and outflow runoffs. Low PO_4-P removal by VFS was observed by Srivastava et al. (1996), and removal efficiency was related to infiltration amount. Lower VFS effectiveness in reducing dissolved nutrients transport was further confirmed by Goel et al. (2004). They reported that average trapping efficiency of P concentration (49.1%) was greater than nitrogen concentration (20.9%). Schmitt et al. (1999) also found lower VFS effectiveness in reducing transport of soluble P and nitrate. These authors found that 24% and 48% of nitrate and 19% and 43% of soluble phosphorous were removed by the 7.5 and 15 m grass strips, respectively. They also observed that VFS was not effective in dissolved pesticide transport reduction.

Despite the ineffectiveness of VFSs in abating the dissolved forms of nutrient transported, limited initiatives have been taken to address this performance limitation. Kim et al. (2005) studied the changes of soluble reactive phosphorous in VFS when they applied milk house waste water. In their study, VFS played a significant role in sorbing P from wastewater when soil remained aerobic. Watt and Torbert (2006) applied gypsum to the VFS and observed increased soluble phosphorous transport reduction (32% to 38%) in the VFS with applied gypsum than that without gypsum (18%). The authors suggested that soluble phosphorous might have been precipitated as insoluble calcium phosphate and were removed from runoff flow. Lindsay (1979) observed that solubility of N, P, and their compounds are largely affected by soil pH condition. Depending on the pH of soil solution, soluble P may be precipitated as water insoluble hydroxyapatite, fluoroapatite, and chloroapatite (Lindsay, 1979; Kanel and Morse, 1978). Thus, changing the pH of soil may influence the solubility of nitrogen and phosphorous species contained in runoff when they flow through the VFS system. However, very limited to no information is available on the impact of pH changes on the buffer performance.

An approach that is considered important in VFS designing is the use of models which can simulate natural conditions and predict outcomes based on inputs. Modeling can make a system or management practice simple, less expensive, and less time consuming. Various regulatory agencies prefer models to assess that any structure or conservation practice under their regulation meets certain set standards. Moreover, simulation of certain practices or natural processes using models helps understand the potential of pollution and helps take preventive precautions.

In a complex situation, e.g., in VFSs, many factors such as vegetation properties, soil properties, hydrologic properties, etc. are involved for filter strip performance, but their contributions to VFS effectiveness are not fully understood yet. To make VFSs effective at field scale, it is important to understand the basic mechanisms, but field studies can help for limited number of cases. Modeling can help study VFS effectiveness under varying set of conditions, understand basic processes involved, and develop design criteria (Abu-Zreig, 2001).

To aid in VFS design through modeling, several studies have been done. Overcash et al. (1981) developed a mathematical model to predict concentration and mass reduction of pollutants of runoff from a VFS installed at the down gradient end of a manure-amended land. In this model, infiltration and dilution were assumed to be the only mechanism for pollutant attenuation. Using Overcash's equation for concentration prediction (Overcash et al., 1981) and an SCS curve number method for runoff prediction, Edwards et al. (1996) developed a VFS design algorithm to design buffer width to meet specific performance requirements. To assess suspended sediment removal effectiveness of a VFS, researchers at the University of Kentucky developed a model, GRASSF, and tested it in a laboratory for an artificial rigid grass media as well as in the field (Barfield et al., 1978, 1979; Hayes et al., 1979; Hayes et al., 1982, 1984; Tollner et al., 1976, 1977). The model used hydraulics of flow and transport and deposition mechanisms of sediments. But, none of the models could successfully model many of the complex situations that may occur in the VFSs.

Munoz-Carpena et al., (1999) developed a vegetative filter strip model (VFSMOD) to simulate complex situation that might occur in a buffer under natural events. VFSMOD is a storm-based, mechanistic, field-scale model that routes incoming hydrograph and sedigraph information from an adjacent field through the VFS and calculates the resulting outflow, infiltration, and sediment trapping efficiency. The model has the capability to account for variable rainfall patterns, time dependent infiltration, and various surface conditions. But, this model has some limitations in that it can predict sediment transport reduction, only. Few studies have been undertaken to include phosphorus (Kuo and Munoz-Carpena, 2009) and pesticide (Sabbagh et al., 2009) transport components. Therefore, there is a need to develop a model that can predict pollutant (sediment and nutrients) trapping efficiencies from a VFS which receives runoff from various source areas including feedlot surface. Based on the above discussion, objectives this research were formulated.

1.1. Objectives

(1) To evaluate the performance of vegetative filter strips installed at the downslope end of feedlots.

(2) To study the effect of pH levels of soil on soluble nutrients reduction from manure borne runoff in VFS.

(3) To develop a model to predict phosphorus and sediment trapping efficiency of VFS from feedlot runoff.

1.2. Dissertation Outline

Chapter 1 of this dissertation presents the rationale of this study and Chapter 2 presents relevant literature related to this research. The following four chapters describe the methodology and results of performance evaluation of vegetative filter strips, effect of pH on soluble nutrient transport reduction from manure borne runoff from VFS, and modeling VFS system for predicting loss and trapping efficiency of sediments and phosphorus from VFS. The final general conclusion chapter (Chapter 7) is based on results found in these studies. All the references were included at the end of Chapter 7.

- **Chapter 3. Efficacy of vegetative filter strips (VFS) installed at the edge of feedlot to minimize solids and nutrients from runoff**

  This study was conducted to evaluate a VFS installed at the edge of a feedlot. This study was conducted in a VFS at Richland County, North Dakota. Performance of the VFS and mechanisms of nutrient reduction are described.

- **Chapter 4. Performance evaluation of three vegetative filter strip designs for controlling feedlot runoff pollution**

This study was conducted to assess the performance of three VFSs with different designs, climatic and management conditions. The three VFSs were located in three counties of North Dakota. Individual as well as their comparative performances are discussed.

- **Chapter 5. Influence of soil pH in vegetative filter strips to reduce soluble nutrients transport**

In this chapter, the VFS performance in reducing soluble nutrient transport from manure borne runoff at different soil pH was evaluated. Effect of soil pH and different mechanisms for transport reductions are discussed.

- **Chapter 6. A model to predict sediment and phosphorus trapping efficiency of vegetative filter strips from feedlot runoff**

In this chapter, a model for predicting sediment and phosphorus transport is described which was developed by incorporating two procedures into existing VFSMOD. Model development and calibration and validation are described.

CHAPTER 2. REVIEW OF LITERATURE

United States Department of Agriculture- Natural Resources Conservation Service (USDA-NRCS, 1998) described vegetative filter strips (VFS), also known as vegetative buffer strips or buffer strips or only buffers, as areas of permanent vegetation established to intercept sediments, nutrients, pesticides, and other pollutants from runoff before the runoff reaches a water body. VFSs are installed at the edge of an agricultural field, alongside any surface water body, or anywhere downstream of a diffused pollutant source. They are effective in attenuating non-point and point sources of pollution which affect surface and ground water quality. In pollution attenuating systems, physical, chemical, and biological processes are involved. Specific processes involved are sedimentation, infiltration, sorption, plant uptake, dilution, volatilization, precipitation, and decomposition (Vanderholm et al., 1979; Vanderholm and Dickey, 1980; Dillaha et al., 1988; KDHE, 1995; Fajardo et al., 2001; Hubbard et al., 1998; Woodbury et al., 2005; Hoffman et al., 2009). When runoff flows through the VFS, its velocity is retarded, and, consequently, the sediment carrying capacity of the runoff decreases and thereby particles settle. As a result, nutrients attached to sediment particles are retained in the VFS, and downstream discharge is eliminated. As velocity decreases, runoff has a longer time to infiltrate into the soil, and soluble nutrients are removed with the infiltrated water. Vegetation also hinders pollutants flowing through it and adsorbed on plant surfaces and soil particles. The vegetation also helps nutrient removal from the VFS by up taking nutrients for their metabolism. However, the effectiveness of the VFS varies widely depending on the vegetation types, buffer physical properties, hydrology, and pollutant properties and some of them are discussed in the subsequent sections.

2.1. Factors Influencing Vegetative Filter Strip Performance

2.1.1. Vegetation type

Dense and standing vegetation is required for efficient filtration effect. Vegetation may increase surface roughness resulting in reduced surface runoff velocity, thus, increased deposition of sediment (Syversen, 2005), and less transport of particulate bound nutrients. Sediment and some nutrients are sorbed on leaves and stems. After decomposition of the root systems, preferential flow paths are created resulting in enhanced infiltration. Nutrient uptake by vegetation and its removal as biomass is also an important way to manage manure nutrients, which are released and transported from the concentrated animal feeding operations. Canopy density, root distribution, and nutrient uptake are all affected by vegetation types.

A few studies have been conducted to find the most effective plant species to control manure borne pollution. Goel et al. (2004) conducted a simulated runoff study with different grasses (e.g., perennial ryegrass, Kentucky blue grass as sod, mixed grass species, and no vegetation) to retain pollutants from mixed slurry. They observed that sod grass (Kentucky blue grass) was the most effective to retain particulate bound nutrients, followed by the perennial rye, and mixed grasses, respectively. But, sod and mixed grasses were equally effective in reducing transport of total suspended solids (TSS). In a similar study, Giri et al. (2008) found a warm season forb (perennial sunflower) and warm season grasses (switchgrass) were most effective to reduce P from runoff, followed by coastal Bermuda grass and cool season grasses. Lee at al. (1999) reported that switchgrass (*Panicum virgatum*) had higher effectiveness for longer periods of time than cool season grasses due to a more uniform distribution of grasses and litter and stem. Similarly, Fasching and Bauder (2001) investigated the sediment reduction effectiveness of eight cool season grasses and found that crested wheatgrass and bromegrass were the most effective due to high basal areas and biomass yield. Comparing the results reported by Lee et al. (1999) and Fasching and Bauder (2001) suggests that density is an important performance factor for selecting a suitable plant for a buffer. Similarly, nutrient uptake, especially P, removal effectiveness was also affected by vegetation as found by Abu-Zreig et al. (2003) and McFarland and Hauck (2004). Their study found that native grass in Elora, ON, Canada was more effective in reducing P than ryegrass and red fescue, and coastal berumda grass was more effective than sorghum and wheat.

In a vegetation composition study, Schmitt et al. (1999) found that young trees and shrubs planted at the lower one half of the VFS had no impact on filter performance. In a modeling study, Zhang et al. (2010) showed that a VFS comprised of grasses or trees only is more effective for sediment control than that mixed with grasses and trees. For N and P reduction, trees are more effective than grasses or a mixture of grasses and trees. Other researchers also agree that the buffer effectiveness was reduced when the buffer was comprised of different species. Duchemin and Hogue (2009) investigated the grass and tree mixed-buffer for filtering runoff and drainage water from a swine manure applied corn-field and Mankin et al. (2007) investigated the grass and shrub mixed-buffer for filtering simulated runoff solution and observed a reduction in effectiveness of total suspended solids (TSS), phosphorous (P), and nitrogen (N) removal. Neither grass-tree nor grass-shrub mixed-buffer systems were found to be more effective than the grassed only filter strips. However, in a similar experiment by Dosskey et al. (2007) found that grass and forest (grass and shrub and tree) vegetation were equally effective as filter strips for sediment and nutrient reduction. Syversen (2005) observed no significant difference between forest buffer zones (grasses and trees) and grass buffer zones (grass only) for nitrogen and phosphorous retention but forest buffer zones had higher particle retention efficiency.

Most of the above studies were conducted under warm climatic conditions and very limited information is available on the vegetation type in cold climatic conditions. Syversen and Borch (2005) studied the retention of soil particle fractions and phosphorus in cold-climate buffer zones in Norway, where they used dominant plant species as a buffer. They found that coarse clay particles were trapped throughout the buffer and independent of width, but silt and sand fractions were trapped mostly in the upper part of the buffer. Although a number of studies have been done to characterize the vegetation effect on VFS performance, limited information is available regarding the effect of vegetation height.

2.1.2. Buffer width/length (in the direction of flow)

Effectiveness of VFSs depends on length of the strips. In general, the greater the length is the higher the trapping efficiency. Longer length increase opportunity for infiltration and sorption to vegetation and organic matter (Barfield et al., 1998). Published literature revealed that buffer widths ranged from 3 m (Chaubey et al. 1994) to 33 m (Kim et al. 2005) for simulated runoff with overland type flow, whereas they can vary from 479 m (Andersen et al., 2009) to 564 m (Dickey and Vanderholm, 1981) for overland and channel type flow with natural rainfall as the runoff source. Researchers observed that the effectiveness of buffer strip increased as length of the strips increased (Edwards et al., 1997; Chaubey et al., 1994, 1995; Dillaha et al., 1998; Srivastava et al., 1996; Magette et al., 1989; Stout et al., 2005; Lim et al., 1998; Young et al., 1980; Goel et al., 2004; Coyne et al., 1998). In contrast, effectiveness decreases as the runoff event and loading rate increase (Magette et al., 1989; Schwer and Clausen, 1989). Concentration and mass transport reductions through VFSs were found to follow the first order exponential decay function (Srivastava et al., 1996; Chaubey et al., 1994, 1995; Edwards et al., 1997; Lim et al., 1998).

Despite the fact that longer width means higher trapping efficiency, the first few meters of a buffer are more effective in reducing sediment and particulate bound nutrients than the remaining buffer length. Lim et al. (1998) found insignificant mass transport of TKN, PO_4-P, TP, and TS beyond 6.1 m of buffer length, except TSS. Dillaha et al. (1998) found that by increasing the VFS length from 4.6 to 9.1 m, sediment transport reduction can be increased by 10%. Srivastava et al. (1996) found no significant reduction of NO_3-N, TKN, and TOC concentration beyond 3 m and NH_3-N, PO_4-P, and TP after 6 m of a buffer width. Coyne et al. (1998) found that a filter strip length of 4.5 m is effective for trapping sediment, and trapping efficiency may be increased slightly if the filter strip width is increased beyond 4.5 m. Basically, larger particles settle quickly in the buffer strip, whereas smaller particles take a longer time to settle and travel a longer distance down the strip. This means that longer filter length needs to be used if pollutant removal has to be maximized from runoff water.

2.1.3. Area ratio (AR)

Area ratio is the ratio of a vegetated buffer area to area that contributes pollutants containing runoff to the buffer area, i.e., VFS area: drainage area. Smaller area ratio results increased volume of runoff onto VFSs. Increased volume of runoff contributes only increased amounts of pollutant mass since longer source length does not have an effect on the pollutant concentrations at the edge of the field (Srivastava et al., 1996; Edward et al., 1996; Edwards et al., 1997). In contrast, when area ratio is greater, runoff will travel through the buffer area for an extended time that will facilitate greater infiltration and sorption (Krutz et al., 2005).

Few studies have investigated the effect of area ratio on pollutant mass reduction. Lee at el. (1999) observed that increasing VFS width from 3 to 6 m increased area ratio from 1:40 to 1:20, while sediment removal efficiency increased by 11%. Having an area ratio of 0.5:1 and 1:1, Webber et al. (2009) did not observe any significant difference in the buffer's effectiveness to reduce nutrients from runoff, which were generated from composting areas. They observed a 98% and a 93% runoff reduction from buffers having area ratios of 1:1 and 1:0.5, respectively, compared to a 1:0 control plot ratio. Mankin et al. (2006) observed positive correlation between area ratio and constituent reductions but negative correlation between constituent reductions and event rainfall depth. Overcash et al. (1981) developed a mathematical model for designing grass filter strips situated downslope of waste-amended land for steady-state rainfall and infiltration. Their model simulation showed that to reduce pollutant mass and concentration at greater levels, either greater buffer to waste area length or increased infiltration to rainfall ratio is required. When no infiltration occurs, pollutant mass reduction does not depend on the area ratio. In VFS design, a minimum length should be specified to give a desired buffer to drainage area ratio.

2.1.4. Filter strip slope

Slope has a predominant effect on velocity of flow. As slope increases, velocity of flow increases resulting in low retention time for sufficient infiltration and sorption. Land slope is also a determining factor for the state of runoff flow, overland sheet flow or concentrated channel flow, through the buffer and land areas. Overland sheet flow is an essential prerequisite for effective buffer strip performance. This type of flow occurs on mild and uniform slopes; however, concentrated channel flow occurs in plots with cross slopes (Dillaha et al., 1988).

Vegetative filter strip slope and soil types determine the lengths of VFSs. Longer VFS length is required on steep slopes and fine textured soil. Liu et al. (2008) conducted meta-analysis of VFS and found that 10 m buffer with 9% slope optimized the sediment trapping capability of a vegetated buffer regardless of area ratio. Hawkins et al. (1998) observed 28% of times of runoff occurrence on 11% slope and less than 11% percent of times on the 5% slope on sandy loam and loamy sand soil, respectively. In terms of nutrient transport reduction, TKN mass was reduced by 93% and 60% on 11% and 5% slopes, respectively. Similarly, K and P masses were reduced by 91% and 92%, respectively, on 11% slope and were slightly lower on a 5% slope. Total solids removed were 37% on an 11% slope and 47% on 5% slope. On the other hand, NO_3-N was decreased by 54% on 5% slope but increased 59% on 11% slope.

2.1.5. Soil type

Since soil is an important component of vegetative filter strips, it is obvious that the performance of VFS will be largely affected by soil physical, chemical, and biological properties. One of the major mechanisms of VFS effectiveness is infiltration which is determined by the type of soil. Amount of runoff and properties of sediment generated are also influenced by the type of soil in the source area (Munoz-Carpena and Parsons, 2004). Unfortunately, performance results of VFSs are scarce showing the variation due to soil types. Importance of soil type in VFS performance is reflected in several modeling studies of VFS. In a simulation study, Munoz-Carpena et al. (1993) compared the performance of VFSs established on sandy-loam and clay soils. They found that sandy-loam soil had less outflow runoff volume compared to inflow, but clay soil showed the opposite trend. Munoz-Carpena and Parsons (2004) developed a VFS model and studied the effect of filter strip length under different soil types and found that filter strip lengths of 1 to 4 m are required for sandy clay soil, whereas, 8 to 44 m filter lengths are required for clay soil to achieve a 75% trapping efficiency of sediment, where design storms of 1 to 10 year return periods were simulated. Infiltration is likely the key mechanism of these variations under different soil types. Since the soluble portion of nutrients is removed from runoff through infiltration, soil type plays an important role in soluble nutrient transport. Moreover, some pollutants are adsorbed on the soil particles' exchange sites. Thus soil types with higher exchange sites play a significant role in pollutant removal.

2.1.6. Pollutant type

Manure borne pollutants in runoff can be broadly classified as two categories: particulate and particulate bound, and soluble. They can be removed from the runoff by using VFSs through different pollution attenuating processes. For instance, particulate and particulate bound pollutants are removed by sedimentation, infiltration, sorption, and other physical processes. But soluble fractions are removed through infiltration, sorption, and other chemical processes. Removal effectiveness of a particular fraction depends on the predominant mechanisms that occur in the VFSs. For most cases, sedimentation is a predominant mechanism, which indicates that pollutants' reduction in terms of mass and concentration is greatest for sediment, followed by sediment bound, and soluble pollutants (Schmitt et al., 1999). Dillaha et al. (1988) suggested that soluble P flows as solution, i.e., independent of suspended sediment, and thereby, difficult to control in transport. Basically, soluble pollutants are less affected by the VFSs. The principle mechanism of soluble pollutant attenuation is infiltration and thus removal effectiveness decreases with duration of flow. Some soluble pollutants are also removed by sorption on soil particles (Schmitt et al., 1999; Mersie et al., 2003) and soil organic matter.

Several research findings suggested that vegetative filter strips are not an effective method for soluble nutrient transport reduction (Dillaha et al., 1988; Lim et al., 1998; Duchemin and Hogue, 2009; Goel et al., 2004). For instances, Dillaha et al. (1988) recommended that mechanisms involved in soluble nutrient transport reduction are infiltration, adsorption, and soil sorption that would decrease with time as infiltration decreases, adsorption capacity of the vegetation is satisfied, and surface soil P sorption sites become occupied. Lim et al. (1998) found that vegetative filter strips removed little dissolved solids as no change in EC values was observed in runoff water flowing through the filter strips. Goel et al. (2004) reported low concentration reduction of soluble phosphorous (49.1%) and nitrogen (20.9%). Sotomayor-Ramirez et al. (2008) observed the trend that dissolved P (DP) to total P (TP) ratio increases with an increase in filter strip width and suggested that the filter strips were more effective in reducing the particulate P fraction relative to the dissolved fraction.

Few studies have been initiated to increase the buffer strips' effectiveness in removing dissolved pollutants. Watt and Torbert (2009) applied gypsum in buffer strips as a soil amendment to reduce the transport of soluble phosphorous. In their study, poultry litter was applied at the upper part of a fescue plot at a rate of 250 kg per hectare, and concentrated runoff water was routed through the buffer at a rate of four liters per minutes. The lower part of the fescue plot was used as a buffer and treated with gypsum. They simulated two runoff events at about four a week interval. The VFS effluent, immediately after poultry litter application, showed higher soluble phosphorous reduction (32% to 40%) as compared to on an untreated plot (18%). No significant difference was observed between gypsum applications rates, but, for the second runoff event, the concentration of soluble phosphorous in runoff was found to be very low although the effect of gypsum had disappeared.

Brauer et al. (2005) conducted an experiment to evaluate the effectiveness of soil amendments including gypsum to reduce the soil test P values. In their study, application of gypsum at a rate of 5 Mg ha^{-1} reduced soil test P levels between 1999 and 2001 but increased between 2003 and 2004. Gypsum reacted with soluble phosphate, which resulted insoluble Ca-phosphate and reduced the P transport. They concluded that if sufficient amounts of Ca can be supplied by adding gypsum into soils, dissolved reactive P levels can be reduced.

According to the pH of a soil solution, Lindsay (1979) suggested that insoluble hydroxyapatite and fluorapatite formed when soluble P reacts with Ca at a higher pH. These Ca-phosphates dissolve in soil solution when pH is lowered. Varying the soil pH, it is possible to get information about soluble phosphorous removal effectiveness from manure borne runoff water flowing through the VFS. However, very limited information is available on VFS performance of soluble pollutants reduction on varying the soil pH.

2.1.7. State of flow

State of flow is a critical factor for buffer strip performance. Flow through the VFS is likely to be overland sheet flow or concentrated channel flow. When concentrated channel type flows occur in actual fields, their effectiveness is reduced as compared to shallow overland sheet flow. In shallow overland sheet flows, high flow resistance and reduced flow velocity occur causing sediment and particulate bound pollutants to be removed by sedimentation and provide more time for sorption and infiltration. On the other hand, in concentrated flow, runoff might flow through a small fraction of the total VFS area, which is likely to decrease infiltration volume (Abu-Zreigh et al., 2001) resulting in reduced effectiveness. Concentrated flows sometimes submerge vegetation causing reduced hydraulic resistance to flow and resulting in decreased in effectiveness.

Few studies have addressed this factor to quantify the effects of flow types on VFS performance. Dillaha et al. (1988) conducted an experiment that kept a 4% cross slope to favor flow concentration. In their study, they found that the VFS that encountered concentrated flow had 40% to 60%, 70% to 90%, and 61% to 70% less sediment, N, and P removal effectiveness, respectively, compared to VFS plots that encountered shallow overland flow. Blanco-Canqui et al. (2006) performed field experiments with the state of flow and observed significant variations of buffer filtering performance for sediment, organic nitrogen, and nitrate nitrogen. For a 0.7 m filter strip, sediment, organic N, and NO_3-N reduction efficiency decreased from 25% to 10%, 62% to 43%, and 34% to 21%, respectively, if flow state changed from interrill flow to concentrated flow. Lower effectiveness for concentrated flow was attributed to the reduction in hydraulic roughness and stiffness of the fescue stems. Concentrated flow occurs through the small parts of the VFS and thus is less conductive to remove pollutants. However, increasing the filter strip width improves the performance of VFS.

2.1.8. Time after establishment

With the time and runoff events, changes in soil properties and vegetation occur within vegetative buffers. Over time, vegetation composition and vegetation density change and plant biomass decomposes and turns into soil organic matter, which affect buffer filtering and sorptive capacity. Decomposing vegetation on soil surfaces and vegetation roots in soil matrices affect infiltration by creating preferential flow paths. Organic matter improves soil structure, increases aeration, and augments activities of microorganisms. Duchemin and Hogue (2009) reported low effectiveness of vegetative filter systems in the first year after establishment due to limited vegetation cover. Similarly, Dosskey et al. (2007) found that the buffer strip performance improved over time and reached full effectiveness within three growing seasons after establishment, and infiltration played a dominant role for pollutant attenuation.

With time, P and N removal efficiencies decrease within the first few meters of filter strips as sediment and nutrients build up from the prior runoff events (Dillaha et al. 1988). As a result, over time, buffer strips may become nutrient rich, and, subsequently, these nutrients may be released in future runoff events. Increase in outflow concentration of nutrient than inflow was led many researchers (Bhattari et al., 2009; Dillaha et al., 1988; Dosskey et al., 2007; Young et al., 1980; Hubbard et al., 1998; Chaubey et al., 1994, 1995; Hawkins et al., 1998; Hay et al., 2006) to assume that nutrient accumulates in buffers, and subsequently releases to next runoff events.

2.2. Pollution Reduction Mechanisms

2.2.1. Sedimentation

VFSs remove pollutants, such as sediment and sediment bound, from the influent runoff water by the process of sedimentation. Sedimentation occurs when vegetation reduces sediment carrier energy of water (Webber et al., 2010). Vegetation increases the hydraulic roughness of the flow and results decreased velocity (Munoz-Carpena, 1999). A decrease in velocity reduces turbulence and increases depth of flow, which results in sediment deposition. Sediment trapping mechanisms in VFSs are influenced by vegetation properties, slope, soil type, size and geometry of VFS, and influent solid concentration (Koelsch et al., 2006). Foster and Mayer (1972) found that sediment trapping efficiency is directly proportional to slope and flow rate. Buffers on steep slopes increase runoff velocity and resulting decrease in sediment trapping efficiency (Liu et al., 2008).

Barfield et al. (1979) described a conceptual model of sediment transport and deposition processes in artificial media after numerous laboratory studies. According to them, impinging sediment-laden flow into the VFS reduces velocity and transport capacity. If the resulting transport capacity is less than the inflow, deposition occurs. The initial deposition causes a sediment wedge to form, and this wedge moves downstream with time. They described four zones in the process of deposition counting from the direction of flow. In the first zone, deposition overtops the vegetation, and all the incoming sediment is transported over the inundated media. The second zone is the deposition zone where sedimentation occurs uniformly along the face of the downstream wedge. The slope of the deposition face is known as the equilibrium slope. In the third zone, sediment is deposited on the VFS floor and fills the depressions. This allows the transport of sediment as bed load in this zone. The fourth zone traps all the sediments that reach into it because surface irregularities are not yet filled. It has been observed from the field studies that the first few meters of VFS is more effective in reducing sediment than the rest of the VFS, which validates the model simulations.

Effective sediment transport reduction by VFS is substantiated by a number of studies. For example, Coyne et al. (1998) found sediment removal effectiveness of 96% for a 4.5 m and 98% for a 9 m long grasses filter strips from simulated runoff. Dillaha et al. (1988) investigated the influence of flow type and length on sediment removal effectiveness. They found 91% and 81% of incoming sediments were removed by 9.1 m and 4.6 m long VFSs, respectively, but performance was low when flow was concentrated. Lim et al. (1998) found significant reduction of TSS from a 18.3 m long VFS. Schellinger and Clausen (1992) investigated the single VFS length effect for reducing solids, N, P, and bacteria in runoff from a dairy barnyard. They observed 22.9 m long VFS on a 2% slope reduced TSS by 27% from runoff that passed through detention basin.

2.2.2. Infiltration

Infiltration that occurs in VFS is an important pollutant attenuating mechanism, especially as a primary mechanism for soluble pollutants (Dillaha et al., 1988). This is an indirect way of attenuation for soluble pollutants, where the volume of water infiltrated determines the degree of pollution reduction. When water infiltrates into a VFS soil, soluble pollutants enter into the soil profile. Pollutants that are attached to very small sediment particles are also reduced if they enter into the soil profile through small soil pores. Moreover, runoff volume itself is reduced though infiltration, thereby pollution potential decreases downstream. Enhanced infiltration is resulted in VFSs through preferential flow path created in the VFS-soil matrix (Mersie et al., 2003). Vegetation increases the hydraulic resistance and retards velocity of flow, allowing more time for infiltration into the soil matrix.

Several studies reported pollutant reductions as a result of infiltration. For example, Roodsari et al. (2005) observed decreased fecal coliform (FC) in the VFS from 68% to 1% on clay loam and 23% to undetectable levels on sandy loam soil compared to bare soil. Hawkins et al. (1998) observed 91% K and 92% P mass retention in a VFS soil on 11% slope due to high infiltration volume. Soluble herbicide (Krutz et al., 2003) and pesticide (Boyd et al., 2003) retention in VFS as attributed to infiltration were also observed. However, infiltration decreases surface water pollution but increases the potential for groundwater pollution as pollutants move through the soil profile as a result of higher infiltration.

2.2.3. Sorption

Some pollutants such as nutrients, pesticides, and herbicides are adsorbed and/or absorbed on organic matter and soil particles. The organic carbon content of the VFS soil might increase sorption of nonionic and weakly basic herbicides and herbicide metabolites, but ionic and or polar herbicides or herbicide metabolites are more controlled by clay mineral particles and iron oxides (Krutz et al., 2005). Clay mineral content, Al and Fe oxides, organic matter content, and calcium carbonate also affect phosphorus sorption (Vought et al., 1994). Other factors such as redox potential, pH, temperature, amount already adsorbed, and reaction time affects sorption (Svendsen, 1992). Sorption is an equilibrium process; therefore, higher concentration of pollutant in water will result in higher adsorption to soil particles (Vought et al., 1994). In VFS soil, particle size of decomposed organic matter influences the amount of sorption. A decrease in particle size increases specific surface area, which increases the sorption (Benoit et al., 2008). It has the implication that the association of organic matter with mineral surfaces increases accessibility of the sorption site (Barriuso et al., 1994). However, pollutants adsorbed onto sediment particles and settled in the VFS may be re-suspended in later hydrologic events and increase outflow pollutant mass and concentration (Hoffmann et al., 2009).

2.2.4. Plant uptake

VFS can be used to produce biomass as a means to control pollution, but plant uptake of nutrients is not a primary pollutant removal mechanism. Periodic vegetative harvesting during the growing periods results in removal of nutrients, and no buildup of nutrients thus occurs in the VFS. Fajardo et al. (2001) found no accumulation of NO_3-N in the soil profile due to mineralization, transport with runoff, and continuous plant uptake. Sanderson et al. (2001) observed a linear increase of switchgrass dry matter yield with increased N application from manure. They found switchgrass recovered 15% of the manure N and less than 20% of manure P. Low N removal accounted for low mineralization of N as it was applied as a solid as because mineralization occurs in solid manure at a slower rate. Since N mineralization is a prerequisite for plant uptake, applying liquid manure will enhance the N assimilation rate versus applying solid manure. Schwer and Clausen (1989) observed 2.5% and 15% removal of total input of P and N, respectively, in the vegetative area when loading rate of wastewater was 2.94 cm/wk. Successful assimilation of N and production of biomass were also observed by other researchers (Woodbury et al., 2003 and 2005; Hubbard et al., 1998), and higher N assimilation by forest than grass buffer was confirmed Hubbard et al. (1998).

2.3. VFS Models

So far, few models have been developed to design and evaluate a VFS system. Attempts have been made to simulate transport of pollutants through the VFS. However, attempts were mostly limited to the sediment and sediment bound pollutants.

Overcash et al. (1981) developed a general equation for transporting mass and concentration of pollutants at a given distance through buffer strips situated at the downslope end of pollution source by using water and pollutant mass balance technique. Infiltration to rainfall ratio and buffer area length to waste area length ratio were found to be important performance parameters.

Using the equation of Overcash et al. (1981), Edward et al. (1996) developed a design algorithm for VFSs design. The algorithm can be used to determine the concentration of pollutants exiting the VFS and is governed by the VFS and the runoff parameters. The model can also be used to determine a length required to meet either a given runoff pollutant concentration or mass transport reductions.

Munoz-Carpena et al. (1999) developed a model called vegetative filter strip model (VFSMOD), which promises successful modeling of pollutants, such as sediment and runoff, transport through VFSs. VFSMOD is a field-scale, mechanistic, storm based model which routes incoming hydrograph and sedigraph information from an adjacent source area through a vegetative filter strip and calculates its retention efficiency. The principal mechanisms which occur in VFSs are described by linking three different sub-models together. The sub-models used were the Petrov-Galerkin finite element kinematic wave overland flow sub-model, the modified Green-Ampt infiltration sub-model, and the University of Kentucky sediment filtration model. The model can effectively handle the inputs similar to those in natural events and provide outputs as outflow water and sediment trapping in the strip. Good prediction of the model output is observed if shallow uniform sheet flow occurs within the vegetative filter strip.

Abu-Zreigh et al. (2001) tested and validated the VFSMOD for its effectiveness in reducing sediment transport in a simulation study. The model was simulated for total width and actual flow width. When total width was used, there was no correlation between observed and predicted infiltration volume, but little correlation was found between observed and predicted sediment trapping efficiencies. On the contrary, when actual flow width was used, the model satisfactorily predicted infiltration volume and sediment trapping efficiency. In another study, Abu-Zreig (2001) investigated factors affecting the sediment trapping in VFSs in which filter length was found to be the most important parameter for the VFS performance. Soil type also played an important role by its infiltration rate in filter performance. Han et al. (2005) used the VFSMOD model to test the effect of performance parameters on TSS removal by VFSs from highway runoff. Larger particles (> 8µm) were found to be efficiently removed by VFS.

Munoz-Carpena and Parsons (2004) developed a design procedure (i.e., selecting construction characteristics such as filter strip length, width, slope, vegetation) of VFS using VFSMOD-W, a windows based version of VFSMOD. A unit hydrograph (UH) method was developed and added to the VFSMOD-W to give necessary source area design inputs for VFSMOD. Using a combination of the NRCS curve number method, the unit hydrograph, the modified universal soil loss equation (MUSLE), and a rainfall hyetograph, a runoff hydrograph and sediment losses from the upland source area were generated for a design storm and provided as inputs to the VFSMOD.

Dosskey et al. (2008) used the VFSMOD model to study the relationship between filter strip width and trapping efficiency of sediment and water. Using the relationships found by model simulations, they developed a simple design aid to select VFS width required to achieve a given trapping efficiency for a site condition of interest. Field conditions were simulated by using different combinations of factors such as slope, soil type, drainage area size, and cropping practices.

In addition to sediment and runoff transport modeling, few studies have approached for modeling transport of pesticide, phosphorus, and microorganism. Sabbagh et al. (2009) constructed an empirical model for pesticide trapping through VFS and linked the model with VFSMOD to simulate pesticide (dissolved and sorbed) trapping. Unlike other empirical equations which used physical characteristics of VFS (width, area ratio, slope, and vegetation type) only, the proposed model was based on characteristics of buffer physical properties, hydrology, and pollutants. The proposed model outperformed the empirical equation which was only based on VFS width (such as in SWAT).

Rudra et al. (2010) developed a model, called GDVFS, to design and evaluate a vegetative filter strip using VFSMOD. In their model, they incorporated phosphorous and bacteria transport components to VFSMOD; however, the hydrology and sediment transport components were retained same in GDVFS model as they were in the original VFSMOD-W model. To calculate sediment bound phosphorous yield and transport from upland agricultural land, the CREAMS model (Knisel, 1980) was used. To estimate the soluble fraction of phosphorous entering into the VFS, a method suggested by Sharply et al. (1981) was used. Similarly, bacterial transports were also divided into sediment bound and free floating in runoff water from upland areas to VFS. Experimental data were used to evaluate the trapping efficiency of sediment, phosphorous, and bacteria as affected by the vegetation, filter strip length, inflow rate, and inflow concentration.

Munoz-Carpena et al. (2007) used the global sensitivity and uncertainty analyses framework for modeling with VFSMOD-W for a phosphate mining region of central Florida. Two filter lengths, 3 and 6 m, and two model structures were used to compare the results from the previous local “one-parameter-at-a-time” analyses. The two model structures considered were VFSM (filter module alone) and UH/VFSM (combined filter and source area component). For both structures, saturated hydraulic conductivity was found to be the most important controlling factor for filter runoff response and explained 90% of total output variances. In the case of the UH/VFSM structure, the source area included three more important factors, i.e., slope of the source area, USLE soil erodibility index, and a runoff curve number in addition to three factors from the VFSM alone. There was no significant parameter interaction for all model outputs except sediment outflow concentration and sediment wedge geometry for this specific application.

Kuo and Munoz-Carpena (2009) investigated the VFSMOD-W model efficiency for modeling vegetative filter strips used for controlling surface runoff pollution from the phosphate mining and sand tailing. Good runoff and sediment predictions resulted in good predictions of particulate phosphorous and total phosphorous transport as apatite was a main component of sediment. Dissolved phosphorous prediction was also found to be satisfactory when considering rainfall impact on dissolved phosphorous, which was dissolved from the apatites in surface soil. The VFSMOD-W and simplified P modeling in a combined approach successfully predicted runoff, sediment, and P transport in phosphate mining sand tailings.

