



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
Montana State University

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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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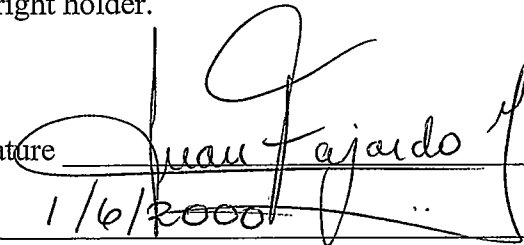
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ABSTRACT

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 ($\text{NO}_3\text{-N}$) 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 $\text{NO}_3\text{-N}$ 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 $\text{NO}_3\text{-N}$ 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 $\text{NO}_3\text{-N}$ 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 ($\text{NO}_3\text{-N}$) 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 $\text{NO}_3\text{-N}$ concentrations above 10 mg L^{-1} and about 7 percent were contaminated with both bacteria and $\text{NO}_3\text{-N}$. 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 50m² 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 NO₃-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 3.3×10^{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 3.86×10^9 million metric tons (about 2 percent of the world's total). On the other hand, soils contain a very small fraction (about 2.4×10^5 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 130×10^9 kg N per year, followed by NH_4^+/NH_3 deposition (19 to 50×10^9 kg N per year), and NO_3^-/NO_x deposition (11 to 33×10^9 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 $\text{NO}_3\text{-N}$ 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 NO₃-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 United States 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 ($\text{NO}_3\text{-N}$) 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 $7,200 \text{ kg ha}^{-1}$ in a 9.1m soil profile while adjacent cropland had just 570 kg ha^{-1} and $\text{NO}_3\text{-N}$ 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 ($0, 60, 120, \text{ and } 180 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, 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 $\text{NO}_3\text{-N}$, 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 $\text{ha}^{-1}\text{yr}^{-1}$ at manure rates of 60, 120, and 180 Mg $\text{ha}^{-1}\text{yr}^{-1}$, 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 coliforms 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 Enterobacteriaceae, 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 -galactosidase 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 brome grass, *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 (NO₃-N) and soluble phosphorus in runoff was moderate, with NO₃-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 m², 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 (*Dactylis glomerata* L.) cultivar Latar, tall fescue (*Festuca arundinacea* Schreb.) cultivar Fawn, meadow brome grass (*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 9m² 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 brome grass 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

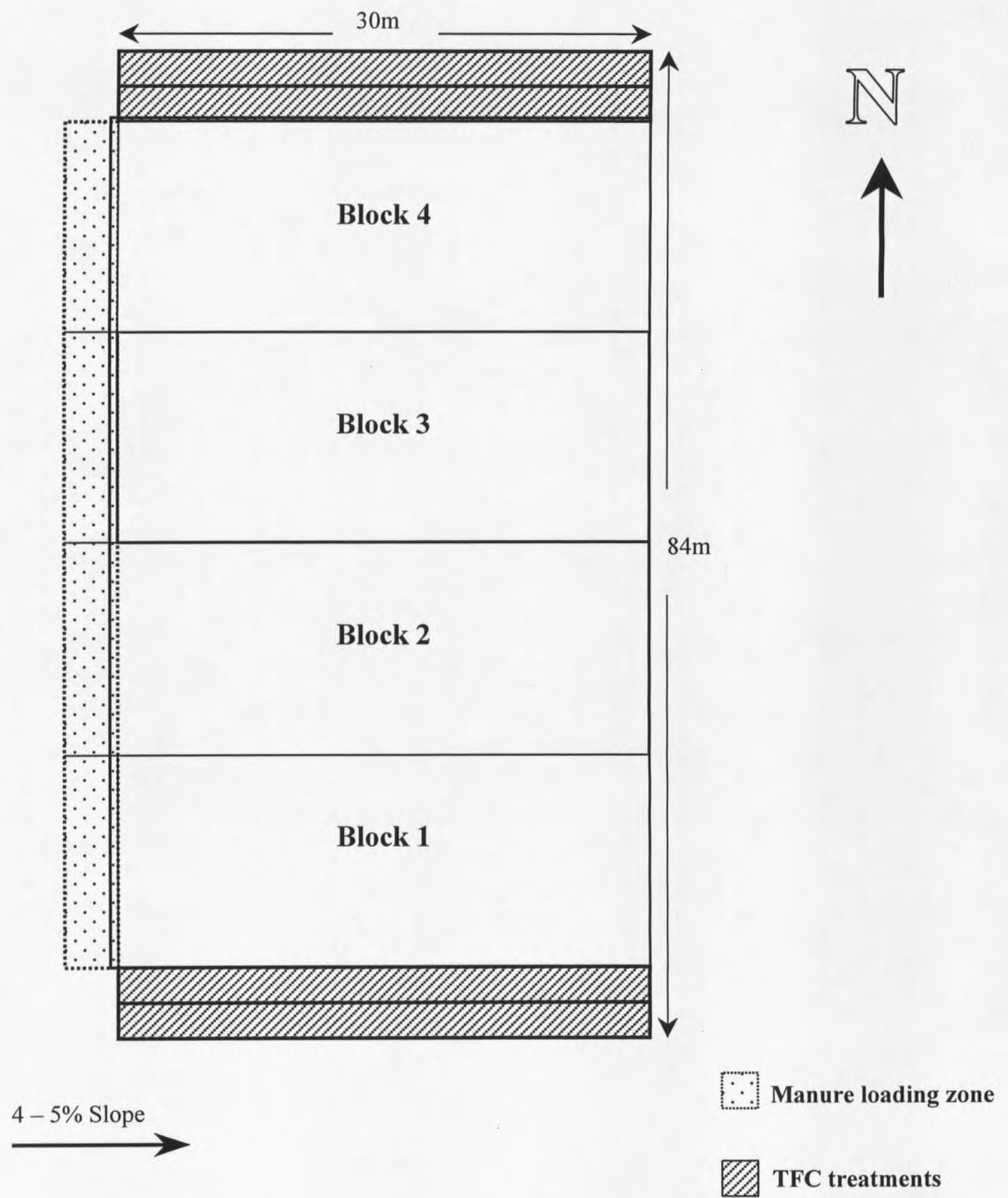


Figure 1. Experimental field plot design.

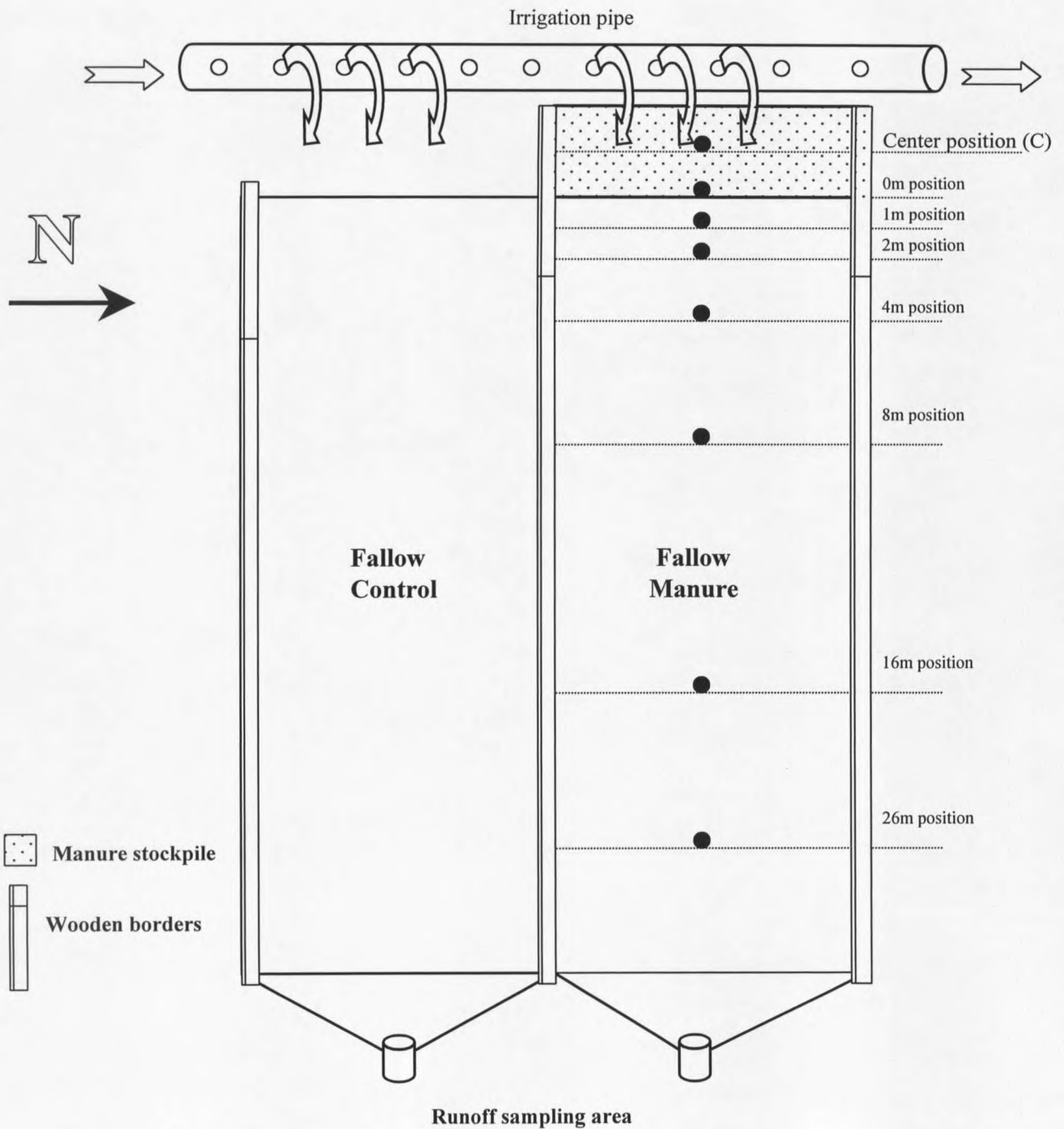


Figure 2. Schematic design of VFS. Sampling positions and modifications introduced are displayed.

treatments (not including FC and TFC treatments), corresponding to approximately 2-metric 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 ($\text{NO}_3\text{-N}$) 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 $\text{NO}_3\text{-N}$ 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 AnalysisNitrate Nitrogen Concentration of VFS Runoff Water

Laboratory determination of $\text{NO}_3\text{-N}$ in runoff was made utilizing the automated cadmium reduction colorimetric-based method, (Clesceri et al., 1989), which cannot detect concentrations of $\text{NO}_3\text{-N}$ below 0.1 mg L^{-1} . Therefore, concentrations of $\text{NO}_3\text{-N}$ 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 ($\text{NO}_3\text{-N}$) concentrations in runoff water for 1997 and 1998 events are presented in Table 1.

Table 1. Summary of ANOVA of nitrate nitrogen ($\text{NO}_3\text{-N}$) 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 $\text{NO}_3\text{-N}$ concentrations in runoff water. These differences were consistent across the runoff measurements made in July and August 1997 and 1998.

The mean $\text{NO}_3\text{-N}$ 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, $\text{NO}_3\text{-N}$ 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 $\text{NO}_3\text{-N}$ 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 $\text{NO}_3\text{-N}$ 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 $\text{NO}_3\text{-N}$ concentrations of 0.12 mg L^{-1} for the TFM and 0.16 mg L^{-1} for the TFC, and $<0.1 \text{ mg L}^{-1}$ for both treatments in August, were not significantly different. In 1998, neither treatment (TFM or TFC) had significant differences in $\text{NO}_3\text{-N}$

concentration in the runoff. All measured $\text{NO}_3\text{-N}$ concentrations were below the threshold of detection (under 0.1 mg L^{-1}).

Table 2. Mean nitrate nitrogen ($\text{NO}_3\text{-N}$) concentration in runoff water of VFS and fallow strips.

Mean $\text{NO}_3\text{-N}$ concentrations (mg L^{-1}) ¹								
Time (min)	July, 1997				August, 1997			
	Treatments ²				Treatments ²			
	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 $\text{NO}_3\text{-N}$ concentrations (mg L^{-1}) ¹								
Time (min)	July, 1998				August, 1998			
	Treatments ²				Treatments ²			
	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).

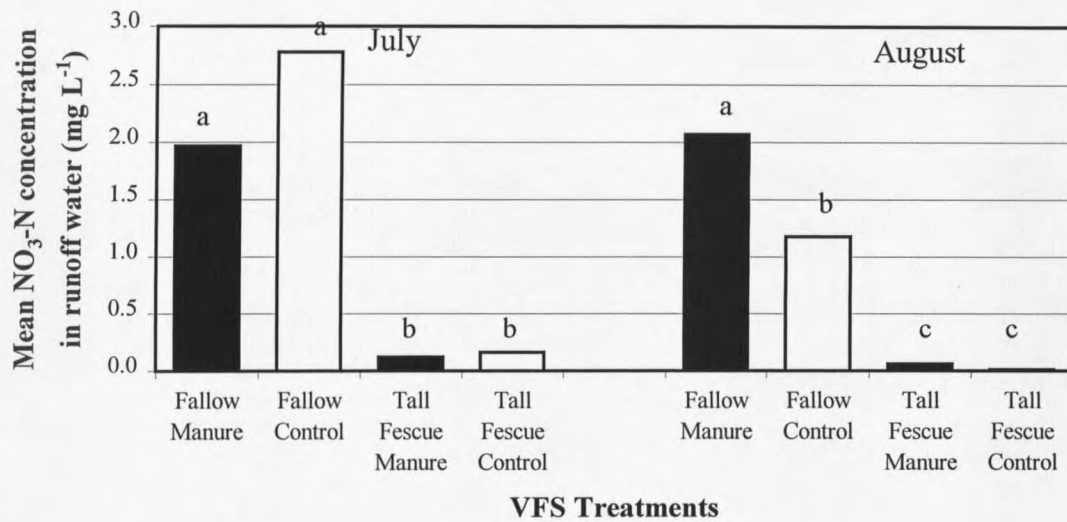


Figure 3. Mean $\text{NO}_3\text{-N}$ concentration in runoff water, 1997. Means were obtained by averaging nitrate concentrations of samples collected at 0, 20, 40, and 60 min after initiation of runoff. Means followed by the same letter are not significantly different at $P=0.05$.

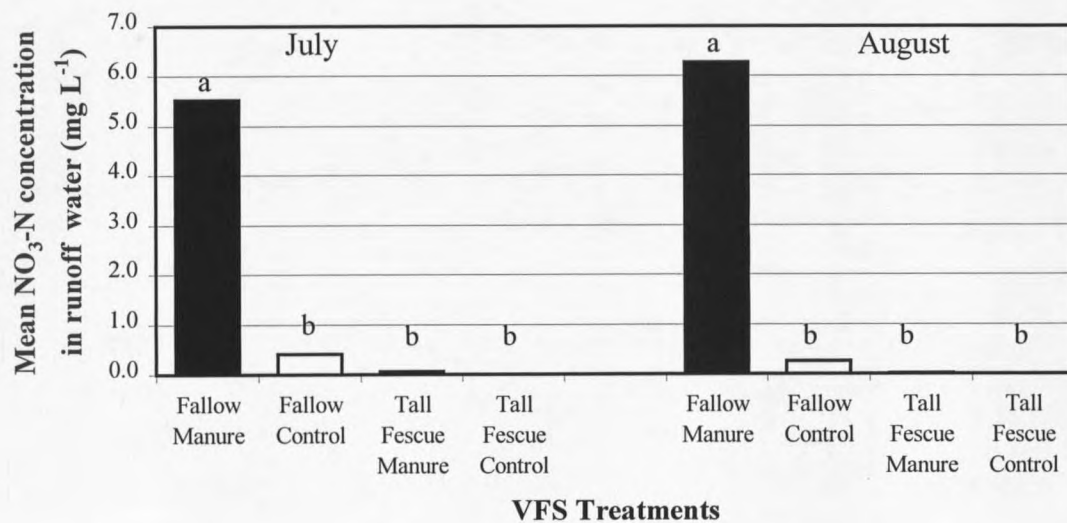


Figure 4. Mean $\text{NO}_3\text{-N}$ concentration in runoff water, 1998. Means were obtained by averaging nitrate concentrations of samples collected at 0, 20, 40, and 60 min after initiation of runoff. Means followed by the same letter are not significantly different at $P=0.05$.

