Effectiveness of grass species for nitrogen recovery from dairy waste
by Valerie Ellen Oksendahl

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Agronomy
Montana State University

Abstract:
Runoff from manure below livestock operations has the potential to increase levels of nitrogen in surface and ground waters. This research determined the relative effectiveness of four grass species: Dactylis glomerata L., Festuca arundinacea Schreb., Bromus biebersteinii L. and Agropyron elongatum L. in reducing the movement of nitrate and ammonium nitrogen from the plant root zone through a vegetative filter system (VFS).

Dairy manure was applied above the replicated vegetative strips and fallow treatments on a 5% slope. Two fallow treatments were maintained, one serving as a control and one as a manure-treated site. Irrigation water was applied two times, each simulating a 24-hour, 25-year storm event of 5.6 cm. Forage was clipped and analyzed for nitrogen concentration and forage yield. Soil was analyzed for both nitrate and ammonium every 6.1 meters throughout the 60-meter VFS.

Total nitrogen (N) removal differed significantly among grass species with the highest yields, N concentration, and total N removal occurring adjacent to the manure application site at 0.0 meters. Low yields and N levels beyond this area suggest little, if any N movement. Total N harvested from this area accounted for 109.1 and 136.4 kg ha-1 for F. arundinacea Schreb. and A. elongatum L., respectively, followed by 95.1 and 109.0 kg ha-1 for D. glomerata L. and B. biebersteinii L., respectively.

Soil nitrate levels for the vegetative treatments were low throughout the soil profile and all distances except on the surface (0 to 30 cm) adjacent to the manure-treated site. Nitrate levels on the fallow manure treatment were four times higher compared to all the vegetative treatment plots at 0.0 meters. Ammonium N did not move further than 6.1 meters downslope of the manure application site during the two storm events.

Following one season of evaluation, all vegetative treatments were effective in their ability to remove N from the soil profile by accumulation of N in grass tissue.

Based on the ratio of N recoveryrresidual soil N levels, D. glomerata L. and B. biebersteinii L. were superior to F. arundinacea Schreb. and A. elongatum L. in absorptive capabilities for removing N from the soil profile. Rapidly growing forage grasses seeded downslope from an excessively loaded manure site are capable of utilizing excess N from N-rich manure runoff. With careful management, VFS can reduce N transport downslope and through the soil profile for conditions common in Montana.

EFFECTIVENESS OF GRASS SPECIES FOR NITROGEN

RECOVERY FROM DAIRY WASTE

by

Valerie Ellen Oksendahl

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APPROVAL

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Valerie Ellen Oksendahl

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

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Date May 7, 1997

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CHAPTER 1

INTRODUCTION

Runoff from livestock feedlots and dairy operations has long been recognized as a potential source of pollution. Direct discharge of manures or animal wastewater from the feeding and surrounding areas may carry high concentrations of solids, nutrients, and pathogens into surface waters, and the leachate may carry pollutants into the groundwater (Butchbaker et al., 1972). Discharge carrying these pollutants must be contained to prevent runoff from a 25-year, 24-hour rainfall from entering a state's waters (MDHES, 1991; Krider et al., 1992).

Disposal of manure and surface runoff have the potential to increase levels of nutrients and fecal coliform in runoff water and thus increase the quantity of agriculturally derived pollution. Montana Public Law and the Montana Water Quality Act state "It is unlawful to cause pollution, as defined in 75-5-103, of any state waters, or to place or cause to be placed any wastes in a location where they are likely to cause pollution to any state waters;" (MDHES, 1991).

As cited in Doyle et al. (1977), Section 208 of the 1972 Amendment to the Federal Water Pollution Control Act, Public Law 100-4 gives each state the responsibility to identify point and non-point sources of pollution and develop plans for controlling these sources. Individual landowner compliance requires a National Pollution Discharge Elimination System (NPDES) permit specifying allowable discharges along with a compliance plan (MDHES, 1991).

Young et al. (1980) examined available technology and found that the best method of controlling runoff was to install holding ponds or lagoons. As lagoons were put into widespread use under existing state and federal feedlot pollution-control regulations, certain problems became apparent: (i) holding ponds were expensive even with cost-sharing and tax incentives; (ii) expensive pumping equipment was often necessary; (iii) enforcement agencies require holding ponds even if the probability of pollutant discharge is remote; (iv) major storms could fill lagoons beyond capacity; and (v) odors could become a nuisance for both operator and neighbors (Young et al., 1980; Pinkowski et al., 1985).

Alternative pollution-control measures or land management techniques that reduce or prevent discharge of pollutants into surface waters are needed. This information is important to the Montana livestock industry for use as an alternative or a supportive practice to costly structures currently being required by various federal and state agencies. One promising alternative to lagoons is the use of vegetative filter systems.

Vegetative filter systems (VFS) are widely used in many areas throughout the United States for manure and slurry pollution control. Recent research in Virginia, Arkansas, and Georgia has shown that VFS are an effective way to reduce pollution from animal wastes (Dillaha et al., 1989; Chaubey et al., 1994; Hubbard et al., 1994). Because they are not commonly used in Montana, the design of VFS remains an intuitive process, and insufficient information is available about species and optimal strip width dimensions for northern conditions.