The literature review suggests limited use of VFSMOD model for evaluating VFSs effectiveness in terms of number of pollutants and pollutant sources. It would be worthwhile to adapt this model in designing and evaluating VFS systems installed for controlling feedlot runoff pollution.

CHAPTER 3. EFFICACY OF VEGETATIVE FILTER STRIPS (VFS) INSTALLED AT THE EGDE OF FEEDLOT TO MINIMIZE SOLIDS AND NUTRIENTS FROM RUNOFF[1]

3.1. Abstract

Runoff from open animal feeding operation is a major source of non-point pollution. Vegetative filter strips (VFS) are one of the effective ways in controlling non-point source pollution. In this study, performance of a vegetative filter strip situated at down slope end of a beef feedlot was evaluated under eastern North Dakota climatic conditions. Two automatic ISCO samplers were installed to collect runoff water entering and leaving the vegetative filter strip. Runoff samples were analyzed for solids, nutrients, pH, and conductivity using standard methods. Results indicated that VFS was effective in reducing concentration of total solids (TS) by 33.7%, total suspended solids (TSS) by 68.0%, total phosphorous (TP) by 29.9%, ortho-phosphorous (ortho-P) by 19.3%, ammonium nitrogen (NH_4-N) by 31.8%, total Kjeldahl nitrogen (TKN) by 35.6%, and potassium (K) by 19.8%. Nitrate nitrogen (NO_3-N) concentrations at the outlet samples increased as expected, and the buffer was not effective in reducing soluble nutrients. Performance of the VFS indicated that a VFS can be used for reducing runoff pollution that comes directly from feedlots into VFSs without passing through the settling basins. Longer buffer lengths might be required for reducing soluble pollutants.

Keywords. Feedlot, nutrients, runoff, solids, vegetative filter strip

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3.2. Introduction

With expanding livestock facilities, animal agriculture is facing increasing environmental concerns, i.e., water and air pollution due to increasing manure volumes from these expanding livestock facilities. Although manure is an excellent source of nutrients for plants and a good soil conditioner, improper manure management, especially from feedlots, can negatively influence water quality. For example, runoff from feedlots may carry significant amount of manure borne nutrients (e.g., nitrogen and phosphorous) to surface water (Swanson et al., 1971) and may cause water pollution. According to Koelsch et al. (2006), runoff from feedlots is a major contributor and will continue to be a contributor to surface and groundwater impairment.

Typically, feedlot runoff is collected and stored in a holding pond or lagoon and usually emptied by pumping and applying to crop land. For an instance, beef cattle feedlots often use a lagoon or settling basin with vegetative filter strips to reduce runoff pollutant concentration and migration to surface water bodies (Mankin et al., 2006). However, holding pond or lagoon construction is expensive, requires large land area and regular maintenance. Moreover, seeping water from the containment structures possesses the risk of contamination of the potential drinking water (Parker et al., 1999). On the other hand, vegetative filter strip (VFS) systems involve spreading and infiltration of runoff, thereby this system do not require any containment structure. The challenge of an effective VFS is to maintain the sheet flow, the systems fails if channelization occurs (Lorimor et al., 2002). While the cost comparison between VFS and settling basin is difficult due to location, topography, and climatic conditions for both systems, but in general the cost involves in a VFS system is lower than other structures due to capital investment and maintenance (Kizil, 2010; Barrett, 1999). As a result, often producers are not interested to construct holding ponds due to high capital investment.

The US Environmental Protection Agency (EPA) has recommended vegetative filter strips (VFS) to minimize the adverse impact of feedlot runoff to surface and groundwater bodies (USEPA, 2001). Vegetative filter strips are a band of planted and/or indigenous vegetation installed at the down slope end of non-point source pollution areas before runoff reaches a water body (Dillaha et al., 1988). Vegetative filter strips provide an environment to reduce pollutants by reducing sediment carrier energy (Webber et al., 2010). In addition, pollutant reduction in the buffer also occurs due to infiltration, adsorption, and plant uptake of nutrients.

During the past three decades, many studies have been conducted, both at field and plot scales, to show the buffer's effectiveness in removing pollutants in runoff from feedlot (Woodbury et al., 2002, 2005; Edwards et al., 1983; Dickey and Vanderholm, 1981; Mankin and Okoren, 2003; Paterson et al., 1980; Young et al., 1980), simulated feedlot (Dillaha et al., 1988; Robinson et al., 1996), simulated pasture (Lim et al., 1998), manure applied pasture (Chaubey et al., 1994, 1995), livestock stockpile (Fajardo et al., 2001), and cropland runoff (Dillaha et al., 1989). In most of these studies, the VFS received runoff either after passing through the settling basin or field applied manure. A wide variability in the VFS effectiveness to remove sediments and nutrients was noticed in all of these studies. Typically, buffer performance depends on soil type and condition, vegetation type and condition, buffer strip length, buffer slope, flow type, influent solids concentration, and particle size distribution (Mankin et al., 2006). Depending on the geographical region, some of these buffer design criteria varied significantly. Recently, significant interest has grown in using VFS without sediment settling basin because of low installation and maintenance costs, as well as eliminating the acreage required for a settling basin. As a result, buffer performance without settling basin needs to be evaluated based on local and regional climatic condition and design criteria. Very limited studies have been conducted to assess the VFS performance at the down slope end of a beef feedlot in mitigating solids and nutrients from feedlot runoff.

The objective of this study was to evaluate the performance of a vegetative filter strip without settling basin in minimizing solids and nutrients concentrations in runoff from a feedlot under eastern North Dakota climatic conditions and management practices.

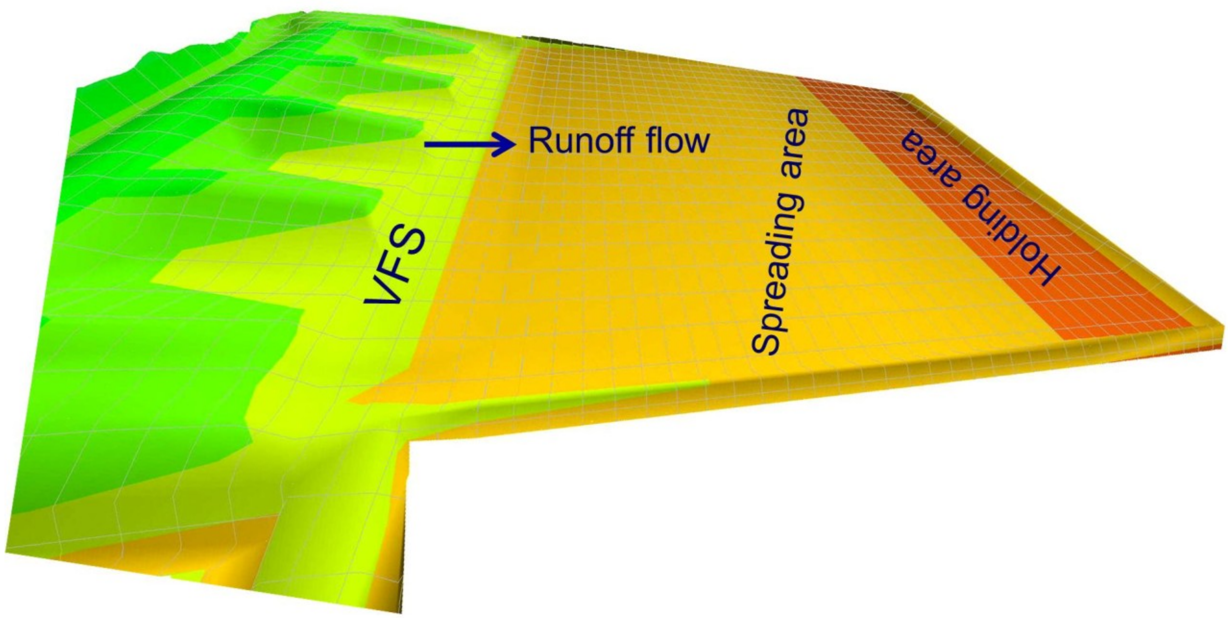
3.3. Materials and Methods

3.3.1. Study site

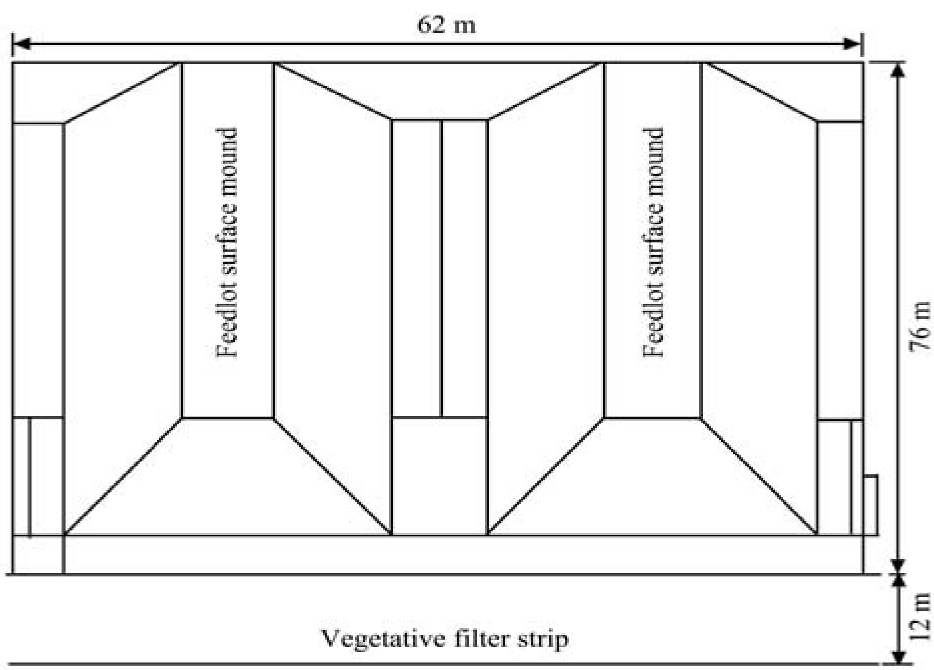
The study site was located in Richland County (46.5637, -97.1406), about 65 kilometers south-west of Fargo, North Dakota. The average annual rainfall in the study area is 468 mm (based on NDAWN). Feedlot soil type is sandy loam and classified as hydrologic soil group A. This feedlot was designed for 500 head of beef cattle with two pens, but only one pen was operational, and runoff samples were collected from that pen only. The length and width of the pen were 76 and 62 m, respectively, and overall aggregate slope of the feedlot about 5% was achieved by incorporating mounds in the pen, with a perception that liquid component will be separated quickly from solids component at a steeper slope, and buffer effectiveness at the end of pen surface will be increased as a result. A 12 m long (in the direction of flow) grass buffer strip was installed down slope of the feedlot with an assumption that runoff from the feedlot will pass through the buffer strip and maximize pollutant retention and then be dispersed evenly throughout the water spreading area. The VFS consisted of mixed vegetation including barnyard grass (*Echinochloa crus-galli),* ladysthumb smartweed (*Polygonaceae persicaria*), common lambs quarter (*Chenopodium berlandiery Moq.)* mares tail weed (*Conyza canadensis),* common ragweed (*Ambrosia artemisiifolin),* yellow foxtail (*Setaria glauca*), and white clover (*Melilotus alba*) and had uniform slope of 2% along the flow direction. The water spreading area was graded with an average slope of less than 1% for the water flowing downslope as shown in figure 3.1. The wastewater is contained in a holding area within a dike system (fig. 3.1), so that no pollutant or runoff is discharging from the feedlot area. This system was designed to contain the runoff from 25- year 24-h rainfall event as state regulations required (NDDoH, 2005).

3.3.2. Experimental procedure

In this study, a section of buffer was selected, and earthen borders were established to collect incoming runoff from the feedlot pen surface to the buffer area and from the buffer to the runoff spreading area (fig. 3.1). The earthen borders were established to separate and prevent mixing of runoff from outside of the buffer areas. Automatic ISCO 6712 samplers (Teledyne ISCO, Inc., Lincoln, NE) were installed to collect feedlot runoff entering into the VFS (hereafter inflow) and to collect runoff leaving the VFS area (hereafter outflow) to spreading area. ISCO samplers were operated with a heavy duty marine battery, which was charged by using a solar panel. A 60 liter bucket was installed at each runoff collection locations to accumulate the flow, and samples were collected from the bucket using the ISCO samplers, which was activated through using a float. The float was installed inside the bucket at a height from the bottom of the bucket to make sure that the bucket had enough water to collect specified sample volume (750 m L). After the first sampling, subsequent samples were collected at hourly as programmed. When the ISCO sampler malfunctioned, grab samples were collected. After a runoff event, runoff collection buckets were emptied and reinstalled to collect runoff from the next rainfall-runoff event during the study period. Immediately after collection, samples were brought back to laboratory and kept refrigerated until analyses were done. Temperature and precipitation data were downloaded from a nearby weather station (<2km) of North Dakota Agricultural Weather Network (NDAWN, 2013) during the study period.



(a)



(b)

Figure 3.1. Layout of the feedlot, buffer, and water spreading area (a) and plan showing dimensions (b).

3.3.3. Sample analysis

Using standard methods (APHA, 2005), runoff water samples were analyzed for nutrients, solids, pH, and electrical conductivity (EC). pH and conductivity were analyzed using a hand held meter (YSI Pro Plus, YSI Inc., Ohio, USA). Solids and nutrients were analyzed at the North Dakota State University Soil Testing Laboratory. Data were pooled and pair-wise means were compared between inflow and outflow using Duncan's multiple range tests at $P<0.05$.

3.4. Results and Discussion

3.4.1. Background information

Runoff samples from seventeen rainfall events were collected during the monitoring period. The effectiveness of the VFS was measured as a function of its capacity to reduce solids and nutrient concentrations. As mentioned previously, all runoff samples were not collected using automatic sampler due to instrument malfunctioning. In that case, grab samples were collected from runoff collection buckets. Total precipitation during each sampling events are presented in appropriate figures. Table 3.1 provides average key soil properties of the VFS area.

Table 3.1. Key soil parameters of the study site.

Parameters	Value
pH	$7.02 \pm 0.34^{\dagger}$
Electrical conductivity, EC (µS/cm)	64.7 ± 39.0
Vertical hydraulic conductivity (cm/s)	$4.34 \times 10^{-4} \pm 4.08 \times 10^{-4}$
Bulk density (g/cm^3)	1.14 ± 0.11

†Standard deviation

3.4.2. pH

Average pH of runoff samples for the different sampling events are shown in figure 3.2, and overall averages during the entire sampling period are reported in table 3.2. The pH values found were in the range observed by others (Miller et al., 2004; Gilley et al., 2007). As shown in figure 3.2, the pH of the inflow and outflow samples varied slightly, but the differences were not statistically significant. Figure 3.2 shows that pH increases after each rainfall and its magnitude varies with rainfall. An apparent increasing trend of pH was observed from the beginning to the end of this monitoring period likely due $CaCO_3$, which is used with feed ration (Gilley et al., 2007). High pH noticed at the beginning and at the end of runoff period was also reported by Hay et al. (2006). In addition, nitrification and denitrification processes may have some effects on the variation of pH, although they were not measured. Overall pH values at the inflow and outflow sampling locations were similar.

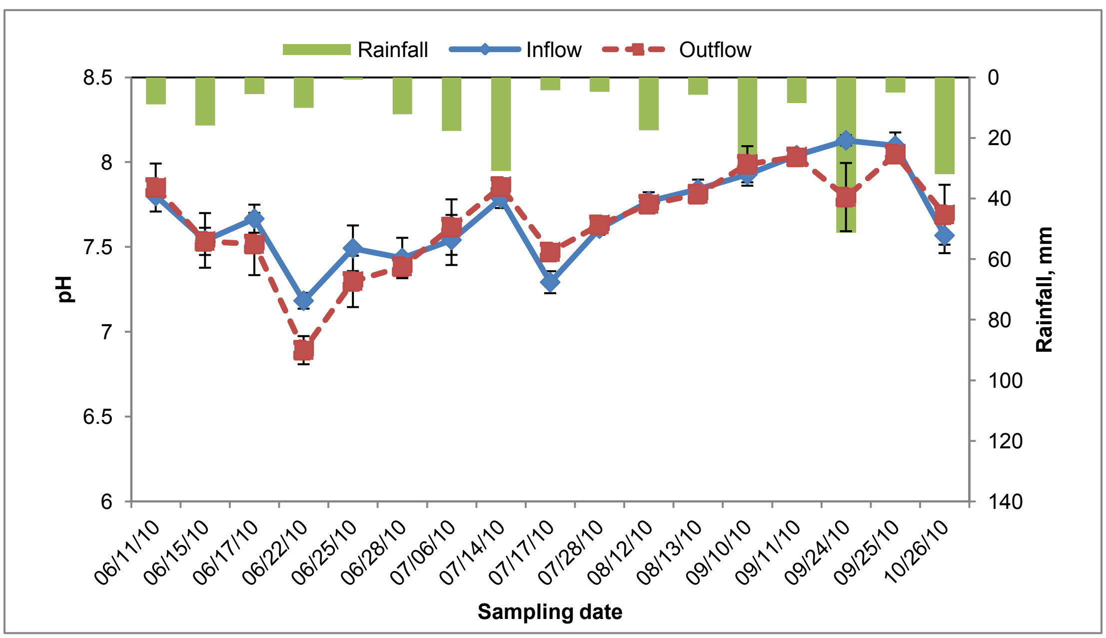


Figure 3.2. pH trend in runoff water samples at different sampling events (Error bars represent standard deviation of mean).

Table 3.2. Overall averages and standard deviations of different parameters measured during the entire sampling period at inflow and outflow runoff samples.

Variable	Inflow	N‡	Outflow	N	% reduction
pH	7.69a†±0.29	187	7.69a±0.29	216	-
Conductivity, μS cm^{-1}	2084a±782	187	1761b±956	217	-
TS, mg L^{-1}	3703a±1937	187	2454b±1422	218	33.73
TSS, mg L^{-1}	1252a±1704	181	401b±686	218	67.97
TP, mg L^{-1}	25.1a±8.8	177	17.6b±10.4	215	29.87
Ortho-P, mg L^{-1}	17.2a±7.4	173	13.9b±8.0	196	19.27
NH_4-N, mg L^{-1}	13.8a±11.4	173	9.43b±10.1	216	31.76
TKN, mg L^{-1}	112a±56.1	177	72.5b±57.1	215	35.56
K, mg L^{-1}	5074a±237	177	406 b±281	216	19.80

† Averages within a row followed by different letters are significantly different at $P \leq 0.05$ according to a Duncan's multiple range tests.

‡N - number of samples

3.4.3. VFS effectiveness in solids transport reduction

Average concentrations of total solids (TS) and total suspended solids (TSS) at the inflow and outflow during sampling events are shown in figures 3.3 and 3.4, respectively. Overall average concentration and concentration reduction of TS and TSS are presented in table 3.2 and figure 3.5, respectively. Total solid concentrations in the inflow and outflow samples fluctuated

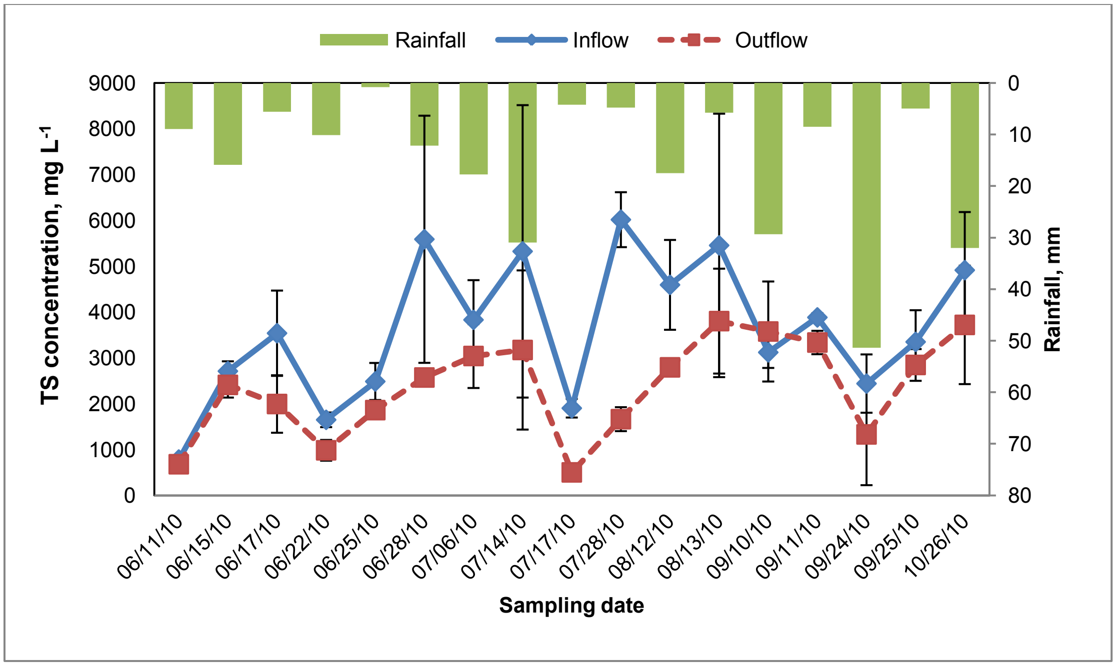


Figure 3.3. Variation in average TS concentration during different sampling events (Error bars represent standard deviation of mean).

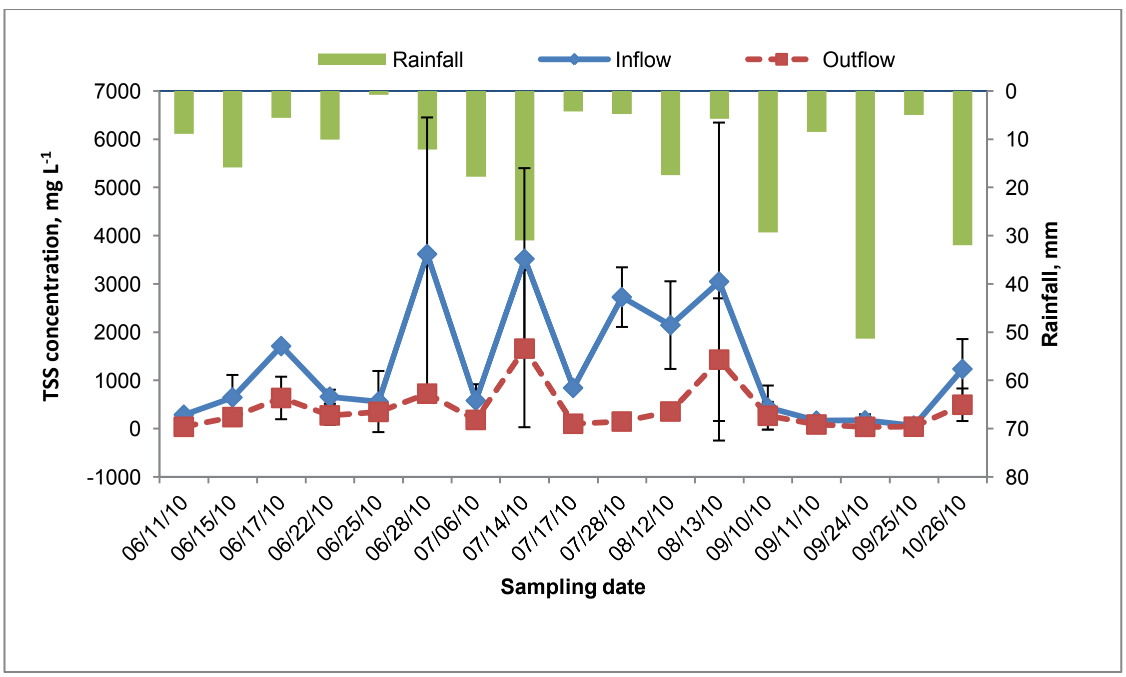


Figure 3.4. Variation in average TSS concentration during different sampling events (Error bars represent standard deviation of mean).

with rainfall as shown in figure 3.3. The vegetative filter strip was effective in reducing TS and TSS concentrations between inflow and outflow samples, except for a few occasions, when inflow and outflow could not be clearly separated due to excessive runoff from specific rainfall events. A similar trend is also observed for TSS (fig. 3.4). Typically, runoff amount and pollutant concentration depend on the antecedent soil moisture condition prior to a rainfall (Duchemin and Hogue, 2009). In this study, following a rainfall event (>5 mm), TS concentration in the runoff samples increased as compared to previous concentrations, which was expected. It is likely that decreased surface water flow resulted in deposition of sediment and absorbed potential pollutants (Stout et al., 2005). Overall, outflow TS and TSS concentrations were significantly lower than the inflow concentrations (table 3.2). This means that the VFS at the end of feedlot pen surface was effective in intercepting sediment. From these observations, it appears that VFS without settling basin might be effective in minimizing sediment-bound nutrients in runoff transport.

Total solids (TS) concentration ranged from 781 to 6017 mg L^{-1} and 501 to 3803 mg L^{-1} in the inflow and outflow, respectively. The results of this study are consistent with other studies. Dickey and Vanderholm (1981) measured TS in effluent runoff from a VFS with dairy facility and a beef feedlot and values reported 996 and 4710 mg L^{-1}, respectively. Similarly, TSS concentrations in runoff samples ranged from 61.9 to 3618 mg L^{-1} at the inflow and 35.5 to 1658 mg L^{-1} at the outflow samples.

When concentration reduction was averaged over the entire sampling period, TS concentration reduction (33.7%) was not as effective as the TSS concentration reduction (68.0%). This might be due in part to concentrated flow and physical obstruction provided by the vegetation because the buffer is effective in removing suspended solids compared dissolved solids. Other researchers observed 73% and 63% TS concentration reductions from a 91 and 61 m long VFSs for dairy facility and beef feedlots (Dickey and Vanderholm, 1981), respectively, and 76.5% TS concentration reduction from a 26 m long VFS (Schwer and Clausen, 1989). In our study, TSS concentration reduction ranged from 37.0% to 94.7%, which agreed with others findings. Schellinger and Clausen (1992) and Schwer and Clausen (1989) observed a 3.6% TSS concentration reduction from a dairy farm barnyard runoff and a 92% reduction from a VFS with milk house wastewater, respectively. It is important to note that in other studies, effluent was captured in a settling basin prior to the runoff entering into a VFS, whereas in this study, runoff from the feedlot directly ran through the buffer. Similarly, Andersen et al. (2009) observed 26% to 95% reduction of TSS concentration in runoff from six beef feedlots in Iowa, USA where settling basins were used for solids separation. Although, in this study, no settling basin was used before the VFS, a 12 m buffer strip itself was effective to retain a significant amount of solids within the buffer area. It is likely that the buffer provides a means of physical separation of suspended solids, reduces transport energy and deposits sediment, and increases infiltration of dissolved constituents into the buffer as was also concluded by Hay et al. (2006).

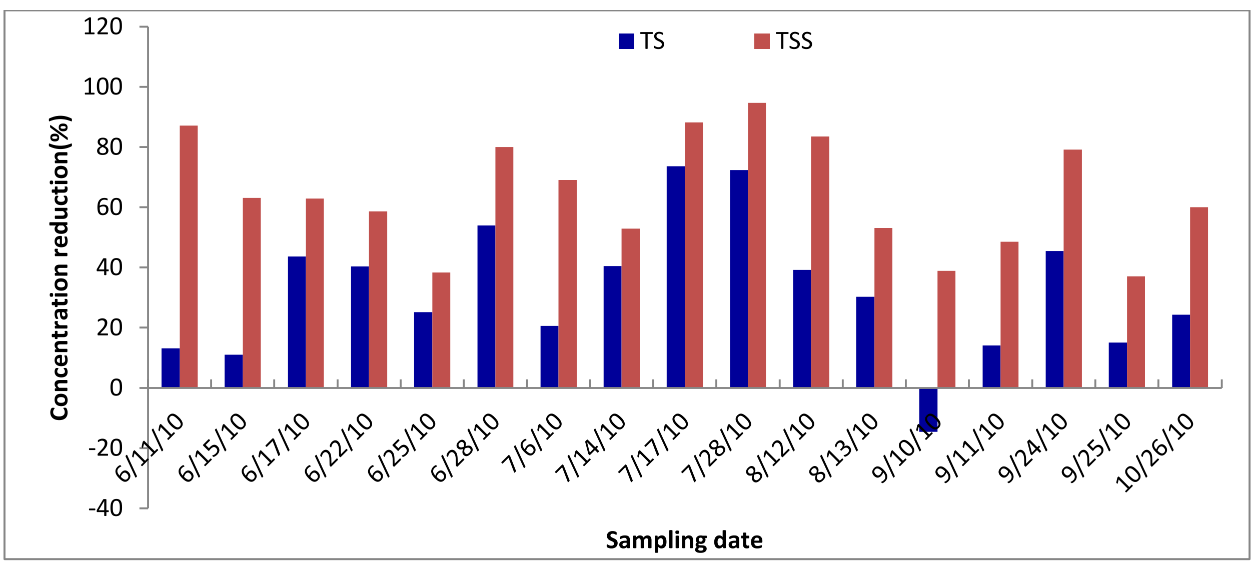


Figure 3.5. Transport reductions of runoff TS and TSS in different runoff events.

3.4.4. VFS effectiveness in nutrients transport reduction

Variations in total phosphorous (TP) and ortho-phosphorous (ortho-P) concentrations in runoff samples are shown in figures 3.6 and 3.7, respectively. Total phosphorus concentration-trends followed the same trend as TS. Total phosphorus concentrations ranged from 5.98 to 36.1 mg L^{-1} and 0.28 to 29.1 mg L^{-1} in the inflow and outflow samples (fig. 3.6), respectively. Similarly, ortho-P concentrations varied from 2.25 to 27.3 mg L^{-1} at the inflow and 0.48 to 23.2 mg/L at the outflow from buffer (fig. 3.7). Other researchers also found that TP concentration in incoming runoff into the buffer varied from 20.0 to 81.5 mg L^{-1} from a dairy facility, whereas ortho-P concentration varied from 16.2 to 54.6 mg L^{-1} (Schwer and Clausen, 1989; Schellinger and Clausen, 1992). Andersen et al. (2009) observed 53 to 222 mg L^{-1} TP and 28 to 101 mg L^{-1} ortho-P concentrations in influent runoff to the VFS. The relatively lower concentrations of TP and ortho-P observed in this study may be due to the differences in feedlot soil types and diet. On an average, both in the inflow and outflow samples, the ratio of ortho-P/TP ranged from 0.21 to 0.94 and 0.65 to about 1.0, respectively, which means that a significant portion of TP was

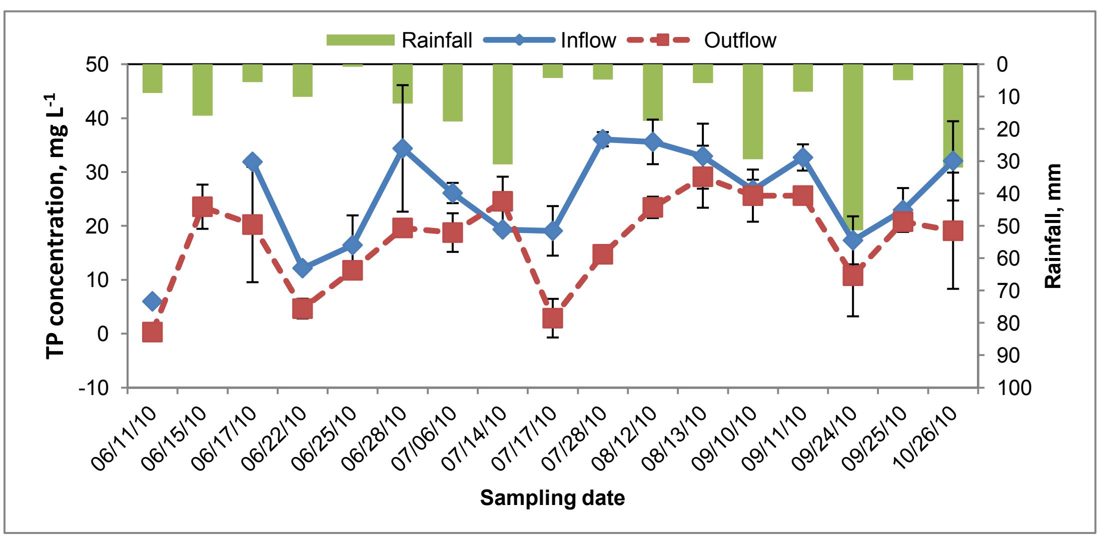


Figure 3.6. Variations in average TP concentration and standard deviation at different sampling events.

soluble phosphorus. It is noted that the ratio of ortho-P/TP was increased in the outflow compared to inflow for most of the runoff events indicating that particulate bound P was retained in the VFS with settled sediments. A small portion of soluble P tended to be captured by the buffer during low runoff flow rates with reduced concentrations at outflow.

Outflow concentrations of TP on 14 July (fig. 3.6) and ortho-P on 6 and 14 July and 26 October (fig. 3.7) were higher than the inflow. This was likely due to grab sampling, as well as flushing effect. For those dates, the buffer area was inundated due to high runoff contributing to flushing that might result in a greater nutrient concentration at the outflow. As waste settled and was retained in the buffer areas, the organic phosphorus may have mineralized to inorganic phosphate compounds (Spellman and Whiting, 2007). Mineralization processes may convert TP into soluble P which mixes with outflow runoff and increased the soluble P contribution in the outflow samples (Dillaha et al., 1988). Moreover, outflow P concentration might be increased

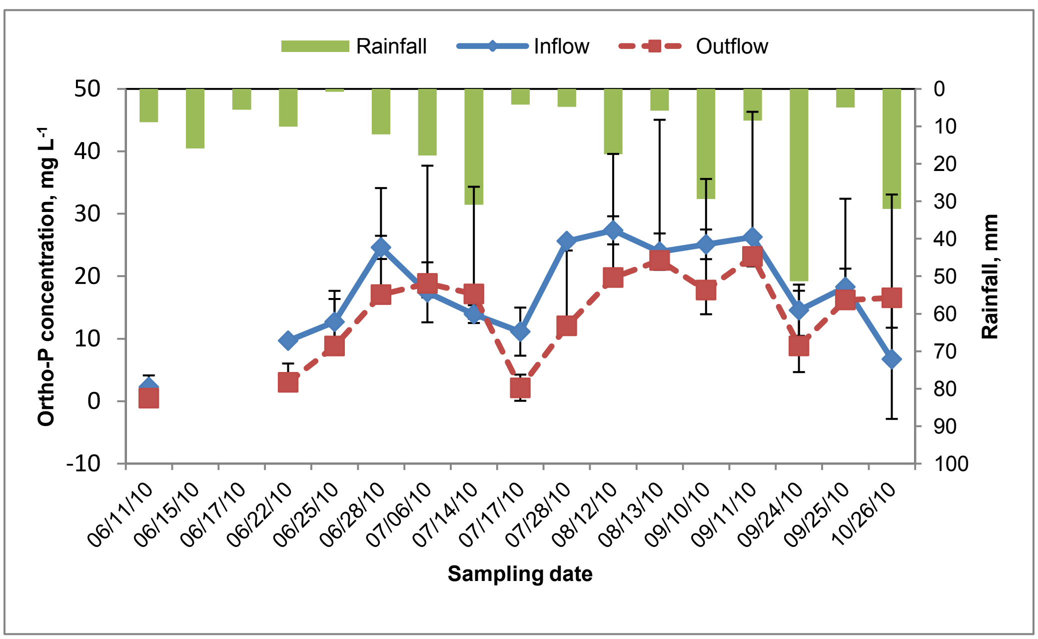


Figure 3.7. Variation in average ortho-P concentration and standard deviation at different sampling events.

due to desorption from the already moist soil, which was previously P enriched. During a low rainfall situation, as runoff passed through the buffer, sediment-bound P is likely to be deposited and soluble P is likely to infiltrate into the buffer soil thereby reducing concentration at the outflow. Other researchers (Schellinger and Clausen, 1992; Hawkins et al., 1998) also observed increased soluble phosphorous concentrations at the outflow sampling location as compared to inflow concentration. Usually, runoff-pollutants dissolved in rainwater is a significant transport mechanism for water soluble pollutants (Spellman and Whiting, 2007) resulting in increased concentration in the outflow.

