Use of vegetative filter strip for controlling nitrate and bacteria pollution from livestock confinement areas

by Juan Jose Fajardo

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Soils
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Abstract:
Point and non-point source pollution of surface and ground water is a major social and environmental concern in the world. Frequently cited and documented sources of nitrate contamination of surface and ground water are livestock waste and storage facilities. Several studies have demonstrated that vegetative filter strips (VFS) can be effective tools used in controlling pollution of waters from cattle feedlots. Vegetative filter strips are highly effective in reducing nutrients, sediment, and suspended solids in surface runoff from feedlots; however, results are variable in controlling bacterial concentration. The present study assessed the role of the cool season grass specie tall fescue (Festuca arundinacea Schreb.) as VFS in reducing contaminants generated by the storage of animal waste from livestock confinement areas under the relatively short growing season and short duration and high intensity rainfall characteristic of southwestern Montana. Specifically, the study evaluated the extent to which livestock manure stockpiles potentially contribute to nitrate-nitrogen (NO3-N) and coliform bacteria contamination of surface water resources. The experiment was conducted during the summers of 1997 and 1998 on Amsterdam silt loam (fine-silty, mixed, superactive Typic Haploboroll) soil. Tall fescue cv. Fawn and bare soil (fallow) strips measuring 3-m wide north-south and 30-m long west-east were established with a slope of 4% west-east, approximately. The treatments consisted of applications of manure in the upland position for the vegetated strip (TFM) and fallow strips (FM). Controls were established without manure in upland position (TFC and FC). Manure was applied annually (approximately 2-metric ton fresh weight per strip treated). Runoff was achieved by applying water to the manure stockpile or the bare border at the head of the treatments (with and without manure stockpile) and then forcing the applied water to pass through the VFS and fallow strips. Runoff water samples were collected in July and August of 1997 and 1998, at intervals of 0, 20, 40, and 60 minutes and analyzed for the presence of NO3-N and coliform organisms (total coliforms and fecal coliforms). Soil samples also were taken (to a depth of two meters) in April of each year of study at seven positions along the strips, before the new manure was applied. Concentration of NO3-N in surface runoff from manure stockpile was reduced significantly by the presence of VFS during the 1997 and 1998 sampling. Although coliform populations in runoff were reduced significantly by VFS in two runoff events, the coliform counts in runoff, even from VFS treatments not receiving manure, remained substantially elevated. Movement of NO3-N down slope within the soil of the TFM and FM treatments does not appear to be significant beyond the direct influence of the manure stockpile for either year of sampling.

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This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

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ABSTRACT

Point and non-point source pollution of surface and ground water is a major social and environmental concern in the world. Frequently cited and documented sources of nitrate contamination of surface and ground water are livestock waste and storage facilities. Several studies have demonstrated that vegetative filter strips (VFS) can be effective tools used in controlling pollution of waters from cattle feedlots. Vegetative filter strips are highly effective in reducing nutrients, sediment, and suspended solids in surface runoff from feedlots; however, results are variable in controlling bacterial concentration. The present study assessed the role of the cool season grass specie tall fescue (*Festuca arundinacea* Schreb.) as VFS in reducing contaminants generated by the storage of animal waste from livestock confinement areas under the relatively short growing season and short duration and high intensity rainfall characteristic of southwestern Montana. Specifically, the study evaluated the extent to which livestock manure stockpiles potentially contribute to nitrate-nitrogen (NO_3-N) and coliform bacteria contamination of surface water resources. The experiment was conducted during the summers of 1997 and 1998 on Amsterdam silt loam (fine-silty, mixed, superactive Typic Haploboroll) soil. Tall fescue cv. Fawn and bare soil (fallow) strips measuring 3-m wide north-south and 30-m long west-east were established with a slope of 4% west-east, approximately. The treatments consisted of applications of manure in the upland position for the vegetated strip (TFM) and fallow strips (FM). Controls were established without manure in upland position (TFC and FC). Manure was applied annually (approximately 2-metric ton fresh weight per strip treated). Runoff was achieved by applying water to the manure stockpile or the bare border at the head of the treatments (with and without manure stockpile) and then forcing the applied water to pass through the VFS and fallow strips. Runoff water samples were collected in July and August of 1997 and 1998, at intervals of 0, 20, 40, and 60 minutes and analyzed for the presence of NO_3-N and coliform organisms (total coliforms and fecal coliforms). Soil samples also were taken (to a depth of two meters) in April of each year of study at seven positions along the strips, before the new manure was applied. Concentration of NO_3-N in surface runoff from manure stockpile was reduced significantly by the presence of VFS during the 1997 and 1998 sampling. Although coliform populations in runoff were reduced significantly by VFS in two runoff events, the coliform counts in runoff, even from VFS treatments not receiving manure, remained substantially elevated. Movement of NO_3-N down slope within the soil of the TFM and FM treatments does not appear to be significant beyond the direct influence of the manure stockpile for either year of sampling.

CHAPTER 1

INTRODUCTION

Point and non-point source pollution of surface and ground water is a major social and environmental concern in the world. Point sources include municipal and industrial wastes, runoff and infiltration from animal feedlots, storm sewer outfalls from cities, and septic tanks, among others. Non-point sources include runoff from agriculture (farm-site fertilizers), runoff from pasture and range, runoff from construction sites (under 2 ha), atmospheric deposition over a water surface, and runoff from urban lands (Carey, 1991; Carpenter et al., 1998; National Academic of Sciences, 1972). Point sources of pollution are continuous discharges that can be relatively easily to monitor and regulate and can be controlled by treatment at the source; on the other hand, non-point sources are more intermittent and associated with seasonal agricultural or other land use activity or heavy precipitation, thus they are difficult to measure and regulate (Carpenter et al., 1998). Sediment, nutrients, and pathogens are the main types of pollutants of concern in surface waters; while pesticides, nitrates, and pathogens are the main types of pollutants in ground waters (Carpenter et al., 1998; Carey, 1991; Goss et al., 1998).

Nitrate contamination of ground water in the USA is closely monitored since ground water is the source of drinking water for about 105 million people. Moreover, about 97 percent of all rural drinking water, 55 percent of water for livestock, and more than 40 percent of all irrigation water originates from groundwater sources (Carey, 1991).

Between 1988 and 1990, the U.S. Environmental Protection Agency completed a national survey, which established that about 52 percent of the community water systems wells and 57 percent of the private wells in the USA contained detectable concentrations of nitrate-nitrogen. About 1.2 percent of the community system wells and about 2.4 percent of the private rural domestic wells nationwide had nitrate-nitrogen (NO_3-N) concentrations above the maximum limits of 10mg L^{-1} established for human consumption (USDA, 1991; USEPA, 1986).