Enhanced infiltration is a significant mechanism that improves performance of VFS in reducing nutrients in runoff (Dickey and Vanderholm, 1981; Schwer and Clausen, 1989; and Yang et al., 1980). Filter strips enhance infiltration and sediment deposition by reducing the velocity of runoff (Yang et al., 1980). Thus, pollutants dissolved in runoff water enter the soil as infiltration takes place. Infiltration may be the mechanism that reduced nitrate-nitrogen concentration in TFM runoff. Thus, nitrates from the manure stockpile and which were carried by the runoff were mainly trapped in the soil profile as infiltration of the runoff occurred. The 30-m long VFS was able to reduce velocity of runoff at least 120 min before the runoff left the VFS, thus increasing infiltration. In addition, background levels of $\text{NO}_3\text{-N}$ in irrigation water used to create runoff were less than the threshold of detection. Therefore, the irrigation water was able to dilute the concentration of nitrates to very low levels.

The $\text{NO}_3\text{-N}$ concentrations in runoff water from the FM and FC treatments for 1997 were 1.97, 2.78 and 2.07, 1.18 mg L^{-1} for July and August, respectively. In 1998, runoff water from the FM treatment had the highest concentration of $\text{NO}_3\text{-N}$. The concentrations were 5.52 and 6.27 mg L^{-1} for July and August, respectively. These values were significantly different from FC. Concentrations of $\text{NO}_3\text{-N}$ in runoff from FC treatment were 0.41 and 0.26 mg L^{-1} for July and August, respectively.

Although sediment concentration in runoff water was not measured, it was apparent from cursory observation that runoff from the fallow treatments (FC and FM) carried considerable sediment. Nitrates may have been associated with this sediment. The successive runoff events most likely eroded away some of the surface layer of soil. This

uppermost soil layer is probably the zone where the higher concentrations of mineralized nitrogen are found in the soil profile. Comparing the runoff events after July 1997 (August 1997, and July and August 1998), it can be seen that the $\text{NO}_3\text{-N}$ concentration in the runoff water for the FC treatment decreased in each subsequent runoff event. This pattern was not observed for the FM treatment, which had a constant supply of nitrates from the manure stockpile. Therefore, it may be postulated that $\text{NO}_3\text{-N}$ measured in the runoff water from the FM treatment may be attributable to the manure application and can be assumed as the total potential $\text{NO}_3\text{-N}$ flushed from the manure stockpile when runoff occurred.

A VFS is potentially a valuable tool to capture the nitrate losses from manure confinement areas. Based on a comparison of the data from the TFM and FM treatments, the VFS reduced nitrate-nitrogen losses in runoff by 94 and 97 percent for July and August, respectively, in 1997; the reduction was 99 percent for both the July and August events in 1998.

Total and Fecal Coliforms from VFS Runoff Water

The runoff samples collected in 1997 were analyzed for total coliforms (C_T); the runoff samples collected in 1998 were analyzed for fecal coliforms (C_F).

Table 3 is a summary of ANOVA of bacteria counts present in runoff water from the 1997 and 1998 events. Total coliform counts differed significantly among treatments (TFC, TFM, FC, and FM) and time of sampling (0, 20, 40, and 60 min) ($P=0.05$) in 1997. In August 1998, only VFS treatment had a significant effect ($P=0.05$) on fecal

coliform counts. Variability in the data was high. This is reflected in the coefficients of variation.

Table 3. Summary of ANOVA for bacteria counts present in runoff water during 1997 (total coliforms) and 1998 (fecal coliforms) sampling events.

July, 1997				August, 1998	
Source of variation	df	F value	Pr > F	F value	Pr > F
Block	3	2.51 ^{NS}	0.0743	4.94 ^S	0.0057
Treatments (TFC, TFM, FC, FM)	3	24.93 ^S	0.0001	5.80 ^S	0.0174
Time (0, 20, 40, 60 min)	3	9.22 ^S	0.0001	9.04 ^S	0.0001
Treatments x Time	9	1.93 ^{NS}	0.0794	1.85 ^{NS}	0.0920
R ²		0.79		0.75	
Coefficient of variation (%)		39.63		124.55	
July, 1998				August, 1998	
Source of variation	df	F value	Pr > F	F value	Pr > F
Block	3	0.66 ^{NS}	0.5809	2.11 ^{NS}	0.1155
Treatments (TFC, TFM, FC, FM)	3	0.53 ^{NS}	0.6720	6.72 ^S	0.0113
Time (0, 20, 40, 60 min)	3	1.59 ^{NS}	0.2086	2.51 ^{NS}	0.0740
Treatments x Time	9	0.62 ^{NS}	0.7735	2.09 ^{NS}	0.0574
R ²		0.41		0.64	
Coefficient of variation (%)		159.13		143.08	

^{NS} Not significant at $P=0.05$

^S Significant at $P=0.05$

The mean coliform bacteria counts in runoff water from the VFS and fallow treatments at four samplings times after the initiation of runoff (0, 20, 40, and 60 min) are shown in Table 4. Bacteria counts in runoff water from TFM treatment were significantly

different among sampling time in 1997; the highest counts of C_T occurred at time 0 min. Bacteria counts did not differ significantly ($P=0.05$) from time 20 through 60 min for either July or August events.

Table 4. Total coliforms (TC) and fecal coliforms (FC) counts in runoff water of VFS and fallow strips.

Mean total coliforms (CFU 100ml ⁻¹) ¹								
July, 1997					August, 1997			
Time (min)	Treatments				Treatments			
	Fallow Control	Fallow Manure	Tall Fescue Control	Tall Fescue Manure	Fallow Control	Fallow Manure	Tall Fescue Control	Tall Fescue Manure
0	20000a	20000a	17000a	14550a	707500a	10500000a	1775000a	6562500a
20	20000a	20000a	10200a	6150b	30000a	4625000b	490000a	1587500b
40	16800ab	20000a	2755b	3435b	47500a	2500000b	405000a	655000b
60	10075b	20000a	1550b	4690b	75000a	2125000b	602500a	255000b
Mean fecal coliforms (CFU 100ml ⁻¹) ¹								
July, 1998					August, 1998			
Time (min)	Treatments				Treatments			
	Fallow Control	Fallow Manure	Tall Fescue Control	Tall Fescue Manure	Fallow Control	Fallow Manure	Tall Fescue Control	Tall Fescue Manure
0	3350a	9750a	2150a	5375a	346000a	1387500a	74500a	88000a
20	2775a	3625ab	1950a	1025a	48500a	517500b	337250a	188250a
40	800a	2400b	3075a	1500a	261500a	422500b	19750a	19500a
60	2350a	1575b	2425a	3125a	265500a	197500b	37750a	36000a

¹Means with the same letter in the same column are not significantly different at $P=0.05$.

Total coliforms counts from the FM treatment in August 1997 and C_F counts in July and August of 1998 differed among sampling times. The highest counts occurred at time 0 min, followed by no significant difference from time 20 min through 60 min.

Significant counts were also measured in runoff from treatments that did not received manure, i. e., FC and TFC. These counts were comparable to the values obtained from the FM and TFM treatments. Populations of C_F in the order of 6×10^3 colony forming units (CFU) per 100ml^{-1} were measured in the water used to force runoff. Most likely, the source of bacterial contamination in the control treatments was the irrigation water. It is possible that cross contamination of coliforms between plots due to rainfall occurred. Moreover, C_T includes species that are commonly found in unpolluted soils and vegetation (Gleeson and Gray, 1997); therefore, their indigenous presence may increase the number of coliforms that were attributed to runoff originating from the manure stockpiles. In addition, wildlife can contribute significantly to coliforms in runoff from land not receiving applications of cattle manure (Faust, 1982; Patni et al., 1985).

Figures 5 and 6 present the average C_T and C_F in runoff water for four times of sampling for each treatment. Concentrations of C_T in runoff from TFM in 1997 were 7.2×10^3 and 2.3×10^6 CFU per 100ml^{-1} , for July and August sampling, respectively; in 1998, the counts were 2.7×10^3 , and 8.3×10^4 CFU per 100ml^{-1} , respectively. The counts in runoff water from TFC treatment were not different significantly from the counts in runoff water from TFM treatment either year.

The bacterial counts in runoff water from the FM treatment were not significantly different from the bacterial counts in runoff water from FC treatment in July of 1997 and July 1998. They did differ significantly in August 1997 and August 1998.

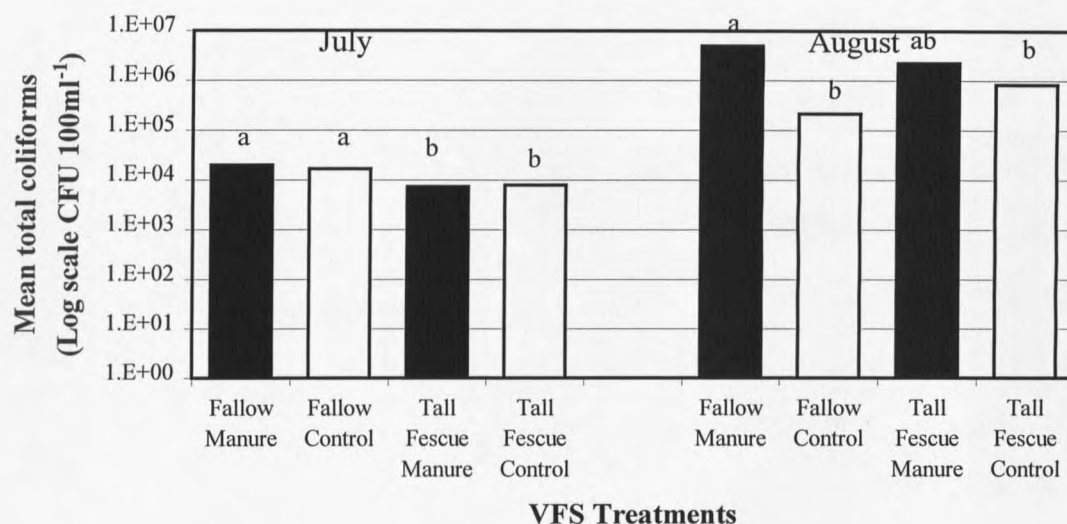


Figure 5. Mean total coliforms present in runoff water, 1997. Means were obtained by averaging bacteria counts of samples collected at 0, 20, 40, and 60 min after initiation of runoff. Means within a single sampling period followed by the same letter are not significantly different at $P=0.05$.

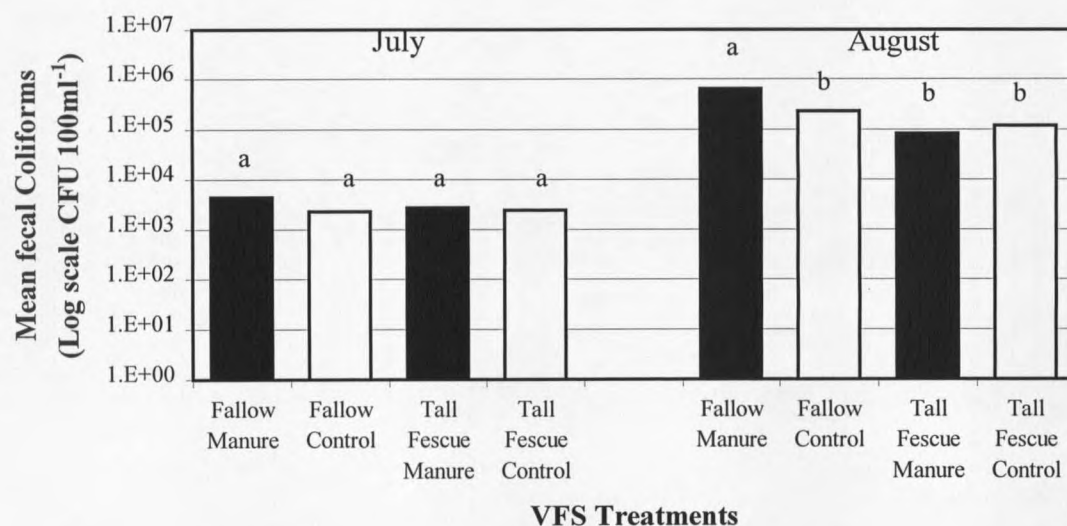


Figure 6. Mean fecal coliforms present in runoff water, 1998. Means were obtained by averaging bacteria counts of samples collected at 0, 20, 40, and 60 min after initiation of runoff. Means within a single sampling period followed by the same letter are not significantly different at $P=0.05$.

Counts of coliforms in the runoff water from FM treatment were 20×10^3 and 5×10^6 CFU per 100ml⁻¹ of C_T in July and August 1997, respectively; and 4.3×10^3 and 6.3×10^5 CFU per 100ml⁻¹ of C_F in July and August 1998, respectively.

The bacterial counts were 16×10^3 and 21.5×10^5 CFU per 100ml⁻¹ of C_T for the FC treatments in July and August 1997, respectively; and 2.7×10^3 and 23×10^5 CFU per 100ml⁻¹ of C_F in July and August 1998, respectively. Between the July and the August sampling each year, bacteria population counts increased by a factor of as much as 100 fold. Presumably, the increase in bacteria counts through the summer in runoff water reflects enhancement of the growth rate of bacteria populations in the manure stockpiles as summer temperatures increased.

In this study VFS did not appear to be as efficient a tool for reducing bacterial contamination contained in runoff from manure stockpiles as it was for reducing nitrates. Assuming, the coliform counts measured in runoff from the FM treatment represented the maximum coliform counts in runoff water, the installation of a VFS reduced bacterial pollution approximately 64 to 87 percent in the runoff of July 1997 and August 1998, respectively. In contrast, VFS treatment did not significantly affect coliform counts from FM treatment in August 1997 or July 1998. The reductions in coliform counts are comparable with values obtained by Yang et al. (1980), which were in the order of 70 percent reductions for C_F and C_T using a 36-m VFS.

Although reductions in coliform counts were relatively acceptable, the final concentrations were still greater than the standards established for bathing waters of 200 counts per 100 ml (USEPA, 1986).

Nitrate Nitrogen in VFS Soil Profile

In April of each study year, following removal of manure from the previous year, soil sampling was completed for the TFM, TFC, FM, and FC treatments. Seven positions (Center (C), 0, 1, 2, 4, 8, 26m down slope from the manure stockpile) within each strip were chosen and sampled to a depth of two meters. Concentration of $\text{NO}_3\text{-N}$ was determined for six consecutive depth increments (0-10, 10-20, 20-40, 40-80, 80-160, and 160-200cm). Nitrate concentration for each depth increment was multiplied by its corresponding depth to obtain an estimated depth-weighted nitrate load, i. e., an expression of mg $\text{NO}_3\text{-N}$ in the soil profile. This calculation was made to facilitate analyses and interpretation of an extensive $\text{NO}_3\text{-N}$ dataset. Bulk density was assumed uniform and estimated as 1.3 Mg m^{-3} . A single value of $\text{NO}_3\text{-N}$ load in the profile for each position was obtained by adding the $\text{NO}_3\text{-N}$ loads of the six incremental depths. A factorial analysis with three factors (VFS, treatment, and position) was used to complete an analysis of variance of these data. The VFS treatment had two levels, fallow (strip without grass cover) and grass (tall fescue); treatment factor had two levels, with manure application and without (control); and position factor had seven levels corresponding to the points where sampling was undertaken.

The ANOVA for $\text{NO}_3\text{-N}$ load in the soil profile calculated by this procedure indicated that $\text{NO}_3\text{-N}$ varied significantly among VFS, treatment, and position ($P=0.05$) for both years of sampling (Table 5). Mean separations and distribution of $\text{NO}_3\text{-N}$ load along the VFS treatments are presented in Table 6 and Figures 7 and 8, respectively. The greatest soil $\text{NO}_3\text{-N}$ loads were measured directly beneath the manure stockpile.

Table 5. Summary of ANOVA of calculated NO₃-N load in soil for 1997 and 1998.