On an average, TP and ortho-P concentrations reduction ranged from 4.02% to 95.3% and 5.91% to 80.9%, respectively (fig. 3.8). A similar TP reduction trend has also been observed by other researchers. Andersen et al. (2009) measured buffer performance from six beef feedlots in Iowa State, USA and observed TP concentration reductions ranged from 38% to 94% and

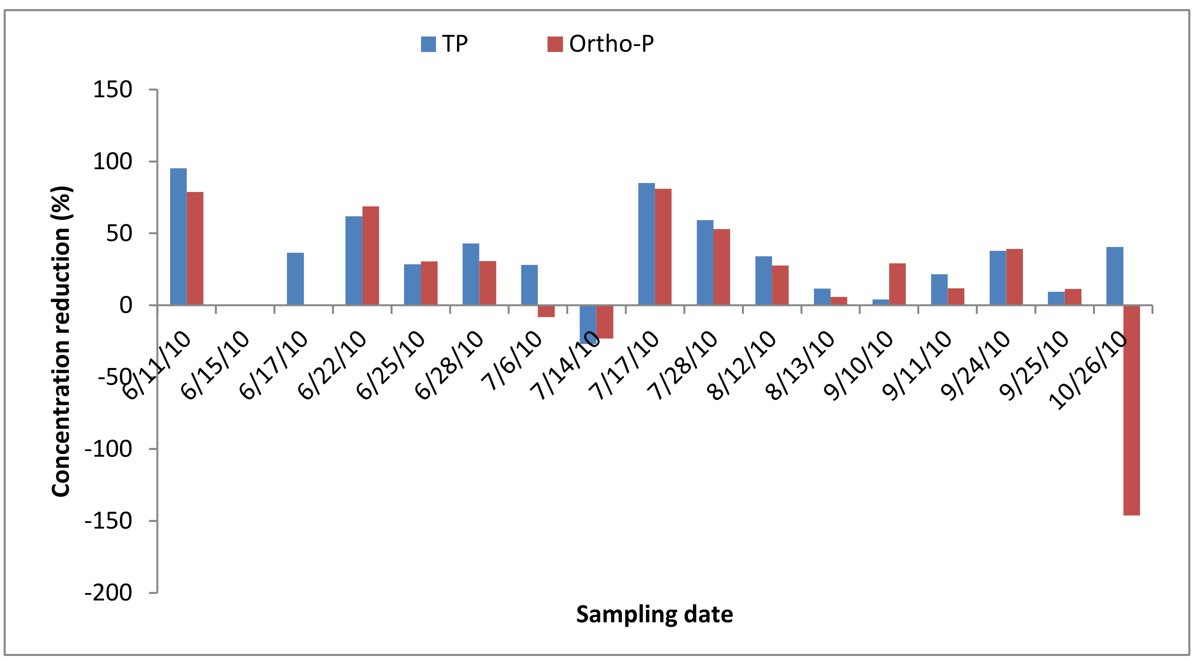


Figure 3.8. Variation in TP and ortho-P concentration reduction averaged over each sampling event.

ortho-P concentration reductions ranged from 33% to 92%. Overall, the buffer was effective in reducing TP and ortho-P concentrations by 29.9% and 19.3%, respectively.

Figure 3.9 shows the variation in NH_4-N concentrations. Significant variation in NH_4-N concentration was observed between inflow and out flow samples (table 3.2). The NH_4-N and NH_3-N are pH dependent. Under acidic condition, the uptake will be NH_4-N and under alkaline condition that of NH_3-N. Although plant biomass samples were not collected and analyzed during the monitoring period, the uptake of NH_4-N by plants and adsorbed in soil might (Koelsch et al., 2006) have contributed to lower NH_4-N concentrations in the outflow runoff, since pH during the monitoring period was slightly alkaline (fig. 3.2).

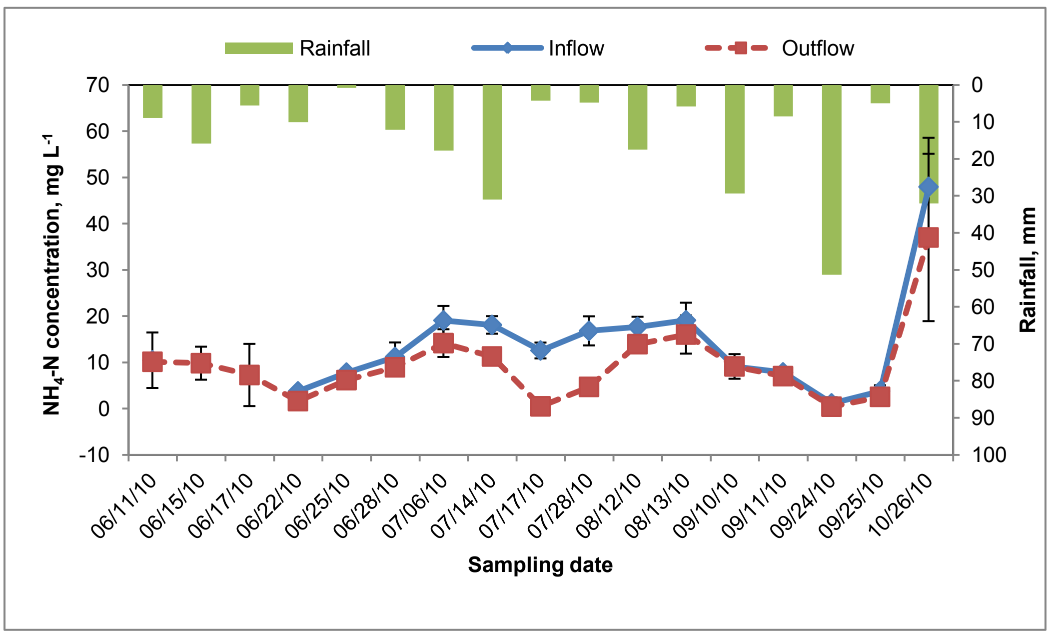


Figure 3.9. Variation in average NH_4-N concentration and standard deviation of mean at different sampling events.

Figure 3.10 shows the variation in NO_2-N + NO_3-N concentrations. Except for anomalies on 15 and 17 June, NO_2-N + NO_3-N concentrations between inflow and outflow were consistent and followed the same trend. The anomalies on 15 and 17 June were unknown. Outflow NO_2-N

+ NO_3-N concentrations were slightly higher than the inflow concentration, but the differences were not statistically significant. Increased nitrate nitrogen at the outflow has been observed in many studies (Dillaha et al., 1988; Mendez et al., 1999; Andersen et al., 2009; Young et al., 1980), which are likely due to mineralization of particulate organic N that is trapped and accumulated in the buffer resulting in increased soluble N over time (Mendez et al., 1999). In this study, except for a few occasions, NO_3-N concentrations were lower than the environmental protection agency (EPA) threshold value (10 mg L^{-1}), meaning that NO_3-N concentration in runoff was not a concern. For soluble nutrients, a longer VFS might be required to enhance infiltration volume within the buffer because NO_3-N reduction primarily occurs due to dilution and infiltration.

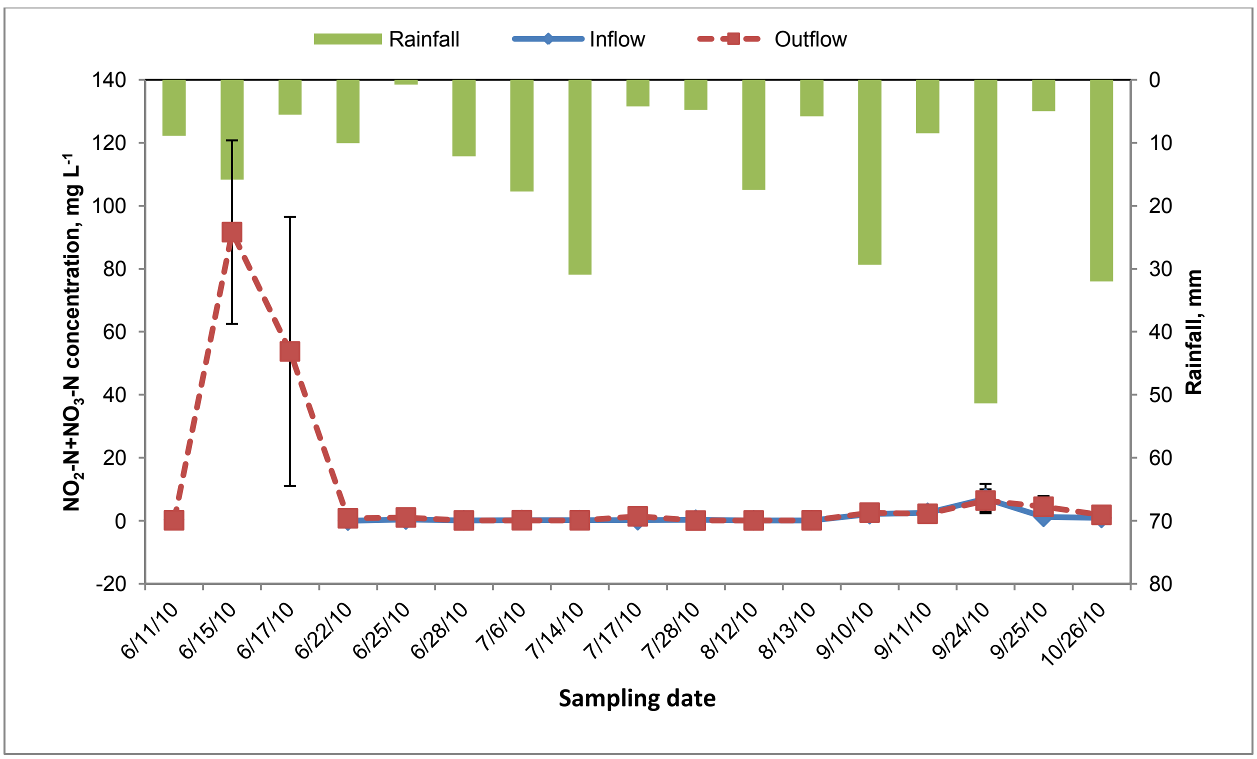


Figure 3.10. Variation in average NO_2-N+NO_3-N concentration and standard deviation of mean at different sampling events.

Average concentrations of TKN during sampling events are presented in figure 3.11, and overall concentrations across all sampling events are presented in table 3.2. Total Kjeldahl nitrogen concentration varied significantly between inflow and outflow samples, and outflow samples had lower concentration than the inflow except for a few occasions. Total Kjeldahl nitrogen was also strongly correlated with total solids (R^2 = 0. 70, data not shown) indicating that reduction of sediment would result in sediment-bound nutrients reduction. Overall, VFS effectively reduced TKN by 35.6%. During the runoff sampling events, the concentration reductions for NH_4-N, NO_2-N + NO_3-N, and TKN are shown in figure 3.12.

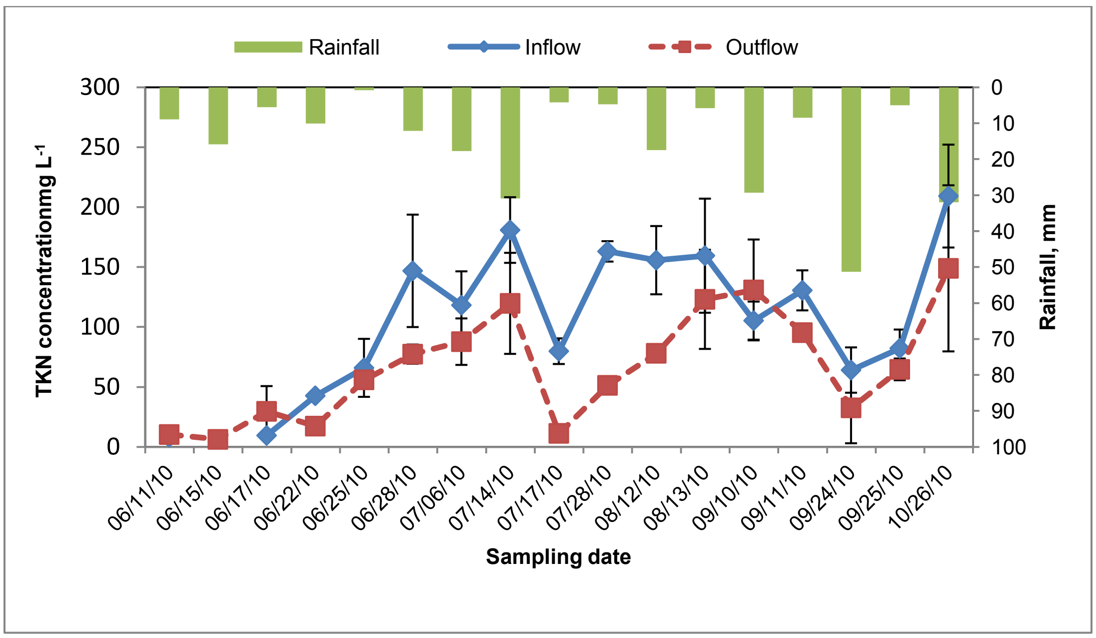


Figure 3.11. Variation in average TKN concentration and standard deviation of mean at different sampling events and corresponding rainfall.

Potassium concentration at the inflow and outflow samples ranged from 43.3 to 854 and 20.7 to 713 mg L^{-1}, respectively (fig. 3.13). It is also evident in figure 3.13 that the potassium concentration at the outflow was higher as compared to inflow on 10 September, which may be due to variation of sampling technique, i.e., grab vs. automatic sampling by the sampler.

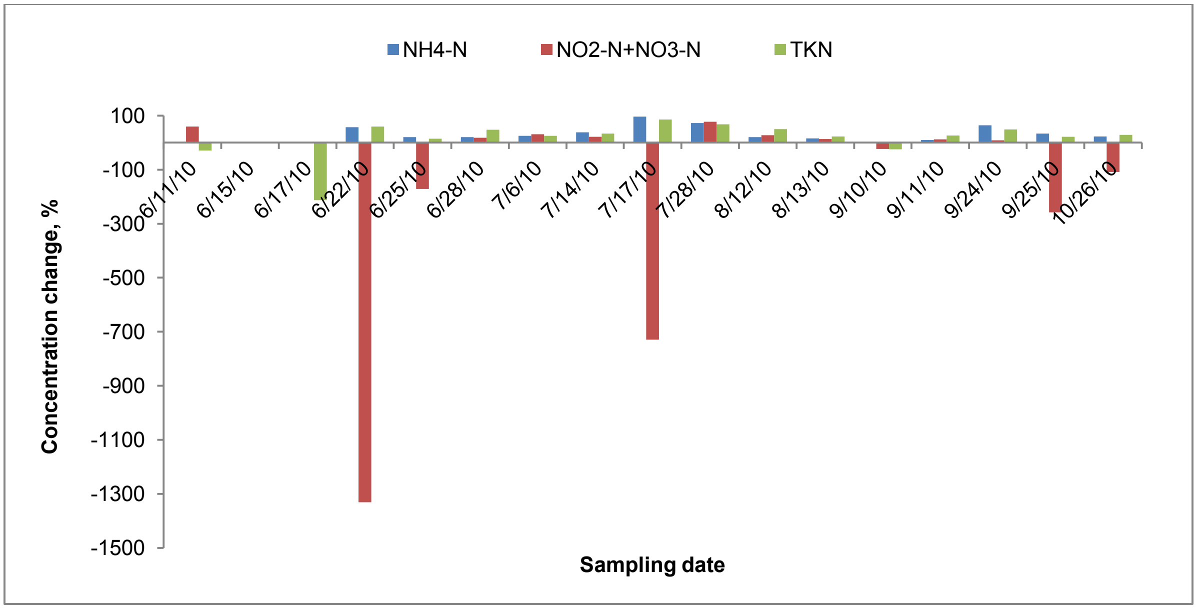


Figure 3.12. Concentration reductions of NH_4-N, NO_2-N + NO_3-N, and TKN at different sampling events.

Dickey and Vandeholm (1981) reported K concentrations at the entry and exit of a VFS were 665 and 168 mg L^{-1}, respectively, and K values in this study were consistent with other studies. Hawkins et al. (1998) conducted VFS studies with swine lagoon wastewater on 11% and 5% buffer slopes and observed K concentration reductions of 5% and -17%, respectively. Since potassium is highly soluble, its concentration reduction potential is usually low. Overall, in this study, K concentration reduction was 19.8%, which was lower than other nutrient concentration reductions.

3.4.5. Conductivity

The average electrical conductivity for inflow and outflow samples of VFS is presented in figure 3.14, where conductivity fluctuated throughout the monitoring period, and the buffer appeared to result in a slight reduction in EC levels. A sharp increase in EC concentration was observed during 6 July and 11 and 25 September, which was likely due to greater amount of nutrients present in runoff at that time compared to the previous sampling since dissolved mineral salts (Stevens et al., 1995; Scotford et al., 1998; Yayintas et al., 2007) change conductivity. Typically, when dissolved matter in soil solution increases, conductivity increases. Conductivity and K exhibited a correlation at inflow (R^2=0.52) and outflow (R^2=0.78) sampling locations. Scotford et al. (1998) observed a stronger correlation (R^2=0.80) between K and EC. Overall conductivity was reduced by 16.3%. Again, the buffer was not very effective in reducing soluble constituents. Probably, buffer length should be increased to enhance infiltration of soluble constituents within buffer; eventually, better buffer performance can be achieved.

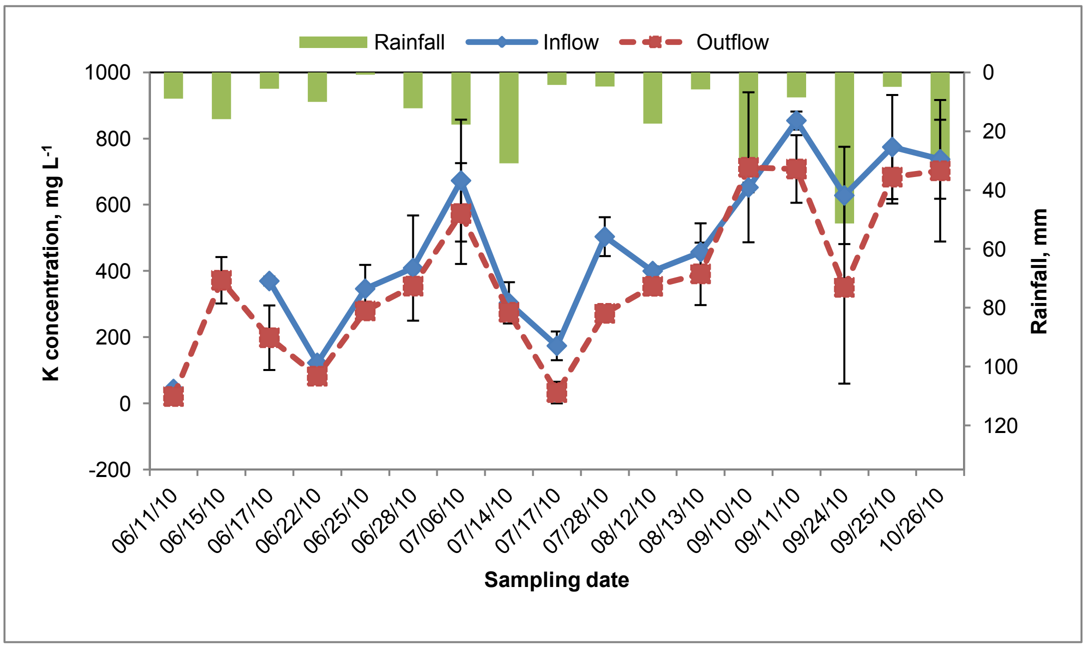


Figure 3.13. Concentration of potassium during different sampling events (Error bars represent standard deviation of means).

3.5. Conclusions

Based on above results and discussion the following conclusions can be made:

- A vegetative filter strip without settling basin was effective in reducing solids and nutrients concentrations from feedlot runoff water, except for soluble nutrients.

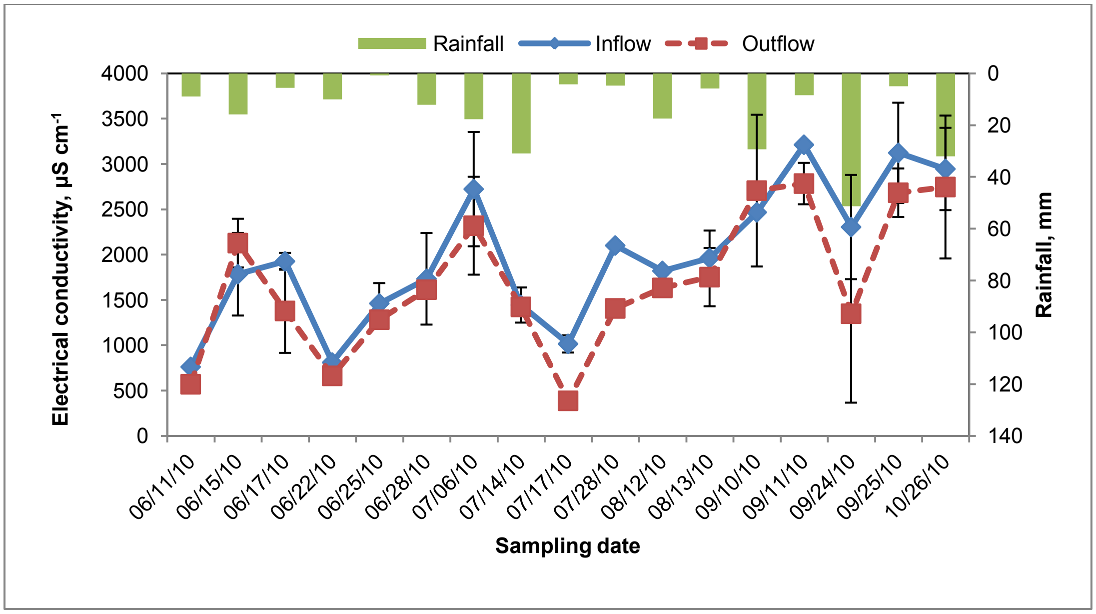


Figure 3.14. Specific electrical conductivity in runoff samples during different sampling events (Error bars represent standard deviation of means).

- On an average, the VFS was able to reduce TS concentration by 33.7%, TSS by 68.0%.
- Total phosphorus and ortho-P concentration reductions were by 29.9% and 19.8%, respectively, whereas potassium concentration reduction was 19.8%.
- Similarly, NH_4-N and TKN concentration reduction was 31.8% and 35.6%, respectively.
- The buffer was not effective in reducing NO_2-N + NO_3-N although the level of these two constituents was very low.
- A longer VFS might be beneficial to enhance infiltration and soluble pollutant removal efficiency.

3.6. Acknowledgements

This project was supported by the North Dakota Department of Health and EPA-ND Section 319 (h).

CHAPTER 4. PERFORMANCE EVALUATION OF THREE VEGETATIVE FILTER STRIP DESIGNS FOR CONTROLLING FEEDLOT RUNOFF POLLUTION[2]

4.1. Abstract

A vegetative filter strip (VFS) is designed to reduce transport of sediments and nutrients downstream mainly through settling, infiltration (into soil profile), adsorption (to soil and plant materials), and by plant uptakes. However, the performance of a VFS greatly depends on a VFS design and climatic conditions of a region. In this paper, relative performance of three VFSs (hereafter Cass County-CC, Sargent County-SC, and Richland County-RC buffers) was evaluated and compared in the context of VFS design for feedlot runoff pollution control and management under agro-climatic condition of North Dakota. The buffer at the CC feedlot was established with broadleaf or common cattail (*Typha latifolia)* grass filter, the SC feedlot buffer had Garrison creeping foxtail (*Alopecurus arundinaceus* Poir.) and reed canary grass (*Phalaris arundinaceus*), and the RC feedlot buffer had mixed grasses. Automatic samplers were installed to collect runoff samples at each inflow and outflow location. Collected runoff samples were analyzed for total suspended solids (TSS), ortho-phosphorus (ortho-P), total phosphorus (TP), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), total Kjeldahl nitrogen (TKN), total nitrogen (TN), and potassium (K). The Cass County (CC) VFS with cattails grass filter had the longest runoff-flow length (65 m) and resulted in a more conducive environment for restricted TSS and TP transports reduction and better adsorption of ortho-P, NH_4-N, and K compared to the SC and RC feedlot buffers. Overall TSS, ortho-P, TP, NH_4-N, and K removal efficacies were 88%, 90%, 89%, 91% , and 90%, respectively, at CC VFS. At the SC feedlot, the VFS resulted

[2] This material was co-authored by Atikur Rahman, Shafiqur Rahman, and Md. Saidul Borahn (Published in J. Civil and Environmental Eng., 3: 124. doi:10.4172/2165-784X.1000124)

in the highest NO_3-N reduction. Relatively poor performance was observed for the RC feedlot which was due to smaller runoff-flow length (12 m). Overall, the CC feedlot outperformed the SC and RC VFSs in respect of TSS, ortho-P, TP, NH_4-N, TKN/TN transport reduction.

Keywords: Vegetative filter strips, Feedlot, Runoff, Nutrients, Buffer performance, Solids, Pollution control

4.2. Introduction

Runoff from open animal feeding operations has long been known as a source of ground and surface water pollution. Runoff from feedlots may carry significant amount of manure borne nutrients (e.g., nitrogen and phosphorous), suspended matter, and pathogens to surface water (Swanson et al., 1971; Laws, 1993; Troeh et al., 2004). According to Koelsch et al. (2006), runoff from feedlots is a major contributor and will continue to be a contributor to surface and groundwater impairment. As per the North Dakota Department of Health 2010 integrated water quality assessment report, a significant portion of the state's surface water is either threatened or does not support the aquatic life use due to excessive nutrient loadings. The report also indicated that primary sources of nutrient loadings in state's surface water are erosion and runoff from cropland, hydrologic modification, and runoff from animal feeding operations (NDDoH, 2010). Nutrient-laden water that enters into surface water causes eutrophication, a condition that decreases dissolved oxygen and kills aquatic animals. Additionally, increased bacterial population, changes in water color, and odor development may affect recreational value. Mitigation of such pollution requires use of some practices or techniques that reduce the downstream discharge of nutrients contained in runoff from feedlots and land application sites.

Vegetative filter strips (VFS), also known as vegetative buffer strips (VBS) or simply buffers, are increasingly viewed as an attractive technology for improving the quality of runoff from pollutant source areas. However, different VFS designs exist either to meet the state regulatory needs or to reduce the installation costs. For an instance, beef cattle feedlots often use a lagoon or settling basin with vegetative filter strips to reduce runoff pollutant concentration and migration to surface water bodies (Mankin et al., 2006). Holding pond or lagoon construction is expensive, requiring large land area and regular maintenance. Moreover, seeping water from the containment structures possesses the risk of contamination of the potential drinking water (Parker et al.,1999). On the other hand, a VFS involves spreading and infiltration of runoff, thereby this system does not require any containment structure. The challenge of an effective VFS is to maintain the sheet flow; the systems fail if channelization occurs (Lorimor et al., 2002). While the cost comparison between a VFS and settling basin is difficult due to location, topography, and climatic conditions for both systems, but in general the cost involved in a VFS system is lower than other structures due to capital investment and maintenance (Kizil, 2010; Barrett, 1999). As a result, producers are often not interested to construct holding ponds due to high capital investment, especially in North Dakota, where annual average precipitation ranged from 305 to 610 mm (http://www.nationalatlas.gov/printable/images/pdf/precip/pageprecip_nd3 .pdf, accessed on 4/5/2013). Instead, significant interest has grown in using VFS without sediment settling basin because of low installation and maintenance costs, as well as eliminating the acreage required for a settling basin. However, limited information is available on the performance of VFS depending on different buffer designs.

The main goal of this study was to evaluate a comparative assessment of three different VFSs for their efficacy in removing solids and nutrients from the feedlot runoff under North Dakota climatic conditions and management practices.

4.3. Materials and Methods

Three existing feedlots were selected from different climatic regions of North Dakota (fig. 4.1), where early VFS design was slightly different and were established at the end of feedlot to control runoff pollutants (fig. 4.2). These feedlots with buffers have been identified as Richland County (RC), Cass County (CC), and Sargent County (SC) buffers in North Dakota (fig. 4.2). The salient features of three VFSs were presented as follows:

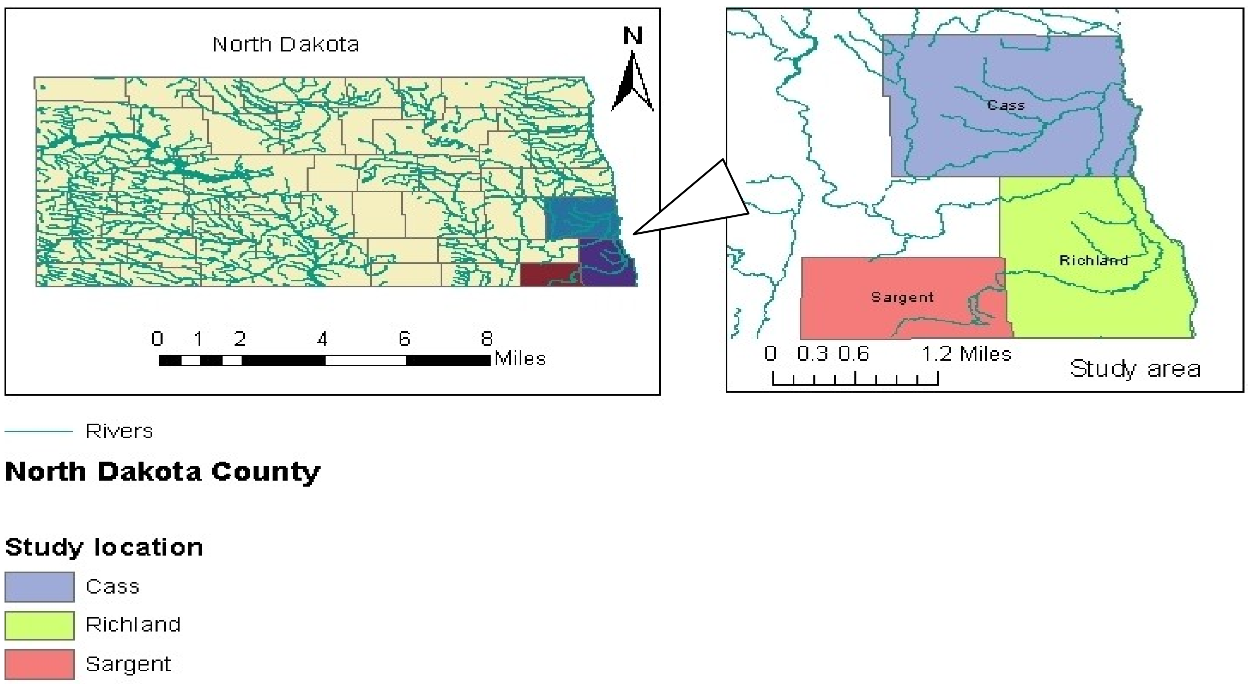


Figure 4.1. Locations of the study area.

4.3.1. RC feedlot buffer

The feedlot was designed for 500 head of beef cattle with two pens, but only one pen was operational, and runoff samples were collected from the operational pen only. The length and width of the pen were 76 and 62 m, respectively, and overall aggregate slope of the feedlot about 5% was achieved by incorporating mounds in the pen. Feedlot has sandy loam soil and classified as hydrologic soil group A. A 12 m long (in the direction of flow) grass buffer strip was installed

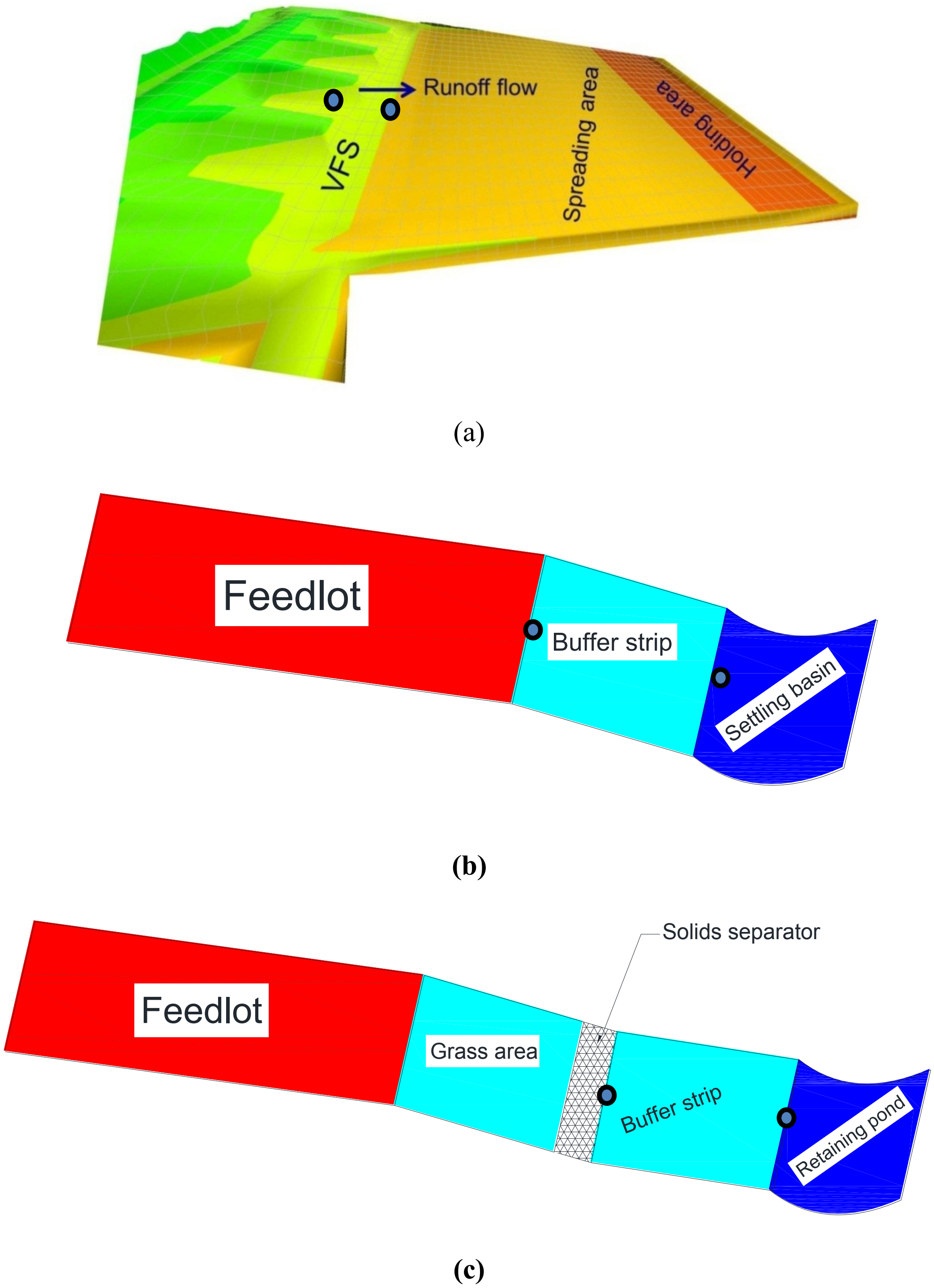


Figure 4.2. Layout of the feedlot, buffer, and water spreading area/settling basin a) RC feedlot without settling basin, b) CC feedlot with settling basin, and c) SC feedlot with solid separator. Small circles represent sampling locations. Figures are not to scale.

down slope of the feedlot with an assumption that runoff from the feedlot will pass through the buffer strip and maximize pollutant retention and then be dispersed evenly throughout the water spreading area (fig. 4.2a). The VFS consisted of mixed vegetation and it had uniform slope of 2%. A detailed description of the VFS has been outlined in a previous paper (Rahman et al., 2012).

4.3.2. CC feedlot buffer

The Cass County (CC) feedlot is located at the North Dakota State University Beef Research Center. This feedlot has a dimension of 115 m × 50 m with a maximum capacity 192 beef cattle. It had total six pens on clay soil and overall slope is about 5%. A 65 m long and 115 m wide vegetative filter strip was constructed immediately after the feedlot pen surface and an alley that ran along the width of the feedlot. The VFS was seeded with common cattails grass and graded to a uniform slope of 2% on clay soil. A settling basin was constructed at the end of the VFS to contain runoff exiting from the VFS (fig. 4.2b).

4.3.3. SC feedlot buffer

The Sargent County (SC) feedlot buffer is a two-stage VFS (fig. 4.2c). At the initial stage, runoff from the feedlot ran through an approximately 165 m long narrow grassed area and reached to a solids separator. Then in second stage, runoff from the solids separator was channeled through a pipe and spread onto a vegetative filter strip. The vegetative filter strip was 40 m long in the direction of flow. Smooth bromegrass (*Bromus inermis*) and western wheatgrass (*Pascopyrum smithii*) were seeded for the grassed area and garrison creeping foxtail and reed canarygrass were seeded for the filter strips. The overall slope of the VFS was 2% and it is established on fine sandy loam soil. At the end, runoff exiting from the VFS is contained in a retaining pond and used for irrigating croplands.

All three systems were designed to contain the runoff from 25 year 24-h rainfall event as state regulations required (NDDoH, 2005). The average annual rainfall for RC, CC, and SC locations are about 468, 494, and 494 mm, respectively, based on average of 21 years of data.