Vegetative filter systems function by promoting filtration, deposition, infiltration, absorption, adsorption, decomposition, and nitrogen volatilization (Dickey and Vanderholm, 1981). Vegetative filter systems reduce sediments and nutrients in runoff by reducing the velocity of surface runoff which decreases particulate transport capacity. The thick, upright vegetation physically filters solid particles that are carried in the runoff. Filtration is probably more significant for the larger particles, aggregates, and manure particles, while absorption is more significant with respect to the removal of soluble pollutants (Dillaha et al., 1986). The soluble nutrients which continue moving through the VFS if flow velocities are high, can be directly absorbed by the plant leaves and stems. Resistance from the vegetation slows surface runoff, facilitating infiltration. Upon infiltration, the pollutants are entrapped by chemical, physical, and biological processes, and to a large extent are transformed into plant biomass or organic and inorganic components of the soil. These processes combine to transform a potential pollutant into vegetation biomass that can be used for forage, fiber, or mulch material (Lemunyon, 1991).

Several problems on the utility and efficiency of VFS remain unsolved. These problems remain site specific, but include: vegetation efficiency, nutrient saturation level, temporary sink adequacy, and optimal width for nutrient reduction (Koviacic et al., 1990). The objective of this research was to design and evaluate VFS for lower rainfall areas in northern latitudes. Criteria of VFS performance effectiveness in Montana were determined by testing soils for leaching and runoff of nitrogen (N) and leaf analyses of grass for total N uptake. State and federal regulations were followed for determining performance criteria.

CHAPTER 2

LITERATURE REVIEW

Vegetative filter systems are areas of crop, grass or riparian vegetation used for removing sediment, organic matter, nutrients and other pollutants from runoff and waste water. They have been shown to be an effective best management practice for the control of many point and non-point source pollutants.

There has been an increased effort to provide information on potential sinks for both point and non-point source pollution within agricultural watersheds. Examples of such sinks include, constructed wetlands, sediment detention basins, grass filter systems, and riparian buffer systems (Hubbard et al., 1994).

Chaubey et al. (1994) and Coyne et al. (1995a) have demonstrated significant removal of solids, N, and phosphorus (P) by VFS from non-point source pollutant sites such as agricultural soils treated with animal waste. Woodard and Rock (1991) and Schellinger and Clausen (1992) demonstrated the effectiveness of VFS for removing sediment and sediment-bound nutrients from point source pollutant sites (construction site runoff and animal waste facilities). Many researchers recommend design criteria for the use of vegetated filter systems. These design criteria include VFS dimensions, species selection, and some measurement of impacts on water quality.

Vegetative Filter System Dimensions

Most VFS design management and research has emphasized sediment entrapment, with only recent attention to the soluble components of the runoff. Current research is now focusing on adequate filter strip width to accommodate the absorption and filtering capabilities of specific VFS vegetation.

Buffer strips of crops are a promising alternative method for controlling pollution from feedlot runoff. Young et al. (1980) seeded corn (*Zea mays* L.), orchardgrass (*Dactylis glomerata* L.), and a mixture of sudangrass (*Sorghum sudanense* L.) and sorghum (*S. vulagre* L.) downslope from a 310-head cattle feedlot. All of the cropping treatments reduced runoff volume and total suspended solids (TSS) transported from the feedlot by 67 and 79%, respectively. Total nitrogen (TN) and total phosphorus (TP) in runoff were reduced by an average of 84 and 83%, respectively. The ammonium-N (NH_4-N) and soluble orthophosphate (PO_4-P) were similarly reduced, however average nitrate-N (NO_3-N) in the runoff increased by about 9%, because NO_3-N was released from the sorghum and sudangrass plots. Buffer strip lengths of at least 36 meters appeared to be sufficient to reduce runoff of nutrients and microorganisms to acceptable levels.

In 1981, Dickey and Vanderholm (1981) reported that VFS installed below feedlots in Illinois retained over 90% of the nutrients, solids, and oxygen-demanding materials from feedlot runoff. The degree of removal was dependent upon the length and type of flow through the system. The VFS studied were 61 or 91 meters (m) in length with varying slopes, rainfall, and amounts of effluent discharged onto the strips. Discharge from the 91 m vegetated filters occurred only during three rainfall events totaling 17.4 centimeters. The researchers speculated that the flow length required to meet state and federal standards would need to be two to four times longer than those evaluated. The VFS were more effective than graded terraces or waterways and required shorter flow lengths for a similar degree of treatment. The overall impact of the VFS filtering capacity appeared to be beneficial, but these researchers suggested further evaluation was required before wide recommendation and use.

Doyle et al. (1977) applied dairy manure to tall fescue (*Festuca arundinacea* Schreb.) plots with a 10% slope. Soluble nutrient concentrations were measured 0.5, 1.5, and 4.0 m downslope in the vegetative filter system. Soluble P and $N0_3$-N were reduced 62% and 68%, respectively in the 4-meter strip. Ammonium-N concentrations increased with increasing filter length, presumably from the release of NH_4-N from decomposing organic nitrogen. The 4-meter VFS were determined to be effective in reducing levels of NO_3-N, P, and potassium (K) in runoff from dairy manure-loaded areas by as much as 94, 100, and 93%, respectively.