Source of variation	df	Nitrate load 1997		Nitrate load 1998	
		F value	Pr > F	F value	Pr > F
Block	3	1.09 ^{/NS}	0.3565	1.11 ^{/NS}	0.3503
VFS (Fallow, Tall fescue)	1	3.99 ^{/S}	0.0429	59.85 ^{/S}	0.0001
Treatment (Manure, control)	1	30.10 ^{/S}	0.0001	71.63 ^{/S}	0.0001
Position (C, 0, 1, 2, 4, 8, 26)	6	22.33 ^{/S}	0.0001	36.64 ^{/S}	0.0001
VFS x Treatment	1	6.42 ^{/S}	0.0132	1.06 ^{/NS}	0.3059
VFS x Position	6	1.51 ^{/NS}	0.1869	1.12 ^{/NS}	0.3599
Treatment x Position	6	23.53 ^{/S}	0.0001	39.85 ^{/S}	0.0001
VFS x Treatment x Position	6	1.65 ^{/NS}	0.1447	0.74 ^{/NS}	0.6173
R ²		0.81		0.88	
Coefficient of variation (%)		82.37		52.75	

^{/NS} Not significant at $P=0.05$ ^{/S} Significant at $P=0.05$.

There appeared to be no significant subsurface movement of NO₃-N along the down slope positions of the TFM and FM treatments beyond the direct influence of the manure stockpile (position C and 0m). The greatest NO₃-N load in the soil profile for the TFM treatment was measured at position C and 0m. There were no significant differences in NO₃-N load among position 1m and any down slope positions to 26m in either year. The greatest NO₃-N load for FM treatment occurred at position C. There were no significant differences among position 0m and any down slope positions to 26m in either year. No significant differences were found between the TFC and FC treatments in 1997. However, in 1998 the differences were significant, indicating greater NO₃-N accumulation in the FC treatment soil profile than in the TFC treatment soil profile. This accumulation of NO₃-N in soil profile from the FC treatment was assumed to be due to

nitrogen mineralized from soil organic matter and not subjected to leaching or root plant uptake. A better observation of this process can be observed in Figure 7 in which accumulation of $\text{NO}_3\text{-N}$ is observed and in Figure 8 where $\text{NO}_3\text{-N}$ is not accumulated between 1997 and 1998.

Table 6. Mean $\text{NO}_3\text{-N}$ load in two meters soil profile along the fallowed and vegetated strips. Sampling corresponds to April 1997 and 1998.

Mean $\text{NO}_3\text{-N}$ (mg) in 2m soil profile, 1997 sampling ¹				
Position	Tall fescue		Fallow	
	Manure	Control	Manure	Control
Center	123.78 a	4.10 c	102.00 a	16.03 c
0 m	53.93 b	2.55 c	15.85 c	19.40 c
1 m	6.33 c	3.55 c	19.40 c	21.93 c
2 m	5.15 c	5.50 c	16.70 c	21.00 c
4 m	4.08 c	5.28 c	12.73 c	18.60 c
8 m	4.43 c	4.88 c	13.65 c	13.75 c
26 m	3.98 c	4.40 c	9.85 c	16.38 c
Mean $\text{NO}_3\text{-N}$ (mg) in 2m soil profile, 1998 sampling ¹				
Position	Tall fescue		Fallow	
	Manure	Control	Manure	Control
Center	135.59 a	5.28 fg	153.17 a	23.14 cdefg
0 m	52.50 b	4.71 g	41.69 bc	29.08 cd
1 m	8.21 defg	4.32 g	30.47 c	32.03 bc
2 m	5.62 fg	3.19 g	33.39 bc	31.61 bc
4 m	5.24 fg	3.82 g	28.03 cd	27.33 cde
8 m	5.27 fg	2.88 g	28.47 cd	26.10 cdef
26 m	6.53 efg	4.23 g	38.31 bc	35.08 bc

¹Means with the same letter within and across columns are not significantly different at $P=0.05$.

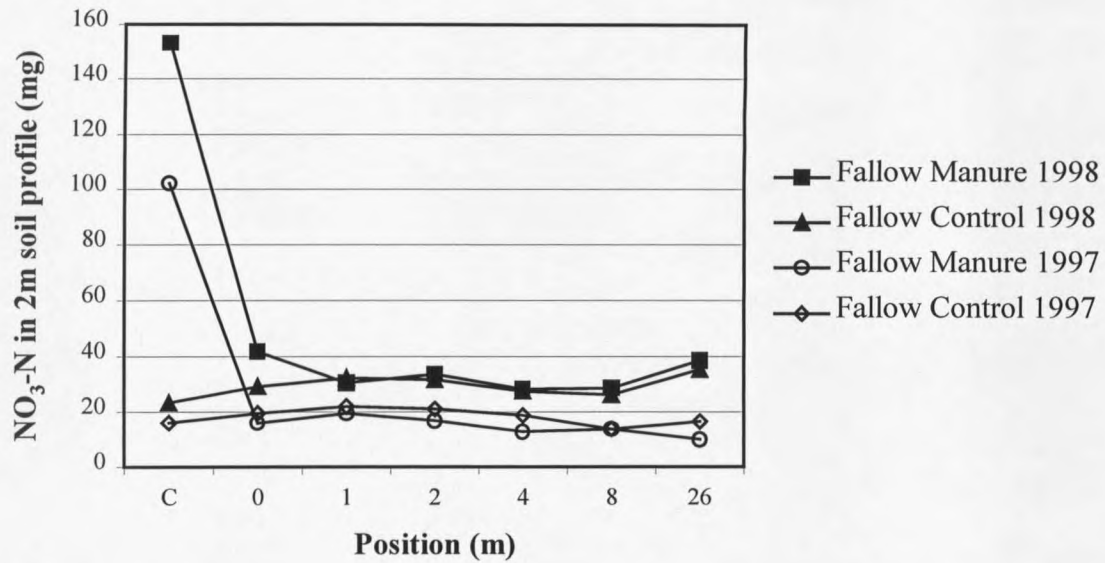


Figure 7. $\text{NO}_3\text{-N}$ load as a function of position along fallow strips for 1997 and 1998 sampling (C= center position).

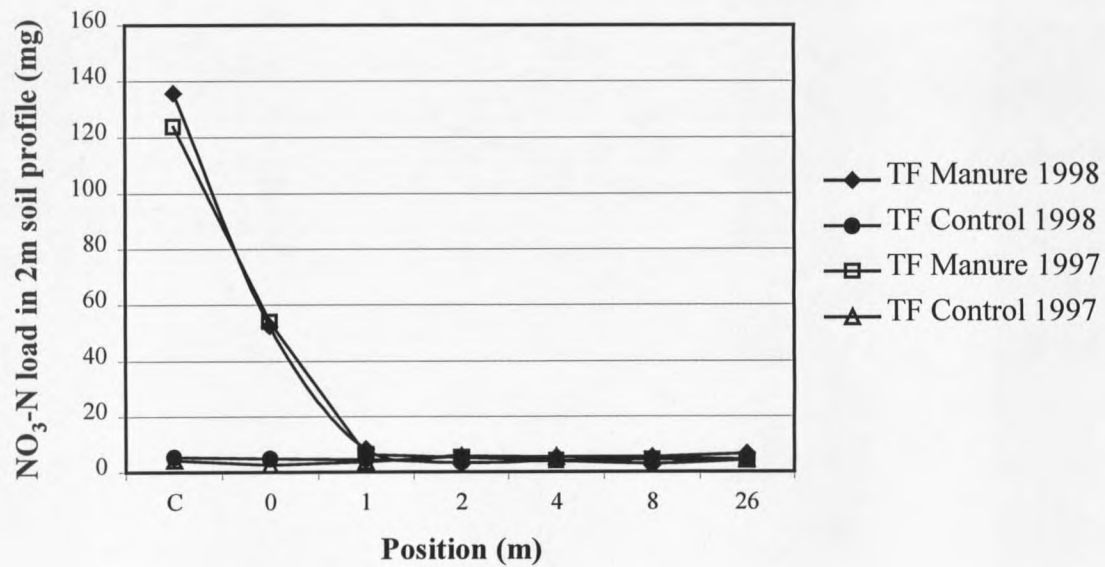


Figure 8. $\text{NO}_3\text{-N}$ load as a function of position along VFS strips for 1997 and 1998 sampling (TF= Tall fescue, C= center position).

The lack of accumulation was assumed to be due to presence of the growing grass cover that continually took up the mineralized nitrogen.

The $\text{NO}_3\text{-N}$ concentrations from the April of 1997 and 1998 soil sampling are shown in Table 7. The values were obtained by averaging the concentrations collected from the six incremental depths in the soil profile at each position. In Table 8 are depicted the $\text{NO}_3\text{-N}$ loads equivalent to the concentrations presented in Table 7. The values are expressed in kilograms of $\text{NO}_3\text{-N}$ per hectare in two meters soil profile. As seen in Table 8, the manure stockpile represented a huge load of $\text{NO}_3\text{-N}$ to the soil profile, in 1997, with approximately 2,000 kg $\text{NO}_3\text{-N}$ per ha in the 2 m depth (at position C), of TFM and FM treatments. In contrast, the average of the seven positions of the FC and TFC strips treatments represented only 217 and 66 kg $\text{NO}_3\text{-N}$ per ha in 2 m depth, respectively, for the same year of sampling.

In 1998, the values were similar with 1,800 kg of $\text{NO}_3\text{-N}$ per ha in 2 m depth at position C for the average of the TFM and FM treatments. Averaging along the seven positions of sampling, the FC and TFC strips were 358 and 76 kg $\text{NO}_3\text{-N}$ kg per ha in 2 m soil depth, respectively. The FC treatment showed an appreciable accumulation of nitrates in the soil profile from 1997 to 1998, as similarly shown in Figure 7. Huge amounts of $\text{NO}_3\text{-N}$ were also found by Mielke and Ellis (1976) from abandoned and active feedlots in Nebraska. These feedlots were occupied by approximately 250 animals. In average 9.1m soil cores from abandoned feedlots contained loads of 7,200 kg ha^{-1} of $\text{NO}_3\text{-N}$. Active feedlots averaged 1,800 kg ha^{-1} of $\text{NO}_3\text{-N}$ and cropland areas 570 kg ha^{-1} of $\text{NO}_3\text{-N}$ in 9.1m soil cores.

Table 7. Mean $\text{NO}_3\text{-N}$ concentrations in two meters soil profile along the fallowed and vegetated strips. Values correspond to April 1997 and 1998 sampling.

$\text{NO}_3\text{-N}$ (mg kg^{-1} soil $^{-1}$), 1997 sampling				
Position	Tall fescue		Fallow	
	Manure	Control	Manure	Control
Center	81.2	2.6	72.4	7.9
0 m	35.3	1.6	10.2	8.7
1 m	3.6	2.2	9.9	9.4
2 m	2.9	2.7	7.6	9.2
4 m	2.5	2.9	5.7	8.2
8 m	2.7	2.9	5.9	6.6
26 m	2.7	2.8	5.1	8.3
$\text{NO}_3\text{-N}$ (mg kg^{-1} soil $^{-1}$), 1998 sampling				
Position	Tall fescue		Fallow	
	Manure	Control	Manure	Control
Center	66.2	4.1	77.7	10.6
0 m	29.1	3.6	25.1	14.2
1 m	7.3	3.1	12.9	14.2
2 m	4.2	2.1	14.7	14.4
4 m	4.1	2.6	13.0	12.5
8 m	3.6	1.9	14.2	13.8
26 m	5.4	3.3	16.9	16.8

Table 8. NO₃-N load per hectare in two meters soil profile along the fallowed and vegetated strips. Values correspond to April 1997 and 1998 sampling.

NO ₃ -N (kg ha ⁻¹ per 2 m soil profile), 1997 sampling				
Position	Tall fescue		Fallow	
	Manure	Control	Manure	Control
Center	2,111	69	1,882	204
0 m	918	43	266	227
1 m	93	57	257	245
2 m	74	71	197	240
4 m	65	76	149	213
8 m	71	76	153	171
26 m	70	72	132	216
NO ₃ -N (kg ha ⁻¹ per 2 m soil profile), 1998 sampling				
Position	Tall fescue		Fallow	
	Manure	Control	Manure	Control
Center	1,721	106	2,019	275
0 m	756	93	653	370
1 m	189	79	334	369
2 m	109	54	383	375
4 m	108	67	339	325
8 m	93	49	370	358
26 m	140	86	439	436

CHAPTER 4

SUMMARY AND CONCLUSIONS

Vegetative filter strips have been widely recognized as a tool for reducing pollutants in runoff from agricultural activities, including runoff from croplands, pastures, and livestock confinement areas. The present study assessed the role of VFS in reducing nitrate and bacteria pollution in runoff water from small livestock confinement areas, as a means of maintaining or improving the water quality.

The concentration of $\text{NO}_3\text{-N}$ in runoff was affected by duration of the runoff event. The greatest concentration was detected in the first runoff leaving the VFS and the fallow strips. Even though several studies suggest that soluble nitrogen is not reduced significantly by VFS, in this study the 30-m VFS was able to reduce levels of $\text{NO}_3\text{-N}$ in runoff below the threshold of detection. Based on the length of time it took runoff to reach the end of the plots, the length of the VFS used in this study promoted high infiltration of the runoff water. Slower water movement allowed infiltration of the nitrates dissolved in the runoff water. Good quality water (no detectable nitrate concentration), combined with a long flow through time of water facilitated dilution of the nitrates in runoff. As in other studies, the manure stockpile was point source of nitrate pollution. Moreover, bare soil was exposed to erosion and contributed to nitrates in runoff. The nitrate pollution generated from these two sources can be reduced up to 99 percent by the installation of a VFS as suggested by the present study.

Bacterial contamination in runoff water was not effectively reduced by the VFS. From the four runoff events monitored, only two events had significant reductions in coliform counts. However, the concentrations of coliforms in runoff from these two events were higher than the standards recommended for water for bathing. Results of this study suggest that high volumes of runoff water may dilute nitrates from manure to undetectable levels but may increase the numbers of coliforms that escaped from the manure stockpile, thus increasing their levels in the runoff. Since one of the purposes of the present study was to force runoff in the VFS strips, it was necessary to apply huge amounts of water, which was not representative of a real rainstorm situation. Therefore, the number of coliforms in runoff could be lower or non-existent, depending on if a real rainstorm event induces runoff able to leave the VFS.

There appeared to be no significant redistribution of $\text{NO}_3\text{-N}$ along the down slope positions beyond the direct influence of the manure stockpile. Moreover, great loads of nitrates were detected under the manure stockpiles. This nitrate, which accumulated in the soil, may be a potential source of contamination to ground water if the same spot is used constantly to accumulate the animal wastes.

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APPENDICES

APPENDIX A
CHEMICAL ANALYSIS OF MANURE APPLIED

Manure was applied annually in April to the upslope position of the treated VFS, after soil sampling was completed. Chemical composition of manure was moderately variable, as illustrated in Table 9, a chemical analysis of the manure applied in April 1998. The analysis indicated high concentrations of potassium, phosphorus, and total nitrogen. The high concentrations of base cations resulted in a relatively high pH of all samples.

Table 9. Chemical analysis¹ of manure applied in April 1998.

Sample	Potassium	Nitrate-N (NO ₃ -N) ²	Olsen Phosphorus	Organic Matter	TKN ³	EC ⁴	pH
	mg kg ⁻¹	mg kg ⁻¹	mg kg ⁻¹	g 100g ⁻¹	g 100g ⁻¹	mmhos cm ⁻¹	
1	36600	283.2	1801	55.8	2.20	11.4	9.6
2	32600	61.8	1688	55.8	2.36	10.0	9.4
3	28000	94.6	1675	48.6	1.79	9.1	9.3
4	36200	11.8	1855	54.0	2.00	11.9	9.7
Mean	33350	112.8	1754.8	53.6	2.1	10.6	9.5

¹ Soil Analytical Laboratory LRES department MSU Bozeman, Montana.

² Samples were analyzed using a 1:10 extraction procedure because of high organic matter content.

³ Total Kjeldahl Nitrogen.

