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Use of Riparian Vegetated Filter Strips to Reduce Nitrate and Fecal Contamination in Surface Water

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USE OF RIPARIAN VEGETATED FILTER STRIPS TO REDUCE NITRATE AND FECAL CONTAMINATION IN SURFACE WATER

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ABSTRACT

This research assessed fecal bacteria trapping in surface runoff by grass filters and their potential to enhance NO_3^- removal via denitrification. Grass filter strips 9.0 m long trapped over 99% of the soil in surface runoff in 1992. Fecal coliform removal was less than 75%. In 1993, 9.0 and 4.5 m grass filter strips trapped 99 and 95% of the sediment, respectively. Fecal coliform trapping efficiency was 90% in 9.0 m grass filters and 75% in 4.5 m filters. Fecal streptococci trapping efficiency was 77% in 9.0 m grass filters and only 56% in 4.5 m filters. Fecal coliform concentration in grass filter strip runoff consistently exceeded 200 fecal coliforms per 100 mL. Grass filter strips which minimized sediment loss did not reduce fecal contamination of water to acceptable levels when runoff occured. Nitrous oxide fluxes were smaller in grass filters than in manured plots. In 1993, N_2O loss ranged from 2050 to 11120 mg N_2O-N m^{-2} h^{-1} in amended soil and 160 to 1060 mg N_2O-N m^{-2} h^{-1} in grass filter strips. Denitrification was not apparently enhanced in the grass filters relative to the manured soil.

DESCRIPTORS

Agriculture, Erosion, Denitrification, Farm Management & Animals, Fecal Coliforms, Fecal Streptococci, Runoff, Sediment Transport, Soil Microorganisms, Surface-groundwater Relationships, Soil Management, Trace Gas, Vegetative Filter Strips, Waste Disposal, Waste Treatment, Water Pollution, Water Quality

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CHAPTER 1 – Introduction

Objectives of Research Proposal

Surface water pollution may increase if animal wastes from expanding livestock industries are disposed of on agricultural soils. Fecal bacteria movement in surface flow is poorly understood as is their persistence once trapped in filter strips. Nitrate is also lost through surface runoff and few studies have looked at its fate in practices recommended to principally reduce soil erosion. This research addressed two needs in water quality: measuring fecal bacteria contamination in surface runoff from poultry manured fields and evaluating grass filter strips as a management practice to trap fecal bacteria from surface runoff and enhance NO_3^- removal from runoff via denitrification.

The specific objectives were:

1. Examine fecal bacteria movement in surface flow from surface applied poultry manure in response to the intensity and volume of simulated rainfall and develop a database which can be used to model this process.
2. Determine the effectiveness of riparian forest and grass filter strips in trapping fecal contaminants from surface flow.
3. Determine the survival of fecal coliforms, fecal streptococci, and *Salmonella* in surface applied poultry manure and the capacity of soils and vegetated filter strips to act as reservoirs for these bacteria.
4. Determine the potential of riparian vegetation to remove NO_3^- from surface runoff via enhanced denitrification.

Variations From the Stated Objectives

There were several variations from the stated objectives. (1) The intensity of simulated rain was maintained at a constant 6.4 cm h^{-1} to improve the replicability of treatment. This intensity mimics a one-in-ten year storm but was necessary to cause surface runoff in a reasonable period. (2) The volume of simulated rainfall was not held constant. Due to uncontrollable plot-to-plot variation, which was a result of inherent plot variability and previous tillage management, the same rainfall intensity did not produce runoff after a uniform period. (3) Rain simulations on grass filter strips illustrated the impracticability of repeating experiments in a riparian forest strip due to the labor and effort required to move the rain simulator to new locations. (4) We intended to enumerate *Salmonella* in this project but two media which we used for this purpose did not prove adequate to selectively isolate *Salmonella* in sediment and water.

Benefits of Research

The data could be used to model movement, survival, and trapping of fecal bacteria in surface water with respect to existing and developing models of sediment flow through filter strips. This research will help efforts to minimize soil erosion losses, show practices for managing soils to prevent fecal contamination, and provide economic and environmental reasons for Kentucky farmers to keep riparian buffers.

Background Information

Runoff from agricultural land is typically studied in the context of soil erosion and its control by tillage practices (Blevins et al., 1990; Wendt and Burwell, 1985). Fecal bacteria transport has principally been studied as a non point-source phenomenon. In grazed watersheds, fecal bacteria in runoff water frequently exceed water quality standards (Jawson et al., 1982; Tiedemann et al., 1988).

Contamination is affected by: the season (Jawson et al., 1982), the timing of rain in a grazed pasture (Jawson et al., 1982), and management intensity (Tiedemann et al., 1988). Centralization of animal production has caused problems with waste disposal, and manuring has been identified as a point-source contributor to surface water contamination (Thornly and Bos, 1985). However, little information is available which describes fecal bacteria transport in surface water and predicts bacteria density in runoff from manured soils in terms of loading rates, fecal age, or the timing and intensity of rainfall (Baxter-Potter and Gilliland, 1988).

Fecal bacteria can enter water supplies through tile drains and channelized flow as well as by attachment to sediment (Bohn and Buckhouse, 1985; Thornly and Bos, 1985). Vegetative filter strips are one technology utilized to trap sediment runoff from soils (Gross et al., 1991). However, that use and the capacity to trap fecal bacteria have not been concurrently examined. Grass filter strips effectively remove most of the solids from animal waste water sources but their performance is affected by the intensity of surface flow (Schwer and Clausen, 1989).

Once fecal bacteria enter vegetative filter strips their fate is ill-defined. Since these strips can promote well-defined soil structure, macropore movement of bacteria through the soil profile may occur (Smith et al., 1985). Repeated irrigation with a source of fecal bacteria could also overwhelm a soil's capacity to trap bacteria since soils have a finite capacity to adsorb and filter bacteria from solution (Tare and Bokil, 1982).

It is also evident that fecal bacteria persist in soils and sediments for extended periods (Bohn and Buckhouse, 1985). Fecal bacteria are readily released from fresh deposits and this release may persist at levels above public health standards for months after deposition (Thelin and Gifford, 1983). Grazing animals must be absent from pastures for prolonged periods before fecal bacteria levels in runoff approach that of ungrazed locations (Jawson et al., 1982). This raises the issue of whether continued manuring of soils promotes runoff which will exceed public health standards and ultimately overcome trapping by vegetative filters. These filters may become reservoirs of fecal bacteria.

Elevated levels of fecal bacteria in almost all agricultural soils have raised questions about the use of indicator bacteria, like fecal coliforms, to accurately reflect real incidents of bacterial contamination (Baxter-Potter and Gilliland, 1988). Attribution of fecal bacteria to human, domesticated or wild animal sources depends on several assumptions about the enumeration and comparison of fecal indicator bacteria (Geldreich, 1976). Poultry waste contains significantly different populations of fecal bacteria than does bovine waste (Geldreich, 1976). Generalizations about fecal bacteria movement, trapping, and survival from poultry manure may not be adequately characterized by studies with bovine waste.

The potential exists for riparian vegetative filters to be more than sediment traps. Groffman et al. (1991) have shown that grass filter strips are a potentially enhance denitrification. Sample cores incubated under optimal denitrifying conditions had a NO_3^- removal efficiency of 25-50% depending on carbon availability (Groffman et al., 1991). Carbon limitation and low denitrifier populations limit denitrification below the root zone in turf and agricultural soils (Exner et al., 1991; Parkin and Meisinger, 1989). Groffman et al. (1991) hypothesized that denitrification in vegetated filter strips could be manipulated by carbon additions. Particulate and soluble carbon from a manured field which are trapped in a riparian filter would fill that need.

Riparian vegetative filters by interdiction, uptake, and increased carbon supply can account for six times as much nitrogen removal as nitrogen output in stream flow (Lowrance et al., 1985). Consequently, a demonstrated function of riparian vegetation in removing soluble nutrients from surface runoff would be an economic incentive for their preservation and maintenance.

CHAPTER II – Research Procedures

1992 - Fecal Contamination in Surface Runoff

Location. We did our experiment at the University of Kentucky Agricultural Experiment Station in Lexington during June and July, 1992. The soil was a Maury silt loam soil (fine, mixed, mesic Typic Paleudalf) with an average natural slope of 9%. The experimental plots consisted of an erosion strip, 4.6 m wide by 22.1 m long, and a grass filter strip, either 4.6 m wide by 9.0 m long or 4.6 m wide by 4.5 m long, which abutted it (Fig. 1). No tillage (Plots 2 and 4), conventional tillage (Plots 1 and 5), and chisel plow tillage (Plots 3 and 6) had been used as the tillage method on the erosion strips since 1984. However, in 1992, the erosion strips used in our study were cultivated using a chisel plow plus disking as the only management practice. The grass filter in each plot was mowed to a height of 4.0 cm and consisted of tall fescue (*Festuca arundinacea* L.) and Kentucky bluegrass (*Poa pratensis* L.) sod.