4.3.4. Sampling runoff

Each experimental site was equipped with automatic samplers (ISCO 6712, Teledyne ISCO Inc., Lincoln, NE) to collect runoff samples sequentially at one hour interval upon activation of the sampler. One sampler was installed to collect runoff at the entry of the VFS (hereafter inflow), and another sampler was installed at the exit of the VFS to collect runoff leaving the VFS (hereafter outflow). Samplers were powered by heavy duty marine batteries, which were charged by solar panels. Runoff in each sampling location was accumulated into a 60 liter bucket, and samples were collected from the bucket using ISCO samplers, which were activated via liquid level actuator (model: 1640, sampler actuator, Teledyne ISCO Inc., Lincoln, NE). The actuator sensor was installed inside the bucket at a height from the bottom of the bucket in such a way that the bucket had enough water to collect specified sample volume (750 mL). When automatic samplers malfunctioned, grab samples were collected from the bucket. After collecting runoff samples, buckets were emptied and reinstalled to collect runoff samples from the next runoff event. However, at the CC location, outflow samples were collected manually from the runoff settling basin. Immediately after collection, samples were brought to laboratory and kept refrigerated until analysis. Temperature and precipitation data for each location were downloaded from a nearby weather station of North Dakota Agricultural Weather Network (NDAWN, 2013) during the study period.

4.3.5. Sample analysis

Standard methods of analysis (APHA, 2005; HACH, 2007) were employed to analyze runoff samples for determining nutrients and solids concentrations, pH, and electrical conductivity (EC). Electrical conductivity and pH were analyzed using a handheld meter (YSI Pro Plus, YSI Inc., Ohio, USA). Solids and nutrients were analyzed at Soil and Water Testing and Waste Management Laboratories at North Dakota State University.

For solids, EPA Method 2540B was used for TS and EPA Method 2540D was used for TSS as described in APHA (2005). Briefly, approximately 200 mL of an unfiltered liquid sample was evaporated in an oven at 105°C for 24 h or until a constant weight was reached to measure TS. Similarly, according to EPA Method 2540D, a well-mixed runoff sample was filtered through a 0.45 micron glass fiber filter, and the unfiltered residue was heated at 105°C for 24h to measure TSS.

For runoff nutrient concentration, runoff samples were measured for ortho-P, TP, NH_4-N, NO_3-N, TKN, TN, and K. Methods/protocols used to analyze nutrient concentration of samples were summarized in table 4.1. When measured concentration exceeded the detection limit of a particular parameter by a particular method/protocol, the runoff samples were diluted and reported values were multiplied by the dilution number. As a measure of quality control, calibration standards and blanks were analyzed along with the samples at every ten samples where appropriate. Later on, the efficacies of the VFSs were judged based on percent reduction of each analyte as measured using the following relationship (equation 4.1):

$$\eta_{red} = \frac{C_i - C_o}{C_i} \tag{4.1}$$

Table 4.1. Method/protocol used to analyze runoff sample from feedlots.

Parameters (mg L^{-1})	Method/protocol used/Measurement range
Ortho-P [a]	QuikChem® Method 10-115-01-1-O (Lachat Instruments, Loveland, CO) Equivalent to EPA 365.1 method; 0-20 mg L^{-1}
NH_3-N [a]	QuikChem® Method 10-107-06-1-J (Lachat Instruments, Loveland, CO) Equivalent to EPA 350.1 method; 0-20 mg L^{-1}
NO_3-N [a]	QuikChem® Method 10-107-04-1-R (Lachat Instruments, Loveland, CO) Equivalent to EPA 353.2 method; 0-20 mg L^{-1}
K [b]	Hach Method 8049 (Tetraphenylborate); 0-7 mg L^{-1};
TP [b]	Hach Method 10127 (Molybdovanadate Method with Acid Persulfate Digestion); 1 -100 mg L^{-1};
TN [b]	Hach Method 10072 (Acid Persulfate Digestion); 2 -150 mg L^{-1};
TKN	APHA 2005 4500-N_{org} C (Semi Micro Kjeldahl Method)

[a] Equivalent EPA methods
[b] USEPA approved for reporting

where η_{red} is reduction efficiency in percent (%), C_i is the inflow concentration of a particular analyte, and C_o is the concentration of the same analyte in the outflow in ppm (mg L^{-1}).

4.3.6. Statistical analysis

The effectiveness of VFSs in controlling/reducing solids (TS and TSS) and nutrients (TN, TKN, TP, K, NH_4-N and NO_3-N) were compared using Analysis of Variances (ANOVA) technique in the SAS environment (SAS, 2009). The null hypothesis tested was that the mean concentrations of a parameter between inflow and outflow runoff for a particular year were equal. Yearly data were pooled and pairwise parameter means between inflow and outflow were compared using the Duncan's multiple range tests at $P \leq 0.05$, if the main effect (inflow and outflow of VFS) was significant at $P \leq 0.05$ for a parameter in the analysis of variance.

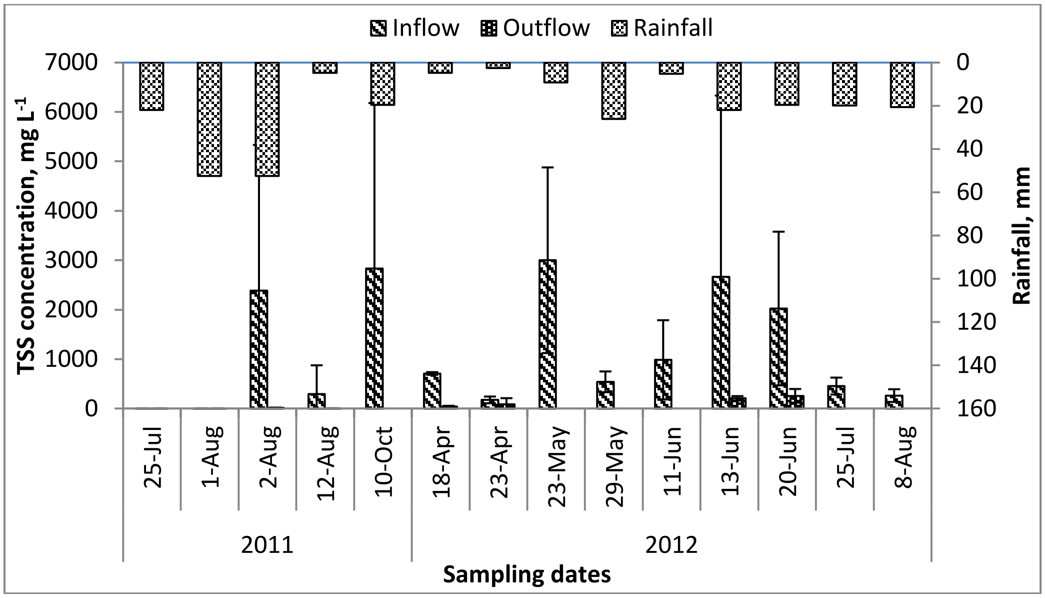
4.4. Results and Discussion

4.4.1. Solids transport reduction

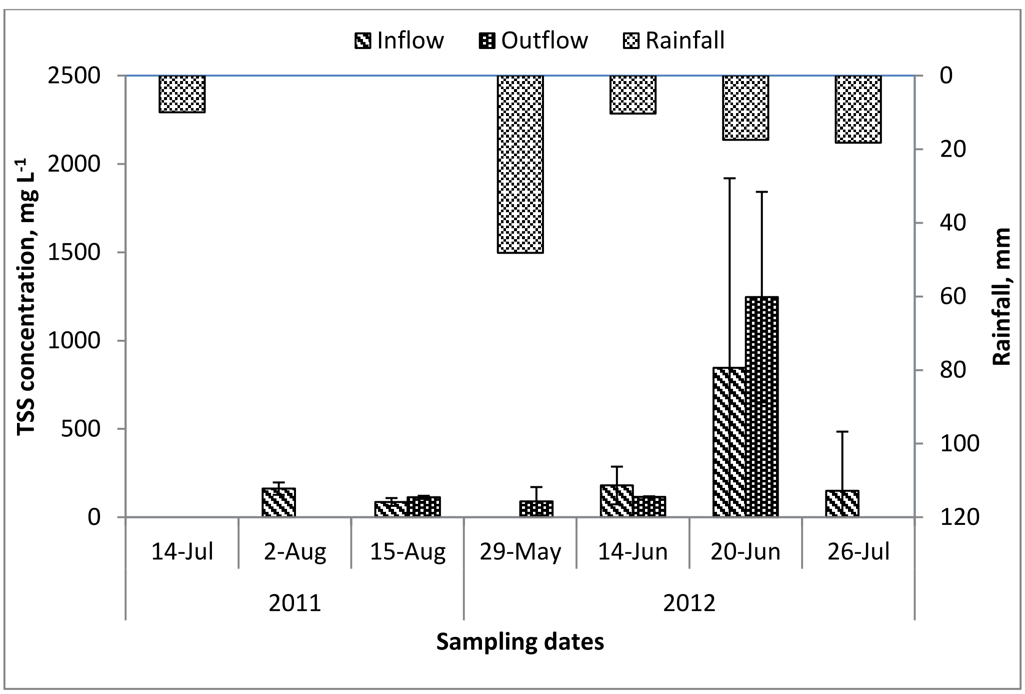
Efficacies of VFSs at the CC and SC feedlot locations in reducing TSS concentration are shown in figure 4.3(a) and 4.3(b), respectively. Concentration of TSS was significantly lower at outflow than at inflow in CC feedlot ($P < 0.05$). The TSS concentration in inflow varied from 0.01 to 3001 mg L^{-1} while at the outflow varied from 0.02 to 259 mg L^{-1}. From figure 4.3a, it was shown that the TSS concentration in runoff fluctuated with rainfall magnitude, which agrees with others findings where median pollutant load varied with rainfall magnitude (Duchemin and Hogue, 2009).

At the CC location, TSS transport reduction was usually high and a maximum 100% concentration reduction was considered when no flow exiting through the VFS following a rainfall event was observed. It is likely that decreased surface water flow resulted in deposition of sediment and absorbed potential pollutants (Stout et al., 2005). The outflow concentrations at CC feedlot in 2011 were low due to the fact that samples were collected from the settling basin in which TSS might have been settled and diluted with runoff from the surrounding areas. Similarly, TSS concentration was significantly lower at outflow than that at inflow in SC feedlot, except for the few rain events. Concentration of TSS in inflow and outflow ranged from 85.7 to 846 mg L^{-1} and 89.3 to 1246 mg L^{-1}, respectively, at SC feedlot.

For two sampling events at the SC feedlot, outflow TSS concentration was higher than the inflow concentration, which may be attributed to grab sampling from the bucket. On August 15 (2011) and June 20 (2012) grab samplings were performed at outflow locations followed by rain events. These grab samples might have contained high TSS because of diminishing runoff accumulated in sampling bucket (fig. 4.3b). In addition, on 29 May (fig. 4.3b), no inflow



(a)



(b)

Figure 4.3. Average TSS concentration at inflow and outflow runoff samples at different sampling dates. Error bar represents standard deviation. (a) CC feedlot and (b) SC feedlot.

runoff sample was collected due to the malfunctioning of the sampler at inflow location and only outflow samples were collected.

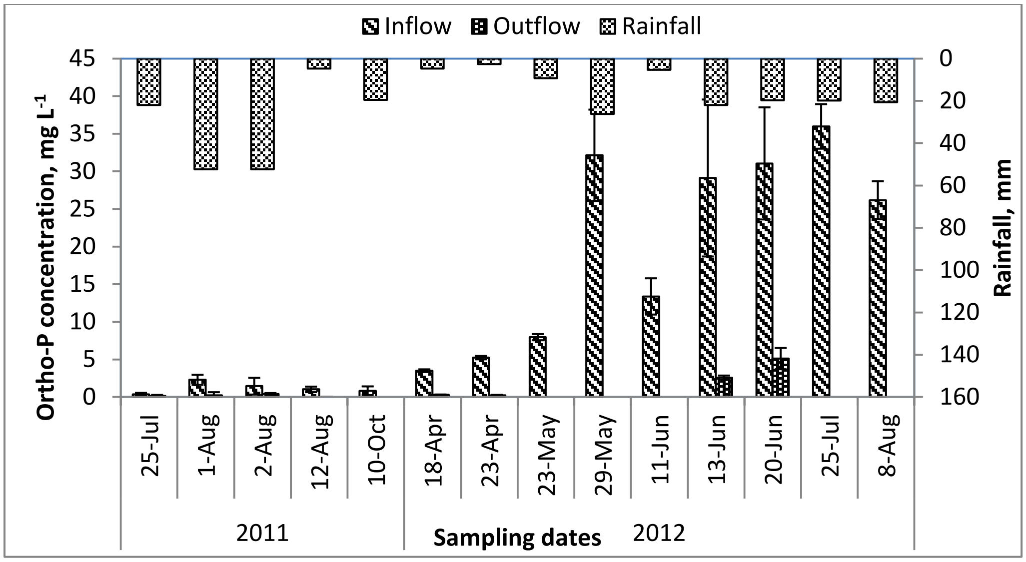
At the RC feedlot, average TSS concentration was significantly lower at outflow than that at inflow with inflow and outflow concentration varied from 61.9 to 3618 mg L^{-1} and 35.5 to 1658 mg L^{-1} (Rahman et al., 2012), respectively. Overall, outflow TSS concentrations were significantly lower than the inflow concentrations. The results observed in this study are consistent with others (Andersen et al., 2009), where they observed 26% to 95% reduction of TSS concentration in runoff from six beef feedlots in Iowa, USA. It is likely that the VFS provides a means of physical separation of suspended solids, reduces transport energy, deposits sediment, and increases infiltration of dissolved constituents into the VFS which was also concluded by Hay et al. (2006).

In the CC feedlot, the buffer had broadleaf cattails which formed dense stands of stems and leaves in various stages of development that might have created rough surfaces, impeding sediment carrier energy, thus increasing separation of solids. However, garrison creeping foxtail and Reed canary grasses at the SC location and mixed vegetation at the RC location were found to be less effective in reducing TSS.

4.4.2. Nutrient transport reduction

Average ortho-P concentration ranged from 0.36 to 36.0 mg L^{-1} at CC and 9.17 to 23.8 mg L^{-1} at the SC feedlot in inflow runoff samples as shown in figure 4.4a and 4.4b. Similarly, average ortho-P concentration ranged from 2.25 to 27.3 mg L^{-1} at the RC feedlot (Rahman et al., 2012). Outflow ortho-P concentration ranged from 0.0 to 5.10 mg L^{-1} at CC, 3.33 to 20.2 mg L^{-1} at the SC, and 0.48 to 23.2 mg L^{-1} at the RC feedlots (Rahman et al., 2012). It was observed from figure 4.4a that the concentrations of ortho-P at the CC location in 2011 are comparatively low than those in 2012. This could be due to fewer animals in the pens in 2011 as compared to 2012, and the fact that the feedlot was commissioned in 2011. When the feedlot was fully operational in 2012, ortho-P concentration in inflow runoff increased significantly, which may have also been due to nutrient contribution from previous year's nutrient accumulation. The ortho-P fractions of TP were less in CC location and were usually below 0.35 compared to SC location where these fractions were up to 0.91 of TP (fig. 4.4 and 4.5). The average ortho-P fraction of TP was higher at the RC location and the highest fraction found was 0.94. It was noted that the ratio of ortho-P/TP increased in the outflow compared to inflow for most of the runoff events indicating that particulate bound P was retained in the VFS with settled sediments. A small portion of soluble P tended to be captured by the buffer during low runoff flow rates with reduced concentrations at outflow.

Inflow TP concentrations ranged from 0.69 to 214 mg L^{-1} at the CC and 11.5 to 97.0 mg L^{-1} at the SC feedlot, and the outflow TP concentrations ranged from 0.22 to 28.5 and 8.03 to 96.8 mg L^{-1}, at CC and SC feedlots, respectively, (fig. 4.5a-b). Rahman et al. (2012) observed TP concentration range at inflow and outflow varied from 5.98 to 36.1 and 0.28 to 29.1 mg L^{-1}, respectively, at the RC location. Higher TP concentrations in runoff samples were likely due to runoff collected immediately after the pen surface, where nutrient concentrations were typically higher. Also, soil characteristics might play some role for high TP concentration in runoff samples. For example, soil at the CC and SC feedlot has greater finer fractions than that of the RC feedlot, which might have carried greater TP load with runoff, as major part of P transport is assumed to occur with transport of finer particles to which they are attached (Sharpley et al., 1994).



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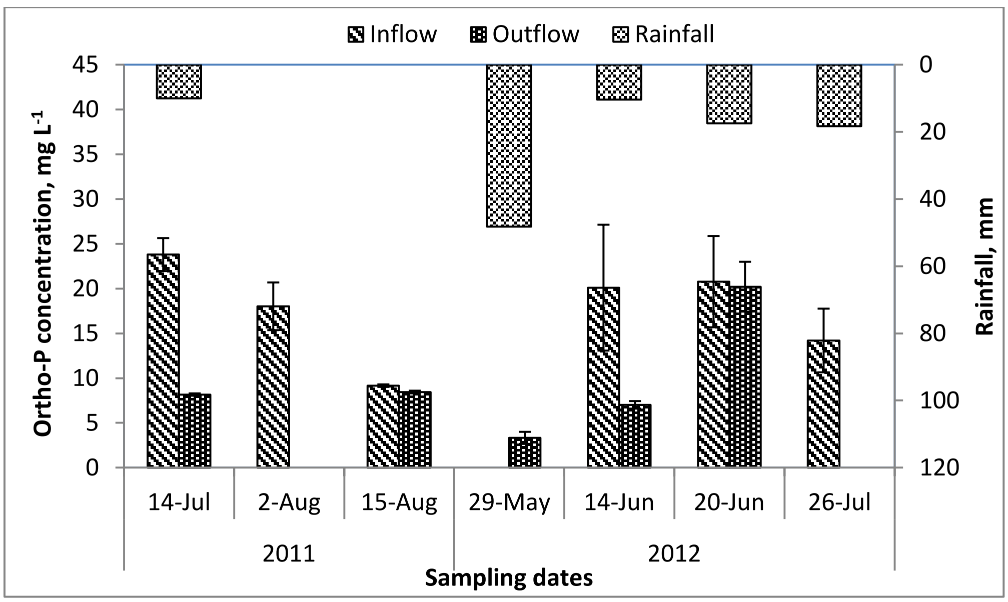
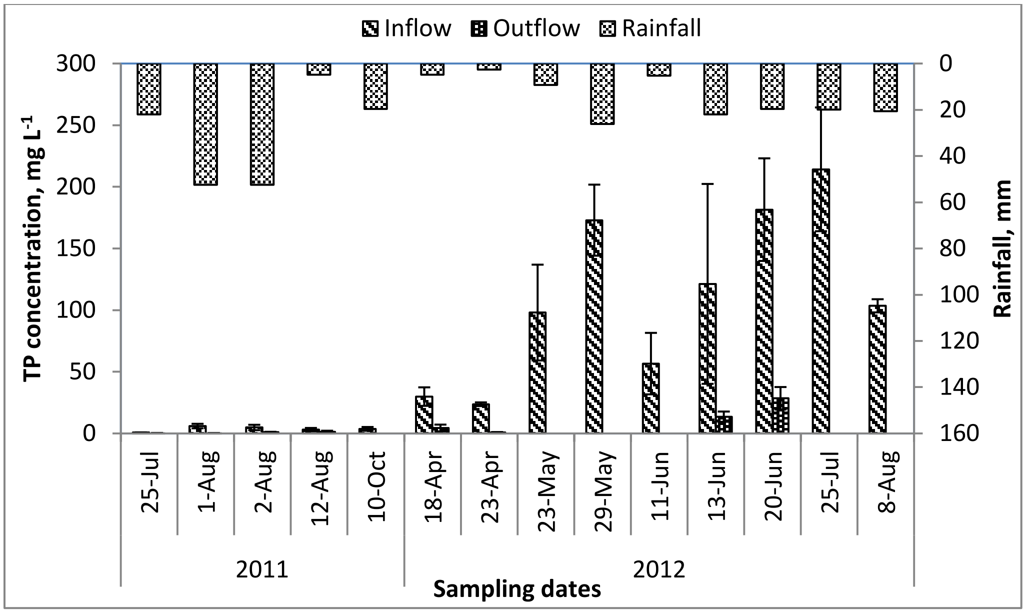


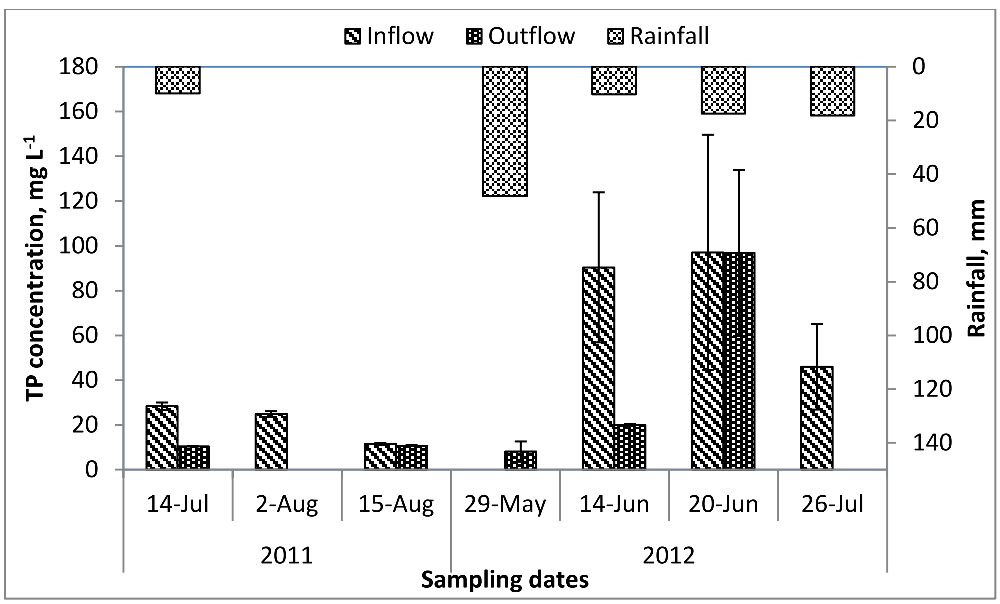
(b)

Figure 4.4. Average ortho-P concentration at inflow and outflow runoff samples at different sampling dates. Error bar represents standard deviation. (a) CC feedlot and (b) SC feedlot.

Other researchers also found that TP concentration in incoming runoff into the buffer varied from 20.0 to 81.5 mg L^{-1} from a dairy facility, whereas ortho-P concentration varied from 16.2 to 54.6 mg L^{-1} (Schwer and Clausen, 1989; Schellinger and Clausen, 1992). Andersen et al.



(a)



(b)

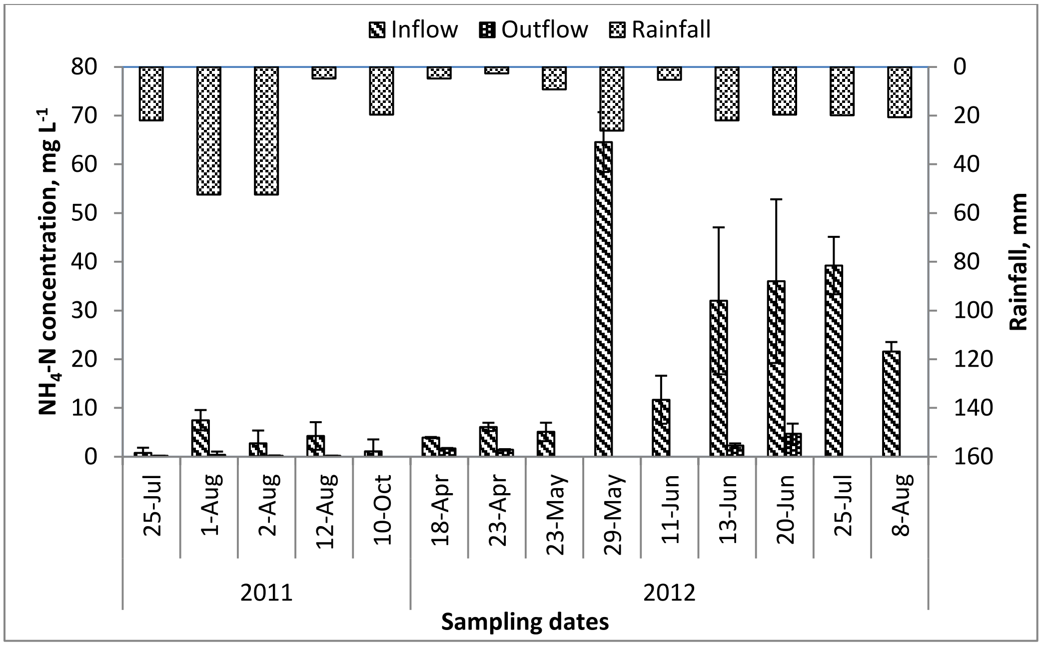
Figure 4.5. Average TP concentration at inflow and outflow runoff samples at different sampling dates. Error bar represents standard deviation. (a) CC feedlot and (b) SC feedlot.

(2009) observed 53 to 222 mg L^{-1} of TP and 28 to 101 mg L^{-1} ortho-P concentrations in influent runoff to the VFS after passing through the settling basins.

Vegetative filter strip in CC feedlot was found very effective in reducing both ortho-P and TP concentrations from runoff compared to filter strips in SC feedlot. Total phosphorus concentration reduction was observed from as low as 57.8%, 0.27%, and 4.02% at the CC, SC, and RC (Rahman et al., 2012) feedlots, respectively, to the highest, 100% where there is no outflow exited the filter strips. Similarly, ortho-P concentration reductions were 65.8%, 2.7%, and 5.9% at the CC, SC, and RC (Rahman et al., 2012) feedlots, respectively, to the maximum, 100%, in the event where no outflow runoff from VFS occurred. Between rainfalls events, when the VFS soil was dry, it did not generate any outflow from the buffer while it received inflow from the feedlot. This indicates that, with time of rainfall occurrence and at low rainfall events, the buffer is more effective due to antecedent soil moisture in the buffer area, which reduces runoff-flow and retains within the buffer area.

Higher ortho-P reductions at the CC feedlot was likely due to sorption to soil particles and plant materials, plant uptake, infiltration, and partly dilution for some runoff events. A similar phosphorus reduction trend has also been observed by other researchers. Andersen et al. (2009) measured buffer performance from six beef feedlots in Iowa, USA and observed TP concentration reductions ranged from 38% to 94% and ortho-P concentration reductions ranged from 33% to 92%.

Figures 4.6a and b show the average NH_4-N concentrations during different sampling dates at the CC and SC feedlots. Similar to ortho-P, concentrations of NH_4-N in runoff at CC location were low in 2011 compared with those in 2012.



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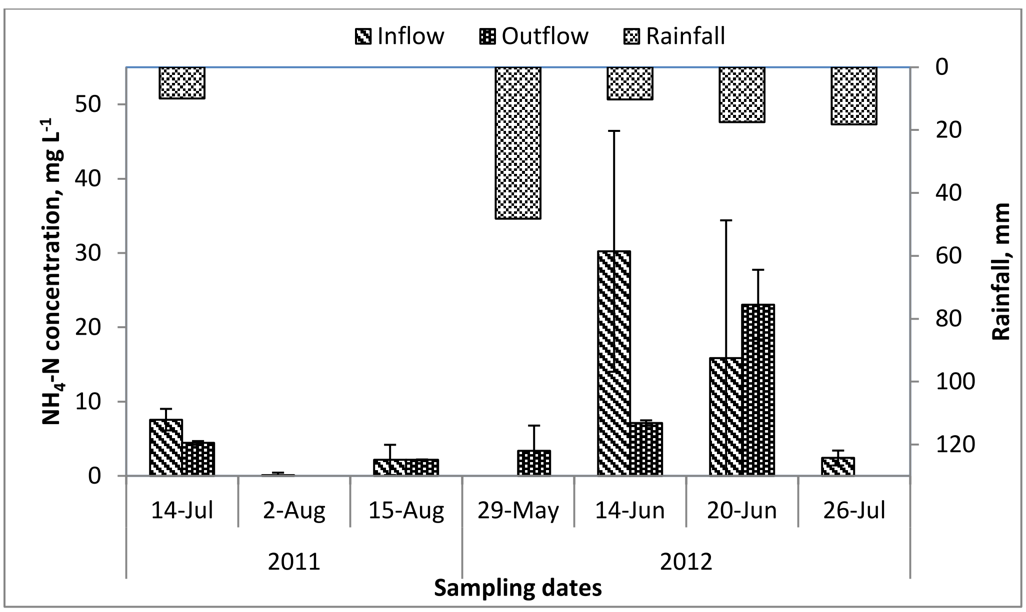

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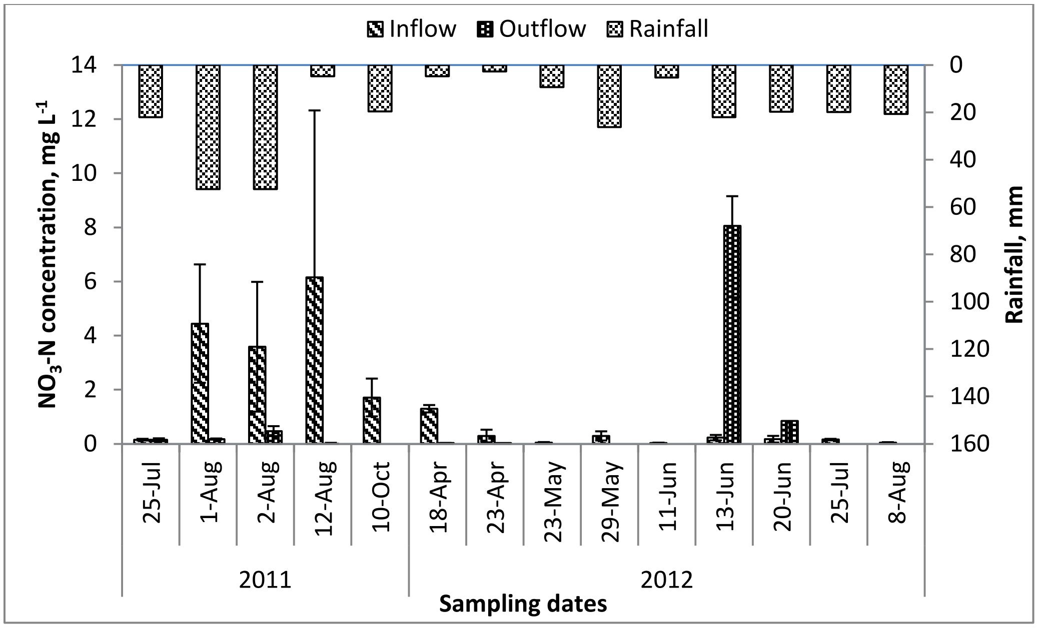
Figure 4.6. Average NH4-N concentration at inflow and outflow runoff samples at different runoff events. Error bar represents standard deviation. (a) CC feedlot and (b) SC feedlot.

Inflow NH_4-N concentrations at the CC and SC locations ranged from 0.78 to 64.6 and 0.09 to 30.2 mg L^{-1} and outflow concentrations ranged from 0.17 to 4.70 and 2.15 to 23.1 mg L^{-1}, respectively. Similarly, inflow and outflow NH_4-N concentrations at the RC location ranged from 1.0 to 48.0 and 0.4 to 37.0 mg L^{-1}, respectively (Rahman et al., 2012). It was observed from both figures that the NH_4-N concentrations in inflow runoff samples were higher towards the end of monitoring period in 2012 than the earlier monitoring period, which might be due to higher microbial activity in manure and soil (Duchemin and Hogue, 2009) at relatively higher temperatures during later part of the monitoring period, although microbial activity was not monitored in this study. Reduction of NH_4-N concentration was found very high in both locations except 20 June, 2012 at SC, which was due to grab sampling. High NH_4-N concentration reductions were likely due to the combined effect of soil sorption, and plant uptake (Rahman et al., 2012).

Figures 4.7a and b show the NO_3-N trends during different sampling dates at the CC and SC feedlot locations, respectively. Comparatively, lower NO_3-N concentration was observed at the CC than that at the SC location for most of the sampling dates. The NO_3-N concentrations in inflow samples ranged from 0.04 to 6.16 and 2.58 to 73.6 mg L^{-1} at the CC and SC feedlot, whereas it varied from 0.01 to 8.05 and 0.13 to 17.8 mg L^{-1} at the outflow for the CC and SC feedlot locations, respectively. The range of measured NO_3-N concentrations at inflow and outflow at the RC feedlot were undetectable limit to 6 mg L^{-1} and undetectable to 54.3 mg L^{-1}, respectively. However, NO_3-N concentrations were always below the EPA minimum allowable effluent discharge concentration level of 10 mg L^{-1} at the CC feedlot location. At the SC location, inflow NO_3-N concentrations were higher than EPA threshold value on several occasions, but only in a few occasions the outflow concentrations were higher than EPA threshold value. This could impact downstream aquatic species and recreational uses.



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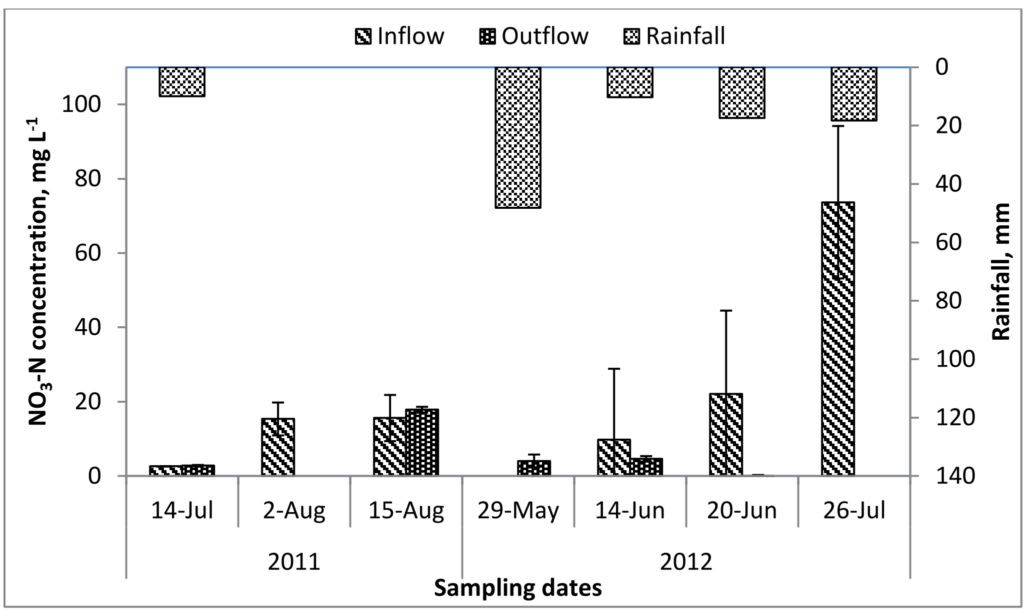


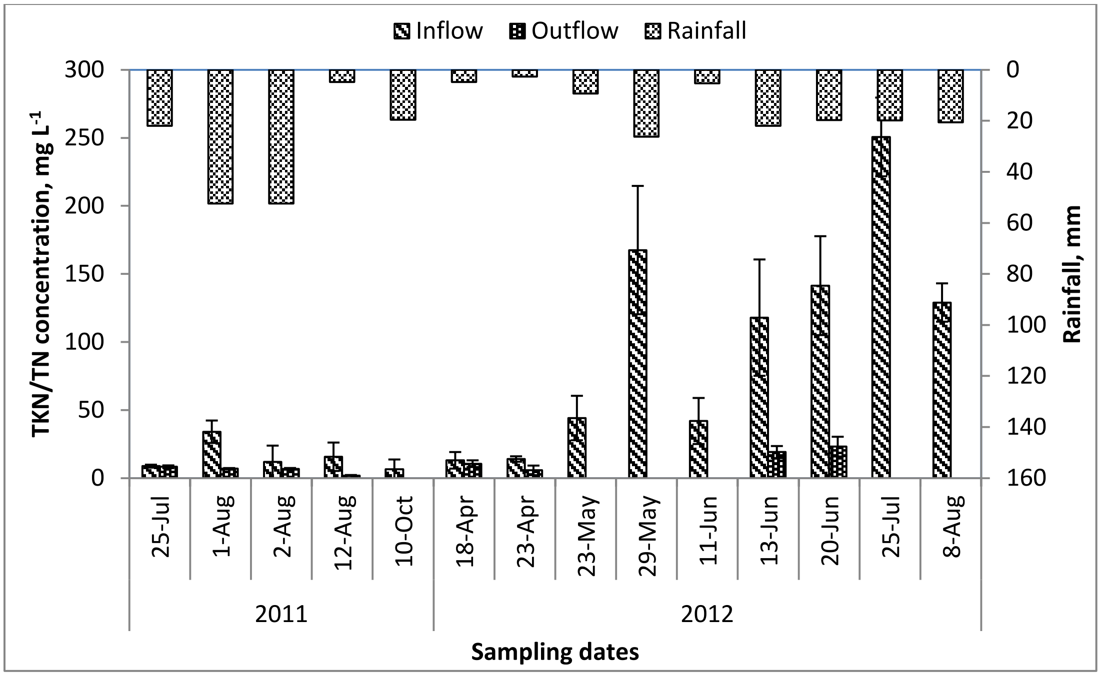
(b)

Figure 4.7. Average NO_3-N concentration at inflow and outflow runoff samples at different rain events. Error bar represents standard deviation. (a) CC feedlot and (b) SC feedlot.