⁴ Samples were mixed 1 (soil) to 4 (water) in order to obtain sufficient supernate to determine both pH and electrical conductivity (EC).

APPENDIX B

SOIL NITRATE CONCENTRATION DATA, APRIL 1997

Table 10. Soil NO₃-N concentration for tall fescue manure treatment, April 1997.

Position	Depths	NO ₃ -N concentration (mg kg ⁻¹)			
		Replications			
		1	2	3	4
Center	0-10	219.0	76.5	152.7	118.7
	10-20	174.3	61.6	105.2	73.8
	20-40	126.2	85.6	104.3	55.9
	40-80	59.1	48.3	83.7	51.4
	80-160	44.2	72.4	65.8	29.0
	160-200	10.2	79.8	30.8	20.1
0m	0-10	17.7	82.3	24.5	186.8
	10-20	6.9	91.4	12.0	47.9
	20-40	2.8	72.4	11.1	21.9
	40-80	1.9	65.7	2.1	12.9
	80-160	2.8	77.3	1.4	16.5
	160-200	11.1	66.6	0.5	11.1
1m	0-10	3.8	8.5	3.7	5.2
	10-20	1.3	8.2	2.6	4.5
	20-40	2.6	7.0	1.4	2.2
	40-80	0.9	5.5	1.1	1.8
	80-160	1.8	6.5	1.7	2.8
	160-200	1.4	6.7	1.7	2.5
2m	0-10	3.0	7.9	3.3	3.4
	10-20	1.8	6.4	2.3	1.6
	20-40	1.1	4.4	1.3	2.2
	40-80	1.2	3.6	1.0	1.7
	80-160	1.3	4.4	1.7	2.3
	160-200	2.2	6.7	2.1	1.7
4m	0-10	3.2	5.9	3.1	3.6
	10-20	1.7	6.7	2.5	3.0
	20-40	1.2	3.1	1.3	2.2
	40-80	1.1	2.4	1.1	1.8
	80-160	0.9	2.7	1.4	1.7
	160-200	0.9	3.4	2.7	2.4
8m	0-10	5.8	4.8	4.3	2.6
	10-20	1.7	5.8	3.0	2.9
	20-40	2.5	3.0	1.5	1.5
	40-80	0.9	1.8	1.7	8.4
	80-160	1.7	1.9	1.4	1.2
	160-200	1.5	2.4	2.1	1.4
26m	0-10	4.7	6.1	3.7	3.2
	10-20	6.1	8.2	1.6	2.8
	20-40	2.1	4.2	1.4	1.7
	40-80	0.7	2.0	1.4	1.4
	80-160	0.9	2.3	1.8	2.3
	160-200	0.3	2.4	1.4	1.7

Table 11. Soil NO₃-N concentration for tall fescue control treatment, April 1997.

		NO ₃ -N concentration (mg kg ⁻¹)			
Position	Depths	Replications			
		1	2	3	4
Center	0-10	5.9	7.3	5.1	4.4
	10-20	2.9	3.2	2.5	2.2
	20-40	1.4	1.8	1.8	1.7
	40-80	1.0	1.8	0.9	1.1
	80-160	0.9	2.7	1.3	1.5
	160-200	1.3	3.0	2.1	5.7
0m	0-10	3.5	3.2	3.1	2.6
	10-20	2.0	1.6	2.4	1.4
	20-40	0.4	2.9	1.2	1.4
	40-80	0.1	0.2	0.6	0.6
	80-160	1.0	0.8	0.5	1.6
	160-200	1.8	1.5	0.4	4.4
1m	0-10	3.4	5.6	3.4	6.3
	10-20	1.2	0.9	1.9	3.6
	20-40	0.6	2.2	0.9	3.2
	40-80	0.4	1.8	0.9	2.8
	80-160	1.0	0.5	0.7	3.5
	160-200	1.8	1.5	1.3	3.6
2m	0-10	2.0	4.0	2.3	4.4
	10-20	0.8	3.0	3.1	3.0
	20-40	0.5	3.6	1.3	3.4
	40-80	0.9	3.0	0.6	2.8
	80-160	2.4	3.6	1.3	3.7
	160-200	1.8	4.6	4.1	5.1
4m	0-10	4.1	5.9	4.2	5.0
	10-20	1.8	2.3	3.6	4.6
	20-40	0.9	2.7	1.3	3.3
	40-80	0.4	2.6	1.3	2.8
	80-160	1.8	3.6	2.1	3.2
	160-200	2.0	3.2	1.6	5.2
8m	0-10	3.1	5.8	6.1	5.3
	10-20	1.3	4.1	3.0	4.8
	20-40	0.6	3.4	1.5	4.0
	40-80	0.5	2.7	0.9	3.2
	80-160	0.8	2.7	1.3	3.7
	160-200	0.9	3.8	1.9	4.4
26m	0-10	5.5	6.0	4.0	4.8
	10-20	2.8	3.4	3.4	2.9
	20-40	2.5	2.3	1.1	2.3
	40-80	2.2	1.8	1.1	1.8
	80-160	0.6	2.7	1.1	2.2
	160-200	2.3	3.2	2.2	3.9

Table 12. Soil NO₃-N concentration for fallow manure treatment, April 1997.

Position	Depths	NO ₃ -N concentration (mg kg ⁻¹)			
		Replications			
		1	2	3	4
Center	0-10	138.1	109.1	161.6	168.3
	10-20	71.1	133.3	153.2	99.2
	20-40	16.2	12.9	98.4	73.1
	40-80	3.2	23.4	76.5	71.4
	80-160	15.4	42.0	60.4	41.0
	160-200	26.7	42.8	28.4	71.4
0m	0-10	20.2	4.1	29.2	14.9
	10-20	8.9	6.5	16.6	9.0
	20-40	10.5	8.9	18.3	5.6
	40-80	1.6	0.8	16.6	3.1
	80-160	3.2	2.4	9.9	1.4
	160-200	1.6	4.1	24.2	23.3
1m	0-10	19.9	7.7	9.2	8.2
	10-20	13.8	3.8	12.3	16.3
	20-40	12.4	5.7	14.8	3.9
	40-80	6.7	3.0	8.5	2.0
	80-160	9.0	16.8	11.9	5.5
	160-200	14.8	7.9	12.0	11.0
2m	0-10	7.5	5.6	9.2	4.2
	10-20	7.5	5.3	9.2	4.6
	20-40	8.2	9.5	7.4	4.9
	40-80	11.8	8.7	5.5	5.7
	80-160	16.8	8.9	10.3	4.2
	160-200	5.5	3.7	11.2	6.1
4m	0-10	9.0	5.5	4.9	5.0
	10-20	3.8	4.4	2.7	2.9
	20-40	2.9	9.0	2.1	7.0
	40-80	6.4	6.1	5.0	7.2
	80-160	1.6	5.7	11.5	10.7
	160-200	2.8	4.5	7.1	9.2
8m	0-10	4.0	4.1	6.8	5.0
	10-20	2.2	3.0	3.5	10.1
	20-40	1.6	8.7	2.5	7.5
	40-80	2.3	11.8	5.3	6.3
	80-160	4.0	15.9	7.9	6.6
	160-200	4.2	4.8	5.8	7.1
26m	0-10	5.7	4.0	5.8	6.1
	10-20	1.8	4.7	4.4	5.9
	20-40	1.6	6.4	4.8	7.6
	40-80	10.6	14.9	4.2	5.4
	80-160	5.0	6.1	2.8	1.5
	160-200	3.5	5.0	3.0	1.4

Table 13. Soil NO₃-N concentration for fallow control treatment, April 1997.

Position	Depths	NO ₃ -N concentration (mg kg ⁻¹)			
		Replications			
		1	2	3	4
Center	0-10	11.9	7.9	6.5	7.2
	10-20	6.2	7.2	4.6	5.0
	20-40	7.5	6.2	7.2	5.5
	40-80	13.2	10.9	14.1	9.6
	80-160	3.9	9.2	8.6	5.6
	160-200	11.2	8.2	5.2	6.0
0m	0-10	15.1	7.5	4.1	9.5
	10-20	9.9	4.5	6.0	5.6
	20-40	13.9	7.0	5.8	4.9
	40-80	13.2	22.4	7.2	2.3
	80-160	12.1	9.6	18.3	6.8
	160-200	5.8	6.0	6.4	5.8
1m	0-10	8.7	6.4	5.7	7.0
	10-20	7.7	4.9	8.2	4.7
	20-40	6.4	7.1	7.0	7.2
	40-80	20.7	10.2	16.6	21.5
	80-160	15.3	10.9	6.1	14.8
	160-200	4.9	6.1	7.6	10.1
2m	0-10	6.8	7.1	8.5	5.6
	10-20	3.8	4.4	9.8	8.0
	20-40	6.5	5.6	10.8	8.8
	40-80	19.0	10.2	23.0	19.8
	80-160	9.9	9.7	10.1	14.8
	160-200	6.0	4.3	5.9	3.3
4m	0-10	7.6	4.5	7.9	7.8
	10-20	6.4	4.1	4.7	9.6
	20-40	7.5	6.7	8.0	6.3
	40-80	20.5	8.9	18.4	16.2
	80-160	7.5	11.6	11.1	12.9
	160-200	2.8	2.1	2.1	1.6
8m	0-10	6.6	6.8	6.6	4.7
	10-20	4.7	6.1	6.4	4.9
	20-40	4.1	4.1	9.7	7.5
	40-80	6.5	1.9	15.8	18.6
	80-160	12.6	1.8	7.2	5.7
	160-200	3.9	2.8	5.8	3.2
26m	0-10	10.7	5.2	5.0	5.5
	10-20	7.0	4.8	4.6	6.3
	20-40	10.1	8.4	7.2	12.1
	40-80	19.1	20.0	11.7	11.2
	80-160	7.9	6.0	2.4	3.6
	160-200	11.8	13.4	2.3	3.3

APPENDIX C

SOIL NITRATE CONCENTRATION DATA, APRIL 1998

Table 14. Soil NO₃-N concentration for tall fescue manure treatment, April 1998.

		NO ₃ -N concentration (mg kg ⁻¹)			
Position	Depths	Replications			
		1	2	3	4
Center	0-10	74.5	26.9	38.6	77.7
	10-20	83.6	30.9	52.9	54.1
	20-40	137.3	37.4	110.1	84.8
	40-80	89.4	35.8	123.6	56.5
	80-160	83.6	32.5	103.0	62.7
	160-200	47.2	27.7	87.1	30.5
0m	0-10	24.9	75.4	24.3	19.4
	10-20	20.7	35.0	19.5	25.0
	20-40	15.8	22.8	11.6	25.7
	40-80	11.6	128.1	9.2	153.3
	80-160	15.8	8.2	11.6	13.1
	160-200	10.8	1.8	2.9	10.8
1m	0-10	29.3	23.9	26.4	6.7
	10-20	5.1	16.4	8.2	3.7
	20-40	5.4	6.6	4.3	6.0
	40-80	2.7	3.9	3.9	4.1
	80-160	2.8	1.5	2.0	1.9
	160-200	4.1	1.3	2.7	1.9
2m	0-10	6.0	8.4	13.7	4.8
	10-20	4.6	10.5	5.5	4.2
	20-40	4.5	4.4	4.7	3.8
	40-80	2.2	1.5	4.9	2.3
	80-160	4.9	0.3	1.6	1.1
	160-200	2.8	0.2	2.2	1.6
4m	0-10	5.0	13.3	13.2	12.6
	10-20	2.8	4.3	6.0	6.1
	20-40	3.0	3.7	4.9	2.3
	40-80	2.7	1.2	2.4	2.2
	80-160	2.1	1.3	1.8	2.1
	160-200	2.6	1.6	1.0	1.4
8m	0-10	4.5	4.7	10.9	6.4
	10-20	5.5	2.1	5.9	4.3
	20-40	4.6	3.0	4.7	4.6
	40-80	2.2	1.3	2.6	4.1
	80-160	2.4	0.4	3.8	1.6
	160-200	2.2	0.7	2.4	1.2
26m	0-10	22.4	5.4	29.7	5.5
	10-20	7.7	2.9	6.6	5.9
	20-40	2.5	7.0	5.1	2.4
	40-80	2.2	2.1	2.1	2.9
	80-160	1.8	3.2	1.7	2.6
	160-200	2.9	1.2	2.1	1.2

Table 15. Soil NO₃-N concentration for tall fescue control treatment, April 1998.

		NO ₃ -N concentration (mg kg ⁻¹)			
Position	Depths	Replications			
		1	2	3	4
Center	0-10	21.2	6.9	3.5	4.2
	10-20	6.0	9.9	3.1	3.5
	20-40	4.7	8.1	3.8	1.5
	40-80	1.3	2.9	1.9	1.6
	80-160	2.0	2.7	1.5	1.2
	160-200	1.2	2.6	1.6	1.5
0m	0-10	19.1	5.8	3.3	3.5
	10-20	8.9	2.6	2.5	6.5
	20-40	6.2	1.4	1.9	3.0
	40-80	1.9	1.4	1.2	1.0
	80-160	2.0	1.9	1.3	1.3
	160-200	2.5	3.8	1.4	1.6
1m	0-10	13.4	6.1	2.6	2.8
	10-20	6.0	4.3	3.3	3.4
	20-40	3.6	2.0	2.7	2.5
	40-80	1.9	1.0	2.5	2.2
	80-160	2.5	1.4	1.7	1.0
	160-200	1.6	1.9	1.4	1.7
2m	0-10	0.9	4.2	4.6	4.0
	10-20	2.8	2.3	3.1	3.2
	20-40	2.2	1.7	1.9	2.9
	40-80	1.7	1.2	2.0	2.8
	80-160	1.7	0.6	1.1	1.6
	160-200	0.9	0.4	1.7	0.5
4m	0-10	4.1	4.5	3.0	4.8
	10-20	2.2	2.7	2.8	6.6
	20-40	3.2	2.7	2.9	3.2
	40-80	2.1	2.0	2.1	2.2
	80-160	0.9	1.2	1.7	2.0
	160-200	1.4	0.5	1.5	1.1
8m	0-10	2.7	3.6	2.2	3.4
	10-20	2.2	2.7	3.1	2.4
	20-40	2.2	1.2	2.7	2.5
	40-80	1.2	0.6	1.9	1.2
	80-160	0.9	0.6	1.5	1.7
	160-200	0.9	1.0	1.5	1.5
26m	0-10	11.0	11.8	3.5	3.5
	10-20	5.3	4.3	4.8	4.4
	20-40	3.9	4.3	2.8	2.1
	40-80	2.1	1.8	2.2	3.2
	80-160	0.9	1.2	1.7	1.5
	160-200	0.9	0.8	1.4	0.8

Table 16. Soil NO₃-N concentration for fallow manure treatment, April 1998.

		NO ₃ -N concentration (mg kg ⁻¹)			
Position	Depths	Replications			
		1	2	3	4
Center	0-10	68.1	118.9	54.4	46.1
	10-20	88.4	99.1	53.6	79.8
	20-40	71.4	52.5	113.7	99.0
	40-80	101.3	39.9	130.5	102.2
	80-160	66.5	47.8	96.8	81.4
	160-200	66.5	51.7	68.0	66.1
0m	0-10	25.2	140.2	42.4	26.9
	10-20	10.6	17.8	35.2	9.2
	20-40	7.4	9.1	44.8	9.2
	40-80	6.6	18.6	47.2	6.0
	80-160	24.4	23.3	23.9	8.4
	160-200	14.7	21.7	18.3	11.6
1m	0-10	10.2	11.7	10.6	9.5
	10-20	6.1	16.5	4.9	12.2
	20-40	3.2	4.2	6.2	20.6
	40-80	3.0	21.5	13.0	19.8
	80-160	12.6	23.3	24.6	13.9
	160-200	18.4	12.1	18.0	12.5
2m	0-10	6.4	5.7	7.6	7.1
	10-20	7.2	6.0	5.2	7.6
	20-40	27.9	15.8	9.9	20.5
	40-80	28.4	25.9	25.5	29.8
	80-160	14.9	11.5	15.7	15.4
	160-200	16.4	15.3	18.8	8.8
4m	0-10	4.6	9.2	7.4	10.0
	10-20	5.9	7.1	7.9	7.9
	20-40	17.9	19.7	14.1	17.4
	40-80	21.2	17.8	35.5	16.3
	80-160	13.7	12.6	10.3	10.7
	160-200	13.0	9.0	9.0	14.2
8m	0-10	7.2	7.3	5.8	10.2
	10-20	4.1	5.1	18.7	9.3
	20-40	19.0	25.7	39.6	18.6
	40-80	27.1	18.8	19.0	19.4
	80-160	9.5	8.6	11.3	16.5
	160-200	8.8	10.2	11.5	9.6
26m	0-10	7.5	5.4	7.8	7.3
	10-20	12.6	8.3	11.3	9.0
	20-40	24.6	17.5	19.5	38.8
	40-80	23.8	24.3	28.4	38.3
	80-160	17.4	19.5	19.7	23.4
	160-200	13.9	9.4	7.5	10.1

Table 17. Soil NO₃-N concentration for fallow control treatment, April 1998.