Chaubey et al. (1994) evaluated tall fescue plots of varying lengths (0, 3, 5, 9, 15, and 21 m) for sediment and nutrient capture. The plots were treated with liquid swine manure equivalent to 200 kg N ha^{-1}. The 21-meter VFS were effective in removing total kjeldahl nitrogen (TKN), ammonia (NH_3-N), PO_4-P, TP, and TSS from incoming runoff by 87, 99, 94 and 92%, respectively.

Srivastava et al. (1994) applied simulated rainfall (50 mm $hour^{-1}$ until runoff occurred for 1 hour) to assess the influence of poultry treated areas to VFS area ratios. Litter-treated lengths of 6.1, 12.2, and 18.3 m had minimal effect on concentrations of NH_3-N, TKN, PO_4-P, and TP entering the VFS. Concentrations of NO_3-N, NH_3-N, PO_4-P, and TP decreased with increasing VFS length. These findings suggest the length of the pollutant contributing area is not an important factor in estimating incoming pollutant concentrations.

Bingham et al. (1980) examined the effect of grass buffer length in reducing pollutant concentration in rainfall runoff from land application areas. He suggested the amount of pollutant transport and volume of runoff from a waste application area will increase as waste area size increases. Caged-layer poultry manure was applied and evaluated for TKN at various distances downslope. A buffer area to waste area length ratio of 1.0, for 12-meter lengths, reduced TKN levels measured in runoff to a level similar to plots receiving no manure. These results contrast with those of Doyle et al. (1977), Young et al. (1980), Dickey and Vanderholm (1981), Chaubey et al. (1994), and

Srivastava et al. (1994). However, conclusions from this study were limited to waste area lengths ranging from 8.7 to 13.0 meters.

Species Selection

The reduction of runoff concentrations in a VFS depends primarily upon infiltration or filtering of pollutants, and dilution by rainfall. Proper management of the filter assures sediment trapping and nutrient infiltration within the strip. The ultimate fate of nutrient and pollutant accumulation in various types of vegetation in the VFS has just recently been given adequate consideration. The type of species selected for the site has a significant effect on trapping and infiltration capabilities.

Koviacic et al. (1990) demonstrated the superiority of a perennial grass buffer in comparison to a corn and soybean (*Glycine max* L.) rotation for reducing losses from an upland system. Dramatic reductions of NO_3-N occurred after passing through 39 m of a reed canarygrass (*Phlaris arundinaceus* L.) buffer. Nitrate concentration ranged from 0.6 to 1.6 mg liter^{-1} for grass and crop treatment, respectively. Mean TKN in the runoff from the grass treatment (161 mg liter^{-1}) was significantly higher ($p < 0.05$) than that found in the crop area (3.9 mg liter^{-1}). The mean volume of runoff ranged from 106 to 2337 ml for grass and crop treatment, respectively. The researchers suggested higher concentrations of NO_3-N at the crop site coupled with these runoff volumes could produce large N loading below this site.

The results of two studies in Rhode Island (Lemunyon, 1991; Groffman et al., 1991) indicate that specific grasses have different rates of N uptake and therefore, abilities to prevent NO_3-N leaching to groundwater. In a two-year study of ten grass species planted as VFS, Lemunyon (1991) found that orchardgrass, tall fescue, and sweet vernalgrass (*Anthoxanthum odoratum* L.) were superior to big bluestem (*Andropogon gerardii* L.) and switchgrass (*Panicum virgatum* L.) in preventing N percolating below the root zone. The mean N recovery of harvested material for the two-year period ranged from 48 kg ha^{-1} for switchgrass to 136 kg ha^{-1} for smooth bromegrass (*Bromus inermis* L.). Cool season species such as bromegrass, orchardgrass, Kentucky bluegrass (*Poa pratensis* L.), tall fescue, and reed canarygrass absorbed over 100 kg N ha^{-1} during the two harvest seasons. Generally, cool season species recovered significantly more N in the plant biomass than warm season species. Plant removal by harvest, N leached, and microbial denitrification or immobilization accounted for the majority of N transformation. Denitrification rates were estimated using data from a microbial study by Groffman et al. (1991). Volatilization rates were assumed to be 5 percent. Immobilization and soil fixation rates were not calculated, although they were presumed to represent a major portion of N not accounted for in the transformation process (Lemunyon, 1991).

Denitrification can be a desirable NO_3-N mechanism since excess NO_3-N is removed from the vegetative filter. Groffman et al. (1991) measured denitrification and microbial immobilization in VFS of tall fescue, reed canarygrass, and two riparian forested sites. One forest site was poorly drained, and the other site had favorable infiltration characteristics. The grass filters had higher denitrification rates, with manure-amended sites having even greater denitrification capabilities. The soils from the grass sites exhibited consistently higher denitrification activities in response to added NO_3-N than soils from the forest sites for both aerobic and anaerobic conditions. These data suggest that the microbial denitrification populations were greater or more active in the grass than in the forest plots. Tall fescue had higher denitrification rates than reed canarygrass plots in aerobic sites (25 and 14%, respectively). Lower levels of NO_3-N in soil and soil water percolate were measured in the tall fescue as compared to the reed canarygrass. Denitrification N-removal efficiencies were as high as 50% when N additions of 30 kg N ha^{-1} were amended with glucose. The data suggests that runoff containing high levels of available carbon (C) from feedlot or manured field runoff may be more amenable to treatment by VFS than runoff that is low in available carbon. Denitrification occurring in VFS when free drainage is restricted allows removal of some of the NO_3-N that infiltrates the grass filters before it reaches groundwater.