Bacterial contamination is also a significant concern. One study found that 34 percent of the wells from 1,292 Ontario farmstead domestic wells had more than the maximum acceptable number of coliform bacteria, which are five and zero colony forming units per 100ml of water for total and fecal coliform, respectively (according to Ontario Drinking Water Objectives). It was also found that 14 percent of the samples contained NO_3-N concentrations above 10 mg L^{-1} and about 7 percent were contaminated with both bacteria and NO_3-N. Significant coliform bacteria contamination was associated with closeness of wells to feedlot or exercise areas (Goss et al., 1998).

A frequently cited and documented source of nitrate contamination of surface and ground water are livestock waste and storage facilities. Livestock waste is normally cleaned from feedlots when animals are marketed (between 90 to 180 days on feed) or once each year. About 50 percent of the nitrogen contained in the manure is lost due to runoff, ammonia volatilization, and denitrification during the storage period and before removal of the manure from the feedlot (Eghball and Power, 1994). An average 500-kg beef animal in a feedlot will produce about 25 kg of wet weight manure (feces and urine) per day (Thompson and O'Mary, 1983). Thus, a conventional feedlot will need to remove approximately 1 metric ton of dry manure (20 to 25 percent moisture content) per animal-year of confinement (Thompson and O'Mary, 1983). Because normal stocking rates in feedlots range from 10 to 50m^2 per animal, large quantities of manure are accumulated in confinement areas, creating potential pollution problems when storm runoff occurs (Khaleel et al., 1980).

A VFS is an area of permanent vegetation established to intercept sediment, nutrients, pesticides, and other contaminants from runoff before the runoff can enter a water body. Water quality can be maintained or improved by placing VFS between contaminant sources and surface waters, such as streams, rivers, and lakes. Vegetative filter strips work by reducing the velocity of runoff water, allowing the settling out of suspended soil particles, infiltration of runoff and soluble pollutants that runoff carries, adsorption of pollutants on soil and plant surfaces, and uptake of soluble pollutants by plants (USDA NRCS, 1998a).

Several studies have demonstrated that vegetative filter strips (VFS) are effective tools when used in controlling pollution of waters from cattle feedlots. Vegetative filter strips are highly effective in reducing nutrients, sediment, and suspended solids in surface runoff from feedlots; however, results are variable in controlling bacterial contamination (Dickey and Vanderholm, 1981; Dillaha et al., 1988; Young et al., 1980). Small feedlot operations can also benefit from the role VFS can play in reducing pollutants in surface runoff (Edwards et al., 1983). Vegetative filter strips (VFS) have been shown to also be an effective best management practice for the control of point and non-point source pollutants. Several studies have demonstrated that VFS are highly effective in reducing sediment and nutrients from cropland runoff (Dillaha et al., 1989; Fasching, 1999; Magette et al., 1989; Robinson et al., 1996) and from surface-applied swine manure (Chaubey et al., 1994; Hawkins et al., 1998).

Objectives

The present study assessed the role of the cool season grass tall fescue (*Festuca arundinacea* Schreb.) as a VFS in reducing contaminants generated by the storage of animal waste from livestock confinement areas under the relatively short growing season and short duration and high rainfall intensity characteristic of southwestern Montana. Specifically, the study evaluated the extent to which livestock manure stockpiles potentially contribute to NO_3-N and coliform bacteria contamination of surface water resources.

Literature Review

Nitrogen in the Environment

Nitrogen (N) is critical for all living organisms on earth. It is an integral component of all amino acids (the building blocks of all proteins) and in plants, it is a constituent of nucleic acids and chlorophyll. Nitrogen is also essential for carbohydrate use within plants. The availability of nitrogen often limits the productivity of plant communities (natural vegetation or agricultural crops). The large need of plants for nitrogen and the limited ability of soils to supply available nitrogen cause nitrogen to be the most limiting nutrient for plant production on a global basis (Brady and Weil, 1999; Foth and Ellis, 1997; Robertson, 1986).

The greatest pool of nitrogen is the lithosphere with 3.3 x 10^{11} million metric tons (about 98 percent of the world's total) component of coal, igneous rocks, sediments, and many other minerals. The atmosphere is the second largest nitrogen reservoir, with 3.86 x 10^9 million metric tons (about 2 percent of the world's total). On the other hand, soils contain a very small fraction (about 2.4 x 10^5 million metric tons) of the total nitrogen, about 90 percent, of which is unavailable complexes in organic matter. Most of the remainder is fixed ammonium in clays. At any single instant, about 1 percent or less of the total nitrogen in soils is available to plants and microorganisms as nitrate or exchangeable ammonium (Foth and Ellis, 1997).

From the overall earth nitrogen cycle two gases (di-nitrogen [N_2] and nitrous oxide [N_2O]) and four forms of nongaseous or combined nitrogen (ammonium [NH_4^+], nitrite [NO_2^-], nitrate [NO_3^-] and amino group) are important. Ammonium is released from either organic matter or urea, or it is synthesized by industrial processes (fixation of atmospheric N_2). Nitrite is formed from nitrate or ammonium by microorganisms in soil, water, sewage, and the alimentary tract. Nitrate is formed by the complete oxidation of ammonium by microorganisms in soil or water (National Academic of Sciences, 1972).

Human activity can influence major portions of the nitrogen cycle at many levels: from a local level passing through a regional level to finally a global level. Modifications to the nitrogen cycling at these levels can affect crop productivity and probably surface water eutrophication; groundwater quality, acid precipitation fluxes, and eutrophication of coastal wetland; and changes in global temperatures, the incidence of ultraviolet light reaching the earth's surface, and patterns of primary productivity in the world's oceans (Robertson, 1986).

Nitrogen-containing compounds in fresh water and marine systems are derived from several sources, both natural and anthropogenic. Biological N_2 fixation is the main source of nitrogen inputs in aquatic systems, contributing about 30 to 130 x 10^9 kg N per year, followed by NH_4^+/NH_3 deposition (19 to 50 x 10^9 kg N per year), and NO_3^-/NO_x deposition (11 to 33 x 10^9 kg N per year) (Robertson, 1986).