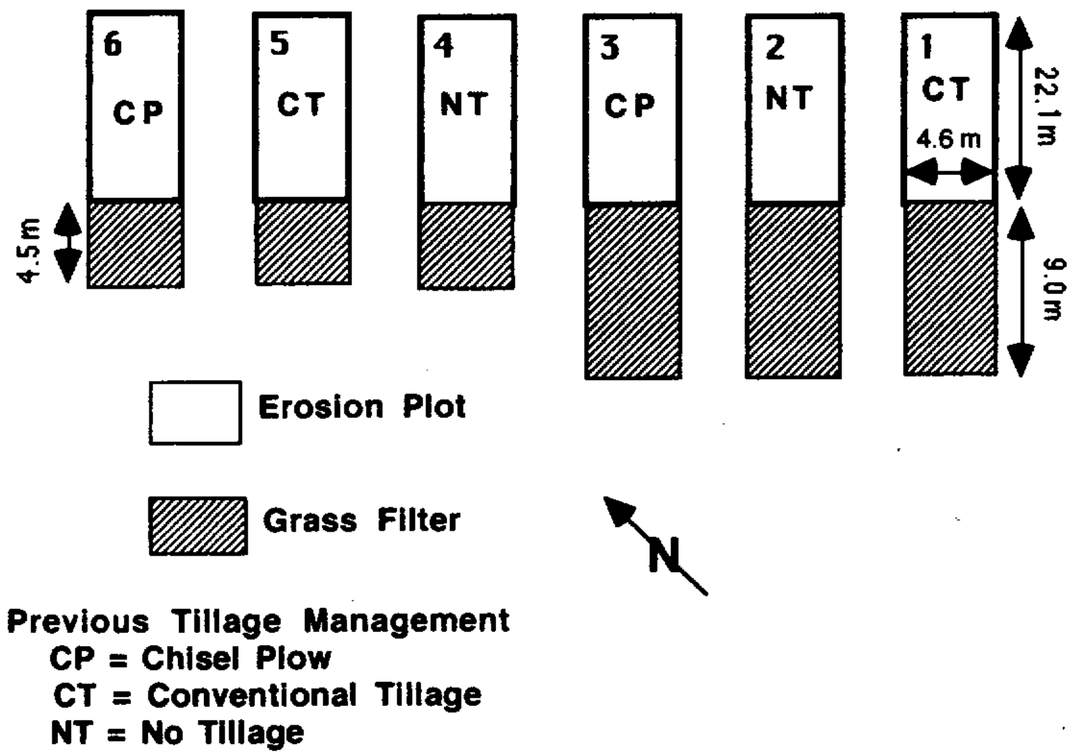

Figure 1. Schematic outline of the study plots used for rain simulation in 1992

Site Treatment. Poultry litter mixed with sawdust and shavings from a laying house was briefly stockpiled and uniformly spread over each erosion strip at 16.5 Mg ha^{-1} (wet weight) on June 30, 1992. It was 60 to 80% incorporated to a depth of 15 cm with a chisel plow and disk as the only tillage practices. The litter contained 2.8% total nitrogen, 2.9% total P, and 1.8% total K (wet weight) at a moisture content of 34.2%.

Moore et al. (1983) have previously described the rain simulator. It has five individual units, hooked in tandem, with dimensions of 4.6 x 6.1 m. Nozzles were set 3.1 m above the erosion strip surface. The simulator mimics natural drop size distribution, impact velocity, and energy. We used it to minimize differences in rain intensity and duration, and to negate the unpredictability of natural rainfall. The five individual units were situated directly over the erosion strip during rain simulations. A metal border, 15 cm high, was inserted at the sides and upper end of the erosion strip to confine runoff. Metal borders were also placed at the sides of the filter strips. To protect the erosion strips from natural rain, we covered them with black plastic tarpaulins. The live grass filters were not covered during the study. Temperature and rainfall during the experiment are shown in Fig. 2.

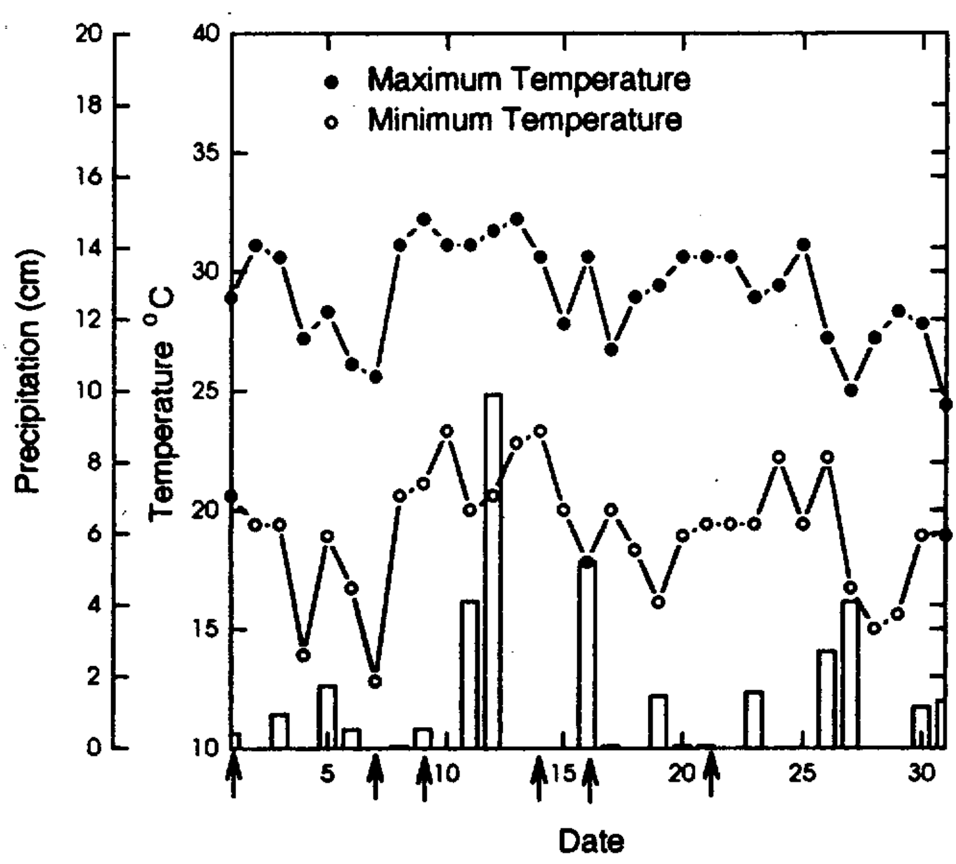

Figure 2. Precipitation and daily temperature at the experiment site during July 1992 (arrows indicate the dates of simulated rain)

We began the rain simulations one week after adding poultry manure to the erosion plots. On July 1, 7, 9, 14, 16, or 21, 1992 rain was simulated on one of the erosion plots beginning with Plot 1. On Plots 2 and 3, which we used for bacterial enumeration, the rainfall simulation dates were July 7 and July 9, respectively. Technical problems prohibited the use of other plots for data analysis. In each plot, we removed the plastic tarpaulin and simulated rain at a rate of 6.4 cm h^{-1} (2.5 inches h^{-1}). This approximates the intensity of a one-in-ten-year storm event in central Kentucky. A storm of this intensity occurred in Lexington on July 16. We did not rain on the grass filters. They were pre-wet by a water hose before each simulated rain. When we observed surface runoff from the grass filter strips for an hour, we stopped simulated rain.

<u>Sampling Protocol</u>. We collected runoff at 5 min intervals in 10 cm wide gutters below both the erosion strip and the grass filter strip (Fig. 3). The gutter below the erosion strip had a manually-operated aluminum slide that could be opened and closed to direct surface runoff onto the grass filter strips or into the gutter for sampling (Fogle and Barfield, 1993).

Runoff from the erosion strip was collected in an 18 L plastic bucket for short periods (10 to 30 seconds) and weighed to determine runoff rate. The contents of the bucket were stirred to uniformly resuspend soil particles and a representative one liter sample was removed for sediment analysis. A second uniformly mixed sample was removed for fecal coliform enumeration and stored in sterile 500 mL plastic bag.

Runoff rates from the grass filter strip were determined at 5 min intervals by the time required to fill an 8 L plastic bucket. As with runoff from the erosion strips, after the collected runoff was uniformly resuspended, subsamples were removed for sediment and fecal coliform analysis.