Position	Depths	NO ₃ -N concentration (mg kg ⁻¹)			
		Replications			
		1	2	3	4
Center	0-10	5.8	9.6	6.2	10.9
	10-20	4.8	5.6	3.8	6.7
	20-40	8.3	22.5	9.3	8.1
	40-80	16.6	14.7	10.7	12.9
	80-160	11.1	10.7	8.7	11.1
	160-200	22.4	18.3	9.3	5.7
0m	0-10	7.3	8.0	4.2	12.3
	10-20	6.1	14.4	3.7	21.6
	20-40	7.4	38.8	14.1	24.6
	40-80	18.5	22.0	18.7	20.0
	80-160	12.4	15.6	11.2	11.1
	160-200	17.5	11.8	10.9	9.0
1m	0-10	5.5	7.3	9.0	2.8
	10-20	4.2	4.8	5.8	5.3
	20-40	27.8	6.1	41.3	21.4
	40-80	24.8	12.9	26.3	12.5
	80-160	10.9	13.8	12.8	23.6
	160-200	11.8	17.5	12.8	20.0
2m	0-10	5.3	6.3	5.5	7.9
	10-20	3.7	7.6	7.2	11.1
	20-40	7.6	10.9	31.0	32.1
	40-80	19.3	17.3	21.9	33.9
	80-160	15.6	14.2	10.9	11.1
	160-200	19.6	13.3	13.0	19.6
4m	0-10	6.5	7.0	6.2	5.1
	10-20	3.4	7.5	12.1	8.2
	20-40	7.7	5.3	33.0	29.6
	40-80	8.2	9.3	12.8	25.7
	80-160	10.5	12.5	14.6	15.1
	160-200	19.2	9.9	20.0	10.7
8m	0-10	7.2	15.1	11.0	8.6
	10-20	3.6	16.4	12.1	24.1
	20-40	10.1	15.3	24.9	31.7
	40-80	9.8	13.8	24.3	23.8
	80-160	9.4	5.3	14.0	16.4
	160-200	9.1	4.8	12.0	7.4
26m	0-10	9.7	7.9	9.6	21.8
	10-20	7.1	12.6	18.2	7.0
	20-40	20.9	32.5	31.3	12.4
	40-80	28.0	23.6	24.3	20.9
	80-160	24.6	19.9	7.7	14.9
	160-200	21.9	14.3	2.9	8.6

APPENDIX D

SOIL NITRATE-NITROGEN CONCENTRATIONS DISTRIBUTED
ALONG VFS AND FALLOW TREATMENTS, APRIL 1997

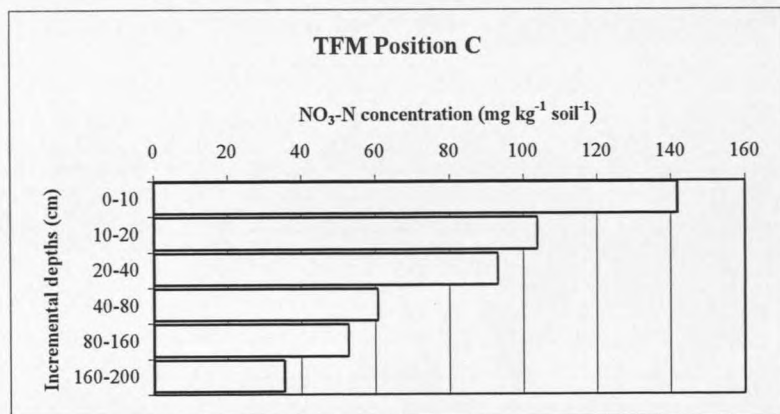
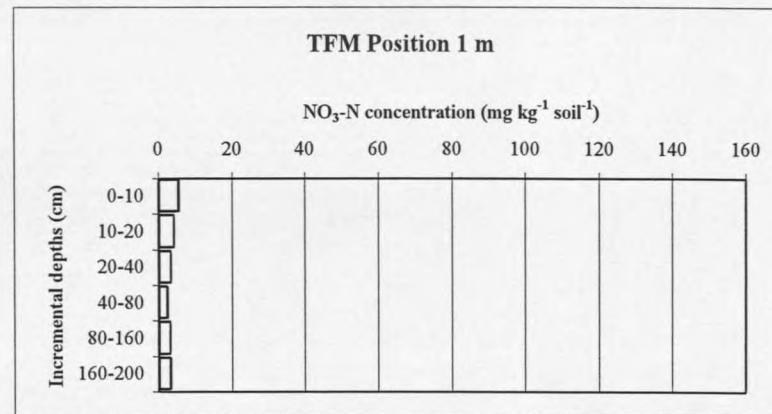
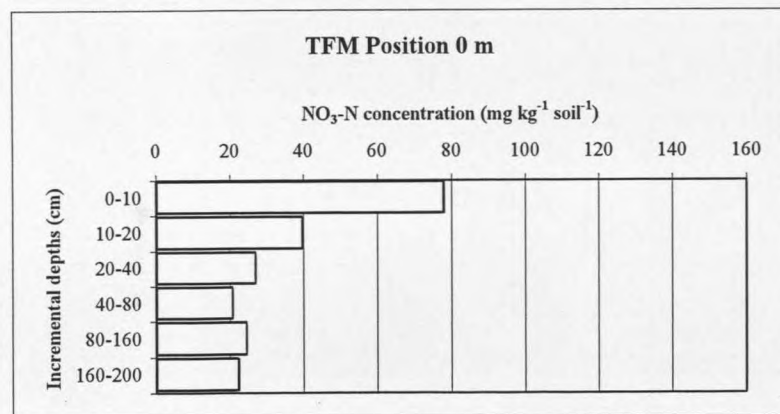
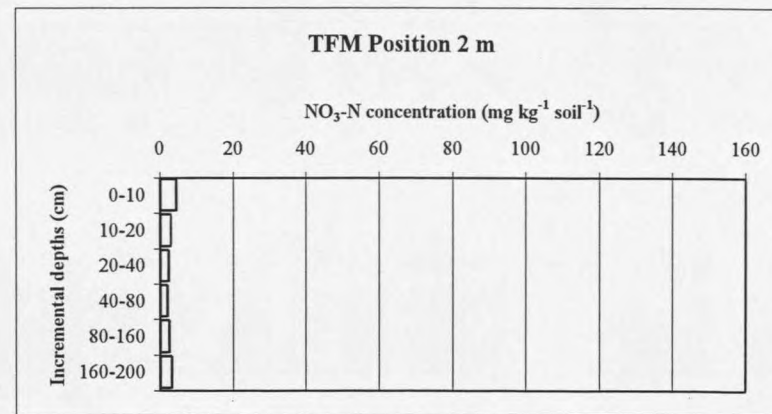


Figure 9. Nitrate-nitrogen concentrations in soil from positions center (C) through 2 m for tall fescue manure treatment (TFM), April 1997.

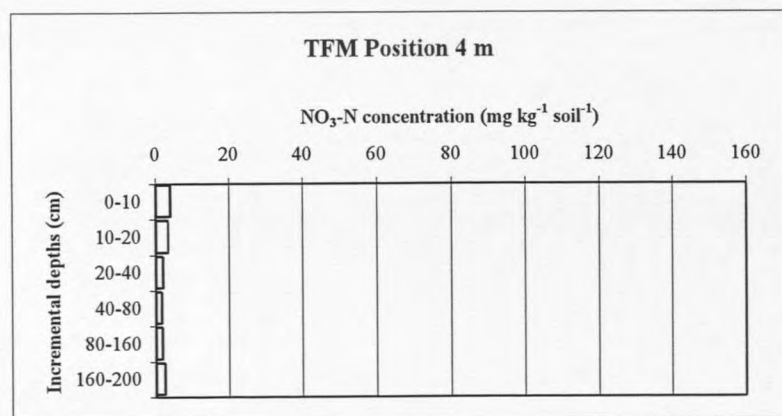
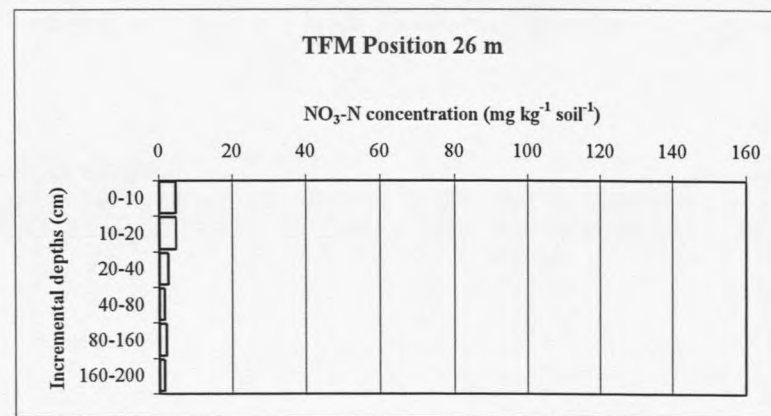
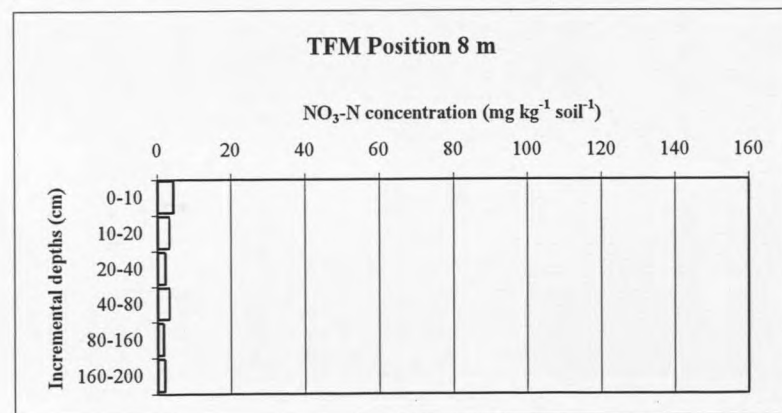


Figure 10. Nitrate-nitrogen concentrations in soil from positions 4 m through 26 m for tall fescue manure treatment (TFM), April 1997.

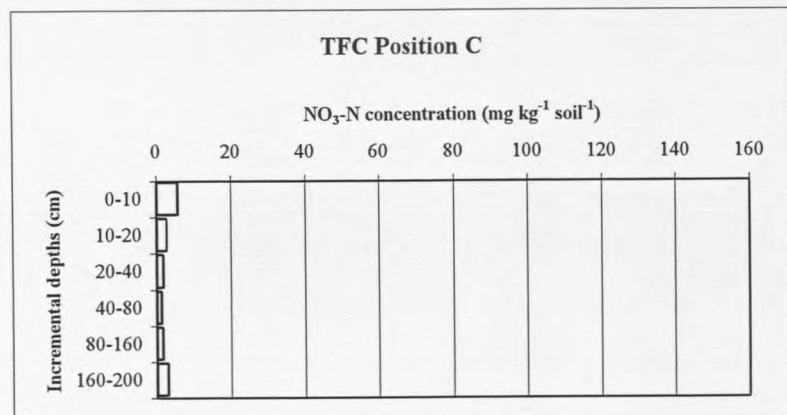
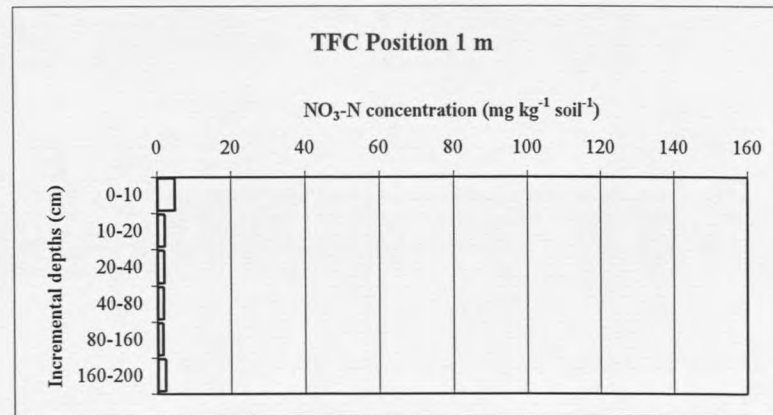
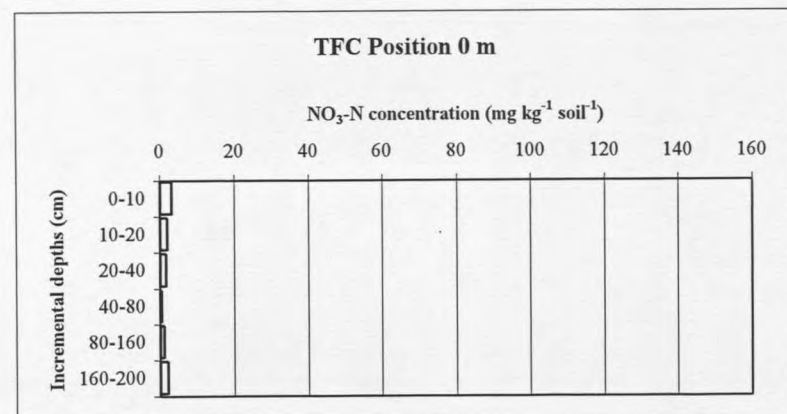
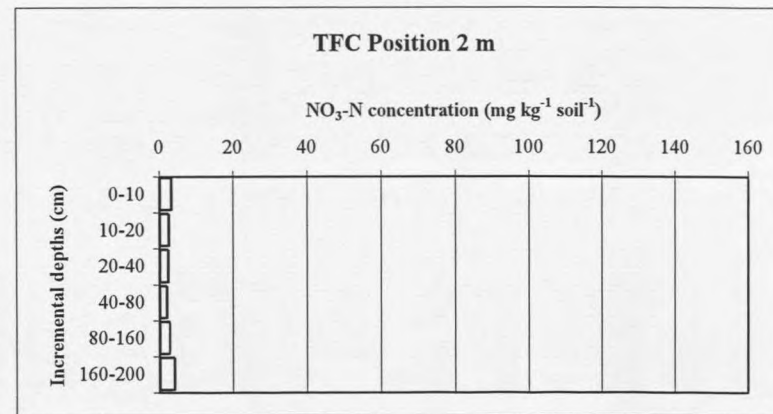


Figure 11. Nitrate-nitrogen concentrations in soil from positions center (C) through 2 m for tall fescue control treatment (TFC), April 1997.