Nitrogen as an Environmental Pollutant and Human Health Problem

One of the problems associated with excess of nitrogen in aquatic ecosystems is eutrophication. Eutrophication describes the biological effects of an increase in concentration of plant nutrients, usually nitrogen and phosphorus, on aquatic ecosystems. Over-enrichment of these nutrients that are limited with respect to a specific nutrient increases the biomass and productivity of phytoplankton (primary producer) and zooplankton (which feed on phytoplankton). This increase in biomass translates into excessive accumulation of dead tissues and feces, which are decomposed by respiring bacteria. Bacterial respiration can cause further problems by depleting the dissolved oxygen (Harper, 1992). The increased growth of algae and weeds restricts use of water for fisheries, recreation, industry, agriculture, and drinking (Carpenter et al., 1998) and depletion of oxygen can cause hypoxia in bottom waters. Hypoxia is a condition that occurs when dissolved oxygen is less than 2 ml L^{-1}. Hypoxia can cause mass mortalities of finfish and shellfish (USDA, 1991). From an extensive literature review, Carpenter et al. (1998) concluded that eutrophication, due to excessive inputs of phosphorus and nitrogen is a widespread problem in rivers, lakes, estuaries, and coastal waters. Both nitrogen and phosphorus contribute to eutrophication in freshwater, although for many lakes excessive inputs of phosphorus are the primary cause. For most temperate estuaries and coastal ecosystems, nitrogen additions cause eutrophication (Carpenter, 1998). They also found that nutrient inputs to aquatic ecosystems are directly related to agriculture, which often uses excess inputs of manure and fertilizer for crop needs. Therefore, excess fertilization and manure production create surpluses of nitrogen, which eventually reach surface and ground waters.

Another problem associated with contamination of aquatic ecosystems with nitrogen is the direct toxic effects to humans. The USEPA established a standard of NO_3-N in drinking water of 10 mg L^{-1}, equivalent to 45 mg L^{-1} as NO_3^- (USEPA, 1986). The standard is set to prevent human health problems especially to infants under three months old, which may suffer from methemoglobinemia (blue-baby syndrome or infant cyanosis) due to nitrate-contaminated drinking water. Methemoglobinemia is due to the presence of methemoglobin in the blood. Methemoglobin is produced when nitrite oxidizes ferrous iron in hemoglobin to ferric iron, making hemoglobin incapable of transporting oxygen. The physiologic effect is oxygen deprivation or suffocation and even death (Walton, 1951; Craun et al., 1981). Early cases of methemoglobinemia were reported in 1945 in infants that had ingested water high in nitrates and specially if the infant was suffering from gastrointestinal problems (diarrhea). Infants under three months old are susceptible of suffering methemoglobinemia because the lack of acidity in their stomach allows the growth of nitrate-reducing bacteria, which reduce nitrate to nitrite before the former is completely absorbed (Walton, 1951). In older children, methemoglobinemia may not be a health problem. A study by Craun et al. (1981) could not document methemoglobinemia incidence in children between ages 1 to 8 years old with ingestions of up to 111 mg L^{-1} of NO_3-N. Cyanosis is easily recognized; methemoglobinemia is readily treated and the condition is rapidly reversible without any known or research-based cumulative effects (USDA, 1991). Clinical reports of methemoglobinemia in the Unites Sates have been virtually nonexistent in recent years and infant deaths now are very rare (USDA, 1991).

Sources of Nitrogen Pollution

One of the most important anthropogenic sources of contamination of surface waters is agriculture (USEPA, 1990; Carey, 1991; Moore, 1991). Nutrients, especially ammonia and nitrate, together with sediment are the main pollutants in surface waters (lakes, reservoirs, rivers and streams) (USEPA, 1990; Carey, 1991). Nitrate leaching, among other pollutants, contaminates groundwater resources (USEPA, 1990). Agricultural nitrogen may enter surface waters because of direct surface runoff and soil erosion activity or through groundwater discharges from unconfined aquifers into streams, especially from shallow alluvial sand or gravel, or glacial residue soils and aquifers (USDA, 1991). Environmental problems caused by nitrogen leaching are mainly associated with movement of nitrate through drainage waters to the ground water. Nitrate in ground water may reach domestic wells, and may eventually flow underground to emerge in surface water. Agricultural activity, such as manure spreading, crop-fallow rotations and irrigation, are the most important contributors of nitrate pollution to groundwater (USEPA, 1990; Brady and Weil, 1999).

One agricultural practice that contributes to nitrate contamination to groundwater is the crop/fallow cereal grain rotation. A study of 3,400 private wells in Montana showed that 6 percent of the all tested wells contained concentrations of nitrate-nitrogen (NO_3-N) that exceeded 10 mg L^{-1}. It was concluded that summer fallow practices increase the accumulation of mineralized nitrogen, which, in turn reaches shallow groundwater (Bauder et al., 1993).

Feedlots and livestock waste disposal areas may also contribute to nitrogen pollution of surface and groundwater. From a compilation of several studies of feedlot runoff in the Great Plains region, it was estimated that the average total nitrogen concentration in runoff water from feedlots ranged from a low of 50 mg L^{-1} to a high of 2100 mg L^{-1} (Khaleel et al., 1980). Another study showed that soil nitrate-nitrogen content from feedlots abandoned for several years averaged 7,200 kg ha^{-1} in a 9.1m soil profile while adjacent cropland had just 570 kg ha^{-1} and NO_3-N concentration in ground water samples from three of four study sites ranged from 0.6 to 77.2 mg L^{-1} (Mielke and Ellis, 1976).

Manure application to cropland can also contribute to nitrate contamination of groundwater. In a study conducted in Lethbridge, Alberta in a Chernozemic clay loam soil between 1973 and 1992, under irrigated and non-irrigated conditions a cereal crop was treated with annual applications of feedlot manure. The manure (1 and 2 year old) was applied at different rates (0, 60, 120, and 180 Mg ha^{-1} yr^{-1}, wet weight) representing zero, one, two, and three times the maximum rate recommended. The results showed that under non-irrigated conditions, all the nitrogen applied in manure was accounted for by crop uptake, soil organic nitrogen, and soil NO_3-N, with minimum losses due to leaching. On the other hand, under irrigation annual leaching losses were appreciable, averaging 0.09, 0.23, and 0.34 Mg nitrogen $ha^{-1}yr^{-1}$ at manure rates of 60, 120, and 180 Mg ha^{-1} yr^{-1}, respectively. Cumulative leaching losses amounted to 1.4, 3.4, and 5.2 Mg nitrogen ha^{-1}, respectively. This difference was attributable to the higher doses of manure applications under irrigation rather than to the effects of increased soil moisture (Chang and Janzen, 1996). Correspondingly, intensive grazing systems, with animal grazing at high stocking densities, have been documented as posing a potential threat for ground water due to nitrate leaching from animal urine and feces (Stout et al., 1997).

Cattle Manure and Bacterial Pollution

Animal wastes can contribute to water contamination. Feedlot operations as well as dairy barnyards contribute to coliform bacteria in runoff (Miner et al., 1966; Young et al., 1980). Activities like spreading and incorporation of manure in cropland and feces deposition on pastures through animal grazing contribute to pollution of surface waters when pathogens contained in the manure are carried by runoff (Faust, 1982; Patni et al., 1985). These studies found also that significant counts of fecal coliforms (indicators of water contamination) may be detected in runoff from areas that did not receive manure applications or not grazed. This contamination is attributed to fecal deposition by wild animals.