On 14 and 20 June, 2012 at the CC and 14 July and 15 August, 2011 at the SC locations, outflow NO_3-N concentrations were higher than the inflow concentrations. This phenomenon has also been observed in many other studies (Andersen et al., 2009; Dillaha et al., 1988; Mendez et al., 1999; Young et al., 1980), which is likely due to mineralization of particulate organic N that is trapped and accumulated in the buffer resulting in increased soluble N over time in outflow (Mendez et al., 1988). Comparing figures 4.6a with 4.7a and 4.6b with 4.7b, it was observed that the concentration of NH_4-N and NO_3-N in runoff has an inverse relationship, increase in one decreases the other, are likely due to biological nitrification (Kim et al., 2008). This could be due to microbial activities, and probably, NO_3-N concentration depends on nitrification.

Concentration of TKN or TN (TN measured for 2012 samples) showed similar trend as TSS (Figures 4.8 a-b), and a correlation was found between the TS and TKN or TN (R^2=0.51 at CC, data and figure not shown). Dillaha et al. (1989) also observed that 90% of TKN transport with sediment. A strong correlation (R^2=0.70) between TKN and TS was also observed at the RC feedlot (Rahman et al., 2012). Vegetative filter strips were very effective for reducing transport of TKN/TN for both the CC and the SC locations except on 20 June, 2012 at SC feedlot, which was due to grab sampling. Typical transport reduction mechanisms of TKN/TN are physical separation by sediment deposition, and infiltration (Vought et al., 19994).

Concentrations of K at different sampling events at the CC and SC feedlots are shown in figure 4.9. Potassium concentration at CC location was very low in 2011 but was very high in 2012 (Figure 4.9a). Inflow concentration of K ranged from 12.3 to 2246 and 227 to 460 mg L^{-1} while at outflow concentrations ranged from 8.03 to 86.5 and 151 to 545 mg L^{-1} at the CC and SC feedlot locations, respectively. Our peak value of 2246 mg L^{-1} is slightly higher than that reported by Clark et al. (1975), where they found the highest K concentration of 1864 mg L^{-1} at Mead, NE. Dickey and Vandeholm (1981) used a settling basin after the beef feedlot and reported K concentrations at the entry and exit of a VFS were 665 and 168 mg L^{-1}, respectively,



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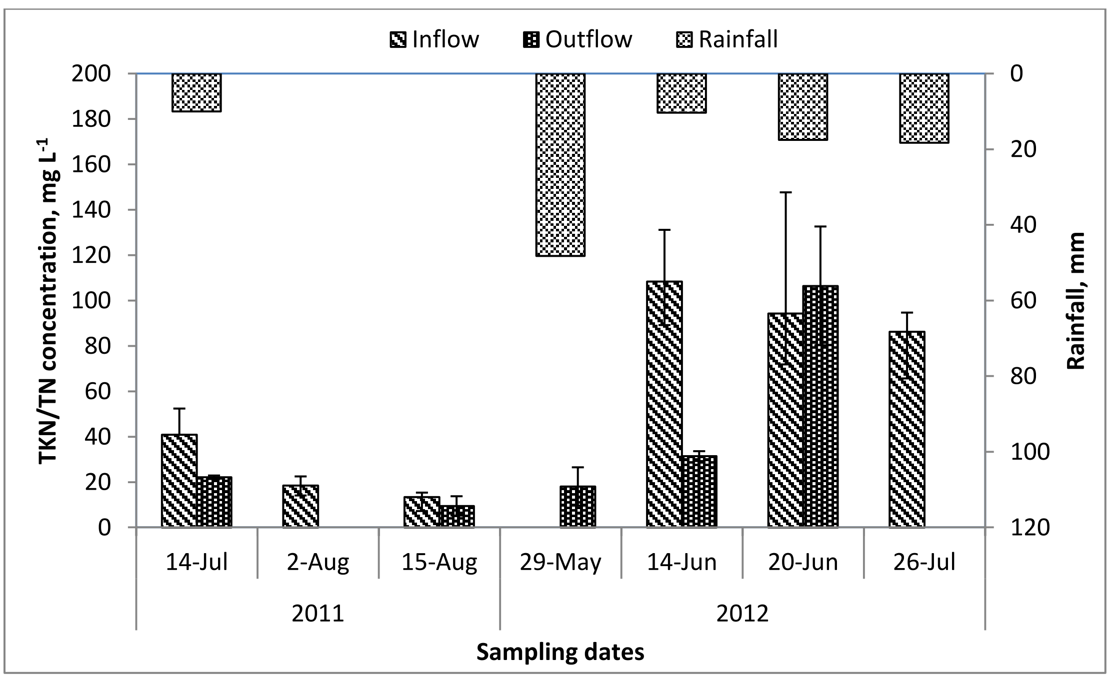
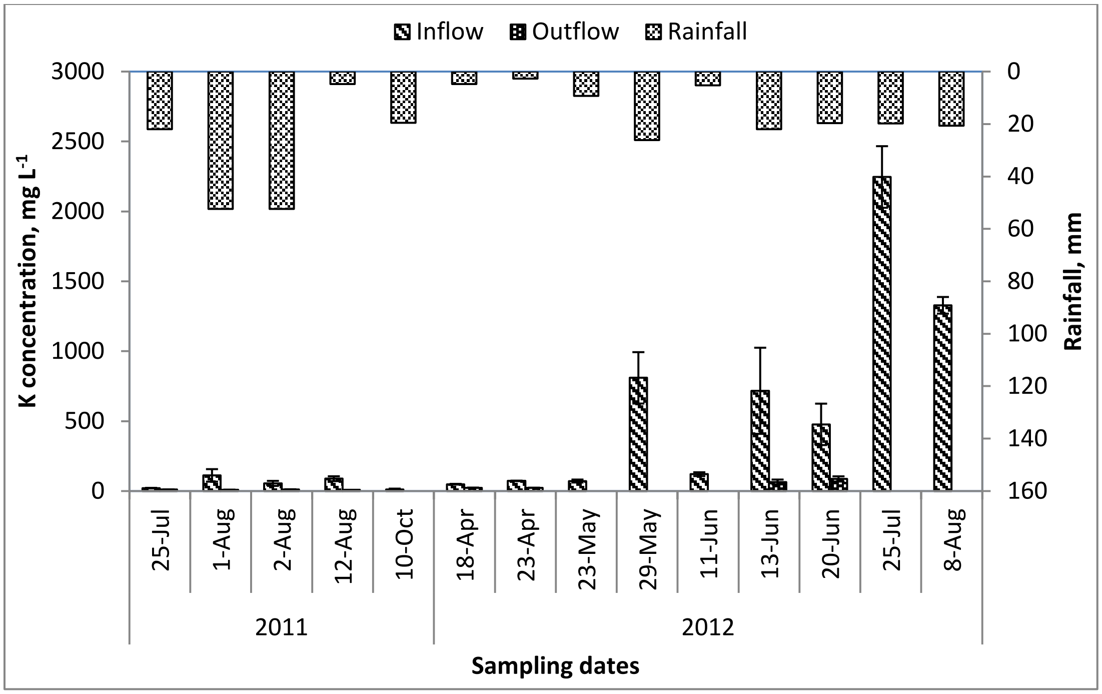


(b)

Figure 4.8. Average TKN/TN concentration at inflow and outflow runoff samples at different rain events. Error bar represents standard deviation. (a) CC feedlot and (b) SC feedlot.



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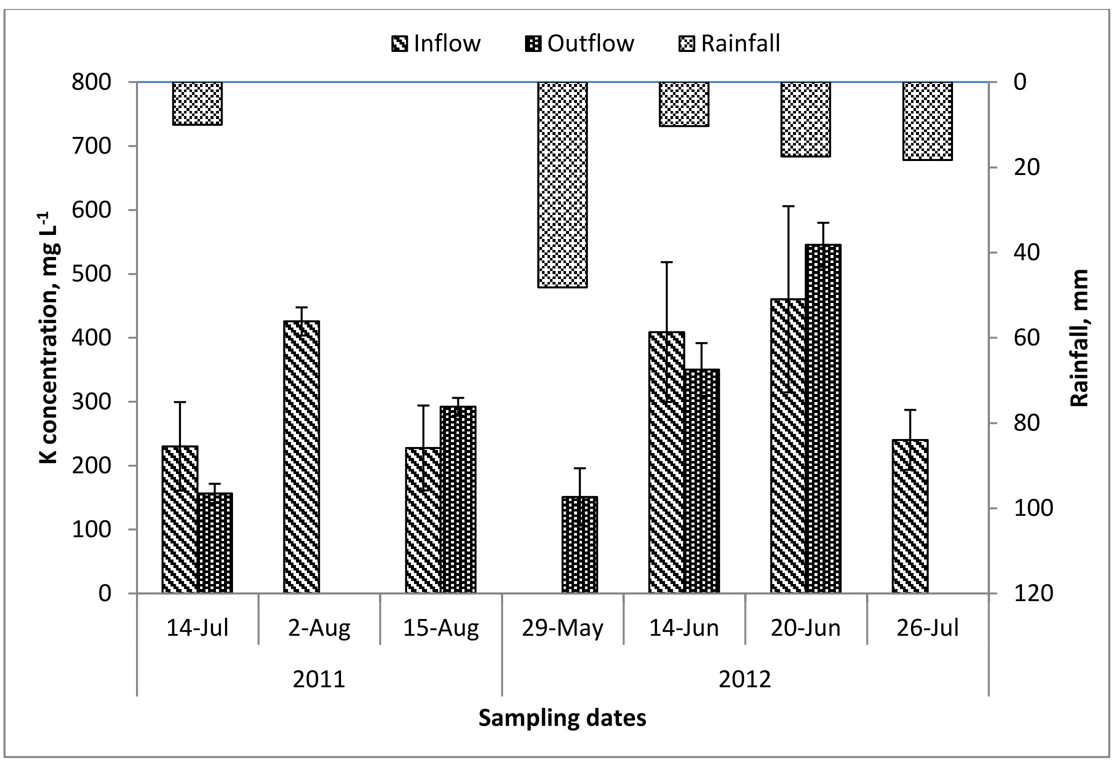

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Figure 4.9. Average K concentration at inflow and outflow runoff samples at different rain events. Error bar represents standard deviation. (a) CC feedlot and (b) SC feedlot.

and these are consistent with the values of K that were found at the SC feedlot. Potassium is highly soluble and a high correlation (R^2=0.83) was found between the K concentration and difference of TSS and TS at the CC feedlot (data not shown). However, a weak correlation (R^2=0.32) exists between K and electrical conductivity in the same location (data not shown). Despite high K concentration in inflow runoff, the VSF system is appeared to be effective in reducing transport of K downstream, except on 20 June, 2012 at the SC feedlot location, which may be due to variation of sampling methods (automatic vs. grab). Potassium is very soluble and its removal mechanism predominantly through infiltration, which is effectively done during some runoff events where there was no outflow beyond the filter strips.

4.4.3. Comparative performance of three different buffer designs

Overall performance of VFSs with different designs is presented in tables 4.2, 4.3, and 4.4 for the CC, SC, and RC feedlot locations, respectively. In terms of solids concentration reductions, the CC VFS system was most effective, followed by the RC and SC VFS systems. Total solids and TSS concentration reductions were 91.7% and 99.7% in 2011 and 72.2 % and 88.3% in 2012, respectively, at the CC VFS system. Concentration reductions of the corresponding parameters at the RC feedlot were 33.7% and 68% and at the SC were 24% and 25.2% in 2011 and -104% and 3.07% in 2012, respectively. High solids removal at the CC VFS system was due to physical separation by vegetation through deposition, settling of solids as time progressed, as well as dilution. The broadleaf cattails used on the CC VFS formed a dense stand of stems and leaves, which increased hydraulic roughness, decreasing water velocity, and hence, reduced sediment carrying capacity of water (Mayer and Wischmeier, 1969). At the SC VFS, low TSS concentration reduction was probably due to the low inflow TSS concentrations as runoff travelled across a 165 m grassed area before entering into the VFS, and the VFS is not very effective when inflow TSS concentration is low (Srivastava et al., 1996). Increase in TS in outflow may be due to the contribution of dissolved salts from soil of VFS, which was supported by an increase in electrical conductivity in outflow runoff (table 4.3).

Ortho-P and TP removal efficacies were the highest for the CC VFS, followed by the SC and RC VFSs. Overall ortho-P and TP removals efficacies were approximately 85% and 90% in 2011 and 2012, respectively, at the CC VFS. At SC VFS, overall ortho-P and TP concentrations reduction were 63% and 68% for 2012 and 55% and 52% during 2011, respectively. However, ortho-P and TP concentration reductions were relatively low (19.3% and 29.9%, respectively) at the RC VFS. It is well known that P adsorption to soil depends on the amount of clay minerals, Al- and Fe-oxides, calcium carbonate, and organic matter (Svendsen, 1992), the CC VFS appeared to be more effective compared to other two locations as the CC VFS was on clay soil. Longer runoff flow-length, dense vegetation, and soil type, could be the factors that made the CC VFS more effective than the SC and CC VFS systems. For the same reason, VFS at RC was less effective than the CC and SC.

Vegetative buffer strips were not always effective for all forms of nitrogen such as NO_3-N. Nitrate nitrogen is highly soluble in water, a negatively charged ion (anion), and not attracted by soil particles or by vegetation to be captured while flowing through a filter strip. For example, NO_3-N concentration at the outflow increased compared to inflow at CC and RC VFSs, which is also reported in many previous studies (Dillaha et al., 1988; Mendez et al., 1999; Chaubey et al., 1995) with similar VFSs configuration. In contrast, the SC feedlot resulted in 19% and 88.6% NO_3-N reduction in outflow runoff in 2011 and 2012, respectively. Thus, this result indicated

Table 4.2. Concentration of different parameters averaged across entire sampling dates followed by standard deviations of the runoff samples at CC feedlot.

	2011					2012				
Variable	Inflow	N	Outflow	N	% reduction	Inflow	N	Outflow	N	% reduction
pH	8.03b±0.5	55	9.50a±0.3	9	-18.2	7.37a±0.3	121	7.16b[†]±0.1	33	2.75
EC, μS cm^{-1}	701a±501	55	366b±46	9	47.8	4740a±2873	121	1074b±314	33	77.3
TS, mg L^{-1}	2445a±3003	65	202b±57.7	14	91.7	4396a±2714	121	1222b±485	33	72.2
TSS, mg L^{-1}	1623a±3024	65	5.13a±7.6	14	99.7	1296a±1631	121	151b±124	33	88.3
Ortho-P, mg L^{-1}	1.21a±0.8	65	0.18b±0.3	14	85	22.0a±13	121	2.21b±2.2	33	89.9
TP, mg L^{-1}	3.94a±2.0	65	0.59b±0.6	14	85.1	121a±73	121	13.0b±12	33	89.2
NH_4-N, mg L^{-1}	3.33a±3.3	65	0.26b±0.4	14	92.3	29.4a±24	121	2.64b±1.7	33	91.0
NO_3-N, mg L^{-1}	3.84a±4.1	65	0.20b±0.2	14	94.8	0.33b±0.4	121	2.44a±3.8	33	-631
TKN/TN, mg L^{-1}	14.70a±13	65	6.10b±2.4	14	58.5	105a±74	121	15.9b±7.7	33	84.9
K, mg L^{-1}	59.4a±43	65	9.59b±1.5	14	83.8	536a±547	121	52.0b±30	33	90.3

[†] Averages within a row followed by different letters for each year are significantly different at $P \leq 0.05$ according to Duncan multiple range tests.

Table 4.3. Concentration of different parameters averaged across entire sampling dates followed by standard deviations of the runoff samples at SC feedlot.

Variable	2011					2012				
	Inflow	N	Outflow	N	% reduction	Inflow	N	Outflow	N	% reduction
pH	8.23a±0.2	29	8.29a±0.1	7	-0.67	7.14a±1.2	45	7.48b[†]±0.2	34	-4.76
EC, μ Scm^{-1}	2120a±234	29	1771b±7.6	7	16.5	2534b±866	45	5544a±2067	34	-119
TS, mg L^{-1}	1750a±526	29	1330a±42	4	24	2735b±1375	44	5584a±1874	33	-104
TSS, mg L^{-1}	150a±43	29	112a±8.6	4	25.2	301a±561	45	292a±500	34	3.07
Ortho-P, mg L^{-1}	23.3a±5.0	29	10.6b±0.3	7	54.7	18.6a±6.4	45	6.86b±6.4	34	63.1
TP, mg L^{-1}	17.4a±4.5	29	8.33b±0.2	7	52.1	79.1a±40	45	25.3b±36	34	68.0
NH_4-N, mg L^{-1}	4.15a±3.2	29	3.36a±2.2	7	19.2	19.5a±19	45	7.40b±8.0	34	62.1
NO_3-N, mg L^{-1}	14.0a±5.9	29	11.4a±8.1	7	19.0	30.2a±34	45	3.45b±2.1	34	88.6
TKN/TN, mg L^{-1}	20.2a±8.7	29	15.0a±7.4	7	25.9	97a±35	45	35.6b±35	34	63.3
K, mg L^{-1}	378a±92	29	234b±74	7	38.1	362a±147	45	253b±160	34	30.3

[†] Averages within a row followed by different letters are significantly different at $P \leq 0.05$ according to Duncan multiple range tests.

Table 4.4. Concentration of different parameters averaged across entire sampling dates followed by standard deviations of the runoff samples at RC feedlot (Rahman et al., 2012).

	2010				
Variable	Inflow	N‡	Outflow	N	% reduction
pH	7.69a†±0.29	187	7.69a±0.29	216	-
EC, µS cm^{-1}	2084a±782	187	1761b±956	217	-
TS, mg L^{-1}	3703a±1937	187	2454b±1422	218	33.7
TSS, mg L^{-1}	1252a±1704	181	401b±686	218	68.0
Ortho-P, mg L^{-1}	17.2a±7.4	173	13.9b±8.0	196	19.3
TP, mg L^{-1}	25.1a±8.8	177	17.6b±10.4	215	29.9
NH_4-N, mg L^{-1}	13.8a±11.4	173	9.43b±10.1	216	31.8
NO_3-N+NO_2-N, mg L^{-1}	1.45a±2.89	173	1.90a±2.59	196	-
TKN, mg L^{-1}	112a±56.1	177	72.5b±57.1	215	35.6
K, mg L^{-1}	5074a±237	177	406 b±281	216	19.8

† Averages within a row followed by different letters are significantly different at $P \leq 0.05$ according to Duncan multiple range tests.
N‡ - number of samples.

that a grassed area (fig. 4.2) located at upstream of a VFS may be appropriate for capturing NO_3-N contained in feedlot runoff inflow.

In contrast, VFS systems were found very effective in reducing transport of NH_4-N, TKN, and TN. Unlike NO_3-N, NH_4-N concentrations were consistently reduced to some extent in all three VFSs since ammonium (NH_4^+) is a positively charged ion and held by the negatively charged soil particles (tables 4.2, 4.3, and 4.4) to be captured by vegetation. The CC feedlot, showed highest reductions in NH_4-N concentration compared to the SC and RC feedlots. This was likely due to densely populated broadleaf cattails vegetation that captured highest TS and TSS attributed to solids borne nutrients capture. The highest TKN/TN reductions were approximately 59% and 85% in 2011 and 2012, respectively, at the CC feedlot VFS (table 4.2). At the SC feedlot VFS, estimated TKN/TN reductions were approximately 26% and 63% in 2011 and 2012, respectively, (table 4.3). Similarly, at the RC feedlot VFS, an estimated TKN reduction was approximately 36% in 2010 (table 4.4).

Very low K transport reduction was observed except in CC feedlot VFS. The highest concentration reduction observed was 90.3% at CC whereas lowest concentration reduction was 19.8% at the RC VFS. Potassium is highly soluble and less effective in transport reductions, which is also indicated by low reduction in EC values. The system which can infiltrate more water is the most effective in reducing K transport. At the CC VFS, longer VFS with low antecedent moisture content was favorable for higher reduction effectiveness.

Nutrient transport reduction depends on deposition, adsorption to soil, infiltration, and plant uptake. Relatively poor performance of the RC VFS was probably due to smallest runoff-flow length (12 m) among three VFSs. If runoff-flow length is longer, runoff will have longer time to travel which will facilitate infiltration and better adsorption to soil. Among the three VFS systems, the VFS system at the CC location had the greatest runoff-flow length and resulted in better performance. Also, a buffer with a dense broadleaf cattails grass might be intercepting runoff flow and depositing solids in the VFS area. Since use of feedlot runoff water is restricted due to high concentration of nitrogen, salinity, or sodium content (Butchbaker, 1973), water that passed through the reasonable buffer length and stored in a settling basin would be suitable for field irrigation. Vegetative filter strips at the RC feedlot might possess some concern at the downstream end due to high nutrient concentration even after passing through buffer strips. Longer buffer strips and better vegetation might improve the situation.

4.5. Conclusions

Vegetative filters reduced solids and nutrients from feedlot runoff to some extent. Degree of pollutants removal was dependent upon the type of vegetation and runoff-flow length of a filter strip. For NO_3-N concentration reduction, the SC feedlot was found more effective than the CC and RC feedlots, which was due to differences in vegetative filter systems. Relatively inferior performance of the RC feedlot buffer compared to the CC and SC was probably due to smallest runoff-flow length (12 m) among three VFSs. Overall, the CC feedlot with longer flow length (65 m), dense broadleaf cattail grass filter bed outperformed the SC and RC VFSs in respect of TSS, ortho-P, TP, NH_4-N, TKN/TN reductions.

CHAPTER 5. INFLUENCE OF SOIL pH IN VEGETATIVE FILTER STRIPS TO REDUCE SOLUBLE NUTRIENTS TRANSPORT

5.1. Abstract

Low efficacy of vegetative filter strips (VFS) in reducing transport of soluble nutrients has been reported in many research articles. It is known that solubility of phosphorus and nitrogen compounds is largely affected by the pH of soil. Changing soil pH, thereby changing nutrient solubility, may result in a decrease in their transportation through VFSs. This study was conducted to evaluate the effect of pH levels of VFS soil on soluble nutrient transport reduction from manure-borne runoff. Soil was treated with calcium carbonate to change pH at different levels (pH range in treatments T1, T2, and T3 were 5.5 to 6.5, 6.5 to 7.5, and 7.5 to 8.5, respectively). Soil with different pH levels was packed into galvanized metal boxes measuring 2.44 m long, 0.50 m wide, and 0.25 m deep. Tall fescue grasses were established in the boxes to simulate the vegetative filter strips. Boxes were placed in an open environment and tilted to a 3% slope. Manure amended water was prepared by diluting fresh manure into tap water. The required amount of manure water (44 L) was applied through the VFS by a peristaltic pump at a rate of 1.45 liter per minute. Water samples were collected at the inlet and outlet as well as from the leachate. Collected samples were analyzed for ortho-phosphorus (ortho-P), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), and potassium (K). Highest transport reductions of ortho-P and K were observed at pH level 7.5 to 8.5 (T3) and were 42.4% and 20.5%, respectively. Ammonium nitrogen transport reduction was highest at pH level of 6.5 to 7.5 and was 26.1%. Surface transport reduction of NO_3-N was 100% regardless of the pH level of the soil, but leachate had the highest concentration of NO_3-N. Mass transport reduction also

confirmed that higher pH in the vegetative filter strips are effective in reducing some soluble nutrient transport reduction.

Key words: Vegetative filter strips, soil pH, soluble nutrients, feedlot runoff, mass transport reduction, pollution control

5.2. Introduction

Concentrated animal feeding operations (CAFOs) are a major source of manure. Manure is rich in nutrients and may be applied to cropland as a nutrient source. However, nutrients and sediments in runoff from CAFOs and land application of greater amounts of manure from increasing agricultural activities are causing degradation of water resources. Runoff of nutrients from CAFOs and land application sites has been identified as the major contributor to surface water pollution. Animal industries account for 16% of surface water quality impairment among total agricultural production sectors (USEPA, 2001). According to USEPA, 45% of river miles, 47% of lake acres, and 32% of estuarine water are impaired because of eutrophication (USEPA, 2002). According to the United States Geological Survey (USGS), large and concentrated animal production facilities are responsible for water quality degradation (Gollehon et al., 2001). To prevent water quality degradation from nutrient runoff, Best Management Practices (BMPs) may be applied.

Vegetative filter strips (VFS) is a BMP that has the capability of reducing pollutant concentrations in runoff. United States Department of Agriculture (USDA) - Natural Resources Conservation Service (NRCS) recommends VFS systems for reducing nonpoint source pollution. Vegetative filter strips have the potential to reduce runoff, decrease erosion, increase infiltration, and give time for sediment and nutrient deposition (Giri et al., 2010). Within VFS, nutrient removal from surface inflows occurs mainly by sediment deposition, thus resulting in the deposition of sediment bound nutrients and exchange of dissolved nutrients with soil and litter surface (Vought et al., 1994). A number of studies have documented the VFS effectiveness in reducing sediment and sediment bound pollutants from runoff in both laboratory and field conditions (Dillaha et al., 1989; Duchemin and Hogue, 2009; Schmitt et al., 1999; Rahman et al,, 2012).

Although removal of sediment and sediment bound nutrients by VFSs is well documented, removal by VFSs of soluble pollutants in runoff is not significant (Dorioz et al., 2006). Dillaha et al. (1988) conducted a field experiment to investigate the effect of filter strip length and flow characteristics on sediment, nitrogen, and phosphate transport. They found that 26% and 19% of total soluble P and soluble nitrogen, respectively, was removed in their experiment. Lim et al. (1998) investigated the effect of VFS length on concentration and mass transport of nitrogen, phosphorus, solids, and fecal coliform from a field treated with cattle manure. They observed the same electrical conductivity (EC) values in both inflow and outflow runoffs and concluded that the VFS was less effective in removing dissolved solids. Low ortho-phosphorus (PO_4-P) removal by VFS was also observed by Srivastava et al. (1996), and they concluded that the removal efficiency was related to infiltration amount. Schmitt et al. (1999) also found low soluble P and nitrate transport reductions amounting to 24% and 48% of nitrate and 19% and 43% of soluble phosphorous by the 7.5 and 15 m grass strips, respectively.

Phosphorus immobility may result due to adsorption, chemical precipitation, bacterial action, plant and algal uptake, and incorporation into organic matter (Xu et al., 2006). Several studies attempted to reduce soluble phosphorus loss in runoff from upslope areas using soil amendments such as lime and gypsum. Stout et al. (1998) found that addition of gypsum effectively reduced the solubility of soil P in runoff from soil with high available P. Watt and

Torbert (2006) applied gypsum onto the VFS, and they observed higher soluble phosphorous transport reduction (32% to 38%) in VFS plots treated with gypsum than that without gypsum (18%). These researchers suggested that soluble phosphorous might have been precipitated as insoluble calcium phosphate, and it was removed from runoff. From a soil amendment study, Brauer et al. (2005) suggested that gypsum might react with soluble phosphorus and precipitate as insoluble Ca-phosphate and decrease P transport in runoff. Lindsay (1979) suggested that solubility of N, P, and their compounds are largely affected by soil pH conditions. Depending on the pH of soil, soluble P may be precipitated as hydroxyapatite, fluoroapatite, and chloroapatite (Lindsay, 1979; Kanel and Morse, 1978; Ugurlu and Salman, 1998), which are insoluble in water. Murphy and Stevens (2010) conducted laboratory experiments to evaluate the effects of lime and gypsum to decrease P loss from soils to water. They found that lime decreased reactive phosphorus solubility somewhat. However, their study was limited to a narrow range of pH (5.8-6.8). Therefore, investigation in a broad pH range would be of great interest from a nutrient transport reduction point of view. However, very limited or no information is available on the impact of soil pH changes on the buffer performance. Therefore, the objective of this research was to study the effect of pH on soluble nutrient reduction from manure borne runoff in vegetative filter strips. Moreover, nutrient loss by leachate was also studied.

5.3. Materials and Methods

This study was conducted in soil boxes. Four soil boxes, each 2.44×0.5×0.25 m, were constructed using galvanized iron (fig. 5.1) to simulate vegetative buffer strips. Each box had a spout to collect surface runoff from the box and holes (5 mm) at the bottom to collect leachate. Holes were provided at the head, middle, and tail sections of each box. The head and middle sections had two holes and the tail section had three holes in a line along the width of the box.

Tygon tubing was used to connect all the holes to a single point to accumulate leachate as shown in figure 5.1. Soil boxes were placed on the top of wooden structures above ground surface to protect the tubing and facilitate leachate collection. A uniform slope (3%) was maintained for each soil box to simulate field condition.

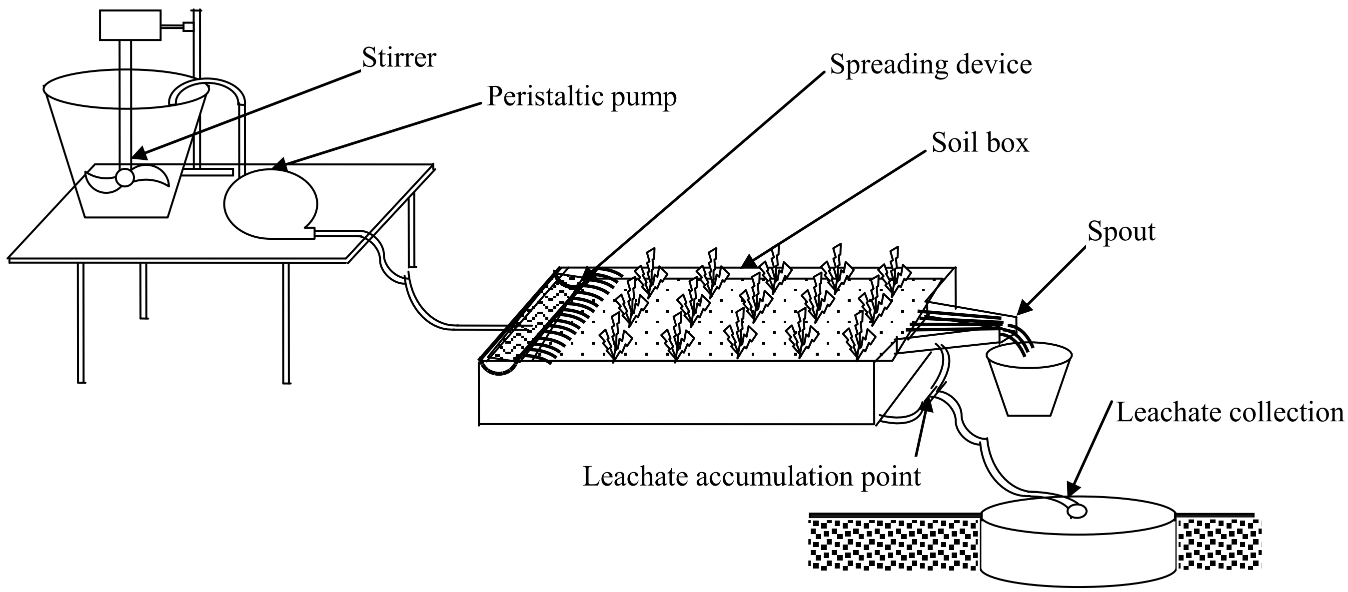

Figure 5.1. Experimental set up for soil box experiment (dimensions are not to scale).

This experiment was comprised of three pH levels: treatment 1 (T1) is at a pH range of 5.5 to 6.5; treatment 2 (T2) is at a pH range of 6.5 to 7.5; and treatment 3 (T3) is at a pH range of 7.5 to 8.5. Soil was collected from the Central Grasslands Research Extension Center, Streeter, North Dakota, from the same field in two different years. Treatment T1 and T2 were conducted using soil 1 and the T3 treatment was conducted using soil 2. Since soil inherent pH was different (Soil 1 pH 6.25±0.05; soil 2 pH 5.38±0.05) additional tests were conducted to find out the sorption capacity for each soil and their similarity to each other based on silt and clay content.

Since the soil 1 inherent pH was 6.2 to 6.3, this pH was considered as treatment T1 and no pH adjustment was made. Treatment T1 was also considered to be the control. Other predetermined pH ranges were adjusted using calcium carbonate ($CaCO_3$). The experiment was conducted in batches and treatments T1 and T2 had four replications while treatment T3 had three replications.

Before adjusting soil pH, soil was sieved through a 6.35 mm sieve to remove crop residue, large soil clods, stone chips, and other foreign materials. To adjust pH, a pre-calculated $CaCO_3$ was added and mixed thoroughly with the bulk soil, and water was sprayed uniformly on the soil. The soil was allowed to go through several wetting and drying cycles until the desired pH was achieved. After achieving the predetermined soil pH range, the soil boxes were packed in layers with a known weight to achieve a bulk density of 1.3 g cm^{-3}. Care was taken to ensure equal compaction throughout the soil box. To prevent overflow during runoff simulation, a free board of 25 mm was provided as shown in figure 5.1.

After packing the soil boxes, fescue grass ('All Pro' cultivar) was manually seeded at a rate of 195 kg ha^{-1} and the boxes were covered with polyethylene sheet for four to five days to facilitate germination. The soil boxes were checked periodically for germination, and the cover was removed from the boxes when the majority of the seeds germinated. It took about two months to establish vegetation on soil boxes, and during this period frequent irrigation was applied to ensure enough moisture for vegetation growth. After establishment of vegetation, runoff experiments were conducted using simulated runoff solution prepared by diluting fresh manure into tap water. The boxes were placed in an open environment subjected to natural rainfall to simulate actual field conditions. Rainfall and temperature data were collected from the

North Dakota Agricultural Weather Network station, which is situated within one kilometer of the experimental site.

Fresh manure was collected from the North Dakota State University (NDSU) dairy barn, and it was diluted with tap water to approximate feedlot runoff nutrient (nitrogen) concentration. About 4 kg of manure was diluted with 44 liters of water to produce a nitrogen concentration of 0.484 mg/L, which was a representative nitrogen concentration in runoff for average feedlot conditions (Alexander and Margheim, 1974). The amount of manure used in this experiment was calculated based on nutrient analysis of manure subsamples, and the volume of water used represents the amount of runoff expected from an average 25 year, 24 hour rainfall at Fargo, North Dakota. Following thorough mixing, the manure water was screened through a 3 mm mesh sieve to remove large particles. The required amount of screened manure water solution was transferred to a separate tank and continuously stirred using an electric stirrer and applied on the simulated buffer strips as shown in figure 5.1. In this study, two runoff events were carried out two to three weeks apart once the vegetation had been established. The second runoff event was initiated to observe if there is any difference between successive runoff events.

A peristaltic pump was used to apply the manure solution uniformly across the soil box with a spreader at the head section of a soil box (as shown in fig. 5.1). The spreader was made from an arc of PVC pipe whose length was approximately equal to the width of a soil box and worked like an overflowing sharp crested weir. Both sides of the spreader were closed with stoppers to prevent leaking through the edges. Before each runoff simulation with manure water, tap water was used to generate runoff for 15 minutes. This runoff was conducted to gather background information and to provide uniform moisture condition for each soil box. One and one half hours after initial runoff, 44 liters of manure water solution was applied in the buffer strip for half an hour at an application rate of 0.024 L s^{-1}. Runoff and leachate samples were collected from both the tap water and manure water solution runoff simulation experiments. For each treatment and replication, runoff samples were collected at 0, 10, 20, and 30 minutes after initiation of the surface runoff. Three leachate samples were collected from each box during runoff experiment. Following collection, samples were stored at 4 ºC until analysis.

Composite soil samples were collected prior to adjusting soil pH and analyzed for pH, specific conductance, cation exchange capacity (CEC), electrical conductivity (EC), calcium (Ca), nitrogen (N), phosphorus (P), and potassium (K). Collected manure samples were sub-sampled and analyzed for nutrients, pH, and EC. Both runoff and leachate samples were analyzed for ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), ortho-phosphorus (ortho-P), K, pH, and electrical conductivity (EC). All samples were analyzed following standard procedures (APHA, 2005). A detailed description of methods/protocols that were used to analyze nutrient concentration of samples is described in Rahman et al. (2013). Similarly, tap water properties were also measured and subtracted from corresponding nutrients. The water sample was filtered through 0.45 µm filter and analyzed for Ortho-P, NH_4-N, and NO_3-N. Ortho-P was measured by ascorbic acid reduction method and NO_3-N was measured by cadmium reduction method. Details of method/protocol were described in Rahman et al. (2013). Phosphorus sorption was estimated by method proposed by Nair et al. (1984). The efficacies of the VFSs were estimated based on percent reduction of each analyte as measured using the following relationship:

$$\eta_{red}=\frac{C_i-C_o}{C_i} \tag{5.1}$$

where η_{red} is reduction efficiency in percent (%), C_i is the inflow concentration of a particular analyte, and C_o is the concentration of the same analyte in the outflow in ppm (mg L^{-1}). Analysis of variance and pairwise comparison of analytes among treatments were done using SAS 9.2 (SAS, 2009).