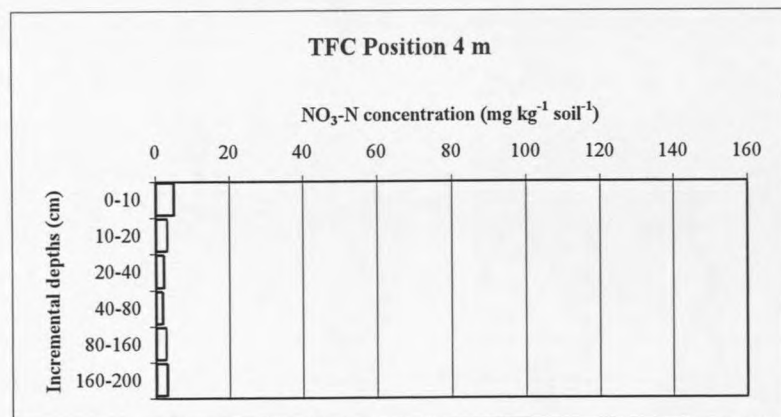
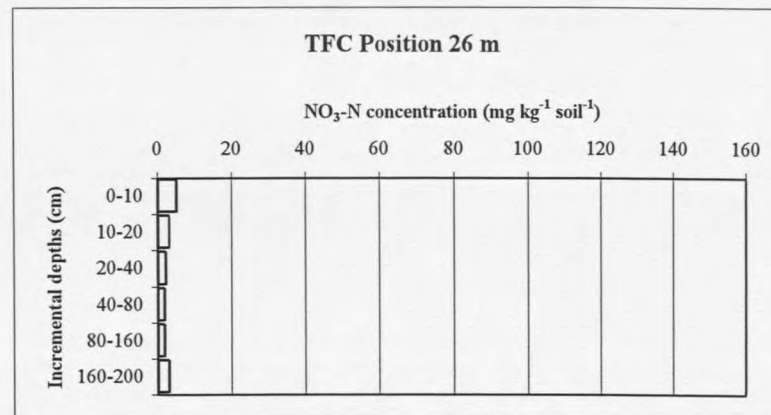
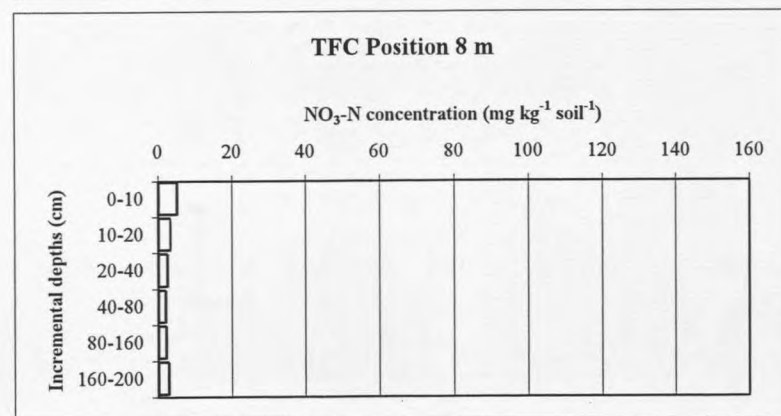


Figure 12. Nitrate-nitrogen concentrations in soil from positions 4 m through 26 m for tall fescue control treatment (TFC), April 1997.

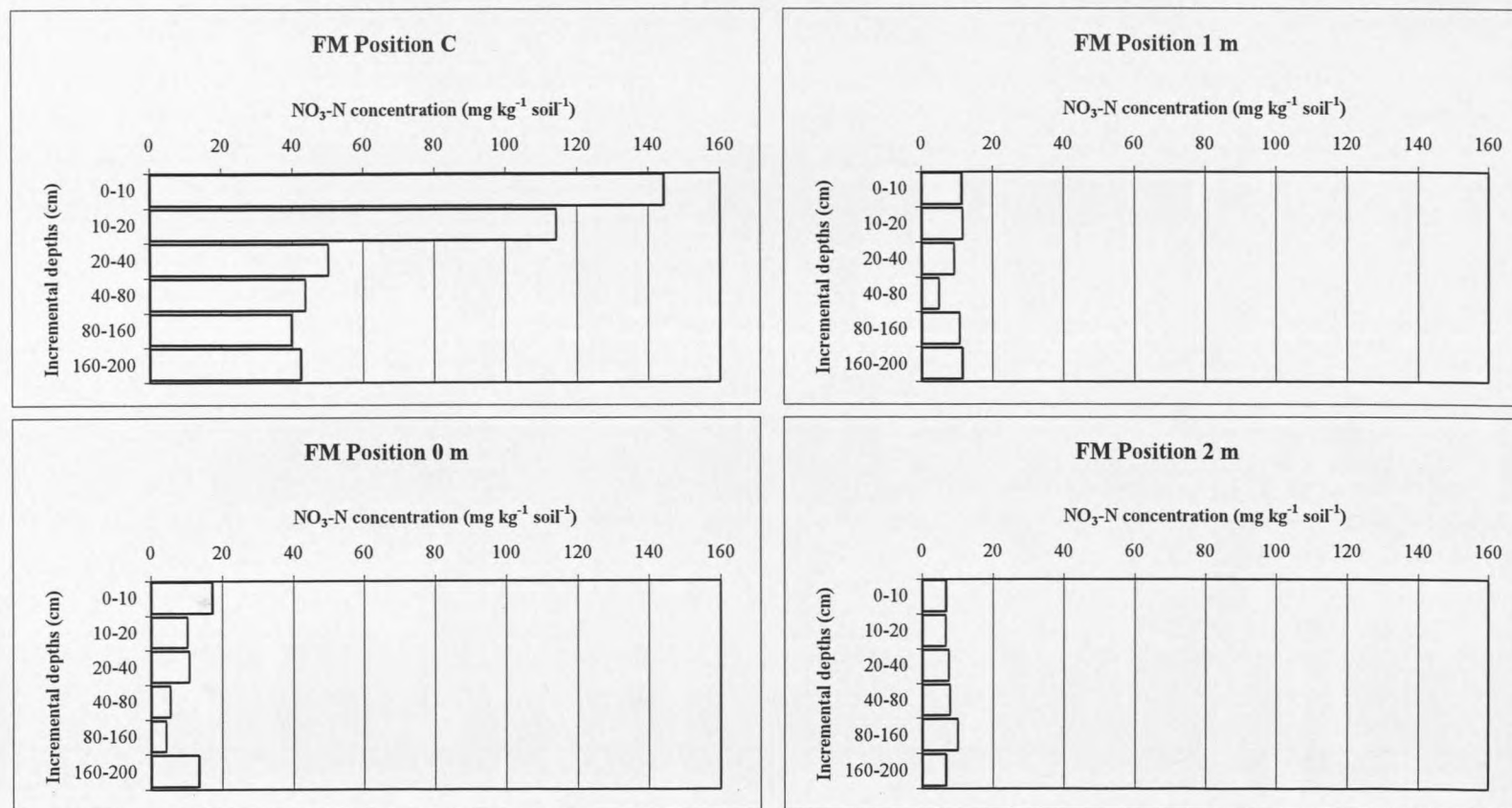


Figure 13. Nitrate-nitrogen concentrations in soil from positions center (C) through 2 m for fallow manure treatment (FM), April 1997.

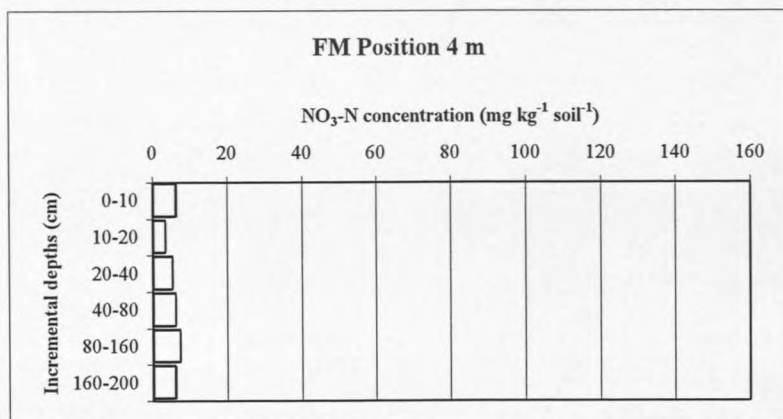
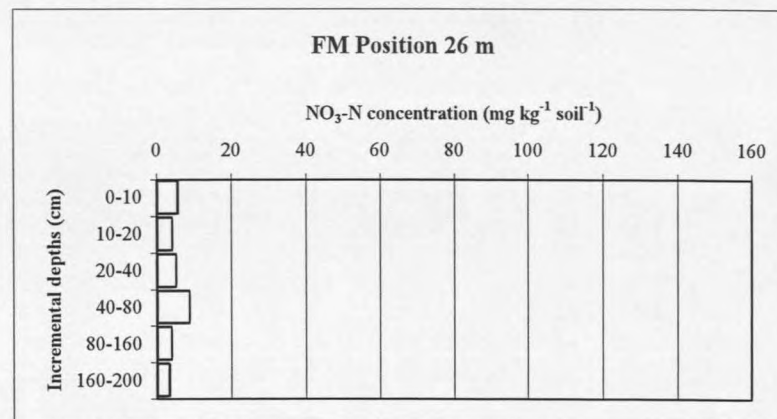
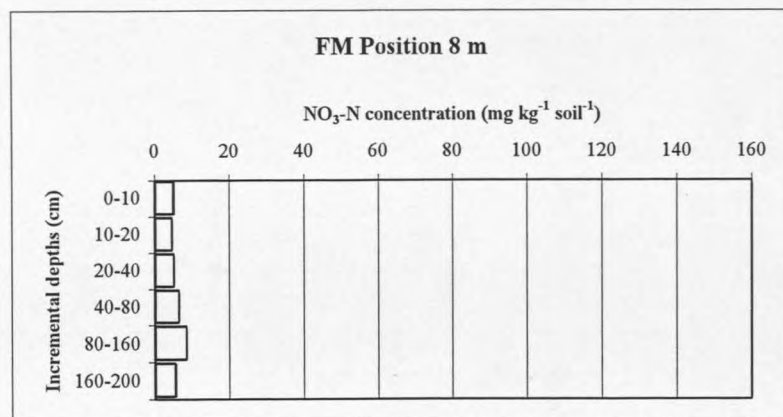


Figure 14. Nitrate-nitrogen concentrations in soil from positions 4 m through 26 m for fallow manure treatment (FM), April 1997.

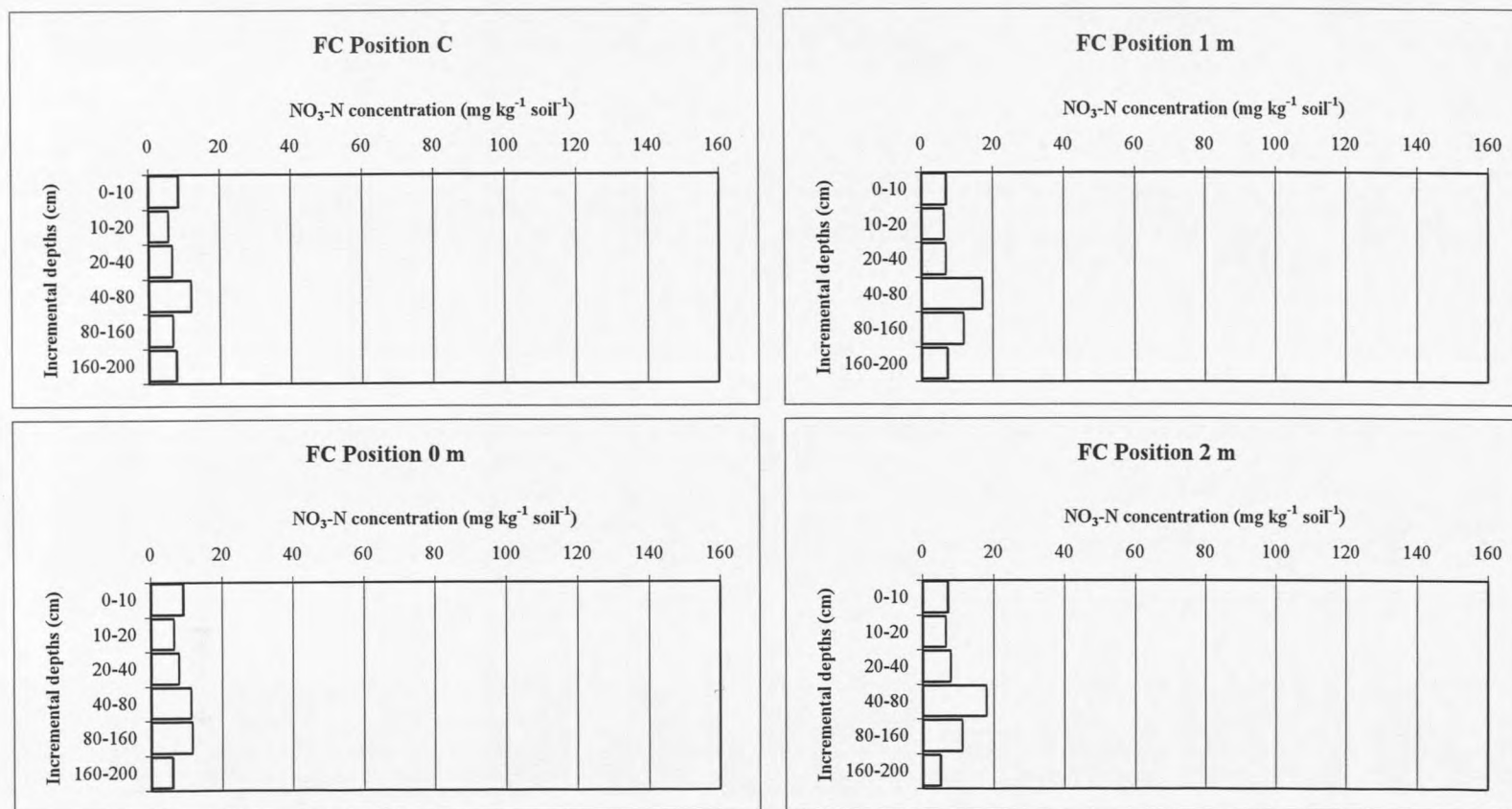


Figure 15. Nitrate-nitrogen concentrations in soil from positions center (C) through 2 m for fallow control treatment (FC), April 1997.

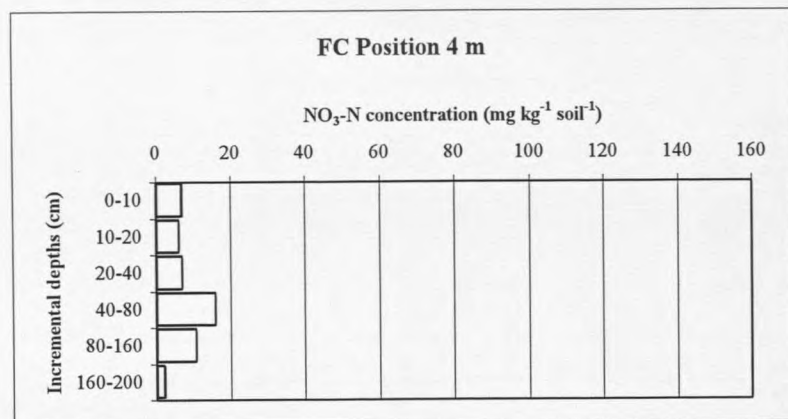
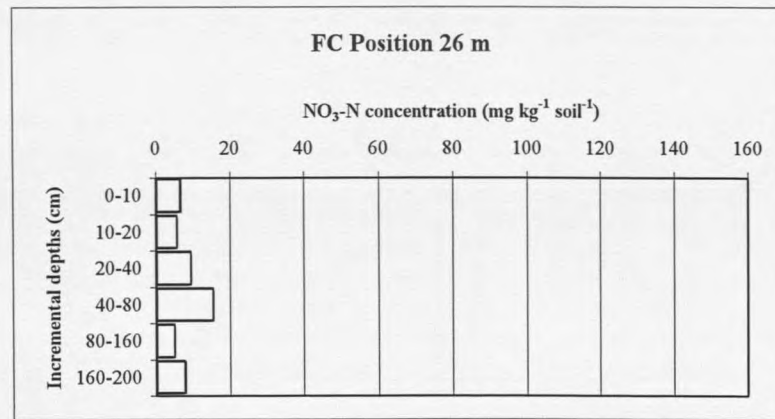
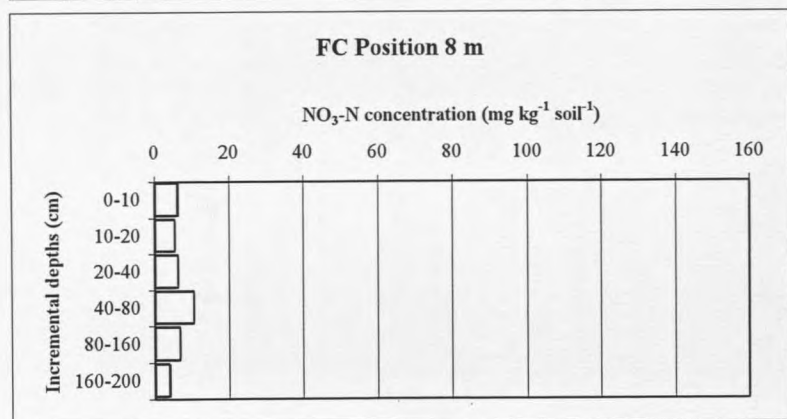


Figure 16. Nitrate-nitrogen concentrations in soil from positions 4 m through 26 m for fallow control treatment (FC), April 1997.