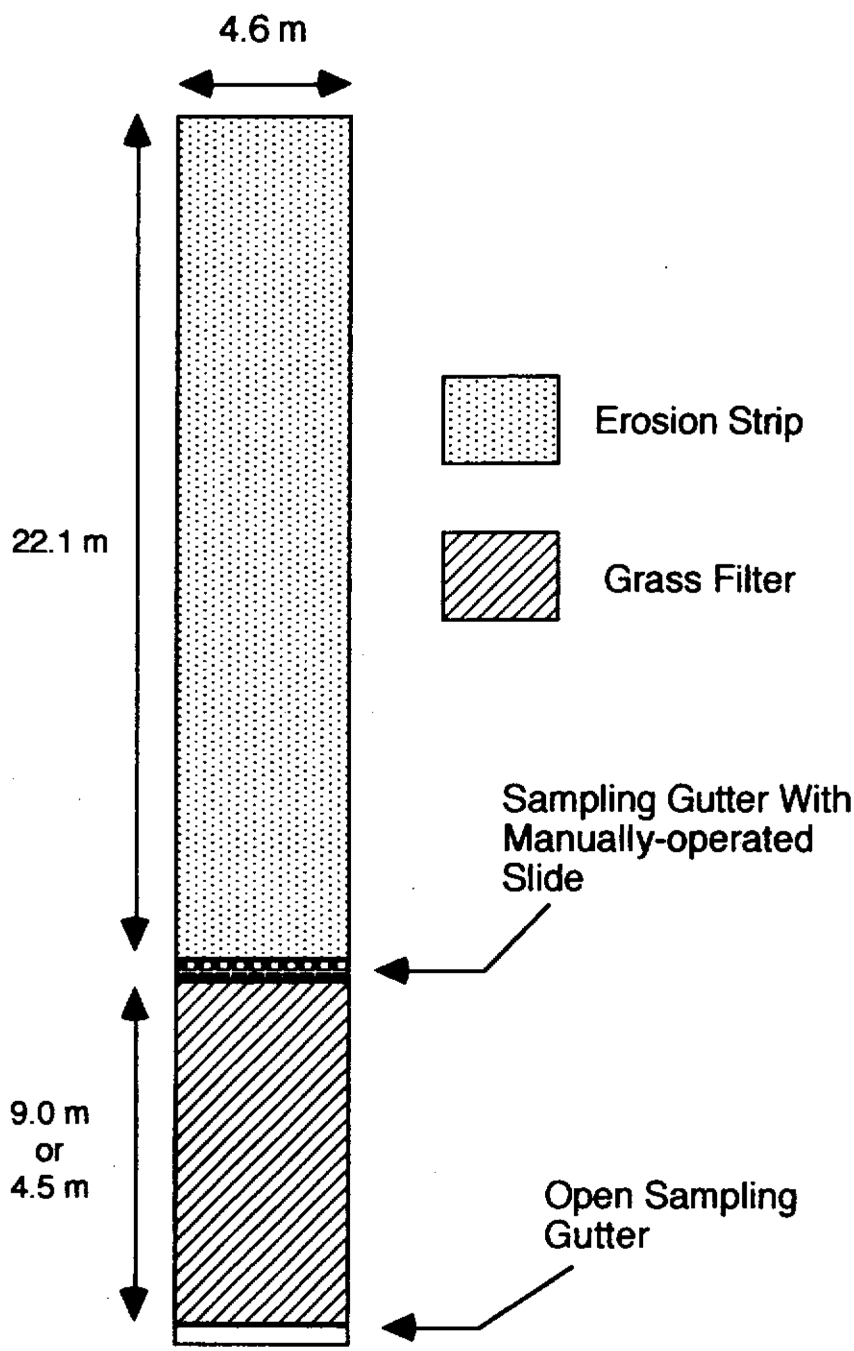

Figure 3. Schematic diagram of an individual plot used in 1992

We collected ten soil samples for fecal coliform enumeration from random locations in both the erosion strips and the grass filter strips to a 15 cm depth immediately before rain simulation and within 48 hours after rain simulation. Most of the grass filter did not receive runoff because variations in elevation diverted surface flow to a few relatively narrow channels (a few cm wide in most cases). Consequently, the soil samples removed from grass filter strips after rain simulation were confined to the upper portion of the filters — within 1 m of the erosion strips. This was where most sediment and presumably bacterial trapping occurred. Soil samples were not removed from the rest of the grass filter to avoid grossly underestimating the fecal coliforms trapped in the filter strips. The ten soil samples from each site (erosion strip or grass filter), at each sample period, were separately pooled and uniformly mixed before analysis.

Chemical and Microbiological Analyses. Chemical analysis of poultry litter was done in the University of Kentucky Regulatory Services soil testing laboratory. Sediment in runoff was determined gravimetrically after water removal and drying at 105 °C. Fecal coliforms (*i.e.* *Escherichia coli*) were enumerated because these are the principal indicator organisms used to assess water quality (APHA, 1992). The fecal coliforms in water samples were stored on ice in the field and at 4 °C in the laboratory and enumerated within 24 hours to minimize cell growth or mortality. Both soil and water samples were diluted in physiological saline (0.8% NaCl in distilled water) prior to their enumeration by membrane filtration technique (APHA, 1992). Fecal coliforms were incubated on mFC agar (Difco, Detroit, MI) for 24 hours at 44.5 °C in an incubating water bath. Typical colonies (dark blue for fecal coliforms) were counted manually after incubation.

Calculation of Trapping Efficiency. The trapping efficiency of the grass filter strips for sediment and fecal coliforms was estimated using a variation of the trapezoidal rule used for hydrographs and sedigraphs (Barfield and Albrecht, 1982). Trapping efficiency, T_r, was estimated by:

$$T_r = \frac{M_i - M_o}{M_i}$$

where M_i and M_o are the total mass of sediment or number of fecal coliforms in the inflow and outflow of the grass filter strip. The mass inflow was estimated from:

$$M_i = \sum_{j=1}^{n} C_{ij}\, q_{ij}\, \Delta t_j$$

where C_{ij}, q_{ij}, and Δt_j are the sediment or fecal coliform concentrations, flow rate, and time interval of the jth measurement of inflow. M_o was estimated by:

$$M_o = \sum_{j=1}^{n} C_{oj}\, q_{oj}\, \Delta t_j$$

where C_{oj} and q_{oj} are the concentrations and flow rate of the jth measurement of outflow and Δt_j is the time interval of outflow. Concentration and flow were conservatively estimated by the average value of C_j and C_{j-1} or q_j and q_{j-1} for the period during which runoff occurred.

1993 - Fecal Contamination in Surface Runoff

Location. We did this study during June and July 1993. Our study site was at the University of Kentucky Agricultural Experiment Station 15 km north of Lexington Kentucky. The plots were on a Maury silt loam soil (fine, mixed, mesic typic paleudalf) with an average natural slope of 9% and a soil permeability that ranged from 5 to 15 cm h^{-1} (Blevins et al., 1990). Six plots were prepared. Three plots (Plots 1, 2, and 3) had erosion strips 13.7 m long and grass filter strips 9.0 m long (Fig. 4). The other three plots (Plots 4, 5, and 6) had erosion strips 18.2 m long and grass filter strips 4.5 m long. From 1984 to 1991, the tillage practices on the erosion strips were conventional tillage (Plots 1 and 5), no-tillage (Plots 2 and 4), or chisel plow tillage (Plots 3 and 6). In 1992 and 1993, the year of this study, we used chisel plowing followed by disking as the tillage management in all of the erosion strips. Each grass filter strip was a mixture of tall fescue (*Festucaarundinacea* L.) and Kentucky bluegrass (*Poapratensis* L.) sod. Before each rain simulation we mowed the grass filter strip to a height of 4.0 cm.

Plot Treatment. Two days before rain simulation we uniformly spread poultry litter from a laying house over an erosion strip. The poultry litter (a mixture of manure, sawdust, and shavings) had a moisture content of 34% and a nutrient analysis of 3% total N, 3% total P and 1.8% total K. The application rate was 16.5 Mg ha^{-1} (wet weight). The litter was shallowly incorporated (85% incorporation to a depth of 15 cm) by chisel plow and two diskings immediately after application.

The erosion strips were not covered until we added the poultry litter. Once we added litter, we only covered an erosion strip if rain was predicted in the evening or if rain occurred during the day. We never covered the grass filter strips during the study. Figure 5 shows the air temperature and rainfall during the study period.