5.4. Results and Discussion

5.4.1. Background information

Soil used in this study had a loamy sand texture with a pH range from 5.38 to 6.25. Electrical conductivity ranged from 90 to 560 µS cm^{-1}. Average cation exchange capacity (CEC) ranged from 8.47 to 20.0 $meq100^{-1}$ g^{-1} of soil. Organic matter content ranged from 2.47% to 6.83%, and available phosphorus (P) concentration ranged from 14.3 to 5.78 ppm.

The manure characteristics (average of 6 samples) are shown in table 5.1 and these values are very representative to those published by the Midwest Plan Service (MWPS, 2000) for dairy cow.

Table 5.1. Dairy manure characteristics on wet-weight basis.

	Moisture	Dry matter	TKN	NH_4-N	P_2O_5	K_2O	Total C	pH	Conductivity 1:5
	--------	--------	--------	%	--------	--------	--------		mmhos cm^{-1}
Mean	83.3	16.7	0.55	0.16	0.18	0.56	40.5	7.40	4.57
SD	1.70	1.63	0.10	0.05	0.04	0.22	0.68	0.41	1.24

Average temperatures during the study periods (August to mid-November) were 13.8, 14.3, and 12.0 °C in 2010, 2011, and 2012, respectively. Total rainfall during the study period was 280, 97, and 84 mm in the 2010, 2011, and 2012, respectively. The sum of the monthly normal rainfall of four months during study period is about 210 mm in this location. Compared to 2010, total rainfall amount in 2011 and 2012 was much lower for the same period. However, between two runoff simulation experiments, no significant rainfall occurred for any batch and rainfall impact on nutrient transport was negligible.

5.4.2. Ortho-phosphorus (ortho-P) transport reduction

Vegetative filter strips had a distinct impact on the ortho-P concentration reduction as shown in table 5.2. Mean inflow ortho-P concentrations were 35.8, 28.5, and 20.1 mg L^{-1} in T1, T2, and T3, respectively, in the first runoff event. Corresponding mean outflow ortho-P concentrations were 28.7, 22.2, and 11.6 mg L^{-1} in T1, T2, and T3, respectively. Reductions of concentrations were 19.8%, 22%, and 42.4% (fig. 5.2) in treatments T1, T2, and T3, respectively, in the first runoff event. This reduction is much higher than the ortho-P concentration reduction reported by Rahman et al. (2013) in field study (<6%), where no soil pH was adjusted in VFS area. In the second runoff event, mean inflow ortho-P concentrations were 21.4, 25.2, and 18.6 mg L^{-1} and outflow concentrations were 17.4, 22.6, and 13.5 mg L^{-1} in treatments T1, T2, and T3, respectively. Ortho-P transport reductions decreased in the second runoff event for all treatments and were 19%, 10.3%, and 27% in treatment T1, T2, and T3, respectively. The lower transport reduction in second runoff event was likely due the fact that most reactive sites of soil were occupied by the previously sorbed P, and freshly applied P was sorbed by less reactive sites (Bowden et al., 1980). Other researchers (Watts and Torbert, 2009) observed increase in soluble transport reduction from 18% to 40% by applying gypsum in the grass buffer strips and effectiveness reduced in the second runoff event conducted one month after the first runoff event.

In table 5.2, no significant effect of sampling times on pollutant concentration was observed in the first runoff event, indicating that reduction was uniform over time. However, a slightly higher reduction of ortho-P was observed during the first sampling times, which

Table 5.2. Mean inflow and outflow and outflow ortho-P concentrations in different sampling times in the first and second runoff event.

Treatment	Inflow	Outflow†				
		Time, min				Mean
		0	10	20	30	
		---------------Runoff event 1, mgL^{-1} ------------------------------------				
T1	35.8	27.9a	29.1a	30.1a	27.7a	28.7
T2	28.5	20.7a	22.6a	22.9a	22.7a	22.2
T3	20.1	10.8a	11.6a	11.0a	12.9a	11.6
		--------------------Runoff event 2, mgL^{-1} --------------------------------				
T1	21.4	23.4a	18.6ab	15.8ab	11.6b	17.4
T2	25.2	22.1a	22.3a	22.9a	23.1a	22.6
T3	18.6	11.5b	13.6a	14.6a	14.4a	13.5

†Means followed by the same letter in a row are not significantly different at 90% significance level.

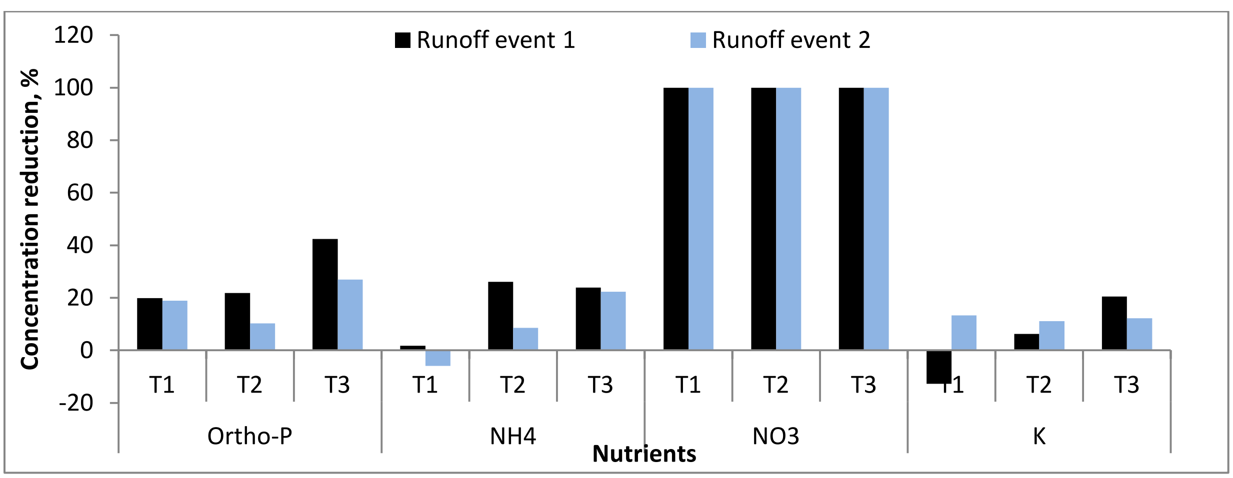

Figure 5.2. Percent concentration reduction of pollutants during first and second runoff events for T1 (pH 5.5-6.5), T2 (pH 6.5 to 7.5), and T3 (pH 7.5 to 8.5).

suggested that the time immediately after initial runoff is important for controlling ortho-P transport in runoff.

Already dissolved P in manure water was immediately adsorbed to soil and vegetation or precipitated. Other researchers also found that the first 10 minutes after initiation of runoff is critical for controlling nutrient concentrations in runoff (Watts and Torbert, 2009). They applied gypsum onto the VFS which received runoff from poultry litter amended field. The concentration reductions in samples after first 10 minutes might be attributed to the cessation of desorption and increase of adsorption on soil particles. In the second runoff event, which was conducted two weeks after the first runoff event, ortho-P concentration increased in outflow except T1 (table 5.2), which was likely due to desorption of P to equilibrate to supplied concentration. Adsorption might have reduced the concentration below that supplied level thereafter (about 20 minutes after rainfall started).

Soluble phosphorus transport reduction increased with increasing pH with calcium carbonate. Highest ortho-P transport reduction observed in T3 treatment was probably partly due to precipitation and sorption at higher pH (Murphy and Stevens, 2010). As pH increases, concentration of Ca increases, which can cause precipitation and sorption of P. However, increase in pH from lower to higher increases the Al and Fe oxides for P sorption, but at higher pH (over 7.5 in T3) their availability to adsorb P decreases (Lindsay, 1979; Litaor et al., 2003). At higher pH, phosphorus forms insoluble Ca-phosphate which precipitated and reduced transport (Dou et al., 2003; Watts and Torbert, 2009; Litaor et al., 2003). In an amendment study by Boruvka and Rechcigl (2003), pH was raised to 7.4 to 7.8 by $CaCO_3$ and resulted in higher P sorption. They concluded that the Ca ion provided must be accompanied with increase in pH for direct precipitation and sorption. In this study, however, sorption could have played a more important role in reducing P transport than the precipitation because Murphy and Sims (2012) observed only 20% reduction of dissolved reactive P through precipitation using lime.

In this study, $CaCO_3$ was used to increase the pH, which implies that $CaCO_3$ can be used to reduce transport of ortho-P from manure borne runoff because higher dissolved amount of Ca may result from the higher total Ca in soil (Bubba et al., 2003). Calcium driven sorption and precipitation was also likely evident in the control VFS under T3 treatment, where soil pH was not adjusted. In the control VFS, ortho-P transport reduction was 11% and 32% in the first and second runoff events, respectively (data not shown). The higher transport reduction in the second runoff event was probably due to higher microbial activities which might cause immobilization of P (Johnston, 1991).

5.4.3. Nitrogen transport reduction

Table 5.3 shows the concentrations of NH_4-N at different sampling times. Like ortho-P, concentrations of NH_4-N did not change significantly with sampling time except treatment T3 in second runoff event. Mean inflow NH_4-N concentrations were 122, 93, and 123 mg L^{-1} in T1, T2, and T3, respectively, in the first runoff event. Mean outflow concentrations observed were 120, 68.6, and 93.4 mg L^{-1} in T1, T2, and T3, respectively, in the same runoff event. In the second runoff event, mean inflow concentrations were 158, 35, and 108 mg L^{-1} in T1, T2 and T3, respectively, where inflow NH_4-N concentration in T2 was much lower than others. The corresponding mean outflow concentrations were 168, 31.7, and 83.8 mg L^{-1} in T1, T2, and T3, respectively, in the second runoff event. However, NH_4-N concentration was reduced by 1.72%, 26.1%, and 24% in the first runoff event for T1, T2, and T3, respectively, when averaged across each sampling time (fig. 5.2). Ammonium nitrogen removal mechanisms from runoff include infiltration, volatilization, nitrification-denitrification, assimilation by plants and microorganisms

Table 5.3. Mean inflow and outflow and outflow NH_4-N concentrations in different sampling times in the first and second runoff event.

		Outflow†				
		Time, min				
Treatment	Inflow	0	10	20	30	Mean
	-------Runoff event 1, mg L^{-1} -------					
T1	122	120a	118a	124a	116a	120
T2	93	66.2a	77.3a	75.8a	55.0a	68.6
T3	123	78.5a	87.4a	102a	106a	93.4
	------- Runoff event 2, mg L^{-1} -------					
T1	158	173a	168.8a	167a	161a	168
T2	35	32.9a	31.2a	32.6a	30.3a	31.7
T3	108	73.3b	89.4a	85.9a	86.5a	83.8

† Means followed by the same letter in a row is not significantly different at 90% significance level.

(Kruzic and Schroeder, 1990), adsorption by negatively charged clay and organic colloids, fixation by clays, and fixation by organic carbon (Lance, 1972). Ammonium nitrogen removal through nitrification in overland flow is low (Kruzic and Schroeder, 1990). Perhaps, major NH_4-N removal was through fixation by adsorption and volatilization as ammonia. Fixation of NH_4-N to an organic fraction is pH dependent and is rapid above pH 7 by adding $Ca(OH)_2$. Moreover, at higher pH, adsorption to exchange sites is inhibited by divalent Ca^+ and Mg^+ ion (Lance, 1972), which is reflected by the higher transport reduction in T2. Volatilization might also slightly contribute to transport reduction of NH_4-N since the runoff was alkaline, however volatilization was not measured. Transport reductions of NH_4-N in the second runoff events were lower than the first runoff event, except in T1 where outflow concentration was higher than the inflow concentration. The higher outflow concentration could be due to desorption from the previous event or microbial ammonification of organic nitrogen. The lower transport reduction in the second runoff event could be due to low availability of surfaces for adsorption and fixation. However, adsorbed NH_4-N is readily nitrified when it comes into contact with oxygen and uptake by the plants.

Nitrate nitrogen concentrations at inflow were very low in all treatments and runoff events (data not shown). Very low NO_3-N concentrations were also observed in feedlot runoff in several rainfall events by Rahman et al. (2013). Low NO_3-N could be due to lack of nitrifying bacteria activity in fresh manure, although the nitrification was not measured in this study. A 100% reduction of NO_3-N concentration was observed in runoff in all treatments and runoff events which confirmed that NO_3-N moves readily into soil through infiltration.

5.4.4. Potassium transport reduction

Potassium concentrations at different sampling times are shown in table 5.4. No significant variation was observed between sampling times except T1 in the first runoff event. In the second runoff event, K concentrations were significantly different by sampling time in all treatments. Since potassium is a very soluble nutrient, its reduction in concentration is very low. Potassium concentration decreased by 6.22% and 20.5% in treatments T2 and T3, respectively, but increased by 12.7% in treatment T1 in the first runoff event. Increased K concentration in the outflow samples was also observed by other researchers (Hawkins et al., 1998). However, in the second runoff event, concentration reductions were almost similar for all treatments, and they were 13.3%, 11.1%, and 12.2% in treatments T1, T2, and T3, respectively. Highest transport reduction of K was observed in higher pH (T3), which could be due to precipitation and/or

Table 5.4. Mean inflow and outflow K concentrations in different sampling times in the first and second runoff event.

Treatment	Inflow	Outflow†				
		Time, min				Mean
		0	10	20	30	
	------ Runoff event 1, mg L^{-1} ------					
T1	183	199b	200b	214a	213a	207
T2	306	285a	288a	286a	290a	287
T3	177	131a	149a	155a	129a	141
	------ Runoff event 2, mg L^{-1} ------					
T1	299	255b	255b	258ab	267a	259
T2	266	226b	237b	248a	236b	237
T3	91.2	74.1b	79.2ab	80.9ab	86.0a	80.1

† Means followed by the same letter in a row is not significantly different at 90% significance level.

adsorption by exchange sites because potassium solubility decreases with increases in pH (Lindsay, 1979). Overall, K concentration reduction due to pH is not significant.

5.4.5. Mass transport reduction

Table 5.5 shows the inflow and outflow mass loads and corresponding transport reductions of nutrients mass. Inflow ortho-P mass loadings were 1293, 1021, and 726 mg m^{-2} in treatments T1, T2, and T3, respectively, in the first runoff event. The corresponding outflow mass loadings were 747, 479, and 264 mg m^{-2} in treatments T1, T2, and T3, respectively, in the same runoff event. With increasing pH, mass transport reductions of ortho-P were 42.2%, 53.1%, and 63.7% for the treatments T1, T2, and T3, respectively. T3 treatment also showed the highest ortho-P

Table 5.5. Inflow and outflow mass loads and mass transport reduction.

Treat-ment	Ortho-P			NH_4-N			K		
	In-flow	Out-flow	Redu-ction	In-flow	Out-flow	Redu-ction	In-flow	Out-flow	Redu-ction
				Runoff event 1					
	----- mg m^{-2} ----		-%-	---- mg m^{-2} ---		-%-	----- mg m^{-2} -----		-%-
T1	1293	747	42.2	4394	3129	28.8	6977	5379	22.9
T2	1021	479	53.1	3328	1445	56.6	10993	6246	43.2
T3	726	264	63.7	4432	2088	52.9	6397	3205	49.9
				Runoff event 2					
	------ mg m^{-2} ----		--%--	--- mg m^{-2} ---		--%--	---- mg m^{-2} ----		--%--
T 1	773	478	38.1	5707	4607	19.3	10766	7109	34.0
T2	909	499	45.1	1252	699	44.1	9609	5236	45.5
T3	685	338	50.7	3893	2199	43.5	3290	2099	36.2

concentration reduction (42.4%). Therefore, by increasing soil pH, ortho-P concentration and mass at outflow may be reduced significantly.

Inflow and outflow NH_4-N loadings were 4394 and 3129, 3328 and 1445, and 4432 and 2088 mg m^{-2} in T1, T2, and T3, respectively in the first runoff event. The highest NH_4-N mass transport reduction was for T2 (56.6%), followed by T3 (52.9%), and T1 (28.8%). Although NH_4-N concentration reduction was not significant, but NH_4-N mass transport reduction was significant at higher pH treatment (T2 and T3) as compare to control (T1).

Potassium loadings for inflow and outflow were 6977 and 5379 mg m^{-2} in T1, 10993 and 6246 mg m^{-2} in T2, and 6397 and 3205 mg m^{-2} in T3, respectively. Like ortho-P, potassium mass transport reductions increased with increasing pH and were 22.9%, 43.2%, and 49.9% in treatments T1, T2, and T3, respectively. Similar to the concentration reductions, mass transport reductions for all pollutants were low in the second runoff event compared to first runoff event. Trends of mass transport reductions with increasing pH were similar in the second runoff event as in the first runoff event for ortho-P and NH_4-N. Although, K concentration reduction due to pH was not significant, but mass transport reduction is significant at higher pH (43.2 and 49.9%) as compared to control (22.9%). Mass transport reductions of soluble nutrients are likely due to infiltration as well as sorption and/or precipitation.

5.4.6. Effect on leaching

Concentration of pollutants in leachate samples was generally low, and no general trend was apparent with the change in soil pH (table 5.6). Very little movement of TP and ortho-P was observed through soil profile because surface runoff is the key transport route for filter strip performance (Hoffmann et al., 2009; Dorioz et al., 2006). The highest TP and ortho-P concentrations were 5.6 and 0.61 mg L^{-1} observed in runoff event one in T3 and runoff event two in T1, respectively. Very low concentrations of NH_4-N and TKN were observed in leachate samples, the highest being observed were 2.92 and 4.4 mg L^{-1}, respectively, in first runoff event of T3. Ammonium nitrogen transported in runoff may have been absorbed by the vegetation and soil (Duchemin and Hogue, 2009) and resulting in low leachate concentration. The highest potassium concentration (13.5 mg L^{-1}) was observed second runoff event in T3. However, a threat to groundwater contamination was observed for NO_3-N because high concentration was observed in leachate in both T2 and T3. Out of the four runoff events in T2 and T3, three were observed to increase leachate NO_3-N concentration. The highest NO_3-N concentration was found in second runoff event of 404 mg L^{-1} in T3. Increased NO_3-N concentrations in leachate samples

Table 5.6. Concentrations of nutrients at supply and in the leachate.

Parameters	Treatments											
	T1				T2				T3			
	Runoff event 1		Runoff event 2		Runoff event 1		Runoff event 2		Runoff event 1		Runoff event 2	
	Inflow	Leachate	Inflow	Leachate	Inflow	Leachate	Inflow	Leachate	Inflow	Leachate	Inflow	Leachate
	mg/L											
Ortho-P	35.8	-	21.4	0.61	28.5	0.32	25.2	0.21	20.1	0.42	18.6	0.36
NH_4-N	122	0.61	158	2.73	92.8	1.64	34.7	0.6	123	2.92	108	2.84
NO_3-N	0.92	0.43	0.38	-	10.3	31.2	5.49	17.0	1.35	-	0.22	404
TKN	253	1.79	358	-	250	1.61	301	1.29	478	4.4	510	-
K	183	-	299	5.29	306	11.0	266	-	177	-	91.2	13.5

from the second runoff event could have been due to nitrification of NH_4-N from the first runoff event. After the first runoff event when soil was dry, it favored the nitrification by supplying oxygen and produced NO_3-N leached through the soil profile. Moreover, higher pH at treatment T2 and T3 favored nitrification (Olsen et al., 1970).

5.5. Conclusions

Effectiveness of vegetative filter strips is influenced by soil pH. As compared to control (T1), higher soil pH increased ortho-P transport reductions by 22.0 and 42.4% in treatments T2, and T3, respectively. Similarly, higher mass transport reduction was also observed at higher pH. The predominant mechanisms for ortho-P transport reduction were assumed to be precipitation and sorption. The concentration reductions of ammonium nitrogen were 1.72%, 26.1%, and 24% in treatments T1, T2, and T3, respectively. The key ammonium nitrogen transport reduction mechanisms were pH driven sorption, fixation, and volatilization. Nitrate nitrogen concentrations reduced in runoff by 100% regardless of treatments and runoff events. Compared to others, potassium transport reductions were lower, but mass transport reduction was much higher. All nutrients concentration in leachate was lower, except nitrate-N, which was due to infiltration. Higher nitrate nitrogen loss through leachate poses risk of groundwater contamination. Overall, changes of soil pH in vegetative filter strip may be an effective way to reduce ortho-P concentration in runoff, but pH treatment might not be very effective for other soluble nutrients.

5.6. Acknowledgements

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CHAPTER 6. A MODEL TO PREDICT SEDIMENT AND PHOSPHORUS TRAPPING EFFICIENCY OF VEGETATIVE FILTER STRIPS FROM FEEDLOT RUNOFF

6.1. Abstract

The objective of this research was to incorporate components in the existing VFSMOD model for predicting total suspended sediment (TSS) and phosphorus (P) trapping efficiency of vegetative filter strip (VFS) from feedlot runoff. In that effort, sub-models for upland phosphorus yield and transport and vegetative filter strip P transport components were coded and incorporated into the VFSMOD model. Later on, the model was calibrated and validated with 17 data points collected from a Richland County, North Dakota feedlot during the study period. Calculated highest average prediction accuracies were -45.8%, 37.5%, and 2.59% for predicting TSS, sediment bound P, and dissolved P, respectively. Similarly, for trapping efficiency prediction, highest accuracies were 76.2%, -29.4%, and 21.4% for TSS, sediment bound P, and dissolved P, respectively. Due to limited data points and difficulties in measuring runoff volume, the model is either under or over predicting. In the future, model predictability may be increased by measuring runoff volume accurately and by incorporating additional data.

6.2. Background

Animal agriculture is one of the major causes of nonpoint source pollution. Organic wastes, for example manure, are the sources of significant amounts of nitrogen and phosphorus, which are transported with runoff. Because of the intensive livestock facilities and land application of manure, increasing amount of nutrients, sediments, and bacteria are being released to the receiving water bodies (Gillingham and Thorrold, 2000).

Adverse impacts of agricultural nonpoint source pollution on surface water can be minimized by implementing best management practices (BMPs). A vegetative filter strip is one of the BMPs that may be installed at the downslope edge of a pollutant generating source or field to reduce transport of pollutant downstream. Various federal and state agencies are implementing different conservation and management practices to minimize pollution transport to water bodies, but there is little quantitative assessment of water quality improvement (White and Arnold, 2009). However, cost effective evaluation of conservation measures is challenging. Predicting pollutant loads and evaluating the effectiveness of any conservation practice such as VFSs, simulation by models has been often used as a cost-effective approach (Abu-Zreig et al., 2001). Modeling as a tool can make a system or management practice simpler, less expensive, and less time consuming. Simulation of certain practices or natural processes using an appropriate model helps to understand the potential of pollution, and preventive measures may be implemented. Additionally, understanding parameter interactions in a certain process is facilitated by model simulations, which might not have been achieved through field studies because of physical and financial limitations, environmental variability and time constraints. Thus, modeling can help study VFS effectiveness under varying set of conditions, understand basic processes involved, and develop design criteria (Abu-Zreig, 2001).

To aid in VFS design and evaluation through modeling, several studies have been conducted to simulate transport of pollutants. Overcash et al. (1981) developed a general mathematical model to predict concentration and mass reduction of pollutants in runoff from a VFS installed at the down gradient end of a manure-amended land. Using Overcash's equation for concentration prediction (Overcash et al., 1981) and the SCS curve number method for runoff prediction, Edwards et al. (1996) developed a VFS design algorithm to design buffer width to meet specific performance requirements such mass or concentration removals. Researchers at the University of Kentucky developed a model, GRASSF, to simulate sedimentation process in grass filter media and tested it in a laboratory for artificial rigid grass media and in the field (Barfield et al., 1978, 1979; Hayes et al., 1979; Hayes et al., 1982, 1984; Tollner et al., 1976, 1977). However, none of the studies were able to successfully model the complex situations that may occur in the VFSs. A more comprehensive mechanistic model was developed by Munoz-Carpena et al. (1999) called vegetative filter strip model (VFSMOD), a modified version of GRASSF, has been shown to effectively simulate complex situations that may occur in natural events. The model was later modified by incorporating an upslope input generating component (UH) and graphical user interface and called VFSMOD-W. The model has the capability to account for variable rainfall patterns, time dependent infiltration, and various surface conditions.

Modeling of VFSs for sediment transport has been successfully performed in several studies, but few studies have been undertaken to address other pollutant transport problems. Sabbagh et al. (2009) and Poletika et al. (2009) coupled an empirical equation with a mechanistic model (VFSMOD) and evaluated the pesticide transport reduction through the VFS. Kuo and Munoz-Carpena (2009) used VFMOD model to predict overland flow and sediment trapping through VFS and linked a simplified algorithm for predicting phosphorus outflow from the VFS from phosphorus mining areas. Rudra et al. (2010) developed a toolkit to design and evaluate VFSs for sediment, phosphorus, and bacteria transport reduction. They incorporated procedures for predicting phosphorus, bacteria, and sediment transport using VFSMOD. More research is still needed to develop a model that is capable to predict suspended sediment and phosphorus from VFS under varying conditions and pollutant runoff generating areas. Therefore, in this study, a procedure was incorporated with the existing VFSMOD model to predict sediment and phosphorus trapping efficiency of VFS from feedlot runoff. The model was calibrated and validated using the field data collected at the end of a buffer.

6.2.1. Objectives

The objectives of this study were:

(1) to develop a model to predict suspended sediment and phosphorus loss from feedlot, and

(2) to develop a model to predict suspended sediment and phosphorus trapping efficiency in a VFS, and

(3) to calibrate and validate the model using field data

6.3. Model Development

The main component of the current process based model is VFSMOD and an associated module called unit hydrograph utility (UH) that produces inputs for the main component. A schematic of the VFSMOD model is shown in figure 6.1.

Use of VFSMOD is facilitated by adding a UH utility to the model for generating inputs such as runoff hydrograph, rainfall hyetograph, and sedimentograph for VFSMOD from upslope source areas. For runoff hydrograph generation, UH utility uses the NRCS curve number method (USDA-NRCS, 1972) and unit hydrograph approach. For estimating sedimentograph, the modified universal soil loss equation is used (Williams, 1975). For a given rainfall amount and duration, a rainfall hyetograph is generated according to a NRCS storm type as selected by users. Using hydrograph, sedimentograph, and hyetograph as inputs, VFSMOD routes the overland flow and sediment through the VFS and calculates respective trapping efficiencies. Vegetative filter strip model (VFSMOD) uses the one-dimensional kinematic wave overland flow equations (Lighthill and Witham, 1955) for routing the overland flow, the Green-Ampt equations for unsteady rainfall (Chu, 1978; Mein and Larson, 1971, 1973; Skaggs and Khaheel, 1982; Munoz-Carpena et al., 1993) for infiltration simulation, and University of Kentucky sediment transport model for sediment transport simulation (Barfield et al, 1978, 1979; Tollner et al., 1976, 1977).

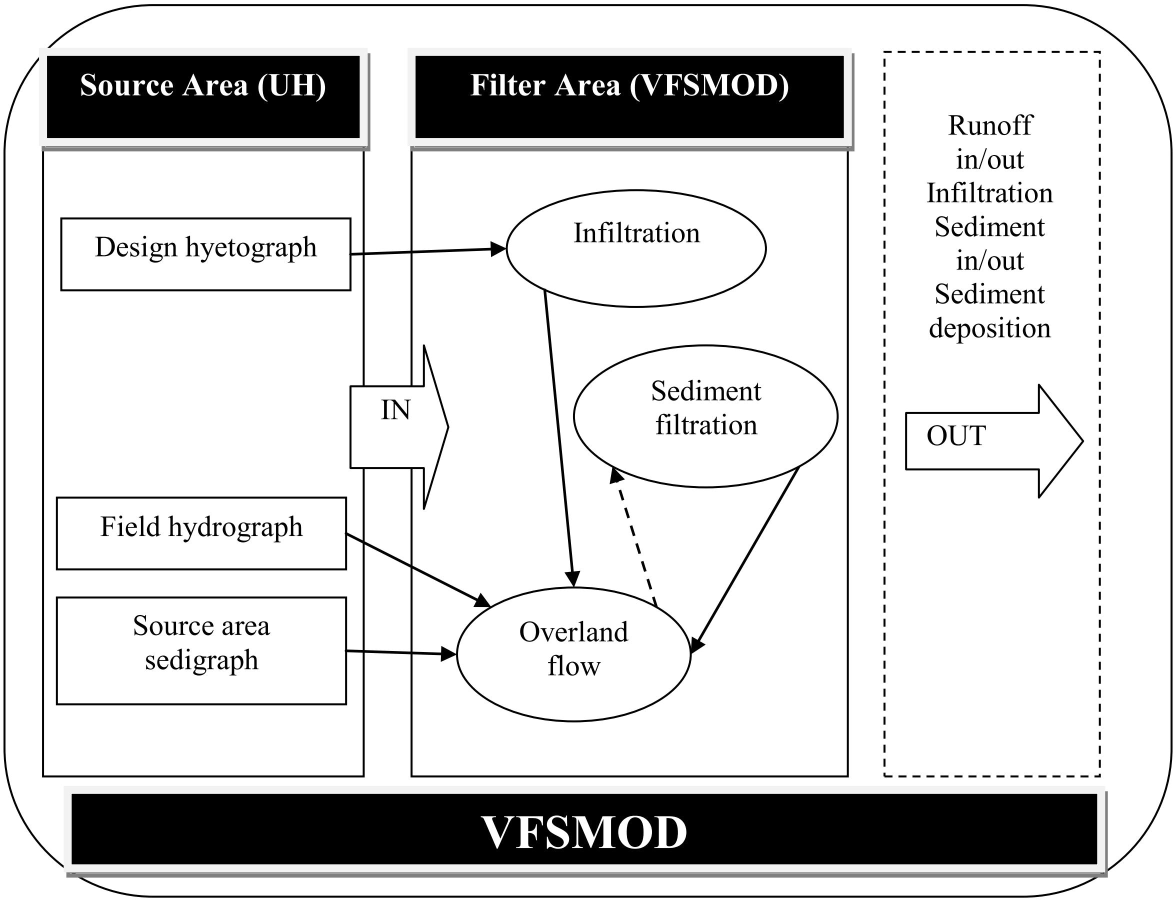

Figure 6.1. Schematic of the model (VFSMOD) for vegetative filter strip (after Munoz-Carpena and Parsons, 2004).

However, as previously mentioned, application of this model to nutrient transport problems is limited.

The present study aims to incorporate a procedure referred to as Upslope Phosphorus Yield and Transport (UH_P) into the UH utility of the existing VFSMOD to estimate upslope phosphorus yield in runoff at the point of entry into VFSs. A procedure referred to as Vegetative Filter Strip Phosphorus Transport Component (VFS_P) was also added into the VFSMOD to estimate P trapping efficiency when runoff is routed through VFSs. The procedure as suggested by Rudra et al. (2010) was used along with some modifications and was discussed in the following sections. The schematic representation of the proposed VFSMOD modules is shown in figure 6.2.

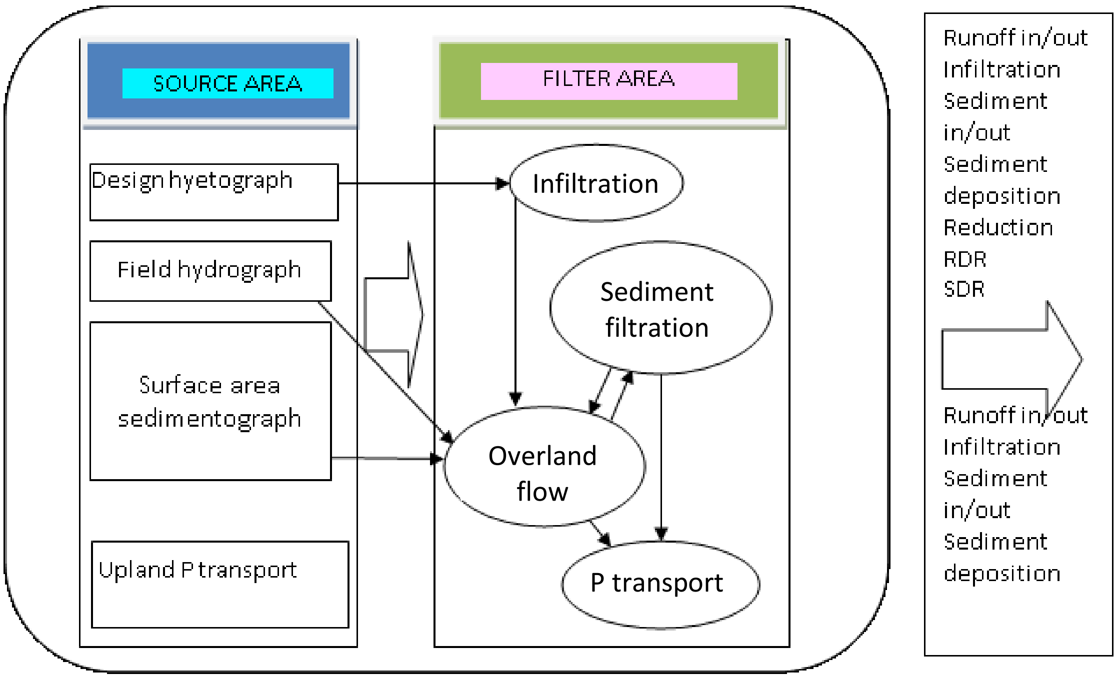

Figure 6.2. Schematic representation of the VFSMOD modules.

6.3.1. Upslope phosphorus yield and transport component (UH_P)

This component predicted the phosphorus yield at the field outlet, which is the best place for VFSs placement. From upland source areas to field outlets, phosphorus is transported as particulate form with sediment and dissolved form with runoff water, both of which enter into VFSs. For both particulate bound and dissolved phosphorus prediction, the EPIC model equations (Williams, 1995) were used. The sediment bound phase of phosphorus at the field outlet was predicted by

$$P_{sed}=0.01\times S_y\times P_o\times PER \qquad (6.1)$$

where,

P_{sed} = Sediment phase runoff P concentration, mg L^{-1}

S_y = Sediment yield, t ha^{-1}

P_o = Feedlot surface P concentration, mg kg^{-1}

PER = Phosphorus enrichment ratio which is the ratio of the specific surface area of the eroded sediment at the field outlet to the specific surface area of the sediment at the point of detachment. The PER ratio may be calculated as follows:

$$PER = \frac{(\text{specific surface area } (SS)_{out})}{(\text{specific surface area } (SS)_{in})} \tag{6.2}$$

The specific surface area of eroded particles can be estimated by knowing the particle size distribution of sediments at the point of interest. In this study, particle size distribution of sediments at the point of detachment (i.e., source area) was estimated by knowing the particle size distribution of the matrix soil. A method proposed by Foster et al. (1985) was used to determine the aggregate size distribution of eroded soil at the point of detachment. By this method, based on the fraction of primary soil particles (clay, silt, and sand), the fractions of particle classes in sediment such as clay, silt, sand, small aggregate, and large aggregate were estimated. Equations 6.3 to 6.9 (Foster et al., 1985) were used to estimate the fractions of particle classes in eroded sediment at the point of detachment.

$$ORcl=0.26PRcl \tag{6.3}$$

$$ORsa=PRsa(1-PRcl)^5 \tag{6.4}$$

$$ORsg=1.8PRcl \text{ when } PRcl <0.25 \tag{6.5}$$

$$ORsg= -0.6(PRcl-0.25)+ 0.45 \tag{6.6}$$

(when $0.25 \leq PRcl \leq 0.50$)

$$ORsg=0.6PRcl \text{ when } PRcl >0.50 \tag{6.7}$$

$$ORsi=PRsi-ORsg \tag{6.8}$$

$$ORlg=1-(ORcl+ORsi+ORsa+ORsg) \tag{6.9}$$

where,

PR = Fraction of primary particles in the soil,

OR = Fraction of particle classes in sediments, and

cl, si, sa, sg, and *lg* represent clay, silt, sand, small aggregate, and large aggregates, respectively.

For determining the phosphorus enrichment ratio using equation 6.2, the particle size distribution of eroded sediment at the field outlet has to be estimated. Because of the selective processes of deposition, a routing function developed by Williams (1980) was used to estimate the particle size distribution of the sediment at the field outlet (feedlot edge). This routing function was based on the aggregate size distribution of eroded soil at the point of detachment. The routing function is given by,

$$\omega_{oi}= \frac{\omega_i e^{-\beta\sqrt{d_i}}}{\left(\frac{q_p}{Q_P}\right)^{0.56}} \tag{6.10}$$

where, ω_{oi} = Portion of particle size d_i contained in the sediment

ω_i = Portion of particle size d_i contained in the soil

q_p = Peak runoff rate at the outlet of the source area, $m^3\ s^{-1}$

Q_P = Peak rate of rainfall excess, $m^3\ s^{-1}$

β = Routing coefficient

The routing coefficient, β, is defined as:

$$\beta= \frac{-\ln\left(\frac{q_p}{Q_P}\right)^{0.56}}{4.47} \tag{6.11}$$

Once particle size distributions are estimated for sediments at the point of detachment and at the field outlet, equation 6.12 (Williams, 1980) was used to estimate specific surface areas at the respective locations.

$$SS_i = 33(d_i)^{-0.1785} + 10.7MM \tag{6.12}$$

where,

SS_i = Specific surface area of the soil particles of diameter d_i, $m^2\ g^{-1}$

MM = Percent montmorillonite clay; can be obtained from literature

d = Particle diameter, μm

To predict the dissolved fraction of P at the outlet of upslope source area, the following equation was used:

$$DP = \frac{0.01 \times P_{sol} \times Q}{k_d} \tag{6.13}$$

DP = Dissolved phase of runoff P concentration, mg L^{-1}

P_{sol} = Feedlot surface dissolved P concentration, mg kg^{-1}

k_d = P concentration in the sediment divided by that of the water, $m^3\ t^{-1}$. The value of k_d of 175 was used in EPIC.