APPENDIX E

SOIL NITRATE-NITROGEN CONCENTRATIONS DISTRIBUTED
ALONG VFS AND FALLOW TREATMENTS, APRIL 1998

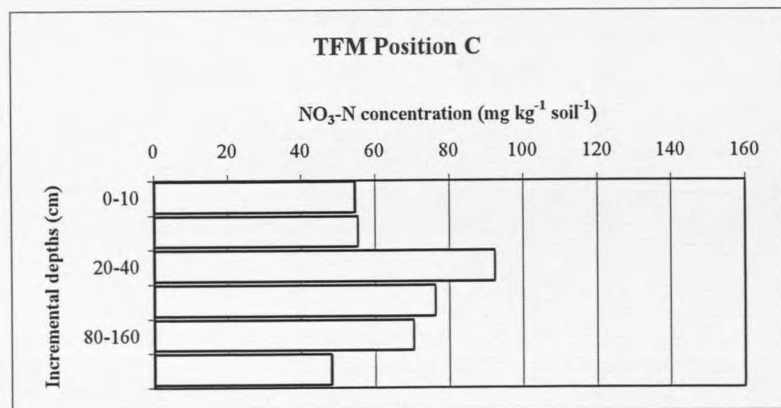
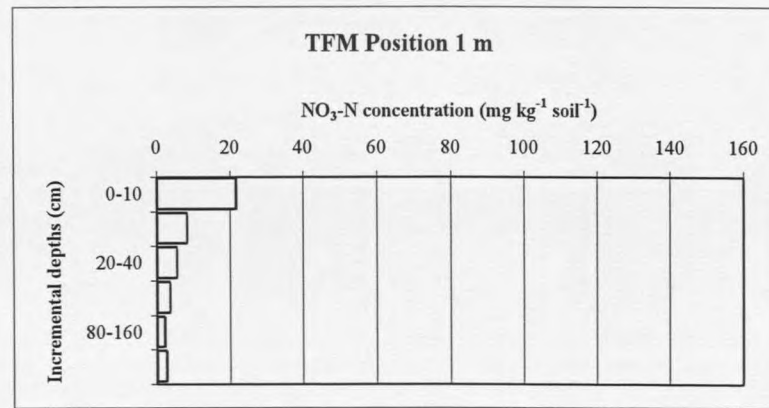
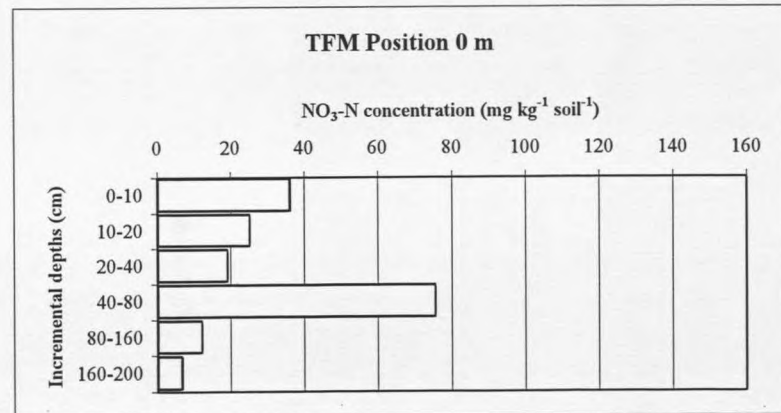
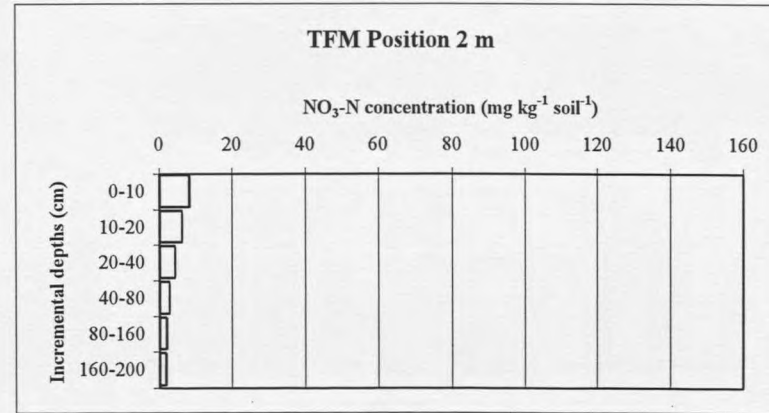


Figure 17. Nitrate-nitrogen concentrations in soil from positions center (C) through 2 m for tall fescue manure treatment (TFM), April 1998.

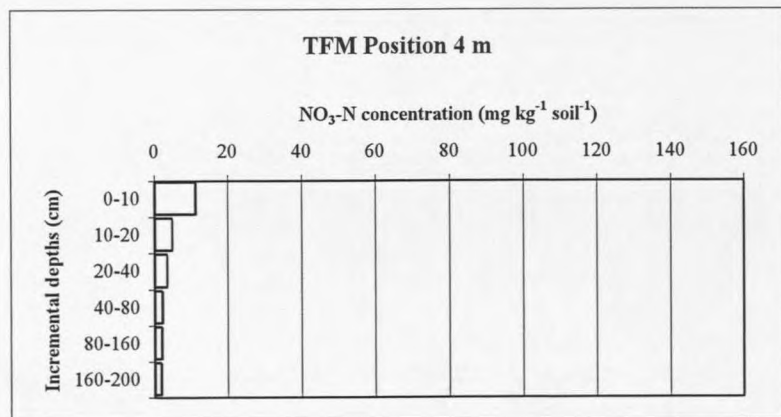
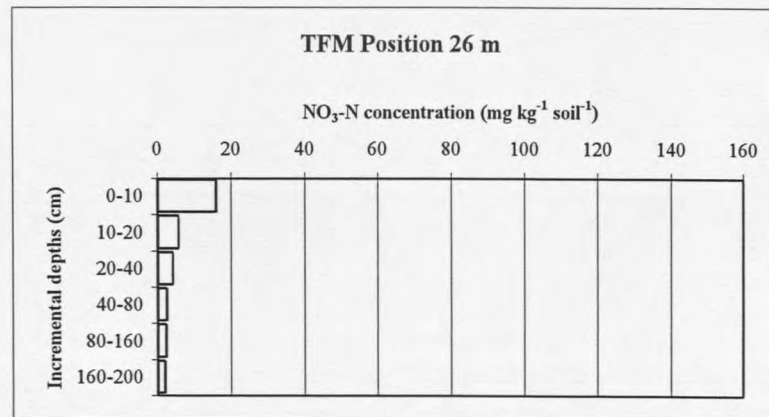
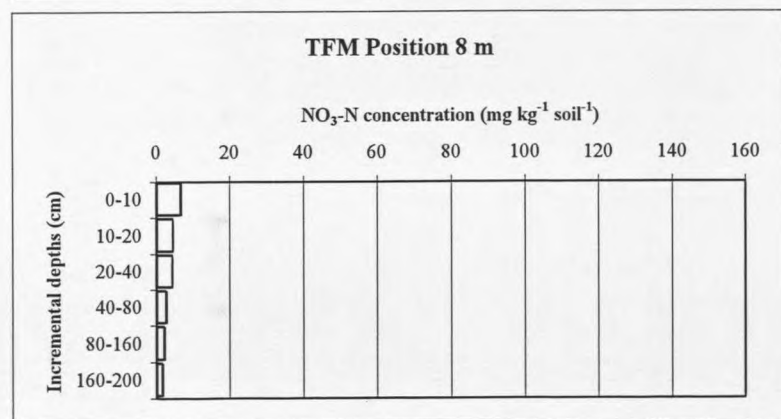


Figure 18. Nitrate-nitrogen concentrations in soil from positions 4 m through 26 m for tall fescue manure treatment (TFM), April 1998.

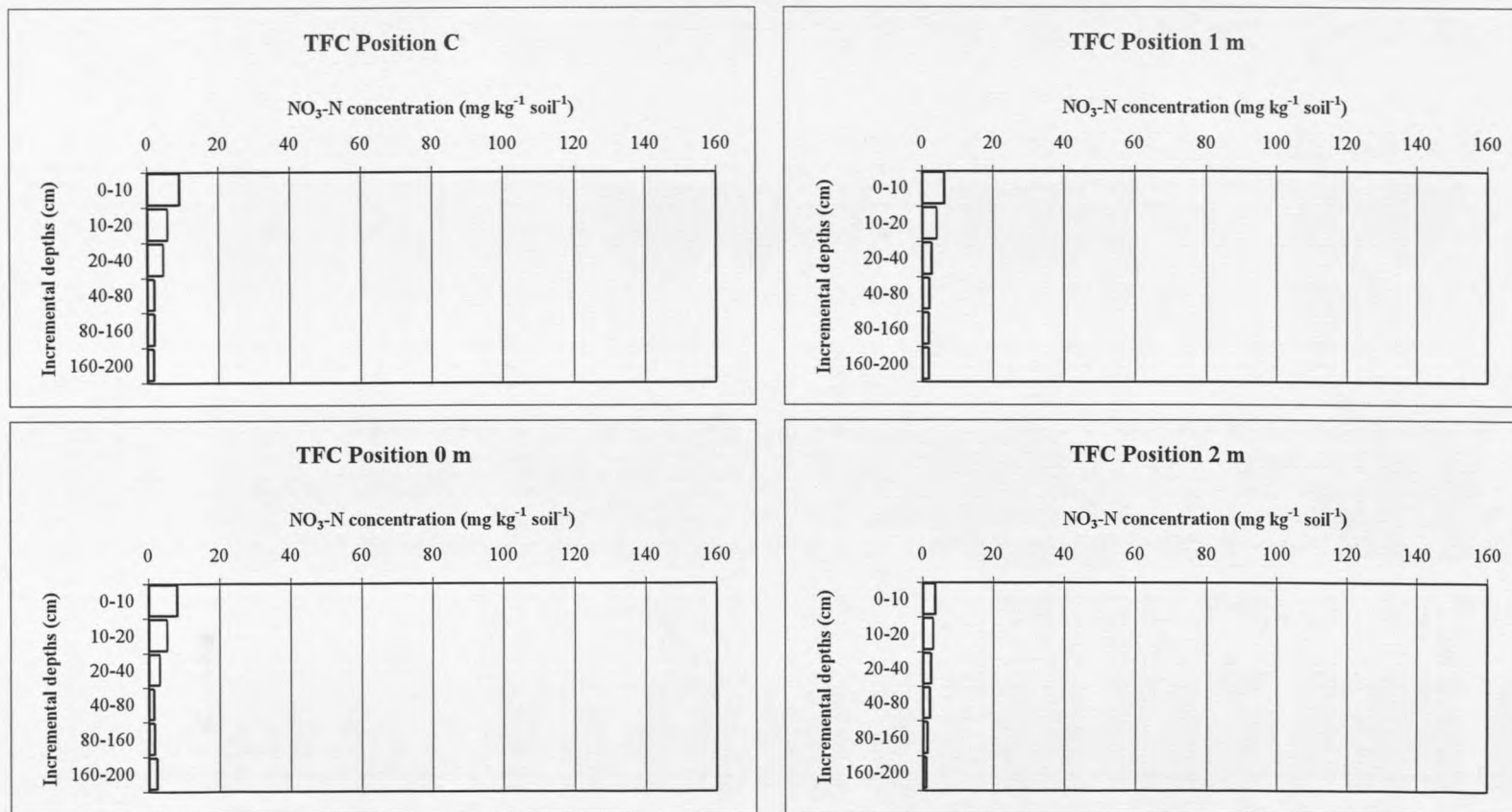


Figure 19. Nitrate-nitrogen concentrations in soil from positions center (C) through 2 m for tall fescue control treatment (TFC), April 1998.

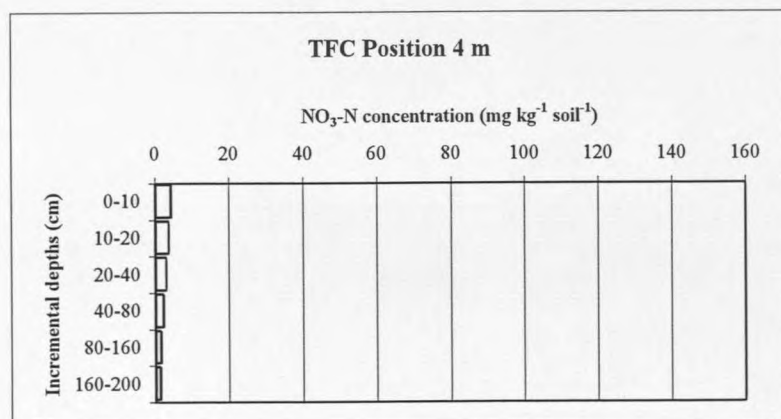
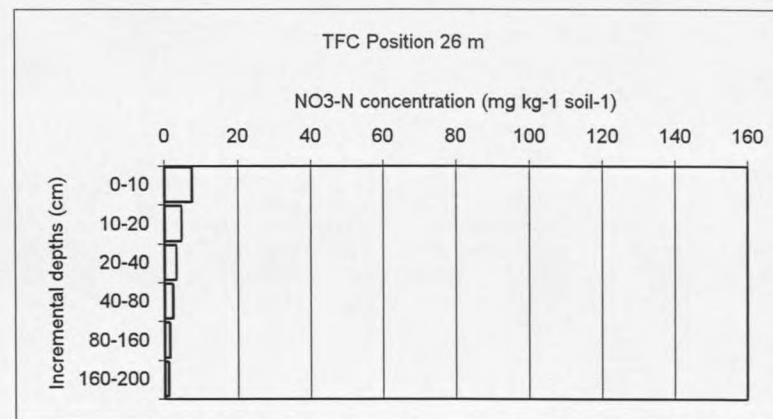
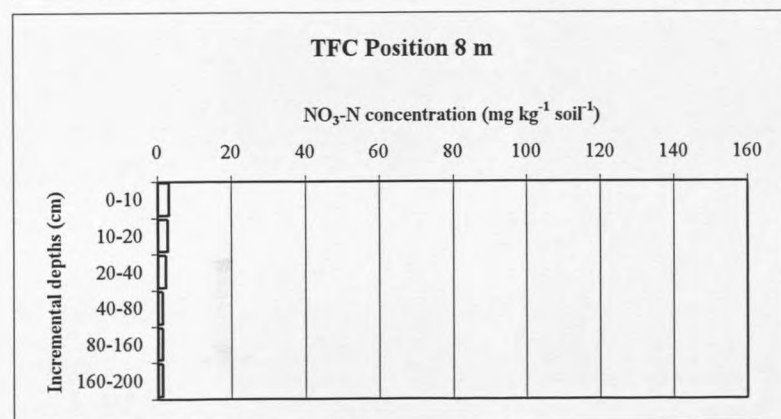


Figure 20. Nitrate-nitrogen concentrations in soil from positions 4 m through 26 m for tall fescue control treatment (TFC), April 1998.