The quantity of bacteria deposited on the land is a function of the type and number of livestock as well as whether or not the waste is stored prior to spreading (Walker et al., 1990). Animal waste application method, attraction of bacteria to soil particles, and rainfall duration and intensity may determine whether bacteria are transported with runoff and eroded soil (Khaleel et al., 1980; Walker et al., 1990). Fecal bacteria out of the digestive tract are affected by changes in moisture, nutrient availability, temperature, pH, ultra-violet radiation, exposure to predators, and exposure to toxic compounds (Walker et al., 1990).

Fecal coliforms can survive in the environment, outside of the animal digestive tract, for a long period. For example, Brown et al. (1980) found that survival of C_F on grass treated with sludge directly depended on climatic factors like light and moisture. They found that 2 to 3 weeks was needed to significantly reduce the population of C_F, suggesting also that moisture would increase the numbers of C_F. Another study suggests that coiliforms can survive in high numbers up to 30 days inside the feces (Thelin and Gifford, 1983). In this study, feces less than 5-days old had C_F counts on the order of millions per 100 ml of water while feces of 30-day old contained about 40,000 C_F per 100 ml of water. Greater periods of survival are reported by Buckhouse and Gifford (1976), with viable C_F in feces at least one grazing season after deposition and even more than one year later in runoff water after cattle were removed from pastures (Jawson et al., 1982).

Coliform Group as Indicator Bacteria

The impact of livestock manure on water quality can be determined by examining water samples for the presence of indicator bacteria. Indicator bacteria are used instead of the actual pathogens because the identified indicator bacteria are usually present in greater numbers than pathogens, and are easier to isolate and much safer to work with. If the indicator organisms are present at a defined threshold, there is good probability that pathogenic organisms are also present (Thelin and Gifford, 1983). Total coliforms (C_T), fecal coliforms (C_F), and fecal streptococci are three groups of indicator bacteria (Thelin and Gifford, 1983). These are also called the coliform group (Gleeson and Gray, 1997).

The coliform group is comprised of several genera of bacteria belonging to the family Enterobacteriacea, which includes *Escherichia*, *Citrobacter*, *Enterobacter* and *Klebsiella*, and non-fecal lactose fermenting bacteria and other species rarely found in feces but capable of multiplication in water (Gleeson and Gray, 1997). Several metabolic and physical characteristics define the coliform group. The general definition for coliforms by the World Health Organization is: gram-negative, rod shaped bacteria capable of growth in the presence of bile salts or other surface active agents with similar growth inhibiting properties, and able to ferment lactose at 35- 37°C with the production of acid, gas, and aldehyde within 27- 48 hours (Gleeson and Gray, 1997). Coliforms are oxidase negative, non-spore forming and show β-galatosidase activity. The production of β-galactosidase is the fundamental characteristic present in Enterobacteriaceae. The coliform group also includes thermo-tolerant fecal coliform (able to ferment lactose at 44°C); with *E. coli,* the true fecal coliform since other thermo-tolerant coliforms

(*Citrobacter*, *Enterobacter* and *Klebsiella*) can be extracted from non-fecally contaminated waters. Growth of these organisms in a water sample may result in high counts that may be interpreted as fecal coliforms (Gleeson and Gray, 1997).

Two standard methods are utilized to identify coliforms from water samples, the multiple tube and the membrane filtration techniques. In the multiple tube coliform test, replicate tubes of lactose broth lauryl tryptose broth are inoculated with a dilute sample of the water. The coliform densities are then calculated from probability formulas that predict the most probable number (MPN) of coliforms. In the membrane filtration technique, a known volume of water is filtered through a membrane filter (MF) of specific pore size, which trap the bacteria. The MF then is placed in a suitable growth media that allows bacteria growth and subsequent counts (Gleeson and Gray, 1997; USEPA, 1975). These two methods are used worldwide and are based on the bacteria production of gas and acid from a lactose based growth media (Gleeson and Gray, 1997).

In relation to pathogens, the USEPA has established a standard that fecal coliforms densities should not exceed 200 counts per 100 ml for bathing waters and should not exceed 14 counts per 100 ml for shellfish harvesting water (USEPA, 1986). Exceeding these standards may lead to a significant increase the risk of fecal-related diseases.

Vegetative Filter Strip Functions

Vegetative filter strip enhance the opportunity for runoff and pollutants to infiltrate into the soil profile, allow deposition of total suspended solids, enhance filtration of suspended sediment by vegetation, provide adsorption on soil and plant surfaces, and enhance adsorption of soluble pollutants by plants. Infiltration is a significant mechanism that affects VFS performance since many pollutants associated with surface runoff enters the soil profile in the VFS as infiltration takes place. Once the pollutants are in the soil profile, they can be trapped by a series of physical, chemical, and biological processes (Dillaha et al., 1988). Vegetative filter strip also reduce pollutants in runoff through deposition. Shallow overland flow velocity is reduced by the grass cover in the VFS immediately upslope and in the filter, thereby reducing surface flow and ultimately sediment transport. Reduction in the transport capacity allows the settling and trapping of suspended solids and sediment bound pollutants (Dillaha et al., 1988). Two other mechanisms associated with VFS effectiveness in reducing pollutants are filtration and absorption. Filtration is probably most significant for larger soil particles, aggregates, and manure particles, while absorption is a significant factor with respect to soluble pollutant removal (Dillaha et al., 1988).

Nutrient and Sediment
Removal by VFS from Croplands

Nitrogen (N), phosphorus (P), and sediment are the primary pollutants associated with surface runoff from cropland areas. Several studies have demonstrated the ability of VFS to remove sediments and nutrients both from manure and inorganic fertilizers (Daniels and Gilliam, 1996; Edwards et al., 1996; Lim et al., 1998; Srivastava et al., 1996; Magette et al., 1989). In these studies, either manure or inorganic fertilizer was applied to bare soil or pasture with variable reductions in sediment in runoff from 60 to 90 percent and up to 80 percent reductions in total nitrogen and phosphorus. For example, an experiment conducted in northeast Iowa, on a silt loam soil, with 7 percent slope, demonstrated that VFS (using bromegrass, *Bromus inermis* L.) of 9.1-m width were capable of remove 85 percent of the sediment in runoff from a 18-m continuous fallow strip. The VFS reduced runoff volumes and promoted infiltration (Robinson et al 1996). In general, VFS are more efficient in trapping sediment and sediment bound nutrients than soluble nutrients. In a study effectuated in Virginia, (silt loam soil) orchardgrass (*Dactylis glomerata* L.) was used as VFS. Plots of bare soil in upland position received 222 kg ha^{-1} of liquid N and 112 kg ha^{-1} of P. Vegetative filter strip of 9.1 and 4.6-m width removed 84 and 70 percent of the incoming suspended solids, 79 and 61 percent of P, and 73 and 54 percent of N, respectively. The 4.6 and 9.1-m VFS removed 53 to 86 percent and 70 to 98 percent, respectively, of the incoming sediment. Total P and total N were removed nearly as effectively as sediment, because 97 percent of the total P and 78 percent of the total N entering the VFS was sediment-bound. The effectiveness of VFS in reducing soluble nitrogen (NO_3-N) and soluble phosphorus in runoff was moderate, with NO_3-N concentration reductions of 27 and 57 percent for the 4.6 and 9.1m VFS, respectively. The overall mean concentrations of inorganic soluble phosphorus in VFS runoff ranged from 0.11 to 0.17 and 0.08 to 0.20 mg-P L^{-1} for the 4.6 and 9.1m VFS, respectively. According to the authors, these concentrations where still high enough to cause eutrophication (Dillaha et al., 1989).