Q = Runoff volume, mm

6.3.2. Vegetative filter strip P transport component (VFS_P)

Upon receiving inputs from the UH and VFSMOD, VFS phosphorus transport component (VFS_P) estimated P removal efficiency by VFS. Two mechanisms are considered for phosphorus removal: removal of particulate bound P with sediment and removal of dissolved P with infiltrating water.

Particulate bound P removal is based on the assumption that the P is attached to the surface of the sediment particles, and the total amount of P is proportional to the total surface area of the sediment. Therefore, P removal efficiency is the ratio of the surface area of the sediment retained in the VFS to the total surface area of the sediment entering into VFSs. With the use of P enrichment ratio (PER) and sediment removal efficiency (equation 6.14) calculated in VFSMOD, the particulate bound P removal efficiency of VFSs is calculated by:

$$SRE=1-\frac{(\text{total sediment})_{out}}{(\text{total sediment})_{in}} \tag{6.14}$$

and

$$PRE=1-\frac{(\text{total surface area})_{out}}{(\text{total surface area})_{in}} \tag{6.15}$$

equation 6.15 can be written as

$$PRE=1-\frac{(\text{specific surface area})_{out}}{(\text{specific surface area})_{in}} \times \frac{(\text{total sediment})_{out}}{(\text{total sediment})_{in}} \tag{6.16}$$

Substituting equations 6.2 and 6.15 into equation 6.16,

$$PRE=1-PER_{VFS} \times (1-SRE) \tag{6.17}$$

where,

PRE = Sediment bound P removal efficiency

SRE = Sediment removal efficiency

PER_{VFS} = Phosphorous enrichment ratio for the VFS. Equation 6.18 was used to calculate PER_{VFS} for the VFS area (Rudra et al., 2010).

$$PER=0.05d_{50}+0.85 \tag{6.18}$$

where d_{50} is the median sediment particle size entering the VFS.

For dissolved fraction of P, water balance was performed assuming dissolved P was removed through infiltration. As dissolved nutrients are removed by VFSs through infiltration, trapping of dissolved fraction of the total phosphorous was calculated based on the total volume of runoff that infiltrated. Assuming that the dissolved P concentration is diluted due to rainfall in VFS and only the diluted dissolved P infiltrates into soil and removed from runoff, outflow dissolved P is computed as suggested by Kuo and Munoz-Carpena (2009), which follows

$$(\text{Dissolved Pmass})_{out} = (\text{Dissolved Pmass})_{in} - \text{Dissolved Pmass}_{infiltrated}$$

$\therefore$ $(\text{Dissolved Pmass})_{out} = DP_{in.} V_{in} - \frac{DP_{in} V_{in}}{V_{in}+V_{rain}} \times V_F$, and $V_F = V_{in} - V_{out} + V_{rain}$, from which

$$(\text{Dissolved P mass})_{out} = \frac{V_{in}\ V_{out}}{V_{in}+V_{rain}} \times DP_{in} \qquad (6.19)$$

where, DP corresponds dissolved phosphorus concentration and V corresponds volume of water. The subscripts in, out, F, and rain represent respective quantities at inflow, outflow, infiltration, and due to rainfall. Therefore, the dissolved P removal efficiency can be expressed as:

$$DPE = \frac{(\text{Dissolved P mass})_{in} - (\text{Dissolved P mass})_{out}}{(\text{Dissolved P mass})_{in}} \qquad (6.20)$$

where, DPE is dissolved P removal efficiency. Substituting equation 6.19 into equation 6.20,

$$DPE = \frac{DP_{in} V_{in} - \frac{V_{in}\ V_{out}}{V_{in}+V_{rain}} \times DP_{in}}{DP_{in} V_{in}} \qquad (6.21)$$

This can be reduced to

$$DPE = \frac{V_{in}(V_{in} + V_{rain} - V_{out})}{V_{in}(V_{in} + V_{rain})} \qquad (6.22)$$

But, $V_{in} + V_{rain} - V_{out}$=infiltration volume. Thus,

$$DPE= \frac{V_{infiltration}}{V_{in}+ V_{rain}} \tag{6.23}$$

6.3.3. Implementation of the model

The model was implemented as a standalone modified VFSMOD program written in FORTRAN (gfortran, from TDM-GCC). For Upslope Phosphorus Yield and Transport component (UH_P), input parameters such as feedlot surface P concentrations, rain event and amount, source soil moisture and soil types, and amount of sediment as well as functions describing both particulate and dissolved P transport mechanisms were coded in a subroutine. Similarly, upon receiving the inputs from UH and VFSMOD, the mechanisms for both sediment bound and dissolved P with infiltrating water were also coded in another subroutine. These subroutines were incorporated in VFSMOD.

The program requires 68 input parameters in seven input files. The input parameters used were collected from various sources including field measurement, literature, and model user's manual. Input parameters to be used in unit hydrograph (UH) utility for storm hyetograph and sedimentograph generation from upland source area are included in one file. Hydrological inputs (overland flow and infiltration) are included in another four files and sediment transport sub-model inputs are distributed into two files.

Many critical parameters were also obtained from published materials. Runoff curve number for the feedlot was obtained from the published data and adjusted to suit present feedlot conditions based on moisture status of the source area. Based on the magnitude of rainfall prior to a rainfall-runoff event, soil was grouped into three antecedent moisture conditions and a corresponding curve number was calculated using equation proposed by Ponce (1989). Critical parameters for infiltration and runoff volume predictions were saturated hydraulic conductivity (K_{sat}), suction depth (SAV), saturated water contents (θ_s) and initial moisture content (θ_o) (Fox et al., 2005). Saturated hydraulic conductivity was measured in the field and saturated water content and suction depth were estimated from a tool called Soil Water Characteristics (SWC) published by USDA (hppt://hydrolab.arsusda.gov/SPAW/Index.htm) based on soil texture, organic matter content, and average compaction. Initial moisture content was estimated based on the prior rainfall amount and field observation. Sensitive parameters for sediment transport were Manning's n, median particle size (Dp), and grass spacing (SS) were selected from the model user's manual and field observation. Slope, length, and widths were measured in the field. Parameters for soil loss were selected from the model manual.

The simulation program outputs the sediment, sediment bound and dissolved phosphorus loss from upland source, trapping efficiency of sediment, sediment bound phosphorus, and soluble phosphorus. Runoff volume and infiltration amount are also the outputs of the model.

6.3.4. Model evaluation/ model testing

6.3.4.1. Data collection, analysis, and preprocessing

For calibration and validation of the model, feedlot runoff data were collected from the Richland County feedlot and a detailed procedure has been described in Chapter 3. During that study, runoff data were collected from 17 rainfall events. Out of these data, 65% were used for calibration and the remaining data were used for validating the model. The data sets were subdivided based on VFS inflow sediment concentrations ensuring similar range of concentrations are used in both processes.

6.3.4.2. Calibration and validation

In the calibration process, sediment concentration was calibrated first. Sensitive parameters for sediment transport and hydrology components were adjusted to match the predicted sediment concentration with observed concentration. Following the same approach, the model was calibrated for phosphorus transport. After calibration, model was validated with the remaining data sets using parameters values optimized in calibration. Several statistics were calculated to measure the goodness of fit between the predicted and observed values. Root means square error (RMSE), average prediction accuracies (APA), standard error of prediction (SEP) (Kramer, 1998) were calculated using following equations:

$$\mathrm{RMSE}=\sqrt{\frac{1}{N}\sum_{i=1}^{N}(m_i-o_i)^2} \tag{6.24}$$

$$\mathrm{APA}(\%)=\frac{1}{N}\sum_{N}^{1}\left[1-\left(\frac{|o_i-m_i|}{o_i}\right)\right]\times 100 \tag{6.25}$$

$$\mathrm{SEP}=\sqrt{\frac{\sum_{i=1}^{N}[(o_i-m_i)-\bar{o}]^2}{N-1}} \tag{6.26}$$

where m_i and o_i are the predicted and observed values, respectively, and $i=1,2,3.......N$; $\bar{o}$ is the mean observed value. Model inputs used in the model are presented in tables 6.1 and 6.2.

6.4. Results and Discussion

6.4.1. Total suspended sediment and phosphorus loss prediction by UH utility

Figure 6.3 shows the predicted and observed TSS concentrations coming out of the feedlot and measured before the entry of the VFS. The lowest and highest concentrations observed were 110 to 3048 mg L^{-1}, respectively; whereas, the predicted lowest and highest concentrations were 377 and 3368 mg L^{-1}, respectively. Very low correlation was observed between the predicted and observed TSS concentrations. Root means square errors calculated were 1169 and 1283 mg L^{-1} (table 6.3), respectively, in the calibration and validation datasets, which are much higher than the lowest TSS concentration observed. Deviation between observed and predicted values was also indicated by low average prediction accuracy (APA), and high standard error of prediction (SEP) values, while negative APA values indicated over prediction. An APA value in the validation phase for TSS indicated 1.45 times over prediction.

Table 6.1. UH input parameters (for upland source area).

Description	Symbol	Value	Unit
NRCS (SCS) Curve Number for the source area	CN	78,89,95	
Area of the upstream portion	A	0.471	ha
storm type (1=I, 2=II, 3=III, 4=Ia)		II	
Length of the source area along the slope	L	70	m
Slope of the source area	Y	0.03	
Soil type		Sandy loam	
Soil erodibility factor (MUSLE)	K	-1	$t.ha.h\ ha^{-1}.MJ^{-1}.mm^{-1}$
C factor (MUSLE)	CFACT	0.4	
P factor (MUSLE)	PFACT	0.5	
Particle size	d_p	0.002	cm
Method to compute the storm R factor in MUSLE	IEROTY	2	
Organic matter (MUSLE)	OM	6	%
Sand fraction		0.754	
Silt fraction		0.165	
Clay fraction		0.081	
Feedlot surface P concentration	P_{sur}	2800	mg kg^{-1}
Montmorillonite clay	MM	5%	%

Table 6.2. Hydrological and sediment filtration model inputs.

Description	Symbol	Value	Unit
Filter length	VL	10.0	m
Filter width	FWIDTH	2.44	m
Filter mean Manning's coefficient	RNA	0.12	s $m^{-1/3}$
Number of nodes	N	57	-
Number of different filter segments	NPROP	1	-
Courant number	CR	0.8	
Order of shape functions	NPOL	3	-
Petrov Galerkin flag	KPG	1	-
Saturated hydraulic conductivity	VKS	7×10^{-6}	m s^{-1}
Average suction at the wet front	SAV	0.1101	m
Water content at saturation	OS	0.45	cm^3 cm^{-3}
Surface storage	SM	0	m
% of coarse particles (d_p>0.0037 cm)	COARSE	0.48	%
Porosity of deposited sediment	POR	0.434	unit fraction
Filter media spacing	SS	3.0	cm
Filter media height	H	20	cm
Grass modified Manning coefficient	VN	0.012	s $cm^{-1/3}$
Manning coefficient for bare soil	VN2	0.04	s $cm^{-1/3}$
Surface changes feedback	ICO	1	-
Incoming sediment particle class	NPART	7	-
Sediment particle density	SG	2.6	g cm^{-3}
Time-weight factor	THWTAW	0.5	

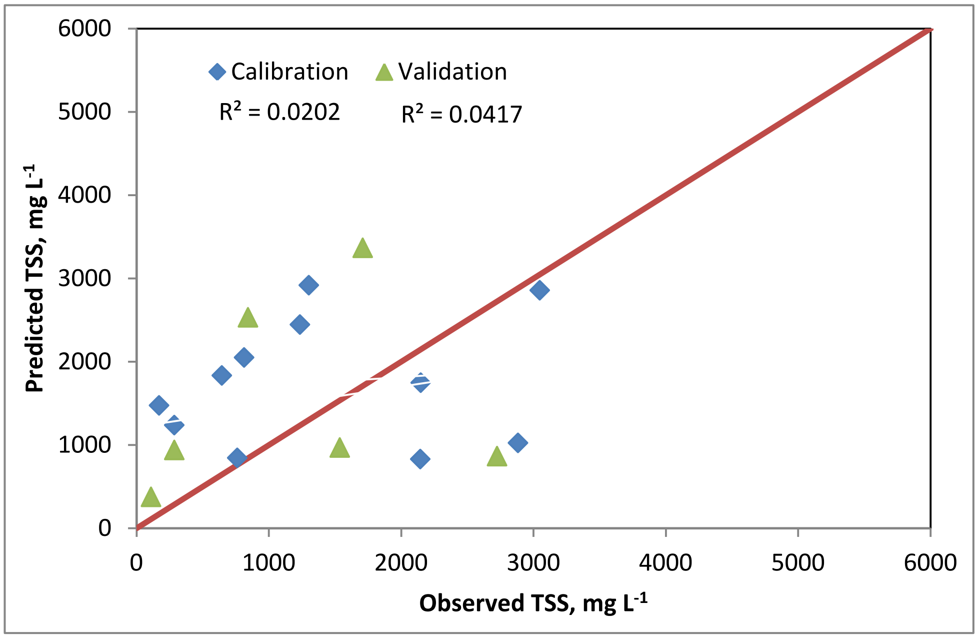

Figure 6.3. Predicted and observed TSS concentrations at the entry of the VFS.

Table 6.3. Calculated statistics used to assess quality of model results for predicting sediment and phosphorus loss from upland source area to VFS.

	TSS		Sediment-P		Dissolved P	
Parameters	Calibration	Validation	Calibration	Validation	Calibration	Validation
APA, %	-65.4	-45.8	37.5	10.7	2.59	0.95
RMSE, mg L^{-1}	1169	1283	14.0	14.1	2.78	1.01
SEP, mg L^{-1}	2077	1957	6.32	5.20	6.79	8.86

Predictability of the UH part was likely to be influenced by the curve number as curve number found very sensitive/critical to sediment concentration during calibration process, which was influenced by moisture content, animal density, animal activity, weight, and hoop size (Kizil et al., 2006). Unfortunately, we were not able to measure the curve number, and not using of appropriate curve number might have resulted in low prediction accuracy.

Figure 6.4 shows the predicted and observed sediment bound P concentrations. The highest and lowest observed sediment-P concentrations were 25.3 and 1.57 mg L^{-1}, respectively, and predicted highest and lowest sediment-P concentrations were 6.45 and 0.03 mg L^{-1}, respectively. Under prediction of sediment-P is also observed from the figure 5.4. Root means square error in calibration and validation processes were 14.0 and 14.1 mg L^{-1}, respectively. Low predictability could be related to low sediment predictability. Moreover, phosphorus transport depends on the sediment transport via enrichment ratio, which is calculated based on soil textural class. In this particular case from feedlot on coarse texture soil, manure particles also contributed to P transport, whose contribution is omitted in enrichment ratio calculation. The under prediction could be associated from not taking into account the manure particle contribution.

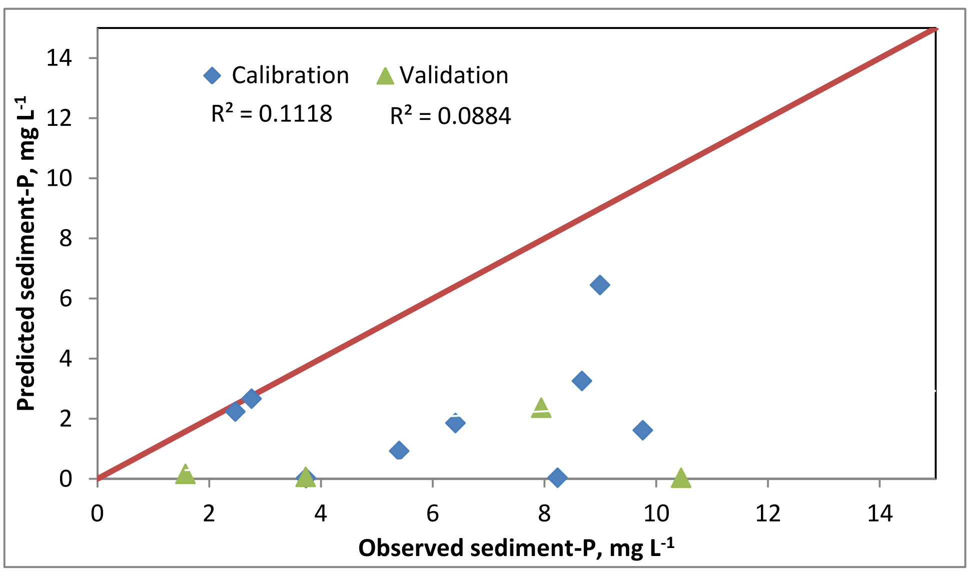

Figure 6.4. Sediment bound P concentrations at the entry of the VFS.

Figure 6.5 shows the observed and predicted dissolved P concentrations. Dissolved-P was under predicted for all rainfall runoff events. The model under predicted about by a factor of 100, indicated that the poor prediction could be related to low runoff volume prediction using the

EPIC model. In the equation 6.14, a constant factor of 175 is used to divide the concentration of soluble P loss in runoff. Low runoff volume from the large area have resulted very low soluble P prediction.

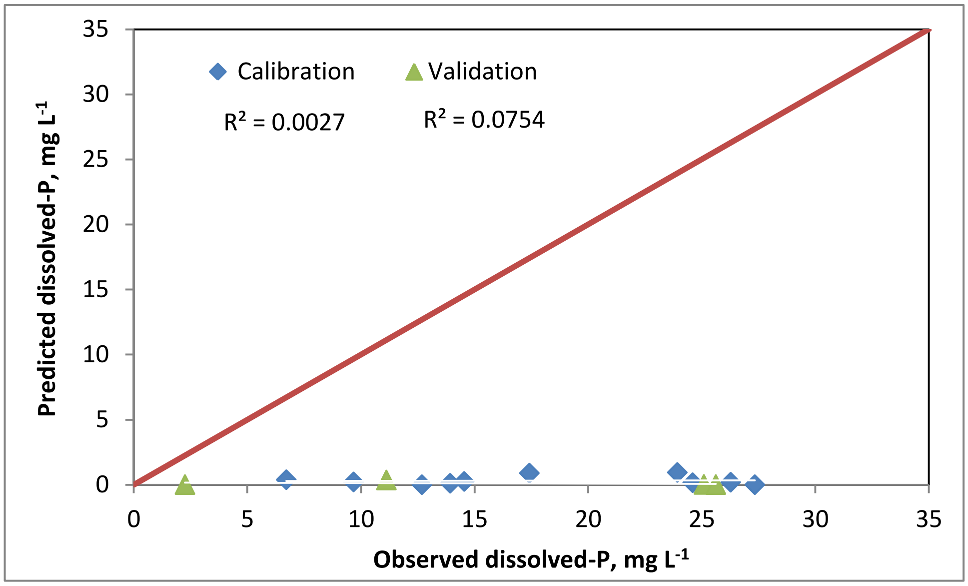

Figure 6.5. Predicted and observed dissolved P concentrations at the entry of the VFS.

6.4.2. Total suspended sediment and phosphorus trapping efficiency by VFSMOD

Figure 6.6 shows the observed and predicted total suspended sediment trapping efficiency through the VFS. Sediment trapping efficiency ranged from 67.6% to 100%, while observed sediment trapping efficiency ranged from 42.5% to 94.7%. The model predicted trapping efficiency well for those events when measured trapping efficiency was over 80%. However, better performance at high reduction efficiency was not reflected by the APA, RMSE, and SEP values in table 6.4, because the overall calibration and validation results are influenced by the numbers of events which resulted in low trapping efficiency. Poor model prediction results were also observed by other researchers (Abu-Zreig et al., 2001). Again in low runoff volume, trapping efficiency could be high (Munoz-Carpena, 2011). The sediment trapping model used was originally developed by Tollner et al. (1976) with the assumption that trapping of sediment is inversely proportional to some turbulence index. This indicates that as flow volume decreases the turbulence decreases resulting more deposition of sediments. However, improvements in model accuracy can be attained by using actual flow area rather than total area (Abu-Zreig et al., 2001). High infiltration volume resulted if total area is used instead flow concentrated area over which runoff actually flows. In sediment transport problem, particle diameter was found to be a very sensitive parameter.

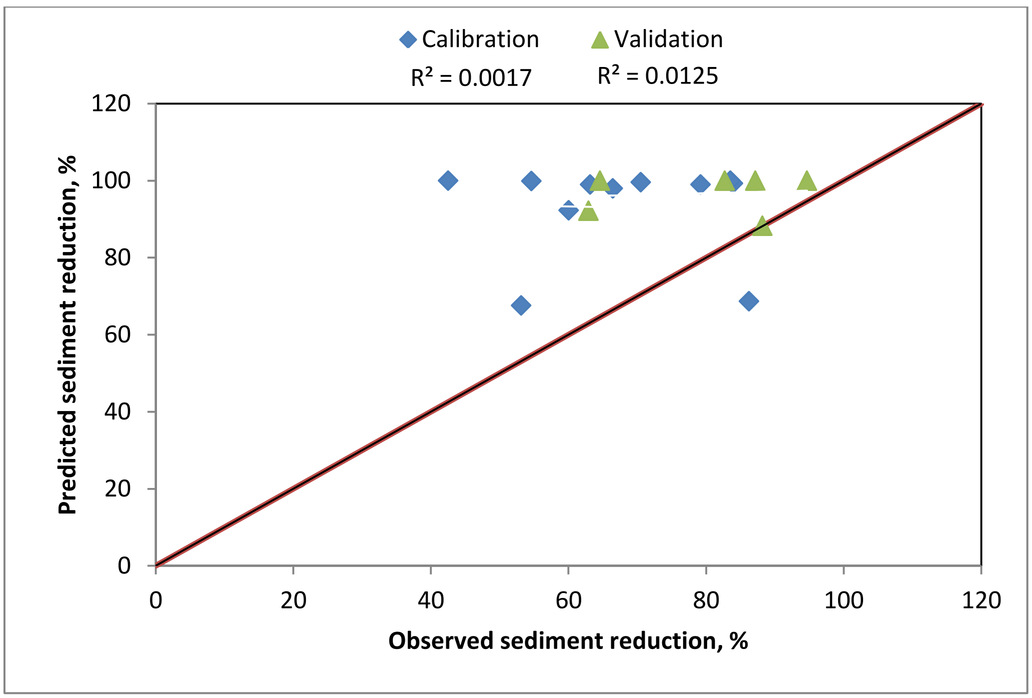

Figure 6.6. Observed and predicted sediment trapping efficiency of VFS.

As we were unable to measure inflow and outflow volume, model mass reduction was compared with observed concentration reductions (fig. 6.7). However, model predicted inflow and outflow volumes of relatively large rainfall events were used calculate mass reduction efficiency. Comparing with concentration reduction, it was found that they were close. Since sediment bound P transport mechanism is related to sediment transport amount, high trapping efficiency resulted from the model simulation value (fig. 6.7). Prediction of model efficiency ranged from 54.1% to 100% when there was no outflow from the filters. Trapping efficiency of observed sediment bound P ranged from 1.85% to 90.6%. All statistical parameters for goodness of fit of model prediction indicate inferior performance of sediment bound phosphorus trapping efficiency (table 6.4).

Table 6.4. Calculated statistics used to assess quality of model results for predicting sediment and phosphorus trapping efficiency of VFS.

	TSS		Sediment-P		Dissolved P	
Parameters	Calibration	Validation	Calibration	Validation	Calibration	Validation
APA, %	52	76.2	-29.4	-980	21.1	-144
RMSE, mg L^{-1}	31.6	20.9	48.7	49.5	39.4	63.5
SEP, mg L^{-1}	95	93.8	94.4	100.4	59.4	99.2

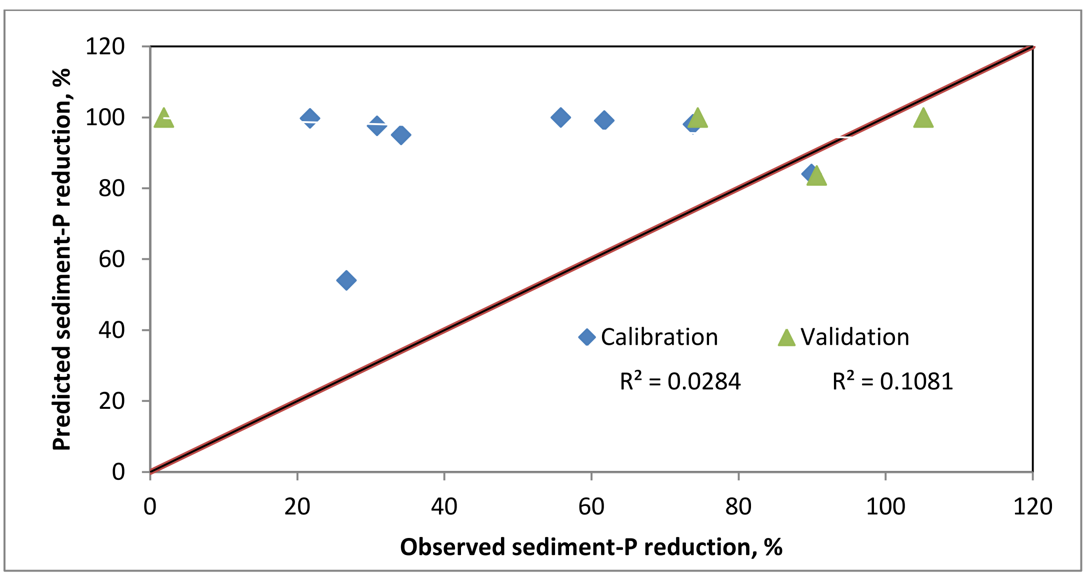

Figure 6.7. Observed and predicted sediment bound phosphorus trapping efficiency of VFS.

Figure 6.8 shows the observed and predicted dissolved phosphorus trapping efficiency. Since transport reduction mechanisms considered for dissolved phosphorus transport was infiltration, its transport reduction efficiency was found very high by model prediction. In this particular model application, infiltration was very high, which caused higher transport reduction.

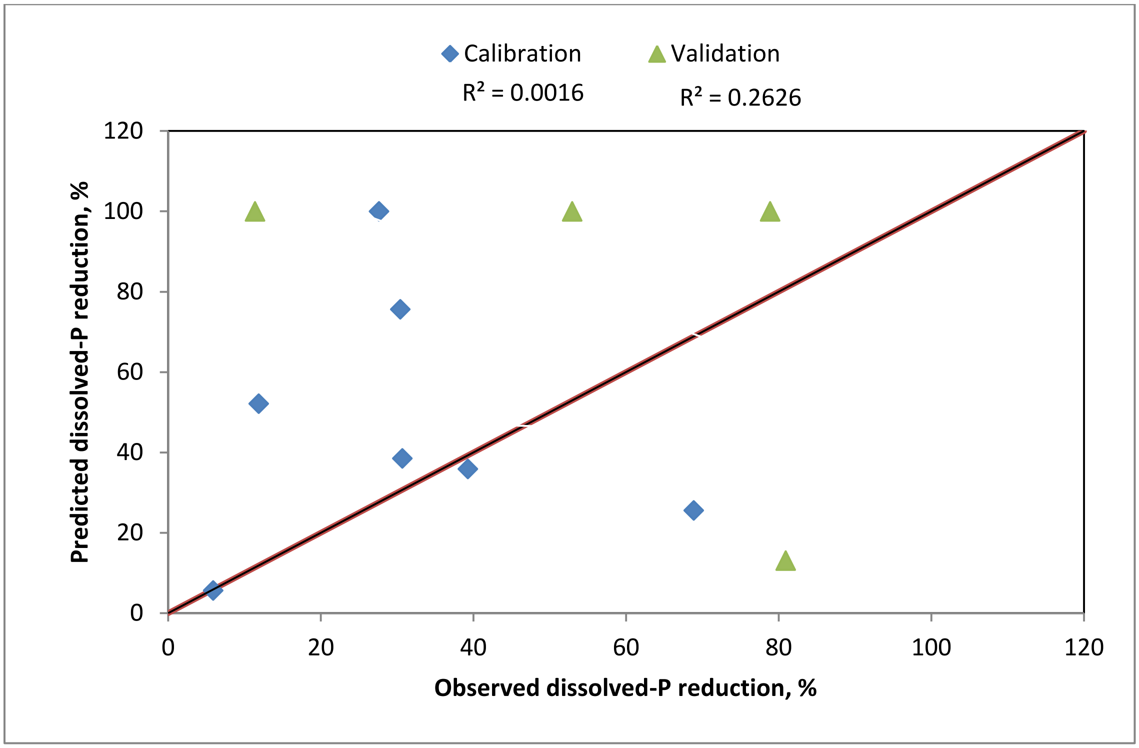

Figure 6.8. Observed and predicted dissolved phosphorus trapping efficiency of VFS.

6.5. Summary and Conclusions

A procedure was developed by coupling a phosphorus transport sub-model into an existing VFSMOD model with a view to predict the sediment and dissolved and sediment bound phosphorus loss from the upland source area and calculate trapping efficiency through vegetative filter strips. EPIC model equations were used to predict the phosphorus loss from the upland source area. Accuracies in predicting TSS concentrations could be increased using an appropriate curve number which could be determined either by direct measurement or choosing from literature. Phosphorus transport prediction could be improved by incorporating the contribution of manure particle or by incorporating measured runoff volume.

For trapping efficiency calculation of sediment bound and dissolved phosphorus, an algorithm based on sediment removal and water balance was used, respectively. The model was calibrated and validated with the field data. The model evaluation suggested inferior performance for sediment, sediment-P, and dissolved P prediction from the source area. However, model sediment trapping efficiency prediction was satisfactory for events with observed trapping efficiency higher than 80%. Trapping efficiency prediction for sediment-P and dissolved-P was inferior to sediment trapping efficiency. Sediment trapping efficiency over predicted for events in which low sediment trapping efficiency was observed. It was observed that model prediction accuracy largely depended on measurement of runoff volumes. This model can be used to predict sediment and phosphorus loss from the upland source area and to predict sediment, sediment bound phosphorus, and dissolved phosphorus trapping efficiency after careful calibration and validation in the field.

CHAPTER 7. GENERAL CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE STUDIES

7.1. Conclusions

The two chapters in this dissertation focused on the evaluation of vegetative filter strips situated at the edge of feedlot surface. The evaluation was aimed to find out if the filter strips were appropriately sited, designed, and managed in an effort to establish VFSs as an alternative technology to baseline technologies (e.g., storage basins, land application etc.) to control point and non-point pollution from runoff generated in animal feeding facilities. It was observed that the transport reduction efficiencies varied from very high to very low. Comparing performance of three VFS systems with different design parameters showed that filter strip with longer length was more effective in reducing transport sediments and nutrients. Broad leaf cattails (*Typha latifolia)* which had dense leaves and stems appeared to be as an important factor for effective VFS performance compared with mixed grasses. Predominant mechanisms for transport reductions were sedimentation, infiltration, precipitation, dilution, adsorption, and volatilization. Transport reduction of soluble pollutants was generally low. For some rainfall events, VFS systems appeared to be as zero discharged systems when soil moisture was very low thereby, pollutant discharge downstream was completely minimized. This indicated the importance of climatic influence on VFSs performance.

It was observed from the present study and from many past studies that VFSs are not as effective in reducing transport of soluble nutrients as sediment and sediment bound nutrients. An attempt was made by varying the VFS soil pH in a broader range to investigate if soil pH would have any effect on reducing transport of soluble nutrients. It was observed from the study that increasing soil pH from 7.5 to 8.5, reduced ortho-P concentration transport by 42.4% from a 2.44

m VFS. The highest ammonium concentration reductions were 26.1% in the 6.5 to 7.5 pH range. Potassium transport reduction was highest in the pH range 7.5 to 8.5. Surface transport reductions for nitrate nitrogen were 100% for all pH ranges but leaching at higher pH was observed, which indicated potential of groundwater pollution from the soil with higher pH. Calcium carbonate showed promise to increase soil pH and thereby reduce transport of soluble nutrients.

In an attempt to establish VFS as an alternative to baseline technologies, a model was developed to attempt to accurately describe the performance of VFSs. The model predicted the sediment, sediment bound phosphorus, and dissolved phosphorus yield from feedlot surface in runoff water. The model also predicted transport reduction efficiency of sediment, sediment bound phosphorus, and dissolved phosphorus from VFSs. The model considers important factors for filter strip performance. The model has the promise that it can be used to simulate VFS performance under varying set of conditions changing the important parameters. Careful and accurate choice of input parameters as well as proper calibration and validation were found most important for applications of the model.

7.2. Recommendations

- Due to variable flow rate, it was difficult to measure the total runoff for feedlots. In any future study, total runoff volume needs to be measured for determining mass transport reduction resulting from buffer.
- Sensitive parameters for hydrology and sediment transport sub-models should be accurately measured from the field.

- Although changing soil pH in buffer area showed promising results for soluble nutrient reduction, further research is needed under different soil conditions, soil types, and pH ranges.
- Since a shallow groundwater table exits in many places of North Dakota, the future performance evaluation should include a groundwater monitoring program to investigate the effect VFS on groundwater quality.
- The proposed model over-or under-predicted results. The accuracy of the model may be improved by incorporating measured total runoff volume. This model can be used as the basis for the future development of a good processed based model.

REFERENCES

Abu-Zreig, M. 2001. Factors affecting sediment trapping in vegetated filter strips: simulation study using VFSMOD. *Hydrol. Process.* 15: 1477-1488.

Abu-Zreig, M., R. P. Rudra, H.R. Whiteley, M.N. Lalovde, and N.K. Kaushik. 2003. Phosphorus removal in vegetated filter strips. *J. Env. Qual.* 32: 613-619.

Abu-Zreig, M., Rudra R. P., and H. Whitley. 2001. Validation of a vegetated filter strip model (VFSMOD). *Hydrol. Process.* 15(5): 729-742.

Alexander, E.L., and G.A. Margheim. 1974. Personal communications with C.E. Fogg.

Andersen, D., R. Berns, L. Moody, I. Khanijo, M. Helmers, C. Pederson, and J. Lawrence. 2009. Performance of six vegetative treatment systems for controlling runoff from open beef feedlots in Iowa. ASABE Paper Number: 097054. ASABE, St. Joseph, Mich.: ASABE.

APHA. 2005. *Standard Methods for Examination of Water Wastewater*. 2nd ed. Washington, D.C.: American Public Health Association.

Arora, K., S. K. Mickelson, J. L. Baker, D. P. Tierney, and C. J. Peters. 1996. Herbicide retention by vegetative buffer strips from runoff under natural rainfall. *Trans. ASABE* 46:2155-2162.

Asmussen, L. E., A. W. White Jr., E. W. Hauser, and J. M. Sheridan. 1977. Reduction of 2, 4-D load in surface runoff down a grassed waterway. *J. Environ. Qual*. 6:159-162.

Barfield, B. J., E. W. Tollner, and J. C. Hayes. 1978. The use of grass filters for sediment control in strip mining drainage. Vol. I: Theoretical studies on artificial media. Pub. no. 35-RRR2-78. Institute for Mining and Minerals Research, University of Kentucky, Lexington.

Barfield, B. J., E. W. Tollner, and J. C. Hayes. 1979. Filtration of sediment by simulated vegetation I. Steady-state flow with homogeneous sediment. *Trans. ASAE*. 22(5):540-545.

Barfield, B. J., R. L. Blevins, A. W. Fogle, C. E. Madison, S. Inamdar, D. I. Carey, and V. P. Evangelou. 1998. Water Quality impacts of natural filter strips in karst areas. *Trans. ASABE* 41:371–381.

Barrett, M. E. 1999. Complying with the Edwards Aquifer Rules: Technical Guidance on Best Management Practices. Austin: Texas Natural Resource Conservation Commission.

Barriouso, E., P. Benoit, and V. Bergheaud. 1994. Role of soil size fractions in retention and stabilization of pesticides in soil. In: *Environmental Behavior of Pesticides and Regulatory Aspects*, 5th symposium, Brussels (eds A. Copinn, G. Houins, L. Pussemier, and J. F. Salembier), pp. 138-143. European Study Service, Rixenhart Belgium.

Benoit, P., I. Madrigal, C. M. Preston, C. Chenu, and E. Barriuso. 2008. Sorption and desorption of non-ionic herbicides onto particulate organic matter from surface soils under different land uses. *European J. Soil Sci.* 59:178-189.

Bhattarai, R., P.K. Kalita, and M.K. Patel. 2009. Nutrient transport through a vegetative filter strip with subsurface drainage. *J. Environ. Mgt.* 90(2009):1868-1876.