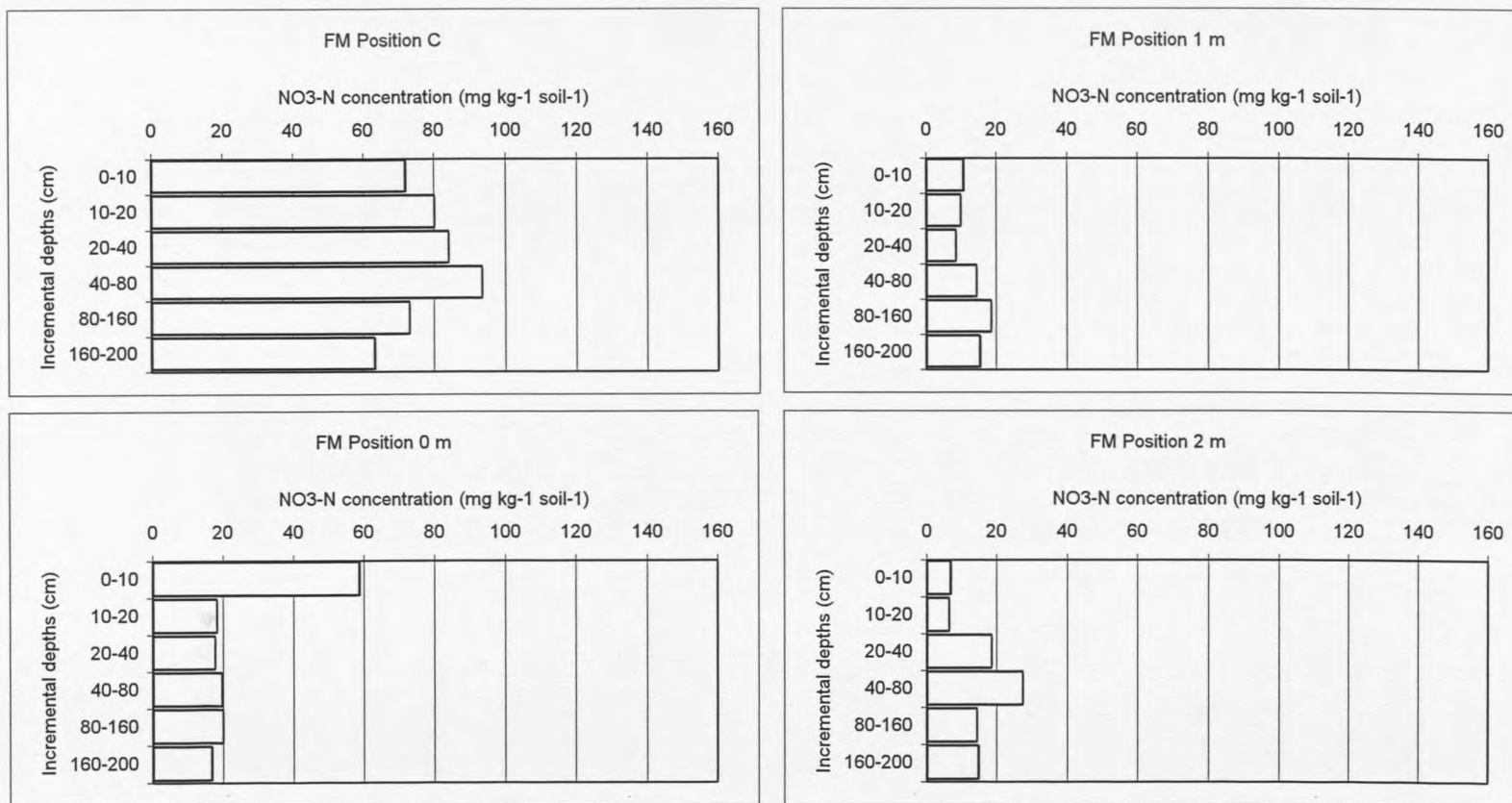


Figure 21. Nitrate-nitrogen concentrations in soil from positions center (C) through 2 m for fallow manure treatment (FM), April 1998.

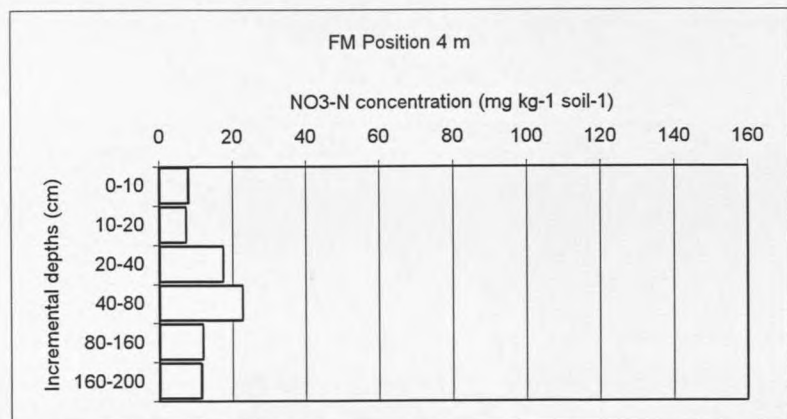
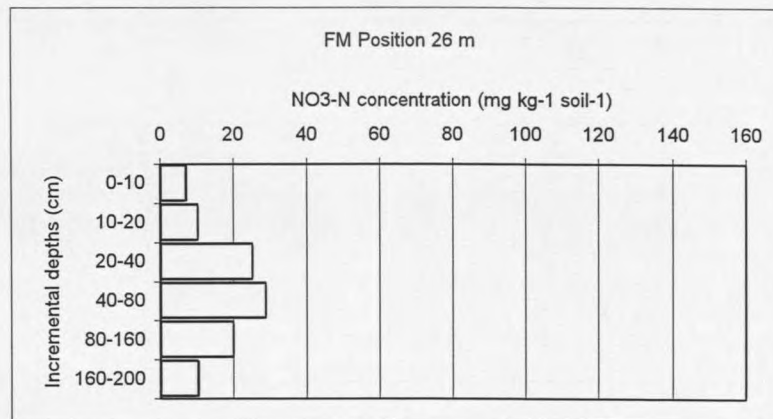
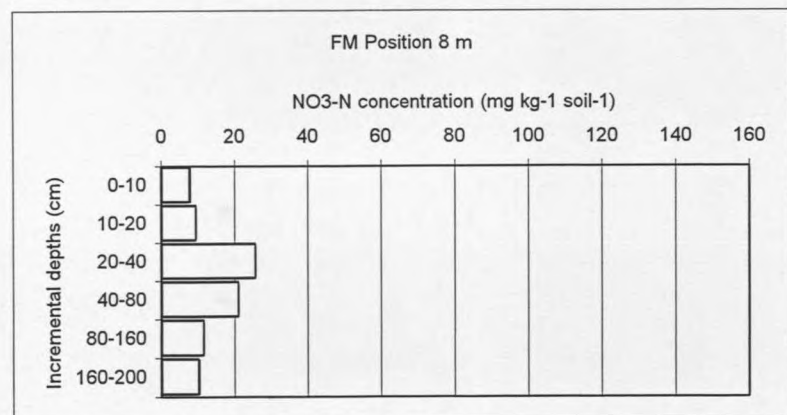


Figure 22. Nitrate-nitrogen concentrations in soil from positions 4 m through 26 m for fallow manure treatment (FM), April 1998.

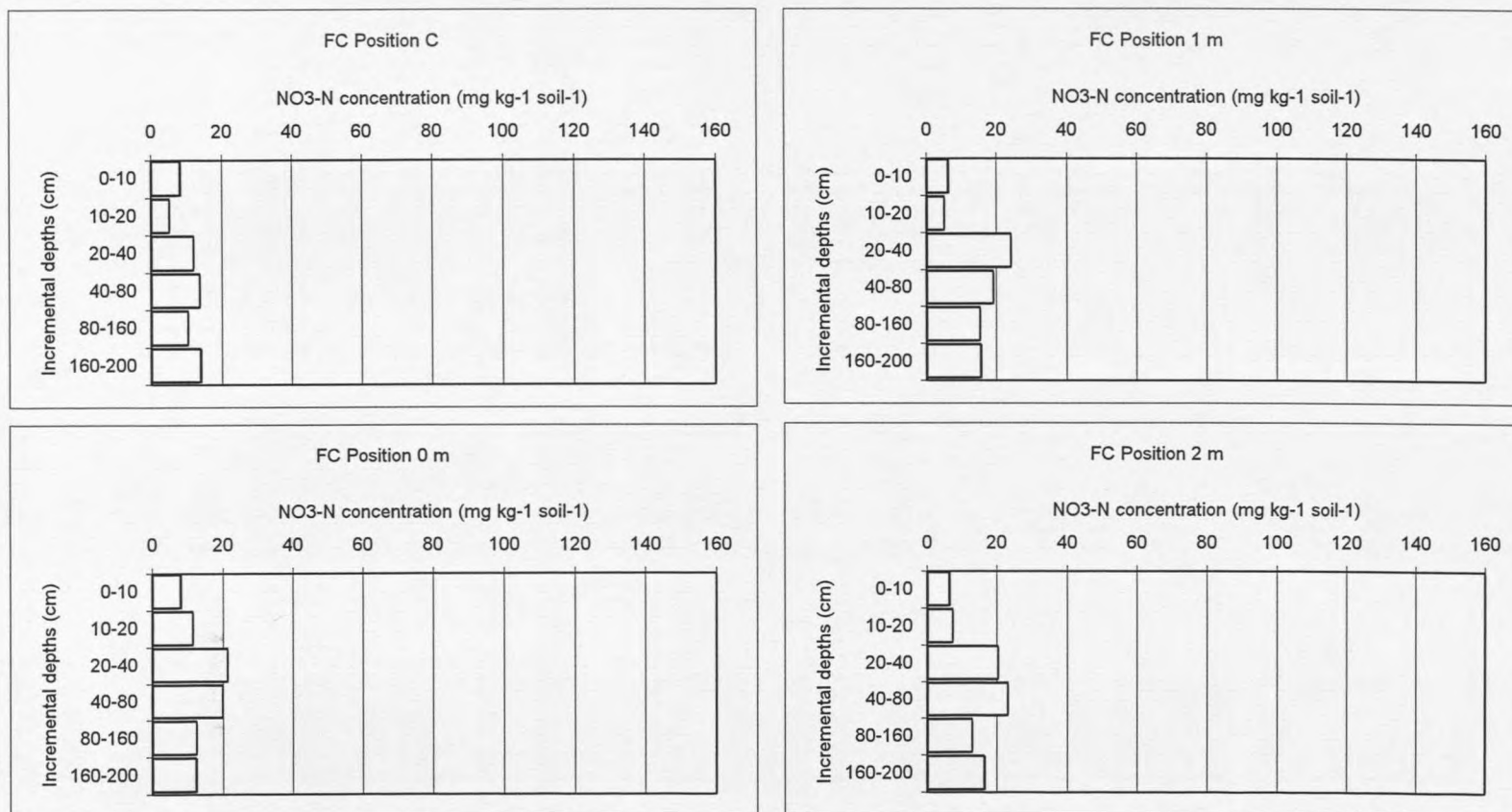


Figure 23. Nitrate-nitrogen concentrations in soil from positions center (C) through 2 m for fallow control treatment (FC), April 1998.

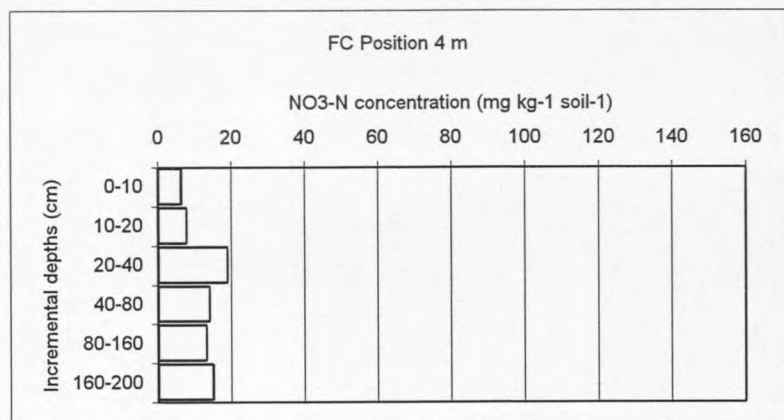
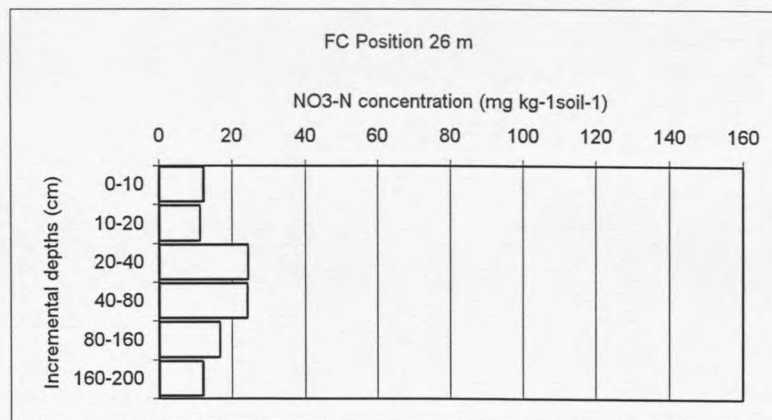
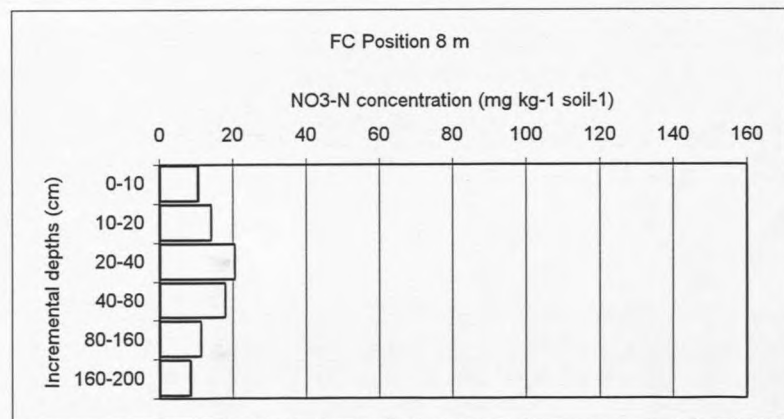


Figure 24. Nitrate-nitrogen concentrations in soil from positions 4 m through 26 m for fallow control treatment (FC), April 1998.

APPENDIX F

STATISTICAL ANALYSIS FOR SOIL NITRATE-NITROGEN CONCENTRATIONS,
APRIL 1997 AND 1998

Table 18. Summary of ANOVA of NO₃-N concentrations in soil for 1997 and 1998.

Source of variation	df	Nitrate concentration		Nitrate concentration	
		1997		1998	
		F value	Pr > F	F value	Pr > F
Block	3	0.77 ^{NS}	0.5069	0.82 ^{NS}	0.4857
VFS (Fallow, Tall fescue)	1	3.63 ^{/S}	0.0574	102.34 ^{/S}	0.0001
Treatment (Manure, control)	1	152.75 ^{/S}	0.0001	188.87 ^{/S}	0.0001
Position (C, 0, 1, 2, 4, 8, 26)	6	104.67 ^{/S}	0.0001	89.38 ^{/S}	0.0001
Depth (0-10, 10-20, 20-40, 40-80, 80-160, and 160-200)	5	10.57 ^{/S}	0.0001	8.02 ^{/S}	0.0001
VFS x Treatment	1	15.56 ^{/S}	0.0001	2.70 ^{NS}	0.1007
VFS x Position	6	4.42 ^{/S}	0.0002	1.54 ^{NS}	0.1637
VFS x Depth	5	1.27 ^{NS}	0.2752	4.20 ^{/S}	0.0009
Treatment x Position	6	106.29 ^{/S}	0.0001	95.04 ^{/S}	0.0001
Treatment x Depth	5	11.01 ^{/S}	0.0001	2.73 ^{/S}	0.0190
Position x Depth	30	5.97 ^{/S}	0.0001	1.39 ^{NS}	0.0832
VFS x Treatment x Position x Depth	101	2.44 ^{/S}	0.0001	1.56 ^{/S}	0.0012
R ²		0.80		0.77	
Coefficient of variation (%)		111.10		81.26	

^{NS} Not significant at $P=0.05$ ^{/S} Significant at $P=0.05$.

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