Effectiveness of VFS in Controlling Pollutants from Livestock Confinement Areas

Nutrients, sediments, and pathogens are associated with livestock confinement areas. Vegetative filter strips can be used to reduce the potential of livestock confinement areas pollutants, to contaminate water supplies. Vegetative filter strips have been used effectively to reduce pollutants from dairy liquid waste discharges (Paterson et al., 1980; Schwer and Clausen, 1989; Yang et al., 1980). For example, in the Schwer and Clausen (1989) study, a 26 by 10-m VFS was able to reduce the concentration of total suspended solids (TSS) by 92 percent, total phosphorus (Pt) by 86 percent and total Kjeldahl nitrogen (TKN) by 83 percent in surface runoff. While total suspended solids were reduced by 97 percent, total phosphorus by 92 percent and TKN by 93 percent in subsurface output, relative to the input liquid wastes. In these studies, the hydraulic loading rate of the liquid wastes was the main factor that affected the effectiveness of VFS to retain nutrients. Therefore, poor performance of VFS is attained if the hydraulic loading rate surpasses the infiltration capacity of the VFS (Schellinger and Clausen, 1992). Maximum efficiency of VFS occurred in the growing period. In addition, soils saturated during summer by heavy rain and wet or frozen soils reduced the infiltration of VFS and increased the pollution potential from surface runoff. Uptake of phosphorus and nitrogen by the vegetation was not a primary removal mechanism, as suggested by the Schwer and Clausen (1989) study.

Studies of VFS effectiveness in controlling pollution from livestock feedlot have demonstrated that VFS are highly effective in reducing concentration of nitrogen, phosphorus, and sediment in the incoming runoff. From a study in Blacksburg, VA, two VFS lengths, 4.5 and 9.1-m were tested in reducing nutrients and sediment, from simulated open feedlots areas of 5.5 by 18.3-m. In these feedlot areas, fresh manure was applied at rates of 7,500 and 15,000 kg per ha (moist weight), equivalent to accumulations of in a feedlot of 7 and 14 days, respectively. Runoff was achieved by using rain simulators applying 50 mm of water per hour. The 9.1 and 4.6-m VFS removed 91 and 81 percent of the incoming sediment, respectively. The 4.6 and 9.1-m long filters reduced total nitrogen by an average of 67 to 74 percent, respectively, but soluble nitrogen concentrations were not effectively reduced. Nitrate-nitrogen reduction in the best situation was 17 percent. Only 69 and 58 percent of the applied phosphorus was removed by the 9.1 and 4.6-m VFS, respectively (Dillaha et al., 1988). Similar results were obtained in west central Minnesota where a 40-m VFS strips of corn, orchardgrass, sorghum, or oat were planted across slope (4 percent) to reduce pollutants in runoff from a feedlot containing 310 head of cattle. A simulated rainfall of approximately 6.4 cm per hour for duration of 70 min was used to force runoff. The study concluded that runoff and total solids were reduced by 67 and 79 percent, respectively; total N was reduced in 84 percent and soluble P was reduced 83 percent (Young et al., 1980).

Bacterial contamination due to fecal coliforms is another pollutant associated with runoff from livestock confinement areas, which may be controlled by VFS, although results are variable. Studies show that reductions of coliform bacteria up to 70 percent in runoff from a feedlot containing 310 head can be attained with a 36m VFS (Young et al., 1980); however, Dickey et al. (1981) did not find significant reduction in coliform counts when runoff from four different types of feedlot passed through a VFS.

CHAPTER 2

METHODS AND MATERIALS

Site Description

The experimental field plots were located at the Montana State University Arthur Post Research Farm, about 8-km west of Bozeman, Gallatin County Montana. The plots were situated about 1000-m north and 20-m west of SE corner of section 7, T.2 S., R. 5E.

The experiment was established on an Amsterdam soil series (fine-silty, mixed, superactive Typic Haploboroll) (USDA-NRCS Soil Survey Division, 1998b). These soils are characterized as being very deep, well drained and of moderately slow permeability. They were formed in alluvium, lacustrine or loess deposited material mixed with volcanic ash. Normally, Amsterdam soils are located on alluvial fans, stream terraces and lake terraces. They are moderately extensive in intermountain valleys of southwestern Montana. Elevation range from approximately 1,200 to 1,800-m and slopes from 0 to 25 percent. The climate is cool, with long cold winters, moist spring, and warm summers. The mean annual precipitation (most as snow in winter, as snow or rain early in spring, or as rain in early summer) and mean annual soil temperature range from 380 to 480 mm and 4 to 7 °C, respectively.

Treatment Design

The present experiment is based on a study initiated by Oksendal (1997). The experiment consisted of six treatments with four replications in a randomized, complete block design. Four grass strips consisting of four different grass species and two bare soil strips (fallow strips) were established in May of 1994. The treatments were established in an area of approximately 2200 m^2, subdivided in strips, each strip 3-m wide north south and 30-m long west east, totaling 24 strips. The strips have a slope of 4.3 to 5.1 percent from west to east and a cross slope ranging from 1.8 to 2.2 percent.

The four grass species evaluated were orchardgrass (*Datylis glomerata* L.) cultivar Latar, tall fescue (*Festuca arundinacea* Schreb.) cultivar Fawn, meadow bromegrass (*Bromus biebersteinii* L.) cultivar Regar, and tall wheatgrass (*Agropyron elongatum* L.) cultivar Alkar.

As part of the original experiment in 1995, manure (solid state) from nearby dairy cattle operations was applied to the upper end of all grass treatments and one of the two fallow treatments in order to simulate a livestock waste disposal area. Manure was applied to an area of 9m^2 for each strip with a composite total of 30 metric tons (fresh weight) applied across all the treated strips. The treatments were designated: 1) orchardgrass with manure application in upland position (OGM); 2) tall fescue with manure application in upland position (TFM); 3) meadow bromegrass with manure application in upland position (MBM); 4) tall wheatgrass with manure application in upland position (TWM); 5) bare soil or fallow with manure application in upland position (FM); and 6) bare soil or fallow with no manure application as control (FC).