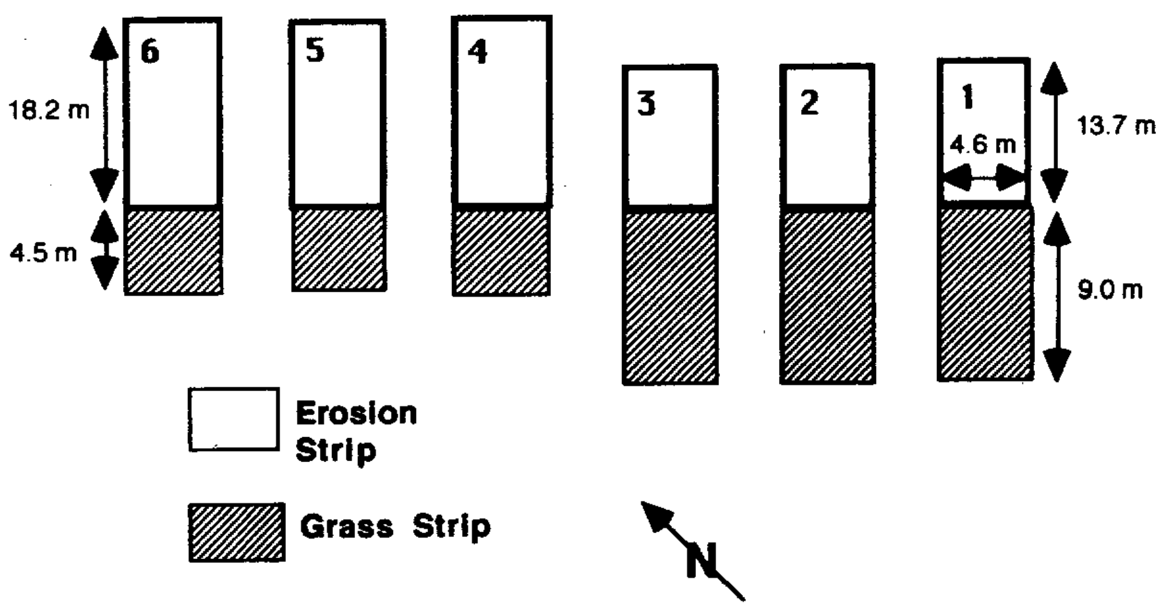

Figure 4. Schematic outline of the study plots used for rain simulation in 1993

We used a rain simulator to minimize differences in rain intensity and free us from the unpredictability of natural rainfall. The simulator mimics natural drop size, distribution, impact velocity, and energy. It has five individual units, hooked in tandem, each with dimensions of 4.6 m by 6.1 m. Each unit has four sections with a total of 12 oscillating 80150 Veejet-type nozzles operated at 41 kPa pressure. We set the nozzles 3.1 m above the erosion strip surface. Moore et al. (1983) give a more detailed description of the rain simulator. Four of the five individual rain simulator units stood directly over the erosion strip on plots with 4.5 m grass filter strips. Three of the five individual units stood over the erosion strip on plots with 9.0 m grass filter strips. The remaining units, in both cases, stood directly over the grass filter strips so that we could simulate rain on the erosion strips and grass filter strips simultaneously. To confine runoff we inserted a metal border at the sides and upper end of the erosion strip. We also placed a similar metal border at the sides of the grass filter strips (Fig. 6).

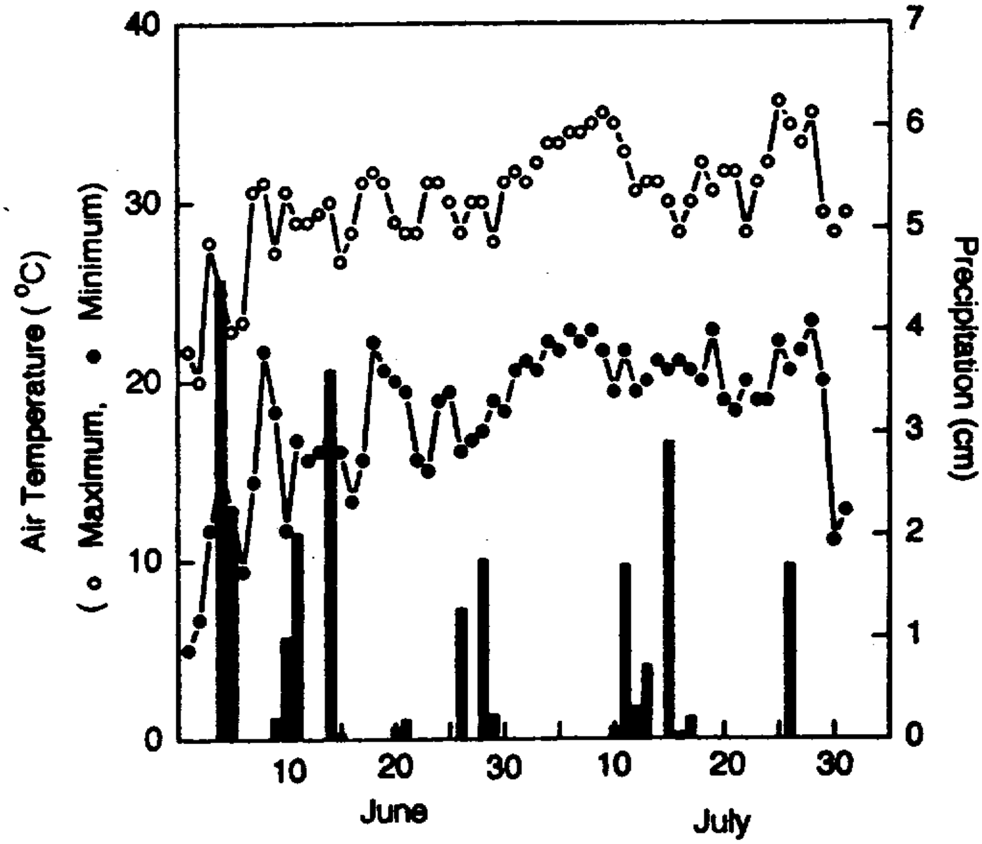

Figure 5. Precipitation and daily temperature at the experiment site during 1993 (arrows indicate the dates of simulated rain)

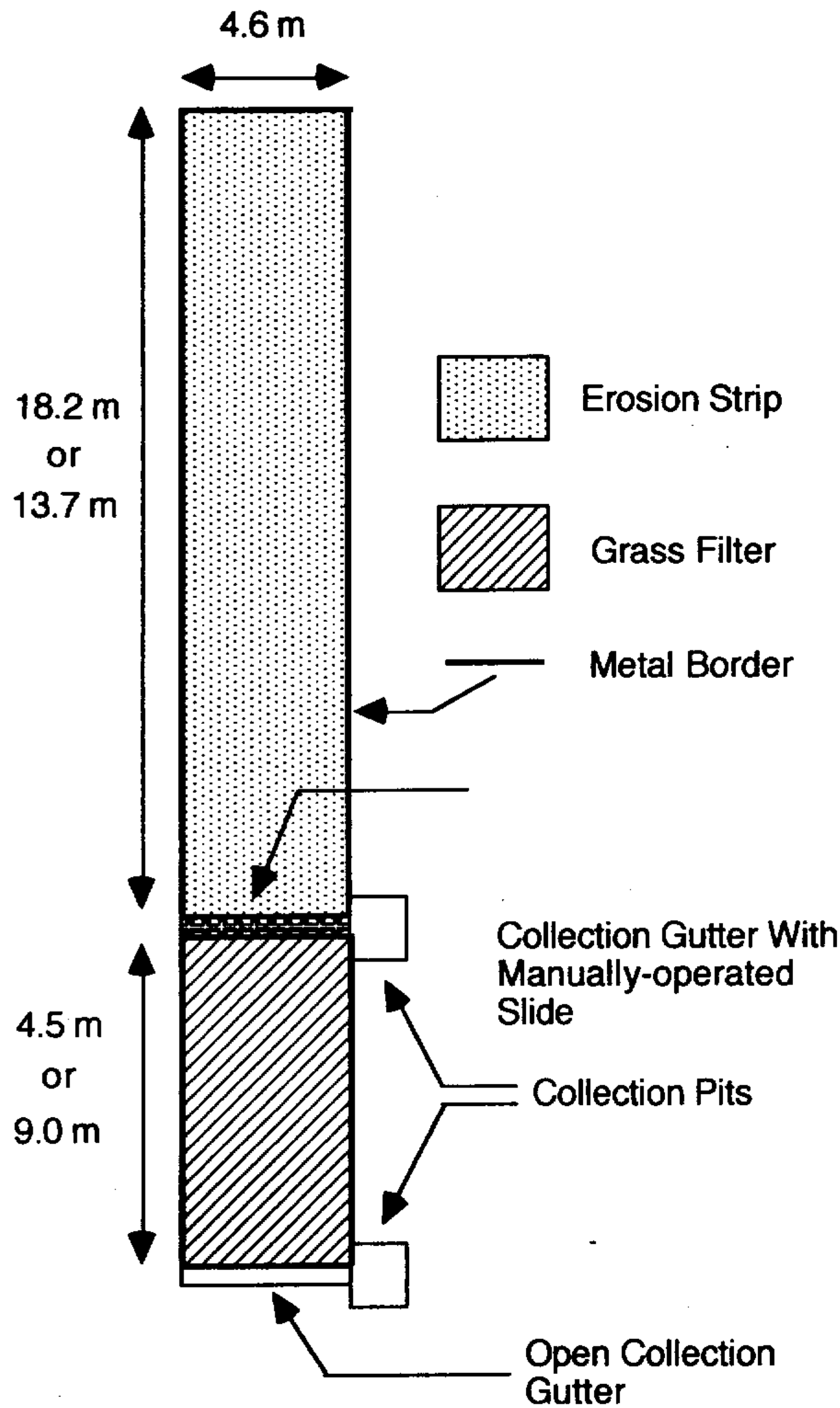

Figure 6. Schematic diagram of an individual plot used in 1993

We simulated rain on the plots two days after we applied poultry litter. Rain simulation dates were: June 9, Plot 6; June 23, Plot 5; July 1, Plot 3; July 8, Plot 2; July 14, Plot 1. We did not use Plot 4, which had a 4.5 m long grass filter strip, because a storm blew off the protective cover and prematurely wet the plot after we added poultry litter. Rain simulation was at a rate of 6.4 cm h^{-1}. This approximates the intensity of a one-in-ten year storm in central Kentucky, but was necessary to cause runoff within a reasonable time after simulations began.