Proper grass management is critical to VFS performance. In VFS surveys conducted in West Virginia by Dillaha et al. (1986), the most common grass species was tall fescue. The survey included 20 landowners implementing VFS as best management practices (BMP). The VFS that were mowed, with residue harvested and removed two or three times per year, promoted thick vegetation with optimum pollutant-removal capabilities. Most respondents in the survey felt that species selection, mowing, and proper fertilization were the principal requirements to ensure long-term VFS effectiveness.

In a document prepared for the Environmental Protection Agency, Newberry (1992) identified conditions under which VFS can be considered a BMP for reducing NO_3-N loads to "receiving" waters. He concluded cool-season grasses appeared to attenuate NO_3-N better than warm season grasses. Similarly, rhizomatous species appeared to attenuate NO_3-N better than bunch grass species. Rhizomatous, cool-season grasses appear to be more opportunistic and optimize the use of environmental factors to reduce groundwater $N0_3$-N. He stressed the importance of choosing grass species that are actively growing during high-rainfall months to minimize leaching. Species adaptation as well as competition, predation, and disease should be considered when appraising sites conditions.

Nitrogen Cycling

Nitrogen is present in many forms in the biosphere. The atmosphere contains vast quantities of inert nitrogen (N_2), about 78% of the atmosphere by volume (Hawkes et al., 1985; Foth and Ellis, 1988; Taiz and Zeiger, 1991). For the most part, this N is not directly available to plants. Acquisition of N from the atmosphere requires the breaking of a very stable triple covalent bond between two N atoms, and higher plants do not have the capacity to carry out this reaction physically (Taiz and Zeiger, 1991; NRC, 1978).

On the other hand, N is quite unstable in soil, and N availability to plants is a function of soil temperature, water content, microbial activity, pH, and method of storage and application. Nitrogen is stored primarily in three forms: NO_3-N and NH_4-N, which are inorganic compounds directly available as plant nutrients, and organic N, contained in live or decaying plant, animal, and microbial biomass (NRC, 1978; Newberry, 1992).

Nitrogen is utilized by plants to synthesize amino acids which in turn form proteins. Nitrogen is also required by plants for other vital compounds such as chlorophyll, nucleic acids and enzymes (Haynes, 1986; Taiz and Zeiger, 1991). Nitrogen exists in a chemically reduced state and commonly constitutes 1.5 to 5% of dry weight of plants (Haynes, 1986). Most of the N taken up by plants is in the NO_3-N form which moves with soil water to plant roots where uptake occurs. Ammonium N does not move to the roots as it is bound to the surfaces of soil particles (NRC, 1978; Hawkes, 1985).

Grassland studies by Legg and Meisinger (1982) determined N cycling and N balance in established grassland pasture systems. The researchers, studying N recoveries in a Mollisol soil, found 36% of the applied N accumulated in grass tops, 28% in the roots to a depth of 28 cm after eight weeks of growth, and 19% in the soil to a depth of 74 centimeters. Since little N was found deep in the soil, 17% loss was attributed to gaseous evolution.

Nitrogen is a principal component of cattle effluent. Depending on the type of livestock, approximately 70-80% of the N fed to animals is excreted in the manure (Porter, 1975; Klausner, 1989). The major factors determining N content and availability are: (i) composition of the feed ration, (ii) amount of bedding and water added or lost, (iii) method of manure collection and storage, (iv) method and timing of land application, (v) characteristics of soil and the crop to which manure is applied, and (vi) the climate (Klausner, 1989; Moffitt et al., 1992; Midwest Plan Service Committee, 1985). There are two forms of organic N in manure, unstable and stable organic N. In either form, the organic N must be decomposed by microorganisms to inorganic N in NO_3-N and NH_4-N form before it can be used by plants (Klausner, 1989). The unstable organic N is present in urine as urea or uric acid. When feces and urine are excreted they contain little, if any NH_3-N. After excretion, urea and uric acid are rapidly hydrolyzed to NH_3-N and carbon dioxide (CO_2) by enzymes from fecal bacteria. With further ammonification during storage, the NH_3-N content moves toward 40-70% of TN depending on manure type, pH, manure drying, and NH_4-N loss (Klausner, 1989; Steenvoorden, 1989).

The decomposition of stable organic N to a plant-available form occurs at two rates. The less resistant organic N decomposes during the year of application, whereas the more-resistant organic N decomposes very slowly in future years. Repeated application to the same field results in an accumulation of a slow-release manure N source (Midwest Plan Service Committee, 1985; Klausner, 1989; Krider et al., 1992).

Protein and allied compounds from manure are broken down into amino acids through a reaction called aminization. Soil organisms acquire energy from this digestion. They also utilize some of the amino N in their own cell structure. Ammonia and ammonium compounds are formed by ammonification (NRC, 1978; Hawkes, 1985).