Modifications to the Original Study

In order to accomplish the objectives of the present study, in 1996 a tall fescue treatment with four replications was established within border areas adjacent to the original experimental plots. In order to function as a grass treatment control (TFC), this treatment did not receive manure. The grass control (TFC) strip dimensions and orientation followed the original design. In order to avoid cross contamination of runoff from one treatment strip to another, the treatment strips were isolated by installing wooden borders 15 cm tall by 4 cm wide along the down slope axis (30 meters) between each two adjacent strips. In addition, on the edges of the positions where manure was applied, 50 cm tall by 1 cm wide wooden borders were installed. Runoff catchment and sampling collectors were installed at the lower end of the strips that were subjects of runoff water and water sampling. They were installed at the down slope terminus of each strip and about 30-m down slope from the manure application area by digging a hole in the center position of the strip and placing a 25-liter plastic bucket in each hole. Diversion dikes, as plastic-lined earthen berms, extending from the outside edge of each strip at the 29-meter position to the plastic bucket, were constructed. For more details see Figure 1, which shows the general design of the experimental plots and the treatments introduced in 1996. Figure 2 shows the specifications of two individual strips.

Manure Application

Following the manure application in 1995, subsequent annual applications were made in April of 1996, 1997, and 1998 in order to simulate a livestock waste disposal area or feedlot. In each year, the manure from the previous year was removed and replaced with fresh manure (approximately 40 metric ton fresh weight; 18 percent dry matter), following the procedures described by Oksendahl (1997). The manure was obtained from nearby livestock operations and was distributed among the five manure

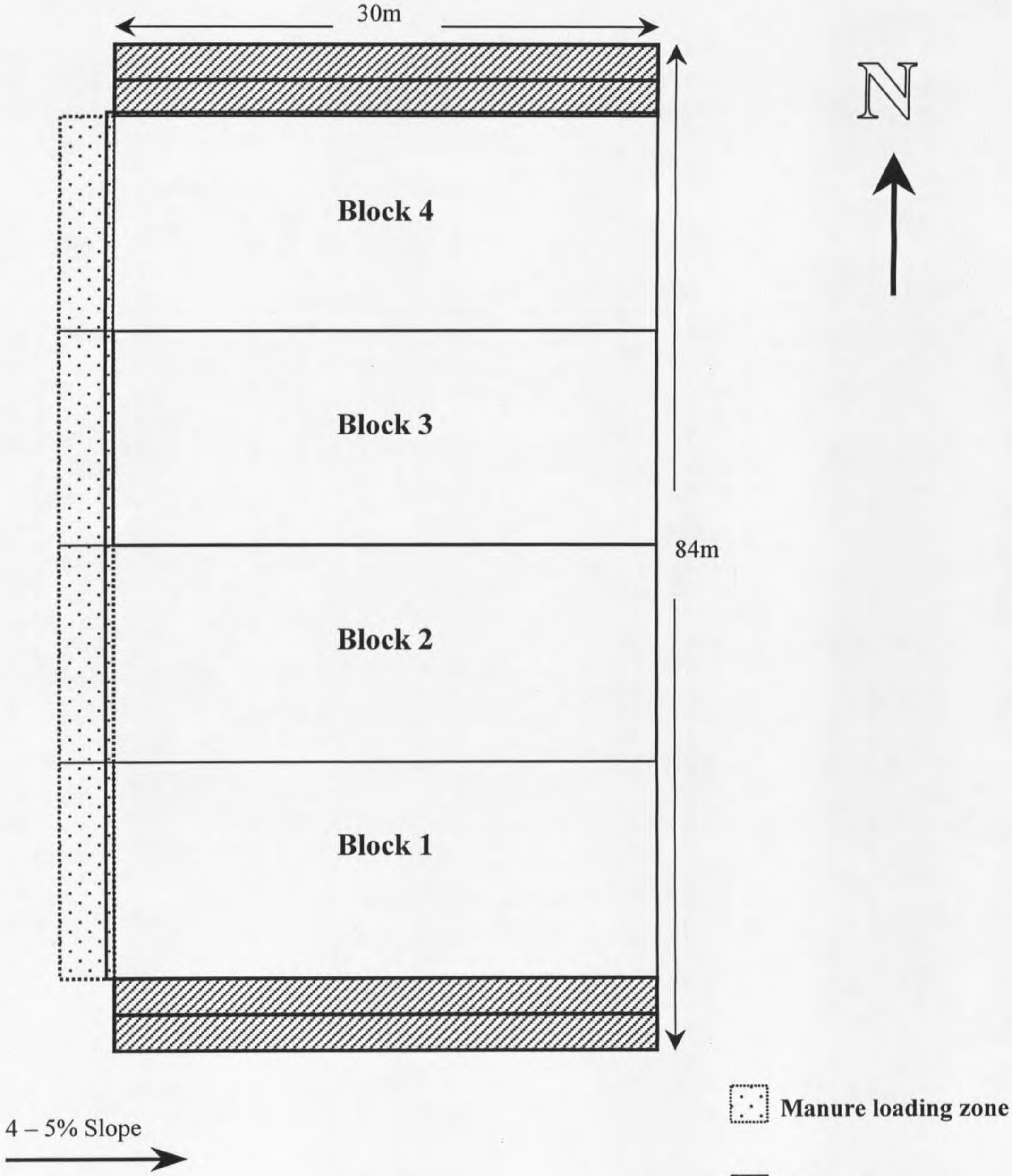

Figure 1. Experimental field plot design.