Sampling Protocol. Surface runoff from the erosion strips usually started 15 to 30 minutes after simulated rain began and runoff from the grass filter strips began 15 to 30 minutes after that. We continued rain simulations until we sampled runoff from the grass filter strips for one hour. Table 6 shows the total period of rain simulation for each plot.

As runoff occurred, we collected it for 10 to 30 seconds at approximately 5 min intervals in 10 cm wide gutters below the erosion strip and grass filter strip (Fig. 6). The gutter below the erosion strip had a manually operated slide that could be opened or closed to direct runoff onto the grass filter strips or into the gutter for sampling (Fogle and Barfield, 1993). We weighed the runoff samples from the erosion strip in a tared plastic bucket to calculate flow. We estimated surface runoff flow from the grass filter strips at 5 min intervals by the time it took to fill an 8 L plastic bucket.

After we stirred runoff samples from the erosion strips to uniformly suspend the sediment, we removed representative 1.0 L samples for sediment analysis and stored them in plastic bottles. Sediment loss in runoff was determined gravimetrically after decanting the water and drying the sediment at 105 °C. We removed a second uniformly mixed sample at each time for bacterial enumeration and stored these in 500 mL sterile plastic bags. We obtained samples for sediment and microbial analysis of grass filter strip runoff directly from effluent leaving the gutters. These were also stored in 1.0 L plastic bottles and sterile plastic bags.

We kept all fecal bacteria samples on ice in the field and at 4 °C in the laboratory to minimize growth and mortality. We counted them within 24 h. We also counted fecal bacteria in soil samples (10 pooled soil cores) collected before and after rain simulations at 0 to 5 and 5 to 15 cm soil depths. The soil cores we took from the grass filter strips after rain simulation were all from within 1 m of the erosion strip since this was the zone of maximum sediment deposition.

Microbiological analyses. To reduce bacterial concentrations to a measurable number we made a ten-fold serial dilution of soil and runoff samples in saline solution (0.8% NaCl in distilled water). We measured fecal coliform and fecal streptococci concentrations based on manual counts of representative colony forming units (CFU) that grew on selective media after spiral plating. We grew fecal coliforms on Difco (Detroit, MI) mFC agar incubated at 44 °C for 24 hours. Fecal streptococci were grown on Difco KFS agar incubated at 37 °C for 48 h.

The spiral plater (Spiral Biotech, Bethesda, MD) puts 48 mL on an agar plate in an Archimedes spiral path that logarithmically decreases volume as the distance from the center of the plate increases. Colony forming units are counted in discrete sections of the plate which contain known sample volumes. Because of the small sample volume plated, the detection limit by this method was approximately 2080 CFU/100 mL in water samples and 208 CFU/g dry soil in soil samples. The high detection limit was not a problem in our study since the bacterial concentrations we observed typically exceeded these levels. When we required greater sensitivity, the serially diluted samples were filtered through sterile gridded 0.45 mm cellulosic membranes using established protocols for water quality analysis (APHA, 1992). Our detection limit for fecal bacteria in soil samples by membrane filtration was 3 CFU/g dry soil.

Calculation of Trapping Efficiency. We estimated the trapping efficiency of the grass filter strips for surface runoff, sediment, and fecal bacteria by applying a variation of the trapezoidal rule used for hydrographs and sedigraphs (Barfield and Albrecht, 1982) as we did for 1992 data.

Statistical Analysis. Trapping efficiency was analyzed using SAS (SAS, 1988) and a completely randomized design. Due to the lack of replication, no set statistical significance level could be used.

1992 - Denitrification in Filter Strips

Site. This study was done at the University of Kentucky Agricultural Experiment Station in Lexington during June and July, 1992. Experimental plots were on a Maury silt loam soil (fine, mixed, mesic Typic Paleudalf) with an average natural slope of 9% and soil permeability ranging from 5 to 15 cm h^{-1} (Blevins et al., 1991). Six individual erosion plots 4.6 m wide by 22.1 m long were used (Fig. 1). A grass filter either 4.5 m or 9.0 m in length abutted each erosion plot. Grass filters were a mixed sod composed of tall fescue (*Festucaarundinacea* L.) and Kentucky bluegrass (*Poapratensis* L.). For 8 continuous years before this study, the tillage management used on these plots was conventional tillage in Plots 1 and 3, no-tillage in Plots 2 and 4, and chisel plow tillage in Plots 3 and 6. In 1992, the tillage management in all plots was chisel plow tillage.

SiteTreatment. Poultry litter mixed with sawdust and wood shavings bedding from a breeder house was briefly stockpiled and then uniformly spread over each erosion plot on June 29, 1992 at 16.5 Mg ha^{-1} (wet weight). The poultry litter contained 2.8% total N, 2.93% total P, and 1.78% total K (wet basis) at a moisture content of 34.2%. Poultry litter was partially incorporated into each plot with a chisel plow as the only tillage practice. Erosion plots were covered with black plastic tarps to protect them from natural rain but which allowed air circulation. Grass filters were not covered.

On July 1, 7, 9, 14, 16, or 21, 1992 rain was simulated on one of the erosion plots beginning with Plot 1. The plastic tarp was removed and simulated rain was delivered to each erosion plot at about 6.4 cm h^{-1}. This intensity approximates a one-in-ten-year storm event in central Kentucky. A storm of such intensity occurred in Lexington on July 16, 1992 (Fig.2). Moore et al. (1983) have previously described the rain simulator used in our study.

The duration of simulated rainfall varied from plot to plot because each erosion plot had different runoff characteristics, and because simulated rain continued until runoff was measured for at least 1 hour at the bottom of grass filters with two different lengths (Fig. 1). Consequently, simulated rain lasted for 115 minutes in Plot 1, 135 minutes in Plot 2, 140 minutes in Plot 3, 136 minutes in Plot 4, 100 minutes in Plot 5, and 110 minutes in Plot 6. Surface runoff from the erosion plots was usually observed 20 to 30 minutes after simulated rain began.

Soil Cover Measurements. After simulated rain ceased, soil covers were immediately inserted to a depth of 2.5 cm in the middle of the erosion plot. The soil covers placed in the grass filters were within 1 m of the erosion plots; this location became saturated by surface runoff during simulated rain. The soil covers were coffee cans 17.1 cm high by 15.6 cm diameter with the bottoms removed and a rubber septum penetrating the original plastic lid. Preliminary experiments with N_2O indicated that the cans remained gas tight for the duration of field measurements (data not shown). Five replicates were used at each location. At 0, 15, 30, and 60 minute intervals, gas samples were removed from the soil cover head space and stored in pre evacuated Vacutainers (Beckton Dickinson, Rutherford, NJ) for N_2O analysis.

Intact Core Denitrification Measurements. Five, randomly distributed, intact soil cores 15 cm in depth were collected from each erosion plot and grass filter strip in 21 cm high x 2.5 cm diameter plastic sleeves about one month after simulated rainfall studies. The cores were saturated with 10 mM KNO_3 and allowed to stand for 30 min before excess solution was drained and the plastic sleeve sealed at both ends with sleeve-type rubber stoppers. The head space was evacuated and flushed three times with N_2. Ten mL of head space gas was removed and replaced by an equal volume of reagent grade acetylene. One mL gas samples were removed from the head space at 0, 2, and 6 h intervals.