Ammoniac forms of N are changed to NO_3-N by a two step process called mineralization. Oxidation of NH_3-N to nitrous oxide (NO_2-N) is carried out by soil bacteria of the *Nitroso* group (*Nitrosomonas*), while the further oxidation to NO_3-N is carried out by bacteria of the *Nitro* group (*Nitrobacter*) (Taiz and Zeiger, 1991). These chemoautotrophic nitrifiers consume oxygen (O_2) while feeding on organic matter which contains C and N (NRC, 1978; Hawkes, 1985; NRAES, 1992). Microorganisms use C for both energy and growth while N is essential for protein and reproduction. In general, biological organisms, including humans need about twenty-five times more C than N (NRAES, 1992).

The ratio of C to N is referred to as the C:N ratio. With C:N ratios below 20:1, the available C is fully utilized but there is excess N available. This N may then be lost to the atmosphere as NH_3 or N_2O and odor can become a problem. Carbon:nitrogen ratios higher than 40:1 require longer composting times for the microorganisms to use the excess C (NRAES, 1992). This immobilization renders inorganic N in the soil unavailable for utilization by the plants.

Nitrification occurs readily under conditions of warm temperature, adequate oxygen and moisture, and optimum pH. At 24 and 10° C, nitrification may be completed in one to two or 10 to 12 weeks, respectively (Hawkes, 1985). The optimum pH for nitrification is 8.0, with activity decreasing rapidly below pH 7 (NRC, 1978; Hawkes, 1985). The oxidation of NH_3 to NO_3 occurs within a few days or weeks, and requires about 4.5 mg of oxygen (O_2) per mg of N. This explains the considerable demands for O_2 of water bodies receiving high loading of NH_3 (NRC, 1978).

Nitrogen may be lost from the soil to the atmosphere by reactions that convert NO_3-N to gaseous compounds of N. Several studies of N balances have indicated a large reduction of N attributed to volatilization and denitrification. These losses range from 1 to 75% of added N (NRC, 1978; Haynes, 1986).

Nitrogen losses from dairy manure range from 30 to 60% when stored in open lots. These losses increase with time, higher temperature, wind, and low humidity (Willrich et al., 1974). Results from wind tunnel experiments show that NH_3-N volatilization loss amounted to 35.5% of the applied NH_4-N for cattle manure from 26 experiments. Ammonia volatilization varied from 12.1 to 56.8% for pig slurry in 36 experiments (Klausner, 1989; Steenvoorden, 1989). Microbial activity almost ceases when the temperature falls below 5° C (Moffitt et al., 1992). Thus, most volatization losses cease in the fall and do not resume again until spring.

High pH levels result in increased loss due to volatilization (Rauschkolb and Hornsby, 1994). Results by Ernst and Massey (1960) and Clay et al. (1990) showed NH_4-N losses of 8 and 50%, after ten days, with soil pH levels of 5 to 7.5, respectively. This volatilized NH_3-N can be absorbed in the vapor phase through open leaf stomata of plant leaf canopies or may also dissolve on the plant leaf surfaces and subsequently be absorbed and metabolized (Haynes, 1986). Since NO_3-N is highly mobile in solution, it may be lost as various gases such as N_2O (Sander et al., 1994), nitric oxide (NO), and N_2. This process, called denitrification, is carried out by bacteria that can use NO_3-N as a terminal electron acceptor under anaerobic conditions (Rauschkolb and Hornsby, 1994).

Nitrous oxide is the main product of denitrification in soils with a low, but finite level of O_2, while N_2 is the main product in anoxic soils. Denitrification occurs at temperatures from less than 5 to 75° C but individual species have narrower temperature limits (NRC, 1978; Haynes, 1986).

Based on an evaluation of denitrification studies by Eichner (1990), daily average emissions for grass sites ranged from 0.4 to 13.4 g N_2O-N ha^{-1} d^{-1} in fertilized and manured soils, respectively. Emissions from a weedy pasture of timothy grass (*Phleum Pratense*) ranged from 0.7 to 1.9 kg of N_20-N ha^{-1} d^{-1}. Most of the N_2O from agriculture land is released during the growing season.

Coyne et al. (1994) support this relatively small fraction of total N loss with N_20 emissions ranging from 0.3 to 0.7% of total N applied in VFS, with rates varying on a daily basis. Coyne et al. (1995b) reported average N_2O loss in VFS from manure-amended, well-drained soils immediately after rain at 4% of the average total N gas flux. Rauschkolb and Hornsby (1994) report no-till treatments increase denitrification activity because of a greater amount of oxidizable C in surface soils compared to conventional tillage conditions.

When NO_3-N is not lost through denitrification and amounts are in excess of crop needs or beyond the crop root zone it may reach surface and groundwater supplies. Leaching is the physical process of downward movement of NO_3-N. Two of the major factors controlling leaching losses are the quantity of water passing through the soil profile and the concentration of NO_3-N in the soil profile (Haynes, 1986). As the soil solution is displaced through the soil profile by rainfall or irrigation in excess of the water holding capacity of the soil, the dissolved ions move with the wetting front. The potential for NO_3-N leaching below the root zone is greater for animal manure than for commercial fertilizer (Rauschkolb and Hornsby, 1994).

Rainfall and irrigation move NO_3-N downward, while evaporation moves it back towards the surface. In the Great Plains, high evaporation and low average rainfall result in low leaching from soils that are not irrigated (NRC, 1978). The evaporation process is usually important only in the upper 30 cm of the soil profile. Summer storms of high intensity may move NO_3-N rapidly out of the root zone of the soils.