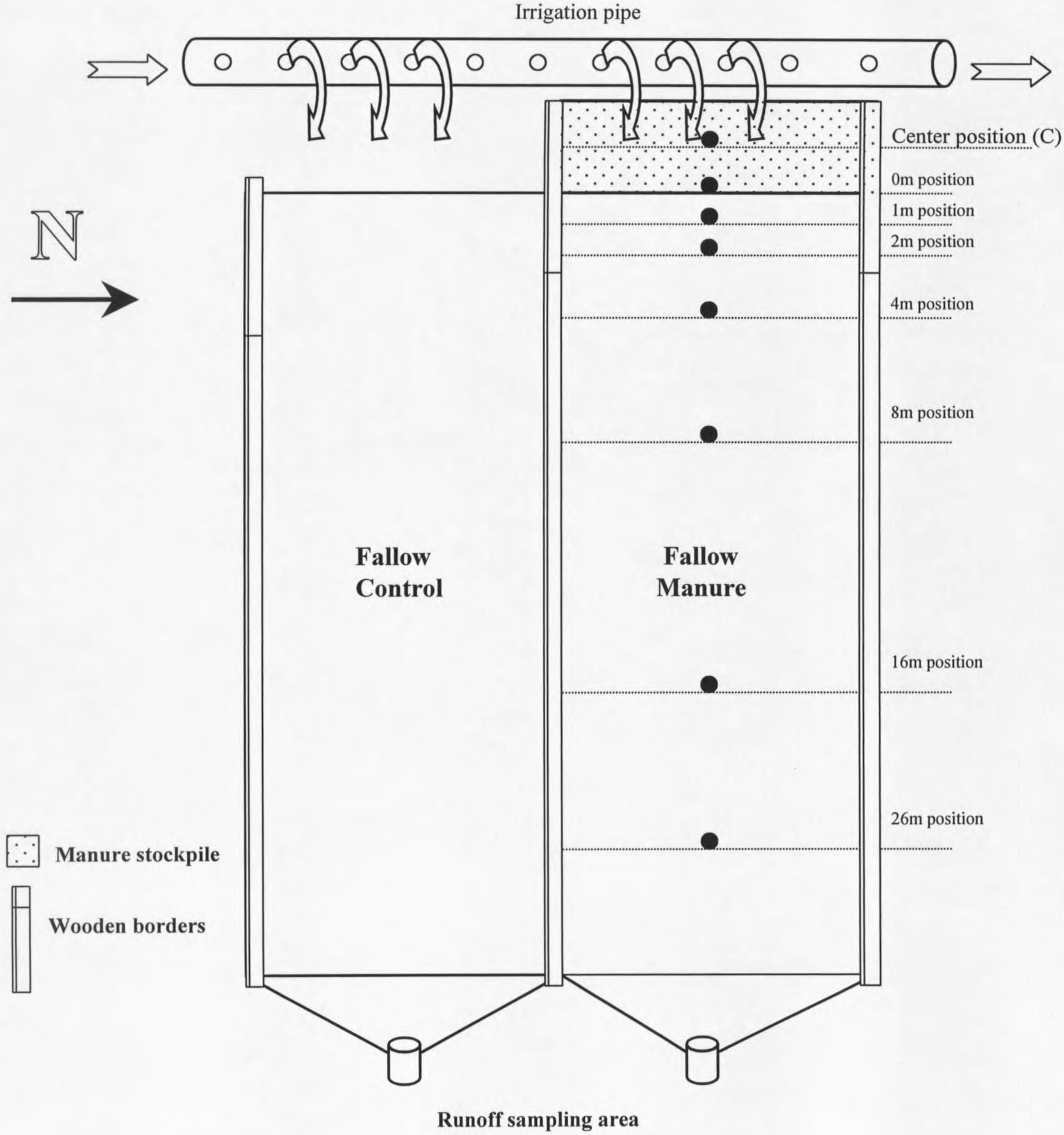

Figure 2. Schematic design of VFS. Sampling positions and modifications introduced are displayed.

treatments (not including FC and TFC treatments), corresponding to approximately 2-metiric ton fresh weight per treated strip.

Sampling and Analysis

Runoff Water Sampling and Analysis

The VFS established in the Oksendal (1997) study and used in the present study were designed in accordance with the USDA-SCS standards. The strip design was based in a peak discharge from a 24-hour, 25-year storm, which is equivalent to 56 mm of precipitation for this area.

Tall fescue manure (TFM), tall fescue control (TFC), fallow manure (FM), and fallow control (FC) treatments were chosen to evaluate the effectiveness of VFS in reducing soluble nitrogen (NO_3-N) and bacterial contamination in runoff water entering the upslope position of the VFS from manure stockpiles. Two runoff events were created each year of the study following grass harvest. The first runoff event for 1997 was imposed between July 8 and 9 and the second on August 22. For 1998, the first runoff event was imposed between July 7 and 10 and the second between August 27 and September 10. Runoff was achieved by applying water to the manure stockpile or the bare border at the head of the treatments (with and without manure stockpile) and then forcing the applied water to pass through the VFS. Irrigation water was applied to FC and FM treatments at a rate and volume only sufficient to produce runoff. The water applied was 1,770 L in a period of 70 min for each strip (90 m^2). The volume of water applied was equivalent to 20mm of precipitation over the entire strip. This precipitation is equivalent to a 2-year 24-hour storm for Bozeman area (Miller et al., 1973). The volume of water applied to TFM and TFC treatment was increased to assure one hour of runoff. The application equaled a total volume of 29,880 L for a period of 180 min, which was equivalent to 330 mm of precipitation applied to each strip. The occurrence of this amount of precipitation is extremely improbable, in much as a 100-year 24-hour precipitation event for the Bozeman area is only 71 mm (Miller et al., 1973). Furthermore, the probable maximum precipitation for the Bozeman area that may occur hypothetically in a thousand years is about 300 mm in a 6-hour precipitation event (USDA-NRCS, 1965).

A sequence of runoff samples was collected from each strip. The first sample corresponded with the time when runoff water begun to leave the filter strip (0 min); three subsequent samples were collected at intervals of twenty minutes: 20 min, 40 min, and 60 min. Two replicate sub-samples, each approximately 200 ml of runoff water, were collected at each sampling. One sub-sample from each sampling time was sent to Montana State University Soil Analytical laboratory for determination of NO_3-N for the sampling completed in 1997 and 1998. The other sub-sample was sent to Environmental Laboratory in Helena, MT in 1997, for determination of total coliform (C_T) and in 1998 to Montana Microbiological Services laboratory in Bozeman, MT for determination of fecal coliform (C_F).

Soil Sampling and Analysis

Soil samples were collected in April of 1997 and 1998 from the TFM, TFC, FM, and FC treatments following removal of the manure stockpile. Soil samples were collected at seven positions along the length of the VFS. The sampling locations corresponded with a position centered directly under the manure pile (or its equivalent in the control treatments), the edge of the manure stockpile, and 1, 2, 4, 8, and 26 meters from the edge of the manure stockpile, respectively. At each location soil samples were obtained in incremental depths of 0-10, 10-20, 20-40, 40-80, 80-160, and 160 to 200cm, respectively, using a truck-mounted hydraulic sampling probe. Each sample was placed in a pre-labeled soil sample bag and transported to the laboratory for further analyses. Soil samples were air dried at 70°C for three days, ground and 2mm sieved and stored until analyses were completed. Nitrate nitrogen was determined using a colorimetric method developed by Yang et al. (1998).

Statistical Analysis

Statistical analyses were performed using the Statistical Analysis System (SAS) version 7.0 (SAS Institute, 1998). Nitrate nitrogen and bacterial counts in runoff water were analyzed using a split-plot design considering time as a subplot. A three factor factorial arrangement was used for the soil samples analysis. Analysis of variance (ANOVA) tables were developed to determine the significance of treatment effects and interactions.