Gas Analysis. Gas samples were analyzed for N_2O on a Varian 3700 gas chromatograph with 2 m Porapak Q columns using an electron capture detector (ECD) for soil cover samples and a thermal conductivity detector (TCD) for intact cores. Analysis conditions for the ECD were: detector temperature, 360 °C; column temperature, 60 °C; carrier gas, 95% argon, 5% methane; carrier gas flow 30 mL min^{-1}; and sample volume, 1.0 mL. Analysis conditions for the TCD were: detector temperature, 120 °C; filament temperature, 140 °C; column temperature, 60 °C; carrier gas, helium; carrier gas flow rate, 35 mL min^{-1}; and sample volume 1.0 mL. Machine response to N_2O was measured and compared to standard curves for N_2O generated from gas standards of known concentration.

Chemical Analyses. Chemical analysis of poultry litter was done in the University of Kentucky Regulatory Services soil testing lab. Soil samples for chemical characterization were taken from the 0 to 15 cm depth interval in both the erosion plots and the grass filters before addition of poultry litter. The pH was measured in a 1:1 soil:water slurry. Percent soil C was measured on a CR 12 Leco Carbon Determinator (Leco Corp., St. Joseph, MI).

Microbial Analysis. Soil samples used to determine denitrifier most probable number (MPN) were removed from the 0 to 15 cm depth interval in both the erosion plots and the grass filters about 24 h after rain simulation in each plot. The denitrifier MPN in erosion plots and grass filters was determined as outlined by Tiedje (1982). A ten-fold serial dilution of soil in physiological saline (8 g L^{-1} NaCl in distilled H_2O) was used to inoculate 5 replicate tubes per dilution. Growth media was Tryptic Soy Broth with 1 g L^{-1} KNO_3. The tubes were incubated 28 days at 26°C and residual NO_3^- was detected with diphenylamine in concentrated sulfuric acid. The MPN denitrifiers was determined using published tables (Alexander, 1982).

Statistical Analysis. ANOVA and t-tests were made using the CoStat® (CoHort Software, Berkeley, CA) statistical software package for personal computers.

1993 - Denitrification in Filter Strips

Site. We conducted this study at the University of Kentucky Agricultural Experiment Station in Lexington during June and July, 1993. Our research plots were on a Maury silt loam soil (fine, mixed, mesic Typic Paleudalf) with an average natural slope of 9% and soil permeability ranging from 5 to 15 cm h^{-1} (Blevins et al., 1991). Six individual erosion strips 4.6 m wide by 13.7 m or 18.2 m long were prepared (Fig. 4). Below each erosion plots was a grass filter either 4.5 m or 9.0 m in length. These grass filters were a mixed sod composed of tall fescue (*Festuca arundinacea* L.) and Kentucky bluegrass (*Poa pratensis* L.). For 8 continuous years before this study conventional tillage was used on Plots 1 and 5, no-tillage was used on Plots 2 and 4, and chisel plow tillage was used on Plots 3 and 6. In 1992, and again in 1993, chisel plow tillage was the only tillage management used for all plots.

<u>Site Treatment</u>. Each plot was treated the same. Poultry litter mixed with sawdust and wood shavings bedding from a laying house was collected during the morning and uniformly spread over one erosion strip at 16.0 Mg ha^{-1} (wet weight). The nutrient composition of the litter was about 3% total N, 3% total P, and 1.8% total K on a wet basis with a moisture content of 34%. The litter was partially incorporated into the erosion strip with a chisel plow as the only tillage practice. We left the erosion strips uncovered after litter application unless rain was forecast. In that event, we covered the strips with black plastic tarps to protect them from rain but allow air circulation. Grass filters were not covered.

Forty-eight hours after litter application, a rain simulation was done. The dates of rain simulation were: June 9, Plot 6; June 23, Plot 5; July 1, Plot 3; July 8, Plot 2; July 14, Plot 1. Simulated rain was delivered to each erosion strip and grass filter strip simultaneously at 6.4 cm h^{-1}. This intensity approximates a one-in-ten-year storm event in central Kentucky. The different lengths of erosion strips used in this study were necessary to accommodate simulated rainfall on both the erosion strip and the filter strip. Moore et al. (1983) have previously described the rain simulator used in our study. The duration of simulated rain varied from plot to plot (Table 4). Each erosion strip had different runoff characteristics, in addition, simulated rain continued until runoff was measured for at least 1 hour at the bottom of the grass filters which differed in length (Fig. 4). Surface runoff from the erosion plots was usually observed 20 to 30 minutes after simulated rain began.

<u>Soil Cover Measurements</u>. After simulated rain ceased, soil covers were immediately inserted to a depth of 2.5 cm in the middle of the erosion strip. Soil covers were placed in the grass filters within 1 m of the erosion plots. The soil covers were coffee cans 17.1 cm high by 15.6 cm diameter with the bottoms removed and a rubber septum penetrating the original plastic lid. Preliminary experiments with N_2O indicated that the cans remained gas tight for the duration of field measurements (data not shown). Five replicates were used in each location. At 0, 15, 30,

and 60 minutes, 4.0 mL gas samples were removed from the soil cover head space for N_2O analysis and stored in pre evacuated Vacutainers (Beckton Dickinson, Rutherford, NJ). In each erosion strip and filter strip three control covers were also used. In the erosion strip the covers were in chisel-plowed soil adjacent to the amended strip. In the grass filter strip, the covers were situated away from areas affected by surface runoff.

Potential Denitrification Assays. Just prior to rain simulation, we removed soil samples from the upper 15 cm of the erosion and filter strips and stored them at 4 °C until rain simulations were completed. We used the soil slurry method of Smith and Tiedje (1979) to measure potential denitrification activity in well-mixed, acetylene-amended samples.

Gas Analysis. We measured N_2O with an electron capture detector (ECD) on a Varian 3700 gas chromatograph fitted with 2 m Porapak Q columns. Analysis conditions for the ECD were: detector temperature, 360 °C; column temperature, 60 °C; carrier gas, 95% argon, 5% methane; carrier gas flow 30 mL min^{-1}; and sample volume, 1.0 mL. Machine response to N_2O was measured and compared to standard curves for N_2O generated from gas standards of known concentration.

Chemical Analyses. Soil samples for analysis of NO_3^- and NH_4^+ concentration before rain simulation were taken from the 0 to 5 cm. The soil was uniformly mixed, extracted with 1 M KCL, and analyzed using a Technicon Auto Analyzer System III.

Microbial Analysis. We determined the denitrifier most probable number (MPN) in composite soil samples of ten cores from 0 to 15 cm depth in both the erosion strips and the filter strips that were sampled immediately before rain simulation. We used the denitrifier MPN procedure as outlined by Tiedje (1982). A ten-fold serial dilution of soil in physiological saline (8 g L^{-1} NaCl in distilled H_2O) was used to inoculate 5 replicate tubes per dilution. Growth media was Tryptic Soy Broth with 1 g L^{-1} KNO_3. The tubes were incubated 28 days at 26 °C and residual NO_3^- was detected with diphenylamine in concentrated sulfuric acid. The MPN denitrifiers was determined using published tables (Alexander, 1982).

Statistical Analysis. Treatment means were analyzed for statistical significance by one and two-way ANOVA using the statistical packages contained in Microsoft ® Excel.

CHAPTER III – Data and Results

1992 - Fecal Contamination in Surface Runoff

Since grass filters are one of the most accessible technologies to control surface runoff, and since we had extensive experience with grass filters as a management tool for soil erosion, we decided to examine whether grass filters were an adequate management practice to control both bacteria and soil runoff from poultry waste amended fields. The objective of this study was to obtain field data to determine if fecal coliform trapping by grass filter strips intercepting runoff from a poultry-manured soil was comparable to soil trapping from the same runoff.

The rain simulation imitated a worst-case event in which waste application was followed after only a brief interval by a high intensity rain. Longer intervals between waste application and potential runoff, and a rain of lesser intensity and duration would decrease erosive loss of fecal coliforms and soil (Crane et al., 1983). Since the grass filter strips were not rained on, our results must be interpreted with some caution since rain falling on the grass filter strips would help keep both soil particles and bacteria in suspension.

Surface runoff from the plots is shown in Fig. 7 and Fig 8. Runoff rates were typically higher in erosion strips abutting 4.5m filter strips. By the end of simulated rainfall maximum runoff rates were between 60 and 100 L min^{-1}. The exception was Plot 1. Trapping efficiency for water was quite variable (Table 1). The average trapping efficiency of 9.0 m filter strips was 84.3% in 9.0 m filter strips and 65.9% in 4.5 m filter strips.

Total soil loss corresponded with increased surface water runoff, and was dramatically reduced in grass filters compared to erosion strips (Figs. 9 and 10). Excluding Plot 1 (in which there were mechanical difficulties during rain simulation) the sediment trapping efficiency of 4.5 m filter strips was 96.3 % while 9.0 m filters trapped sediment with 99% efficiency (Table 2).