The intensity and seasonality of precipitation may also create surface runoff. In terms of mass emission, the amount of N lost in surface waters is relatively small, but its concentration is of most concern (Rauschkolb and Hornsby, 1994). Magdoff et al. (1977) monitored a low-cost manure storage facility for runoff volume and nutrient concentration. During the winter and early spring, runoff rates were as high as 69% of the total annual runoff, even though only 26% of the annual precipitation occurred during these months. Monthly losses of solids and nutrients were correlated with volume of runoff. The annual loss of solids, N, P, and K in the runoff amounted to 82.0, 6.0, 0.4 and 9.3 kg ha^{-1}, respectively. These results indicate the runoff had definite potential to pollute small streams and ponds. Approximately 73% of the N lost in runoff was NH_4-N. The total loss of N during storage was likely enhanced by NH_3-N volatilization during the warmer part of the year.

Ultimate disposal of feedlot waste on agricultural land should make use of the high levels of NH_4-N and NO_3-N available in manure. Butchbaker et al. (1972) reported loading rates of about 178.5 kg N ha^{-1} would need to be applied to agricultural land for disposal of available solid wastes. The increased production of animal manures and concern regarding environmental risks associated with its use necessitate using VFS as a best management practice (BMP). A better understanding of manure as a nutrient source and disposal alternatives which consider ways to avoid concentration in small areas will benefit the Montana livestock industry.

VFS Design Criteria for Montana

Vegetative filter strips have been shown to be effective in removing N from runoff water, yet there are no available studies reported for effectiveness of VFS for N recovery in Montana. The ultimate goal of this study was to evaluate the ability of newly established VFS to reduce N transport downslope and through the soil profile for sites and climatic conditions common in Montana.

Methods and techniques for achieving high N containment in VFS have been evaluated extensively in higher precipitation areas. Design criteria for VFS for treating waste are based on the peak discharge from a 24-hour, 25-year storm (USDA-SCS, 1989; Krider et al., 1992). The design storm for VFS in Montana differs from overland flow (OF) systems typically used in higher precipitation areas. These OF systems are designed to meet criteria based on waste application discharge occurring on a daily basis or at peak discharge rates much greater than storm events that typically occur in Montana (USEPA, 1984).

The evaluation of individual plant species to provide optimal N attenuation under dry, northern climates also requires further study. Investigations and reports that clearly demonstrate the N recovery capabilities of various grass species are available in areas where precipitation is higher but rainfall intensities are not as extreme as in Montana. This research is directed towards determining N recovery capacities of meadow bromegrass, orchardgrass, tall fescue, and tall wheatgrass. The question of whether grasses are a viable BMP for dairy and feedlot operation management will be addressed along with VFS management options pertinent to Montana agriculture.