CHAPTER 3

RESULTS AND DISCUSSION

Runoff Water Analysis

Nitrate Nitrogen Concentration of VFS Runoff Water

Laboratory determination of NO_3-N in runoff was made utilizing the automated cadmium reduction colorimetric-based method, (Clesceri et al., 1989), which cannot detect concentrations of NO_3-N below 0.1 mg L^{-1}. Therefore, concentrations of NO_3-N that were less than 0.1 mg L^{-1} were assigned a value of zero (0 mg L^{-1}) in order to complete the appropriate statistical analyses. It is important to recall that in order to create runoff within fallow treatments (FC and FM), it was necessary to apply water at a rate of 25 L min^{-1} to each strip. At this rate of application, the runoff reached the end of the strip in 10 min. In contrast, the rate of water applied to the VFS treatments (TFC and TFM) was 170 L min^{-1}. At this rate of application, the runoff reached the end of the strip in 120 min. Under these conditions, the total water applied to obtain 60 min runoff was equivalent to 1.77 m^3 and 30 m^3 for each strip of fallow and VFS treatments, respectively. Because of this disparity in the water application rate, the results were analyzed independently for the runoff from the VFS treatments and fallow treatments. Comparisons were then made between them to estimate the impact that VFS had in mitigating nitrate pollution from fallow and manure stockpile.

The results of the analysis of variance (ANOVA) of nitrate nitrogen (NO_3-N) concentrations in runoff water for 1997 and 1998 events are presented in Table 1.

Table 1. Summary of ANOVA of nitrate nitrogen (NO_3-N) concentration of runoff water from VFS, 1997 and 1998.

		July, 1997		August, 1997	
Source of variation	df	F value	Pr > F	F value	Pr > F
Block	3	0.46/NS	0.7109	2.12/NS	0.1142
Treatments (TFC, TFM, FC, FM)	3	22.65/S	0.0002	55.32/S	0.0001
Time (0, 20, 40, 60 min)	3	10.19/S	0.0001	14.68/S	0.0001
Treatments x Time	9	2.27/S	0.0391	5.37/S	0.0001
R^2		0.77		0.86	
Coefficient of variation (%)		96.06		74.51	
		July, 1998		August, 1998	
Source of variation	df	F value	Pr > F	F value	Pr > F
Block	3	3.72/S	0.0198	1.54/NS	0.2206
Treatments (TFC, TFM, FC, FM)	3	22.79/S	0.0002	14.73/S	0.0008
Time (0, 20, 40, 60 min)	3	29.42/S	0.0001	8.30/S	0.0003
Treatments x Time	9	23.98/S	0.0001	8.01/S	0.0001
R^2		0.95		0.83	
Coefficient of variation (%)		75.68		166.09	

/NS Not significant at P=0.05

/S Significant at P=0.05.

The nitrate-nitrogen concentration differed significantly among the VFS treatments (TFC, TFM, FC, FM) and among the sampling times after the initiation of runoff (0, 20, 40, 60 min). The interaction of both main treatment effects resulted in highly significant differences (P=0.05) in the NO_3-N concentrations in runoff water. These differences were consistent across the runoff measurements made in July and August 1997 and 1998.

The mean NO_3-N concentrations of runoff water from VFS and fallow treatments at four samplings periods (0, 20, 40, and 60 min) are shown in Table 2. Clearly, NO_3-N concentration in runoff was affected by duration of the runoff event. The first runoff sample (0 min) in the FM treatment, for July and August for both 1997 and 1998, had the highest NO_3-N concentrations. Correspondingly, the concentration of the initial runoff was significantly different from the concentration of subsequent samplings. This pattern was also measured in runoff events for FC treatment in July and August 1997. Nitrate concentration of runoff from VFS (TFC and TFM treatments) over time did not follow the pattern observed in the fallow treatments. Nitrate-nitrogen concentration in runoff water from TFC and TFM treatments did not differ significantly among the different times of sampling for July and August of 1997 and 1998.

The effect of the VFS and fallow treatments on NO_3-N concentrations of the runoff water is shown in Figure 3 for 1997 and Figure 4 for 1998. Values for nitrate-nitrogen concentrations presented in Figures 3 and 4 are the averages of the concentrations of the samples collected at 0, 20, 40, and 60 minutes after initiation of runoff.

The July 1997 mean NO_3-N concentrations of 0.12 mg L^{-1} for the TFM and 0.16 mg L^{-1} for the TFC, and <0.1 mg L^{-1} for both treatments in August, were not significantly different. In 1998, neither treatment (TFM or TFC) had significant differences in NO_3-N concentration in the runoff. All measured NO_3-N concentrations were below the threshold of detection (under 0.1 mg L^{-1}).

Table 2. Mean nitrate nitrogen (NO_3-N) concentration in runoff water of VFS and fallow strips.

	Mean NO_3-N concentrations (mg L^{-1})/1							
	July, 1997				August, 1997			
	Treatments/2				Treatments/2			
Time (min)	Fallow Control	Fallow Manure	Tall Fescue Control	Tall Fescue Manure	Fallow Control	Fallow Manure	Tall Fescue Control	Tall Fescue Manure
0	5.6a	4.03a	0.40a	0.45a	2.25a	4.30a	0.03a	0.18a
20	2.75b	1.98b	0.25a	0.03a	1.13b	1.85b	0.00a	0.00a
40	1.53b	1.03b	0.00a	0.00a	0.73b	1.28b	0.03a	0.05a
60	1.23b	0.85b	0.00a	0.00a	0.63b	0.85b	0.00a	0.00a
	Mean NO_3-N concentrations (mg L^{-1})/1							
	July, 1998				August, 1998			
	Treatments/2				Treatments/2			
Time (min)	Fallow Control	Fallow Manure	Tall Fescue Control	Tall Fescue Manure	Fallow Control	Fallow Manure	Tall Fescue Control	Tall Fescue Manure
0	1.05a	13.88a	0.00a	0.10a	0.43a	17.75a	0.00a	0.00a
20	0.23a	4.23b	0.00a	0.03a	0.23a	3.78b	0.00a	0.03a
40	0.18a	2.28c	0.00a	0.03a	0.20a	2.08b	0.00a	0.03a
60	0.18a	1.70c	0.00a	0.05a	0.18a	1.48b	0.00a	0.03a

/1Concentrations below 0.1 mg L^{-1} were assumed equal zero.
/2Means with the same letter in the same column are not significantly different at P=0.05.

Several mechanisms are described in the scientific literature as being responsible for trapping sediment and nutrients in runoff through vegetated filter strips (Dillaha et al., 1988).