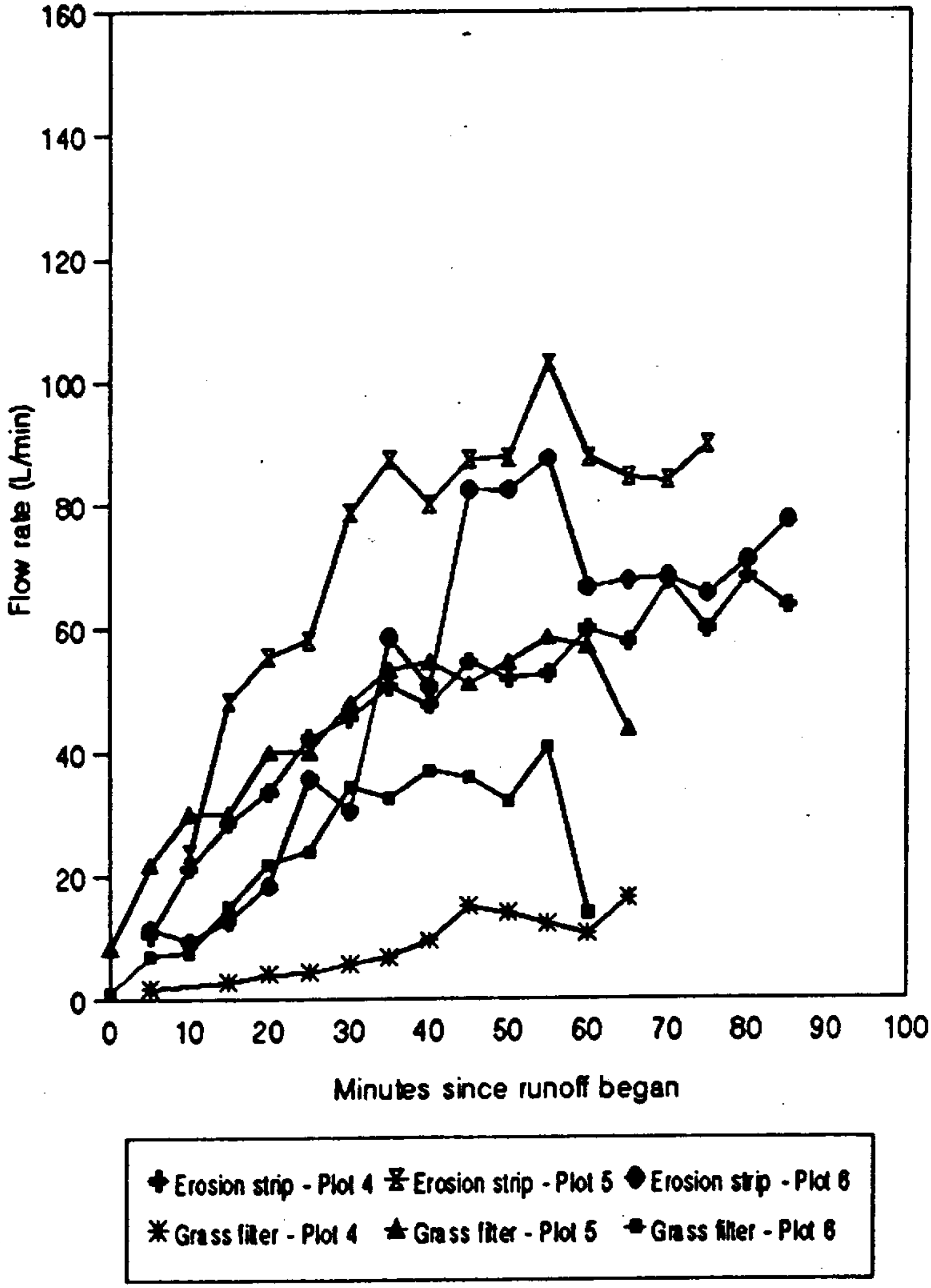

Figure 7. 1992 — Surface flow from plots with 4.5 m filter strips.

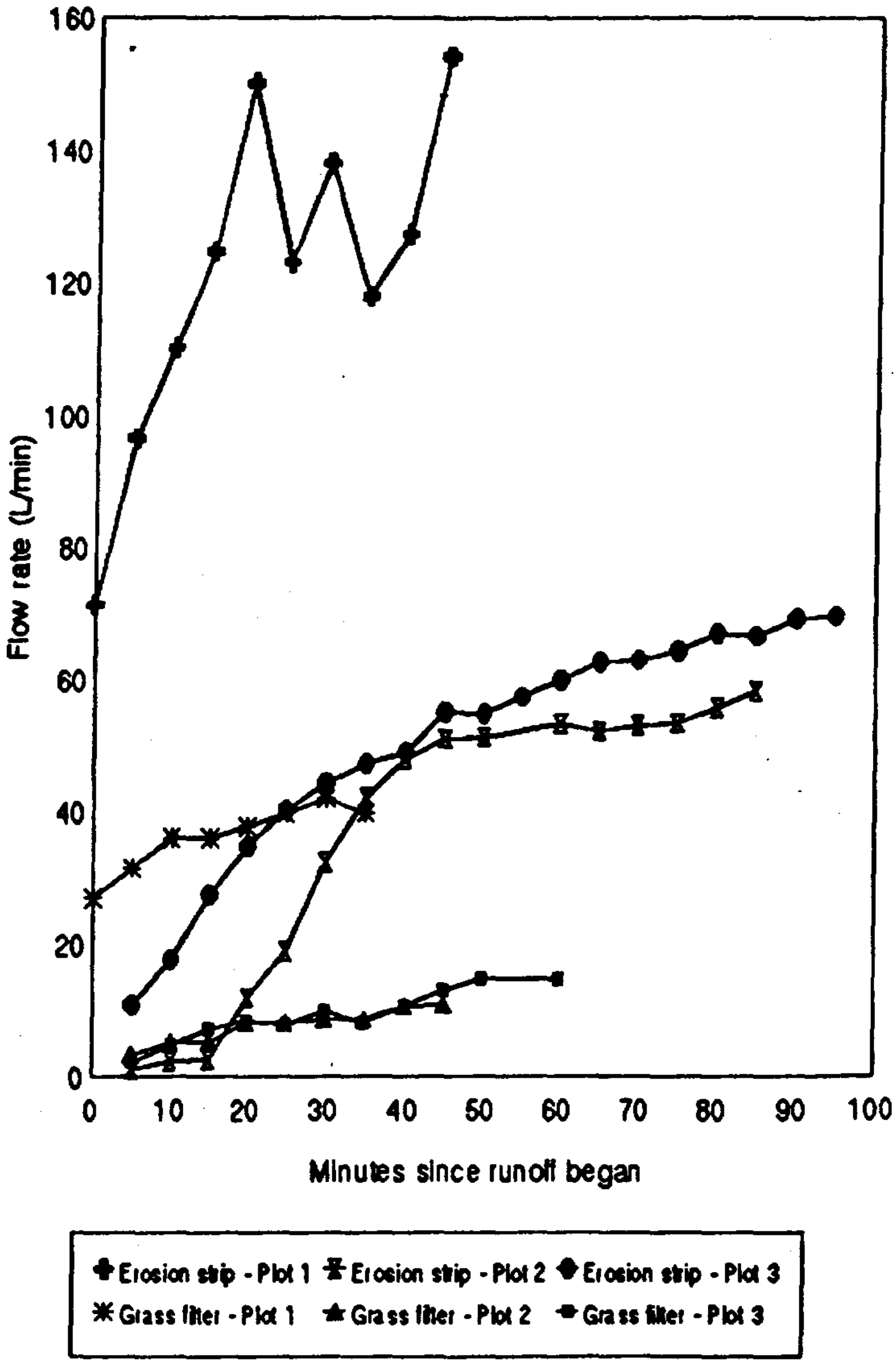

Figure 8. 1992 — Surface flow from plots with 9.0 m filter strips.