CHAPTER 3

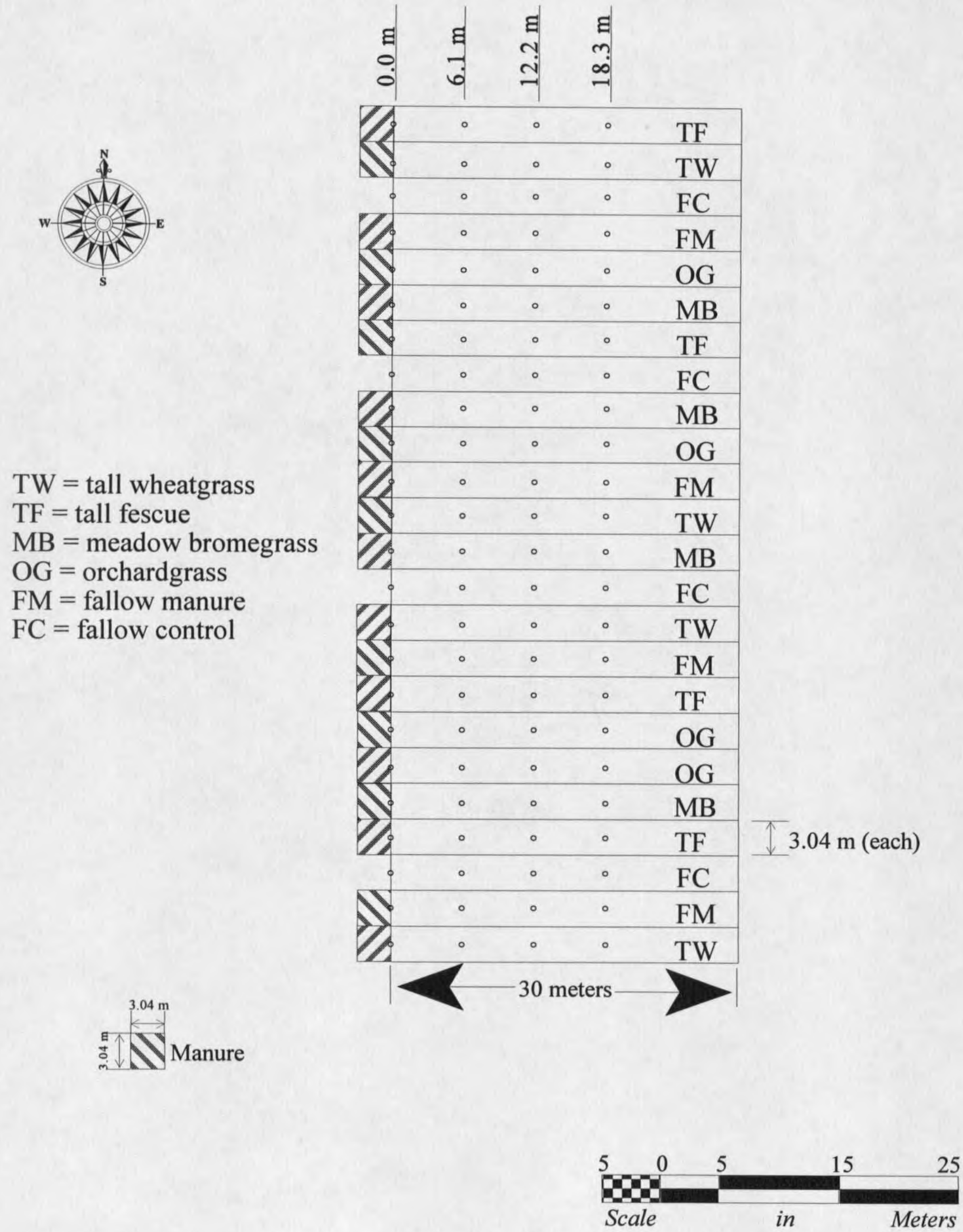
MATERIAL AND METHODS

Site Description

Field plots were established in 1994 at the Montana State University Arthur Post Research Farm eight kilometers west of Bozeman. The plots are laid out in plot p-19 of section 8, running in an east-west direction. Each individual plot measures 3.0 m north and south by 30.4 m east and west (92.4 m^2). The site is 0.2 hectares with a slope of 4.3 to 5.1% falling west to east (Figure 1). Cross slope of the plots measure from 1.8 to 2.2%.

The experimental plots were established on an Amsterdam-Quagle silt loam soil. This very deep, well-drained soil overlies relict stream terraces. The soil is classified as a fine-silty, mixed Typic Haploboroll. The surface layer has 28 cm of dark silt loam with light brown silt loam to a depth of 76 to 102 cm. There are areas with gravel at 76 to 102 cm which intergrade to stream floodplains below this slope. Soil permeability is moderate with soil available water holding capacity ranging from 26 to 30 cm. Soil pH averages 7.8. This soil is representative of sites throughout the Gallatin Valley.

Figure 1. Field plot layout of vegetated filter system experiment.



The site had five years of perennial grass followed by two years of summer fallow prior to field plot establishment. The northern section of the site had not been fallowed. In spring of 1994 the area was rototilled three times. The area was surveyed and laid out to ensure minimal cross slope occurred in the plot area.

Six treatments with four replications in a randomized complete block design were used for this study. The six treatments per replication consisted of two fallow strips and four grass species. `Regar' meadow bromegrass (*B. biebersteinii* L.), `Latar' orchardgrass, `Fawn' tall fescue, and `Alkar' tall wheatgrass (*Agropyron elongatum* L.) were seeded 6 May 1994 at recommended rates and allowed to establish throughout the year. The grass was drilled and packed using a plot seeder with 15.2-cm row spacing.

Tall fescue and orchardgrass were chosen for the VFS as they meet vegetation selection requirements for OF systems (USEPA, 1984). Tall wheatgrass and meadow bromegrass were chosen because of their adaptation to climatic conditions for this region. These cool-season species green up early in the spring. Early spring growth is an important trait as typical storm events for this area normally occur in early and late spring. Additionally, these species can tolerate prolonged dry conditions.

Two fallow plots were included in each replication. One fallow plot received manure, and one fallow control plot did not receive manure. The plots were irrigated once during the 1994 establishment period. Broadleaf weed control treatments of Curtail-M (clopyralid + MCPA, at 0.785 kg active ingredient per hectare) were applied in 1994 and in the spring of 1995. Excellent vegetative cover was established, and forage was removed once in 1994 to prevent buildup of organic material and to reduce nutrient concentration in the soil profile. Additional weed control was accomplished by cultivation on the summer fallowed strips and mowing the grasses in late spring and fall.

Dairy Waste Additions to the VFS

Dairy manure was surface applied to an area measuring 3.0 x 3.0 m along the upper edge of each VFS plot. The total length along the upper portion of the site measured 72.0 meters. Those plots receiving no manure were covered with polyethylene sheeting to protect the area from contamination during application. The manure, 11 and 18 metric tons (as-is basis), was applied on 5 June and 31 July, respectively. The dairy manure, obtained from a loafing area in an open lot, contained fresh fecal material and straw bedding. To ensure uniformity of the manure it was loaded in 454 kg lifts by a front-end loader. Six trips were required to haul the manure to the site where it was deposited in one pile prior to site application. The manure was deposited in six-242 kg lifts by a front-end loader above each vegetated strip. Manure was raked and spread over each 9.2 m^2 area to ensure uniformity and mixing.