Blanco-Canqui, H., C. J. Gantzer, and S.H. Anderson.2006. Performance of grass barriers and filter strips under interrill and concentrated flow. *J. Environ. Qual.* 35:1969-1974.

Boruvka, L. and J. E. Rechigl. 2003. Phosphorus retention by the Ap horizon of a spodosol as influenced by calcium amendments. *Soil Sci.* 168 (10): 699-706.

Bowden, J. W., A. M. Posner, and J. P. Quirk. 1980. Adsorption and charging phenomena in variable charge soils. In '*Soils with variable charge*'. (Ed. BKG Theng) pp. 146-166. (New Zealand Society of Soil Science: Lower Hutt, New Zealand).

Boyd, P. M., J. L. Baker, S. K. Mickelson, and S. I. Ahmed. 2003. Pesticide transport with surface runoff and subsurface drainage through a vegetative filter strip. *Trans. ASAE* 46(3):675-684.

Brauer, D., G. E. Aiken, D. H. Pote, S. J. Livingston, L. D. Norton, T. R. Way, and J. H. Edwards. 2005. Amendment effects on soil test phosphorous. *J. Environ. Qual.* 34:1682-1686.

Bubba, M.D., C. A. Arias, and H. Brix. 2003. Phosphorus adsorption maximum of sands for use medial in subsurface flow constructed reed beds as measured by the Langmuir isotherm. *Water Res.* 37: 3390–3400.

Butchbaker, A.F. (1973). Feedlot runoff disposal on grass or crops. L 1053. in GPE-7521, Great Plains Beef Cattle Feeding Handbook, Texas Agricultural Extension Service, Texas A&M University.

Chaubey, I. D. R. Edwards, T. C. Daniel, P. A. Moore Jr., and D. J. Nicols. 1994. Effectiveness of vegetative filter strips in retaining surface-applied swine manure constituents. *Trans. ASAE.* 37(3): 845-850.

Chaubey, I., D. R. Edwards, T. C. Daniel, P. A. Moore, and D. J. Nichols. 1995. Effectiveness of vegetative filter strips in controlling losses of surface-applied poultry litter constituents. *Trans. ASAE* 38(6):1687-1692.

Chu, S.T. 1978. Infiltration during unsteady rain. *Water Resour. Res.* 14 (3): 461–466.

Clark, R.N., Gilbertson C.B., and Duke H.R. (1975). Quantity and quality of beefyard runoff in the great planins. . in Managing Livestock Wastes, Proceedings of the Third International Symposium on Livestock Wastes, Proc-275 1975. American Society of Agricultural Engineers, St. Joseph, MI. .

Coyne, M. S., R. A. Gilfillen, A. Villalba, Z. Zhang, R. Rhodes, L. Dunn, and R.L. Blevins. 1998. Fecal bacteria trapping by grass filter strips during simulated rain. *J. Soil and Water Cons.* 53(2) 140-145.

Dickey, E. C., and D. H. Vanderholm, 1981. Vegetative treatment of livestock feedlot runoff. *J. Environ. Qual.* 10:279-284.

Dillaha, T. A., J. H. Sherard, D. Lee, S. Mostaghimi, and V. O. Shanholtz. 1988. Evaluation of vegetative filter strips as a best management practice for feed lots. *J. Water Pollut. Control Fed.* 60(7): 1231-1238.

Dillaha, T. A., R. B. Reneau, S. Mostaghimi, and D. Lee. 1989. Vegetative filter strips for agricultural non point source pollution control. *Trans. ASAE* 32(2):513-519.

Dorioz, J., M. D. Wang, J. Poulenard, and D. Trevisan. 2006. The effect of grass buffer strips on phosphorus dynamics- A critical review and synthesis as a basis for application in agricultural landscapes in France. *Agric. Ecosyst. Environ.* 117: 4-21.

Dosskey, M. G., M. J. Helmers, and D. E. Eisenhauer. 2008. A design aid for determining width of filter strips. *J. Soil and Water Cons.* 63(4):232-241.

Dou, Z., G. Y. Zhang, W. L. Stout, J. D. Toth, and J. D. Ferguson. 2003. Efficacy of alum and coal combustion by-products in stabilizing manure phosphorus. *J. Environ Qual.* 32:1490–7.

Dousset, S., M. Thévenot, D. Schrack, V. Gouy, and N. Carluer. 2010. Effect of grass cover on water and pesticide transport through undisturbed soil columns, comparison with field study (Morcille watershed, Beaujolais). *Environ. Poll.*158: 2446-2453.

Duchemin, M., and R. Hogue. 2009. Reduction in agricultural non-point source pollution in the first year following establishment of an integrated grass/tree filter system in southern Quebec (Canada). *Agric., Ecos. and Environ.* 131: 85-97.

Edward, D. R., T. C. Daniel, and P. A. Moore Jr. 1996. Vegetative filter strip design for grassed areas treated with animal manures. *Applied. Eng. in Agric.* 12(1):31-38.

Edwards, D. R., P. A. Moore Jr., T. C. Daniel, P. Srivastava, and D. J. Nichols. 1997. Vegetative filter strip removal of metals in runoff from poultry litter-amended fescue grass plots. *Trans. of the ASAE* 40(1):121-127.

Edwards, W. M., L. B. Owens, and R. K. White. 1983. Managing runoff from a small, paved beef feedlot. *J. Environ. Qual.* 12(2): 281-286.

Edwards, W. M., L. B. Owens, R. K. White, and N. R. Fausey. 1986. Managing feedlot runoff with a settling basin plus tiled infiltration bed. *Trans. of the ASAE* 29(1): 243-247.

Fajardo, J. J., J. W. Bauder, and S. D. Cash. 2001. Managing nitrate and bacteria in runoff from livestock confinement areas with vegetative filter strips. *J. Soil and Water Cons.* 56(3): 185-191.

Fasching, R. A., and J. W. Bauder. 2001. Evaluation of agricultural sediment load reductions using vegetative filter strips of cool season grasses. *Water Environ. Res.* 73(5):590-596.

Federal Register. 2003. 40 CR Parts 9, 122, 123, and 412, National Pollution Discharge Elimination System Regulation and Effluent Limitation Guidelines and Standards for Concentrated Animal Feeding Operations; Final Rule. 12 February 2003. 68(29):7176-7274.

Foster, G. R. and L. D. Meyer. 1972. Transport of soil particles by shallow flow. *Trans. ASAE* 15(1):99-102.

Foster, G. R., R. A. Young, and W. H. Neibling. 1985. Sediment composition for non-point source pollution analyses. *Trans. ASAE* 28(1):133-139,146.

Fox, A. L., D. E. Eisenhaur, and M. G. Dosskey. 2005. Modeling water and sediment trapping by vegetative filters using VFSMOD: Comparing methods for estimating infiltration parameters. ASAE Paper No. 052118. ASAE St. Joseph, Mich.

Gilley, J. E., B. Eghball, and D. B. Marx. 2007. Nutrient concentrations of runoff during the year following manure application. *Trans. ASABE* 50(6):1987-1999.

Gillingham, A. G., and B. S. Thorrold. 2000. A review of New Zealand research measuring phosphorus in runoff from pasture. *J. Environ. Qual.* 29(1): 88-96.

Giri, S., S. Mukhtar, and R. Wittie. 2010. Vegetative covers for sediment control and phosphorus sequestration from dairy waste application field. *Trans. ASABE* 53(3): 803-811.

Giri, S., S. Mukhtar, R. Wittie. 2008. Vegetative covers to control sediment and phosphorous in runoff from dairy waste application fields. ASABE paper number: 084132. ASABE, St. Joseph, Mich.: ASABE.

Goel, P. K., R. P. Rudra, B. Gharabaghi, S. Das, and N. Gupta. 2004. Pollutants removal by vegetative filter strips planted with different grasses. ASAE/CSAE paper number: 042177.

Presented at the ASABE annual international meeting from 1-4 August at Ottawa, ON, Canada.

Gollehon, N., M. Caswell, M. Ribaudo, R. Kellogg, C. Lander, and D. Letson. 2001. Confined animal production and manure nutrients. AIB-771, June. USDA, Econ. Res. Serv., Washington, DC.

Han, J., J. S. Wu, and C. Allan. 2005. Suspended sediment removal by vegetative filter strip treating highway runoff. *J. Environ Sci. and Health* 40:1637-1649.

Hawkins, G. L., D. T. Hill, E. W. Rochester, and C. W. Wood. 1998. Evaluation of vegetative filter strips for swine lagoon waste water. *Trans. ASAE* 41(3):639-643.

Hay, V., W. Pittroff, E. E. Tooman, and D. Meyer. 2006. Effectiveness of vegetative filter strips in attenuating nutrient and sediment runoff from irrigated pastures. *J. Agric. Sci.* 144:349-360.

Hayes, J. C., B. J. Barfield, and R. I. Barnhisel. 1979. Filtration of sediment by simulated vegetation II. Unsteady flow with non-homogeneous sediment. *Trans. ASAE.* 22(5):1063-1967.

Hayes, J. C., B. J. Barfield, and R. I. Barnhisel. 1982. The use of grass filters for sediment control in strip mine drainage. III. Empirical verification of procedures using real vegetation. Report No. IMMR82/070.

Hayes, J. C., B. J. Barfield, and R. I. Barnhisel. 1984. Performance of grass filters under laboratory and field conditions. *Trans. ASAE* 27 (5), 1321–1331.

Hoffmann, C. C., C. Kjaergaard, J. Uusi-Kamppa, H. C. B. Hansen, and B. Kronvang. 2009. Phosphorus retention in riparian buffers: review of their efficiency. *J. Environ. Qual.* 38:1942-1955.

Hubbard, R. K., G. L. Newton, J. G. Davis, R. Lowrance, G. Vellidis, and C. R. Dove, 1998. Nitrogen assimilation by riparian buffer systems receiving swine lagoon waste water. *Trans. ASAE* 41(5): 1295-1304.

Johnston, C. A. 1991. Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. *Crit. Rev. Environ. Control* 21:491-565.

Kanel, J. D., and J. W. Morse. 1978. The chemistry of orthophosphate uptake from seawater on to calcite and aragonite. *Geochimia et Cosmochimica Acta* 42:1335-1340.

Kanneganti, V. R., and S. D. Klausner. 1994. Nitrogen recovery by orchard grass from dairy manure applied with or without nitrogen fertilizer. *Commun. Soil Sci. Plant Anal.* 25:2771-2783.

Kansas Department of Health and Environment (KDHE). 1995. Design standards for confined livestock feeding operations. Topeka, KS.

Kim, J.R., Zuo Y., Regan J.M., and Logan E. 2008. Analysis of ammonia loss mechanisms in microbial fuel cells treating animal wastewater. . *Biotechnol. and Bioeng*. 99: 1120-1127.

Kim, Y., L. D. Geohring, and T. S. Steenhuis. 2005. The fate of phosphorous from milkhouse waste discharged to a vegetative filter strip. ASAE paper number: 052177. St. Joseph, Mich.

Kizil, U. 2010. Comparison of alternative feedlot runoff management systems. *African J. Ag. Res.* 5(3): 236-242.

Kizil, U., J.A. Lindley, and G. Padmanabhan. Verification of Nutrient Transport Modelling of a Bison Feedlot. *Biosys. Eng.* 44(3): 453–460.

Knisel, W. G. 1980. CREAMS: a field-scale model for chemicals, runoff, and erosion from agricultural management systems. Conservation Research Report No. 26. USDA-SEA, Washington, DC.

Koelsch, R. K., J. C. Lorimerr, and K. R. Mankin. 2006. Vegetative treatment systems for management of open lot runoff: review of literature. *Applied. Eng. in Agric.* 22(1):141-153.

Kramer, R. Chemometric Techniques for Quantitative Analysis. New York, N.Y.: Marcel Dekker.

Krutz, L. J., S. A. Senseman, M. C. Dozier, D. W. Hoffman, and D. P. Tierney. 2003. Infiltration and adsorption of dissolved atrazine and atrazine metabolites in buffalograss filter strips. *J. Environ. Qual.* 32: 2319-2324.

Krutz, L. J., S. A. Senseman, R. M. Zablotowicz, and M. A. Matocha. 2005. Reducing herbicide runoff from agricultural fields with vegetative filter strips: a review. *Weed Sci.* 53:353-367.

Kruzic, A. P. and E. D. Schroeder. 1990. Nitrogen removal in the overland flow wastewater treatment process- removal mechanisms. *J. Water Pollut. Control Fed.* 62: 867 -876.

Kuo, Y., and R. Munoz-Carpena. 2009. Simplified modeling of phosphorous removal by vegetative filter strips to control runoff pollution from phosphate mining areas. *J. Hydrol.* 378: 343-354.

Lance, J. C. Nitrogen removal by soil mechanisms. 1972. *J. Water Environ. Control. Fed.* 44(7): 1352-1361.

Laws, E. 1993. Aquatic Pollution: An Introductory Text (2nd ed.): Wiley and Sons, New York.

Lee, K-H., T. M. Isenhart, R. C. Schultz, and S. K. Mickelson. 1999. Nutrient and sediment removal by switchgrass and cool-season grass filter strips in Central Iowa, USA. *Agroforestry Syst.* 44:121-132.

Lighthill, M.J., and C. B. Whitham. 1955. On kinematic waves: flood movement in long rivers. *Proc. R. Soc. London Ser.* A 22, 281–316.

Lim, T. T., D. R. Edwards, S. R. Workman, B. T. Larson, and L. Dunn. 1998. Vegetated filter strip removal of cattle manure constituents in runoff. *Trans. ASAE* 41(5): 1375-1381.

Lindsay, W. L. 1979. Chemical equilibria in soils. The Black Burn Press, New Jersey, NJ, USA. Pp. 162-204, 268-279.

Litator, M. I., O. Reichmann, M. Belzer, K Auerswald, A. Nishri, and M Shenker. 2003. Spatial analysis of phosphorus sorption capacity in a semiarid altered wetland. *J. Environ. Qual.* 32:335-343.

Liu, X., X. Zhang, and M. Zhang. 2008. Major factors influencing the efficacy of vegetated buffer on sediment trapping: a review and analysis. *J. Environ. Qual.* 37:1667-1674.

Lorimor, J.C., Shouse S., and Miller W. (2002). Vegetative filter strips for open feedlot runoff treatment. in Extension Bulletin PM 1919,: Iowa State University Press, Ames, Iowa. .

Magette, W. L., R. B. Brinsfield, R. E. Palmer, and J. D. Wood. 1989. Nutrient and sediment removal by vegetated filter strips. *Trans. ASAE* 32(2):663-667.

Mankin, K. R. and C. G. Okoren. 2003. Field evaluation of bacteria removal in a VFS. ASAE Paper No. 032150. St. Joseph, Mich.: ASAE.

Mankin, K. R., D. M. Ngandu, C. J. Barden, S. L. Hutchinson, and W.A. Geyer. 2007. Grass-shrub riparian buffer removal of sediment, phosphorous, and nitrogen from simulated runoff. *J. American Water Res. Assoc.* 43(5) 1108-1116.

Mankin, K. R., P. L. Barnes, J. P. Harner, P. K. Kalita, and J. E. Boyer. 2006. Field evaluation of vegetative filter effectiveness and runoff quality from unstocked feedlots. *J. Soil and Water Cons.* 61(4): 209-217.

Mayer, L.D. and Wischmeier W.H. (1969). Mathematical simulation of the process of soil erosion by water. . *Trans. ASAE* 12: 754-758.

McFarland, A. M. S., and L. M. Hauck. 1999. Relating agricultural land uses to in stream Storm water quality. *J. Environ. Qual.* 28: 836-844.

McLeod, R.V., and R. O. Hegg. 1984. Pasture runoff quality from application of inorganic and organic nitrogen sources. *J. Environ. Qual.* 13(1):122-126.

Mein, R.G., and C. L. Larson. 1971. Modelling the infiltration component of the rainfall-runoff process, Bulletin 43, University of Minnesota, MN, Water Resources Research Center.

Mein, R.G., and C. L. Larson. 1973. Modeling infiltration during a steady rain. *Water Resourc. Res.* 9 (2):384–394.

Mendez, A., T. A. Dillaha, and , S. Mostaghimi. 1999. Sediment and nitrogen transport in grass filter strips. *J. Am. Water Resourc. Assoc.* 35(4): 867-875.

Mersie, W., C. A. Seybold, C. McNamee, and M. A. Lawson. 2003. Abating endosulfan from runoff using vegetative filter strips: the importance of plant species and flow rate. *Agric. Ecosyst. and Environ.* 97:215-223.

Midwest Plan Service (MWPS).2000. Manure characteristics: manure management systems series. MWPS-18 Section-1. Ames, Iowa: Midwest Plan Service.

Miller, J. J., B. J. Handerek, B. W. Beasley, E. C. S. Olsen, L. J. Yanke, F. J. Larney, T. A. McAllister, B. M. Olsen, L. B. Selinger, D. S. Chanasyk, and P. Hasselback. 2004. Quantity and quality of runoff from a beef cattle feedlot in southern Alberta. *J. Environ. Qual.* 33(3):1088-1097.

Muñoz-Carpena, R. 2011. VFSMOD-W. Vegetative Filter Strips Modeling System. Model documentation &user's manual. Agricultural & Biological Engineering, University of Florida.

Munoz-Carpena, R., and J. E. Parsons. 2004. A design procedure for vegetative filter strips using VFSMOD-W. *Trans. ASAE* 47(6):1933-1941.

Muñoz-Carpena, R., C. T. Miller, and J. E. Parsons. 1993a. A quadratic Petrov-Galerkin solution for kinematic wave overland flow. *Water Resour. Res.* 29(8):2615-2627.

Munoz-Carpena, R., J. E. Parsons, and J. W. Gilliam. 1993. Numerical approach to the overland flow problem in vegetative filter strips. *Trans. ASAE* 36(3): 761-770.

Munoz-Carpena, R., J. E. Parsons, and J.W. Gilliam. 1993. Numerical approach to the overland flow problem in vegetative filter strips. *Trans. ASAE* 36 (3): 761–770.

Munoz-Carpena, R., J. E. Parsons, and J.W. Gilliam. 1999. Modeling hydrology and sediment transport in vegetative filter strips. *J. Hydrol.* 214: 111-129.

Munoz-Carpena, R., J. E. Parsons, and W. Gilliam. 1992. Vegetative filter strips: modeling hydrology and sediment movement: ASAE paper No. 92-2625. St. Joseph, Mich.:ASAE.

Munoz-Carpena, R., Z. Zajac, and Y. M. Kuo. 2007. Global sensitivity and uncertainty analyses of the water quality model VFSMOD-W. *Trans. ASABE* 50(2): 1719-1732.

Murphy, P. N. C. R. J. Stevens. 2010. Lime and gypsum as source measures to decrease phosphorus loss from soils to water. *Water Air Soil Pollut.* 212:101–111.

Murphy, P. N. C. and J. T. Sims. 2012. Effects of Lime and Phosphorus Application on Phosphorus Runoff Risk. *Water Air Soil Pollut.* 223: 5459–5471.

Nair, P. S., T. J. Logan, A. N. Sharpley, L. E. Sommers, M. A. Tabatabai, and T. L. Yuan. 1984. Interlaboratory comparison of a standardized phosphorus adsorption procedure. *J Environ. Qual.* 3:591–595.

NDAWN, 2013. North Dakota Agricultural Weather Network. Rainfall data available at http://ndawn.ndsu.nodak.edu/, accessed on 3/15/2013)

NDDoH. 2010. North Dakota 2010 Integrated Section 305(b) Water Quality Assessment Report and Section 303(d) List of Waters Needing Total Maximum Daily Loads.

NDDoH. 2005. North Dakota livestock program design manual. North Dakota Department of Health, Bismarck, North Dakota.

Olsen, R. J., R. F. Hensler, and O. J. Atto. 1970. Effect of manure application, aeration, and soil pH on soil nitrogen transformations and on certain soil test values. *Soil Sci. Soc. Am. J.* 34(2): 222-225.

Overcash, M. R., S. C. Bingham and P. W. Westerman. 1981. Predicting runoff pollutant reduction in buffer zones adjacent to land treatment sites. *Trans. ASAE.* 24(2): 430-435.

Parker, D. B., D. E. Eisenhauer, D. D. Schulte, and D. L. Martin. 1999. Seepage characteristics and hydraulic properties of feedlot runoff storage pond. *Trans. ASAE* 42(2): 369-380.

Patterson, J. J., J. H. Jones, F. J. Olsen, and G. C. McCoy. 1980. Dairy liquid waste distribution in an overland flow vegetative-soil filter system. *Trans. ASAE* 23(4):483-488.

Poletika, N. N., P. N. Coody, G. A. Fox, G. J. Sabbagh, S. C. Dolder, and J. White. 2009. Chlorpyrifos and atrazine removal from runoff by vegetated filter strips: experiments and predicting modeling. *J. Environ. Qual.* 38:1042-1052.

Ponce, V. M. 1989. Engineering Hydrology: Principles and Practices. Prentice Hall, NY.

Rahman A, S. Rahman, and M.S. Borhan. 2013. Performance Evaluation of Three Vegetative Filter Strip Designs for Controlling Feedlot Runoff Pollution. *J. Civil Environ. Eng*. 3: 124.

Rahman A., S. Rahman, and L. Cihacek. 2012. Efficacy of vegetative filter strip to minimize solids and nutrients from feedlot runoff. *Agric Eng Int: CIGR J.* 14(4): 9-21.

Robinson, C. A., M. Ghaffarzadeh, and R. M. Cruse. 1996. Vegetative filter effects on sediment concentration in cropland runoff. *J. Soil and Water Cons.* 51(3): 227-230.

Roodsari, R. M., D. R. Shelton, A. Shirmohammadi, Y. A. Pachepsky, A. M. Sadeghi, and J. L. Starr. 2005. Fecal coliform transport as affected by surface condition. *Trans. ASAE* 48(3):1055-1061.

Rudra, R. P., B. Gharabaghi, S. Sabti, N. Gupta, and A. Moharir. 2010. GDVFS: a new toolkit for analysis and design of vegetative filter strips using VFSMOD. *Water Qual. Res. J. Can.* 45(1): 59-68.

Sabbagh, G. J., G. A. Fox, A. Kamanzi, B. Roepke, and J. Z. Tang. 2009. Effectiveness of vegetative filter strips in reducing pesticide loading: Quantifying pesticide trapping efficiency. *J. Environ. Qual.* 38(2):762-771.

Sanderson, M. A., R. M. Jones, M. J. McFarland, J. Stroup, R. L. Reed, and J. P. Muir. 2001. Nutrient movement and removal in a switchgrass biomass-filter strip system treated with dairy manure. *J. Environ. Qual*. 30:210-216.

SAS. (2009). SAS version 9.1Cary, NC: SAS Institute Inc.

Schellinger, G. R. and J. C. Clausen. 1992. Vegetative filter treatment of dairy barnyard runoff in cold regions. *J. Environ. Qual.* 21(1):40-45.

Schmitt, T. J., M. G. Dosskey, and K. D. Hoagland. 1999. Filter strip performance and processes for different vegetation, widths, and contaminants. *J. Environ. Qual.* 28:1479-1489.

Schwer, S. B. and J. C. Clausen. 1989. Vegetative filter treatment of dairy milkhouse wastewater. *J. Environ. Qual.* 18: 446-451.

Scotford, I. M., T. R. Cumby, R. P. White, O. T. Carotn, F. Lorenz, U. Hatterman, and G. Provolo. 1998. *J. of Agric. Eng. Res*. 71(3): 291-305.

Sharpley, A. N., L. R. Ahuja, M. Yamamoto, and R. G. Mengel. 1981. The release of soil phosphorous in runoff in relation to the kinetics of desorption. *J. Environ. Qual.* 10: 386-391.

Sharpley, A.N., Chapra S.C., Wedepohl R., Sims J.T., Daniel T.C., and Reddy K.R. (1994). Managing agricultural phosphorus for protection of surface waters - issues and options. *J. Environ. Qual.* 23(3): 437-451.

Skaggs, R.W., R. Khaheel. 1982. Infiltration, Ch. 4, In: Hydrologic modeling of small watersheds. C.T. Haan, H.P. Johnson, D.L. Brakensiek, Eds., St. Joseph, MI, ASAE. pp. 121–168.

Sotomayor-Ramirez, D., G. A. Martinez, J. Ramirez-Avila, and E. Mas. 208. Effectiveness of grass filter strips for runoff nutrient and sediment reduction in dairy sludge-amended pastures. *J. Agric. Univ. P.R.* 92(1-2):1-14.

Spellman, F. R., and N. E. Whiting. 2007. Environmental Management of Concentrated Animal Feeding Operations (CAFOs), Boca Raton, Florida, USA: CRC Press.

Srivastava, P., D. R. Edwards, T. C. Daniel, P. A. Moore Jr., and T. A. Costello. 1996. Performance of vegetative filter strips with varying pollutant source and filter strip lengths. *Trans. ASAE.* 39(6):2231-2239.

Stevens, R. J., C. J. O'Bric and O. T. Carton. 1995. Estimating nutrient content of animal slurries using electrical conductivity. *J. Agric. Sci., Cambridge*. 125(2): 233-238.

Stout, W. L., A. N. Sharpley, and H. B. Pionke. 1998. Reducing soil phosphorus solubility with coal combustion by-products. *J. Environ. Qual*. 34: 111-118.

Stout, W. L., Y. A. Pachepsky, D. R. Shelton, A. M. Sadeghi, L. S. Saporito, and A. N. Sharpley. 2005. Runoff transport of faecal coliforms and phosphorus released from manure in grass buffer conditions. *Letters in Appl. Microbio.*. 41: 230- 234.

Svendsen, L. M. 1992. Dynamics of phosphorus, organic nitrogen and organic matter in watercourses-methods, retention, transport and models. Ph.D. Thesis. Ministry of the Environment, National Environmental Research Institute, Denmark, p. 260.

Swanson, N. P., N. L. Mielke, J. C. Lorimor, T. M. McCalla, and J. R. Allis. 1971. Transport of pollutants from sloping cattle feedlots as affected by rainfall, intensity, duration, and reoccurrences. In: *Livestock Waste Management and Pollution Abatement*. Pp 51-55. ASAE, St. Joseph, MI 49085: ASABE.

Syversen, N. 2005. Effect and design of buffer zones in the Nordic climate: The influence of width, amount of surface runoff, seasonal variation and vegetation type on retention efficiency for nutrient and particle runoff. *Ecol. Eng*. 24:483-490.

Syversen, N., and H. Borch. 2005. Retention of soil particle fractions and phosphorus in cold-climate buffer zones. *Ecol. Eng*. 25:382-394.

Syversen, N., and M. Bechmann. 2004. Vegetative buffer zones as pesticide filters for simulated surface runoff. *Ecol. Eng.* 22(3): 175-184.

Tollner, E. W., B. J. Barfield, C. T. Haan, and T. Y. Kao. 1976. Suspended sediment filtration capacity of simulated vegetation. *Trans. ASAE* 19(4):678-682.

Tollner, E. W., B. J. Barfield, C. Vachirakornwatana, and C. T. Haan. 1977. Sediment deposition patterns in simulated grass filters. *Trans. ASAE*. 20(5):940-944.

Troeh, F.R., Hobbs J.A., and Donahue R.L. (2004). Water quality and pollution. in Soil and Water Conservation: for Productivity and Environmental Protection (4th edn)Prentice Hall: Upper Saddle River, New Jersey, USA. 523–550.

Ugurlu, A., and B. Salman. 1998. Phosphorus removal by fly ash. Environ. Int. 24: 911–918.

US Department of Agriculture, Natural Resources Conservation Service. 1998a. Montana field office technical guide. Section VI. Practice Standard 393A: Filter Strip. Bozeman: Natural Resources Conservation Service.

USDA-NRCS. 1989. Filter strip standard. Section IV, AL-393. Washington, D.C.:US Department of Agriculture-Natural Resources Conservation Science.

U.S. NRCS (Formerly Soil Conservation Service), National Engineering Handbook, Hydrology, Section 4 (1972) and USDA ARS 41-172 (1970).

USEPA (US Environmental Protection Agency). 2001. Funding Agricultural Best Management Practices with the Clean Water State Revolving Fund (EPA 832-F-01-006).

USEPA. 2001. National pollutant discharge elimination system permit regulation and effluent limitations guidelines and standards for concentrated animal feeding operations; proposed rule. Fed. Regist. 66:2960-3145.

USEPA. 2002. National water quality inventory: Report to congress 2002 Reporting cycle. Chapter 2, p. 14, and Chapter 3, p. 21. Office of Water, Washington, DC.

USGS, 2011. United States Geological Survey. Climate of North Dakota: temperature. Available at (http://www.npwrc.usgs.gov/resource/habitat/climate/temp.htm, accessed 02/09/2011).

Vanderholm, D. H., and E. C. Dickey. 1980. Design of vegetative filters for feedlot runoff treatment in humid areas. *Trans. ASAE* 23:681-684, 687.

Vanderholm, D. H., E. C. Dickey, J. A. Jacobs, R. W. Elmore, and S. L. Spahr. 1979. Livestock feedlot runoff control by vegetative filter strips. EPA-600/2-79-143. U.S. Environ. Protect. Agency, Ada, OK.

Vought, L. B.-M., J. Dahl, C. L. Pedersen, and J. O. Lacoursiere. 1994. Nutrient retention in riparian ecotones. *Ambio* 23(6):342-348.

Watts, D. B., and H. A. Torbert. 2009. Impact of gypsum applied to grass buffer strips on reducing soluble P in surface water runoff. *J. Environ. Qual.* 38:1511-1517.

Webber, D. F., S. K. Mickelson, S. I. Ahmed, J. R. Russell, W. J. Powers, R. C. Schultz, and J. L. Kovar. 2010. Livestock grazing and vegetative filter strip buffer effects on runoff sediment, nitrate, and phosphorus losses. *J. Soil and Water Cons.* 65(1): 34-41.

Webber, D.F., S.K. Mickelson, T.L. Richard, and H. K., Ahn. 2009. Effects of a livestock manure windrow composting site with a fly ash pad surface and vegetative filter strip buffers on sediment, nitrate, and phosphorous losses with runoff. *J. of Soil and Water Cons.* 64(2):163-171.

Westerman, P. and M. Overcash. 1980. Dairy open lot and lagoon-irrigated pasture runoff quantity and quality. *Trans. ASAE* 23:1157-1164.

White, M. J. and J. G. Arnold. 2009. Development of a simplistic vegetative filter strip model for sediment and nutrient retention at the field scale. *Hydrol. Process.* 23: 1602-1616.

Williams, J. R. 1980. SPNM, a model for predicting sediment, phosphorus, and nitrogen yields from agricultural basins. *Water Res. Bullet.* 16 (5): 843-848.

Williams, J. R. 1995. The EPIC model. In *Computer Models of Watershed Hydrology*, 909-1000. V. P. Singh, ed. Highlands Ranch, Colo.: Water Resources Publications.

Williams, J. R. 1975. Sediment-yield prediction with the Universal equation using runoff energy factor. In: Present and prospective technology for predicting sediment yields and sources. ARS-S-40. USDA-Agricultural Research Service, ppp. 244-252.

Woodbury, B. L., J. A. Nienaber, and R. A. Eigenberg, 2002. Operational evaluation of a passive beef cattle feedlot runoff control and treatment system. *Applied Eng. in Agric.* 18(5): 541-545.

Woodbury, B. L., J. A. Nienaber, and R. A. Eigenberg. 2005. Effectiveness of a passive feedlot runoff control system using a vegetative treatment area for nitrogen control. *Applied Eng. in Agri.* 21(4): 581-588.

Xu, D., J. Xu, J.Wu, and A. Muhammad. 2006. Studies on the phosphorus sorption capacity of substrates used in constructed wetland systems. *Chemosphere* 63:344–352.

Yayintas, O. T., S. Yilmaz, M. Turkoglu, and Y. Dilgin. 2007. Determination of heavy metal pollution with environmental physicochemical parameters in waste water of Kocabas Stream (Biga, Canakkale, Turkey) by ICP-AES. Environ. Monitoring Assessment. 127(1-3): 389-397

Young, R. A., C. A. Onstead, D. D. Bosch, and W. P. Anderson. 1987. AGNPS: Agricultural non point source pollution model. A watershed analysis tool. Conservation research report 35, 80p. Springfield, VA:USDA.

Young, R. A., T. Huntrods, and W. Anderson. 1980. Effectiveness of vegetative buffer strips in controlling pollution from feedlot runoff. *J. Environ. Qual.* 9: 483-487.

Zhang, X., X. liu, M. Zhang, and R. A. Dahlgren. 2010. A review of vegetated buffers and a meta-analysis of their mitigation efficiency in reducing nonpoint source pollution. *J. Environ. Qual.* 39:76-84.

APPENDIX

A.1. Subroutines

A.1.1. Upslope sediment and P transport component

```
      subroutine per(sand,silt,clay,qpeak,bigqp,sy,bigq)
      Implicit double precision (a-h, o-z)
      real ORsand, ORclay, ORsilt, ORsg, ORlg, ratio
      real beta,MM,Po,Kd,Psol
      real agtype(5),d(5)
      Data d(1),d(2),d(3),d(4),d(5)/2.0,10.0,200.0, 27.5,300.0/
      tss=0
      tssi=0
      MM=5
      Po=2800
!      Psol=2237
      kd=175
6500  format(f3.1,1x,f4.2,1x,f4.2)
      ORclay=0.26*clay
      agtype(1)=ORclay
      ORsand=sand*(1-clay)**5
      agtype(3)=ORsand
      if(clay.LT.0.25) then
       ORsg=1.8*clay
       agtype(4)=ORsg
```

```
      else if((clay.GE.0.25).AND.(clay.LE.0.50)) then
       ORsg=-0.6*(clay-0.25)+0.45
       agtype(4)=ORsg
      else
       ORsg=0.6*clay
       agtype(4)=ORsg
      endif
      ORsilt=silt-ORsg
      agtype(2)=ORsilt
      ORlg=1-(ORsand+ORclay+ORsilt+ORsg)
      agtype(5)=ORlg
      DO 15 i=1,5
          ss=33*d(i)**(-0.1785)+10.7*MM
          tss=tss+ss*agtype(i)
15        continue
          ftss=tss
       write(20,6511)ftss
       write(20,6510)ORsand,ORsilt,ORclay,ORsg,ORlg
6510  format('Calculated sediment fraction', 5f6.3)
      ratio=(qpeak/bigqp)**0.56
      a=log(ratio)
      beta=-a/4.47
      Do 16 i=1,5
```

```
        b=exp(-beta*sqrt(d(i)))
        omega=(agtype(i)*b)/ratio
        ssi=33*d(i)**(-0.1785)+10.7*MM
        tssi=tssi+ssi*omega
16      continue
        ftssi=tssi
! Phosphorus enrichment ratio
        er=ftssi/ftss
        write(20,6514)er
6511    format('Sp. srfc at the detachment,ftss=',f8.4)
6512    format(5f8.4)
6514    format('Phosphorus enrichment ratio, PER=',f8.4)
!-----------------------------------------------------------
! calculating P loading
!-----------------------------------------------------------
     Psed=0.01*Sy*Po*er
     Write(20,6515)Psed
!------------------------------------------------------------
! calculating dissolved P concentration from feedlot
!-----------------------------------------------------------
     Pdis=(0.01*Po*bigq)/kd
     write(20,6516)pdis
```

```
6515   format('Sediment phase P yield from
      1 the feedlot is=',f10.2,1x,'mg/L')
6516   format('Dissolved phase P yield from the feedlot
      1 is=',f10.2,1x,'mg/L')
6517     format('sand fraction is=',f6.6)
6518     format('sand, silt, clay',3f3.4)
     end
```

A.1.2. VFS sediment and phosphorus trapping efficiency component

```
      subroutine vfsp (VIN,VF,TOTRAIN,SMIN,SMOUT,dpsoil)
      Implicit double precision (a-h,o-z)
      d=dpsoil*10
      write(21,4640)d
       inflo=VIN
       ifil=VF
       rain=TOTRAIN
!      write(21,4650)inflo,ifil,rain
!-----------------------------------------------
!Calculating dissolved P removal efficiency
!-----------------------------------------------
      dpe=VF/(VIN+TOTRAIN)
      f=dpe*100
      write(21,4630)f
!-----------------------------------------------
!Calculating sediment bound P removal efficiency
!----------------------------------------------
      vfser=0.05*d+0.85
      c=vfser
      write(21,4600)c
      sre=1-SMOUT/SMIN
      a=sre*100
```

```
      write(21,4610)a
      pre=1-vfser*(1-sre)
      b=pre*100
      write(21,4620)b
      write(6,*)f,dpe,b
4600  Format('P enrichment ratio in filter=',f8.3)
4610  Format('VFS sediment removal efficiency=',f5.2,'%')
4620  Format('VFS sediment bound phosphorus
     1      removal efficiency=',f5.2,'%')
4630  Format('VFS dissolved P removal
     1     efficiency=',f8.3,1x,'%')
4640   Format('Particle dia Dp=',f8.6,1x,'mm')
!4650    Format('VIN,VF,TOTRAIN are',3f10.3,'m3')

        END
```