Storm Criteria and Climatic Data

The average annual precipitation during the entire period on record between 1963 to 1995 is 48 cm. Nearly two-thirds of the precipitation falls during the period from April through September. Normal precipitation for May and June amounts to about one-third of the total annual precipitation. A secondary maximum precipitation occurs in September, normally much less than that in the spring (USDI, 1960). The precipitation from year to year, however, is characterized by many departures from average.

The VFS at the Arthur Post Experiment Farm was constructed following USDA-SCS design standards to pass the peak discharge from the 24-hour 25-year storm of 5.59 cm for this area (Miller et al., 1973; USDA-SCS, 1984; Midwest Plan Service Committee, 1985; USDA-SCS, 1989). A runoff curve number (CN) of 90 was used to determine the volume of water that would run off a nearly impervious area, such as a feedlot. This CN represents soils with high runoff potential and very low infiltration rates when thoroughly wetted (0.00 to 0.13 cm h^{-1}). A runoff curve number of 58 was used to simulate runoff for the VFS. This represents a soil with low runoff potential and a moderate transmission rate (0.38 to 0.76 cm h^{-1}) when thoroughly wetted (McCuen, 1989).

The SCS Curvilinear Unit hydrograph (McCuen, 1989) was used to estimate volume of excess runoff from a 60 x 72 m confinement area, equivalent to the average size of a 100 cow-feedlot, and the 30 x 72 m VFS. Runoff generated from the confinement area for this design storm totals 3.4 cm and would enter each strip at a discharge rate of 8.5 m^3 $hour^{-1}$. The probability that this system would receive this application rate is four percent (USDA-SCS, 1984).

To simulate or exceed the 25-year event, irrigation water was applied at a rate of 10.4 cm on July 21 and 15.8 cm on 5 August over 8 and 12 hour time periods, respectively. Rates exceeding 5.6 cm were applied so minimum antecedent soil moisture conditions ranging from 3.6 to 5.3 cm would be met (Table 1). This soil condition refers to the 5-day total growing season rainfall occurring prior to the 25-year event (McCuen, 1989). The predicted runoff for the 25-year event was based on antecedent soil moisture condition II. For any given probable maximum precipitation, the antecedent soil moisture and thus the rate of infiltration determine the volume of runoff. Irrigations during 1995 were designed to examine the VFS performance exceeding 25-year storm events.

Table 1. Seasonal rainfall limits for three antecedent soil moisture conditions (McCuen, 1989).

Antecedent moisture condition	Total 5-day antecedent rainfall	
	Dormant season	Growing season
	----------cm----------	
I	< 1.3	< 3.6
II	1.3–2.8	3.6–5.3
III	> 2.8	> 5.3

Cutting Treatment and Dry Matter Collection

Forage from all grasses was harvested and removed as hay on 29 June and 10 October 1995. Each grass treatment was also hand-clipped on five dates during the regrowth period (4, 18 August and 1, 15, 29 September 1995). These bi-weekly clippings occurred at 0.0 (adjacent to the manured site), 6.1, 12.2, and 18.3 m distances downslope from the manure pile. A different microplot, measuring 0.1 m^2, was sampled during each clipping to estimate the highest N removal period and optimal second harvest date for each grass species. Samples were weighed and dried at 50° C in the drying ovens at the Plant Growth Center, Montana State University. Samples were then ground with a Wiley Mill to pass a 40 mesh screen. Samples were sent to Montana State University Soil Analytical Laboratory for chemical analysis of total nitrogen.

Soil Collection

Prior to manure application, twelve sites in the test area were sampled 26 May 1995 for soil fertility status at 0 to 30, 31 to 60, and 61 to 122 cm. A total of 384 samples were taken again in November with a 3.81 cm diameter soil core to determine N levels throughout the soil profile and the surface area downslope of the manured site. Soil samples for NO_3-N and NH_4-N were taken within each plot at 0.0, 6.1, 12.2, and 18.3 m downslope of the manure site. Soil samples for NO_3-N were also taken at four depth increments of 0 to 30, 31 to 60, 61 to 90, and 91 to 122 cm. Fallow manure plots were pre-sampled 29 September 1995 to a depth of 183 cm to characterize N movement. All samples were dried at 50° C, weighed, and ground to pass through a 2.0 mm screen. Soil analyses were completed by the Montana State University Soil Analytical Laboratory.

Soil And Plant Analyses

Plant samples were analyzed for plant total N using the TKN procedure (Bremner and Mulvaney, 1982). Digestate was analyzed for NH_4-N following the standard method for examination of water and wastewater (Clesceri et. al. 1989). Nitrate was extracted (Keeney and Nelson, 1982) from the soil samples using potassium chloride (KCl). The extract was reduced through a copper-cadmium column and NO_3-N concentration determined by colorimetric ally (Sims and Haby, 1970; Clesceri et al., 1989).

Manure samples were analyzed for total N, NH_4-N, and NO_3-N following the same procedures as the soil and plant analyses. The NH_4-N and NO_3-N manure extract ratio were 1:10 instead of the normal 1:5 because the high organic matter content.

Statistical Analyses

All data were subjected to appropriate statistical analyses utilizing the Statistical Analysis System (SAS, 1985). Analysis of variance (ANOVA) were used to test treatment effects. The level of significance for controlling risk was set at alpha = 0.05, and mean separation was accomplished by Least Significant Differences.