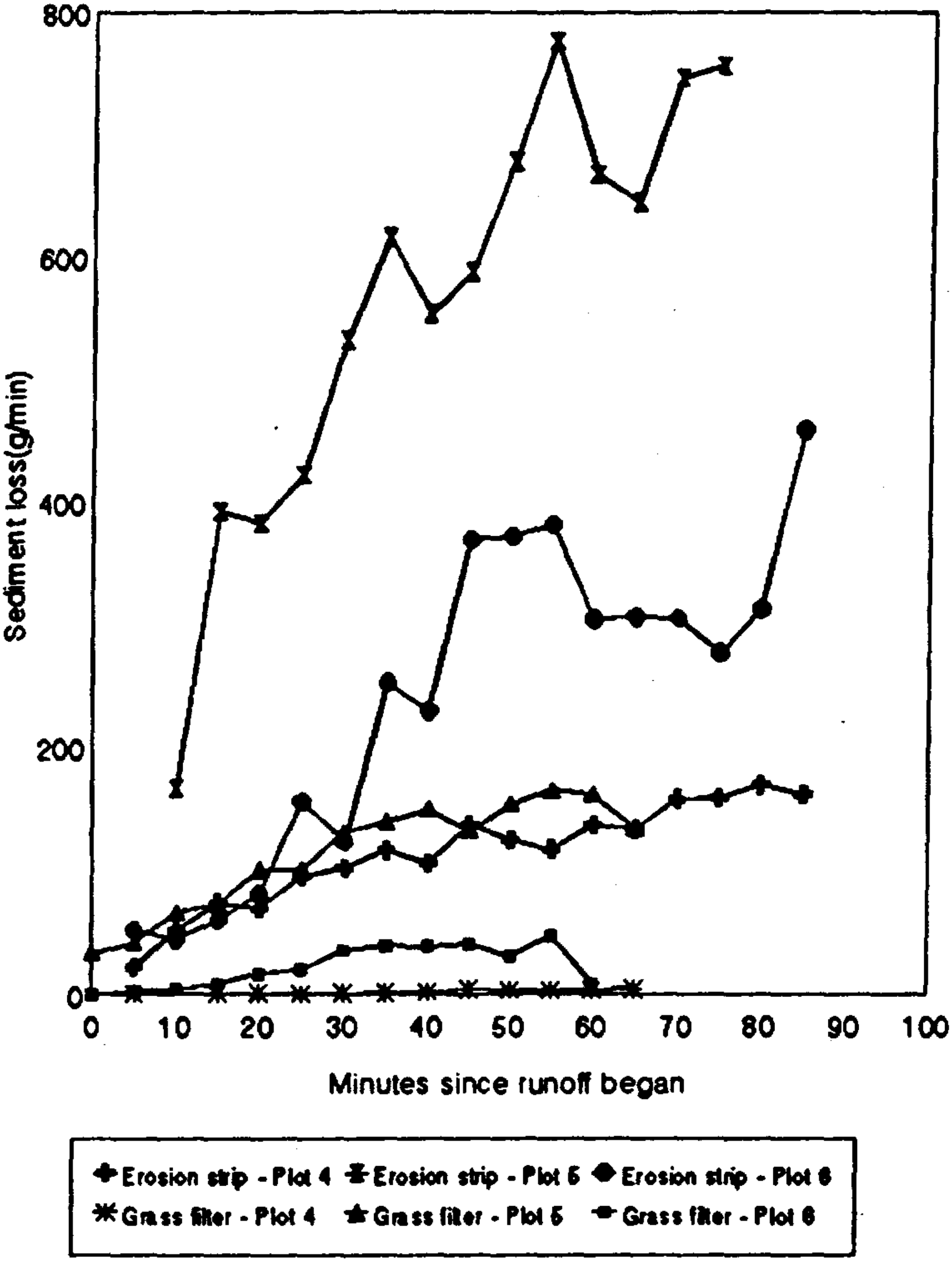

Figure 9. 1992 — Sediment runoff from plots with 4.5 m filter strips.

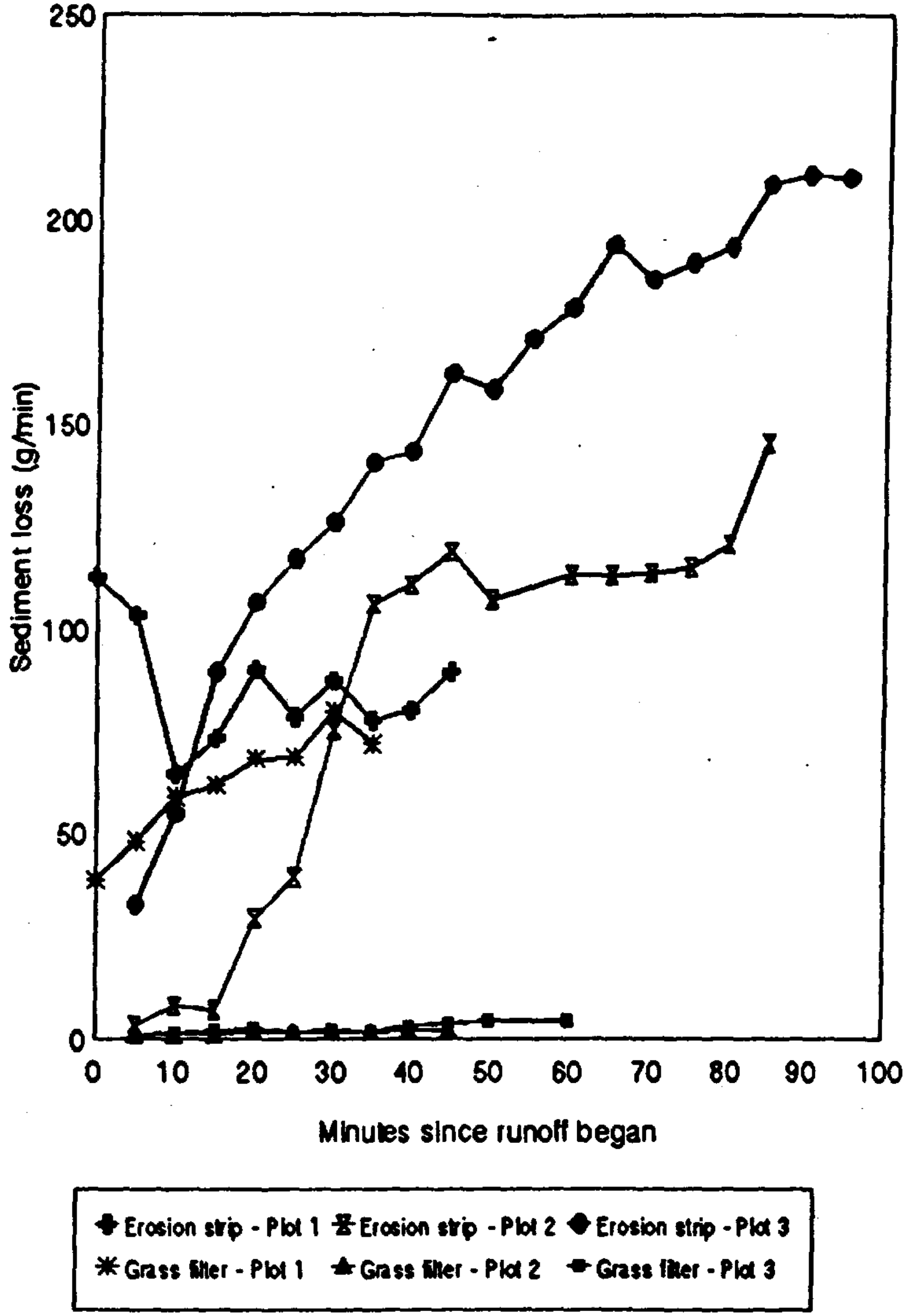

Figure 10. 1992 — Sediment runoff from plots with 9.0 m filter strips.

Table 1. 1992 - Total water inflow, outflow, and trapping efficiency.

Plot	Filter length (m)	Liters Inflow	Liters Outflow	Percent Trapped
1	9.0	6063	1445	76.2
2	9.0	3195	343	89.3
3	9.0	4570	582	87.3
4	4.5	4080	516	87.4
5	4.5	5390	3005	44.0
6	4.5	4480	1514	66.2

Table 2. 1992 - Total sediment inflow, outflow, and trapping efficiency.

Plot	Filter length (m)	Grams Inflow	Grams Outflow	Percent Trapped
1	9.0	4274	2472	42.2
2	9.0	7186	66	99.1
3	9.0	14344	155	98.9
4	4.5	9734	158	98.4
5	4.5	365915	7969	97.8
6	4.5	20494	1497	92.7

Madison et al. (1992) previously observed efficient soil trapping on these plots under similar rainfall conditions. The soil trapping efficiency is potentially overestimated, however, because the grass filter strips were not rained on simultaneously with the erosion strips. Although not directly comparable with our experiment, Hayes et al. (1984) reported trapping efficiencies of 94-99% in a saturated filter strip which received a sediment plume, while Albrecht and Barfield (1981) observed soil trapping efficiencies of greater than 98% in grass filters down slope of surface mines which received natural rainfall. Gross et al. (1991) indicate that even low density turf stands greatly affect soil runoff.

Technical difficulties prevented us from obtaining representative data for all but fecal coliforms in Plots 2 and 3. In Plot 2, maximum fecal coliform loss in erosion strip runoff occurred between 30 and 40 minutes after simulated rain began (Fig. 11). The data for fecal coliform loss in Plot 3 showed that fecal coliforms had peaked or were already declining within 10 minutes (Fig. 12). Fecal coliform trapping was not as effective as soil trapping. In Plot 2, the apparent trapping efficiency of the grass filters for fecal coliforms was 74%. In Plot 3 the apparent trapping efficiency was only 43% (Table 3).

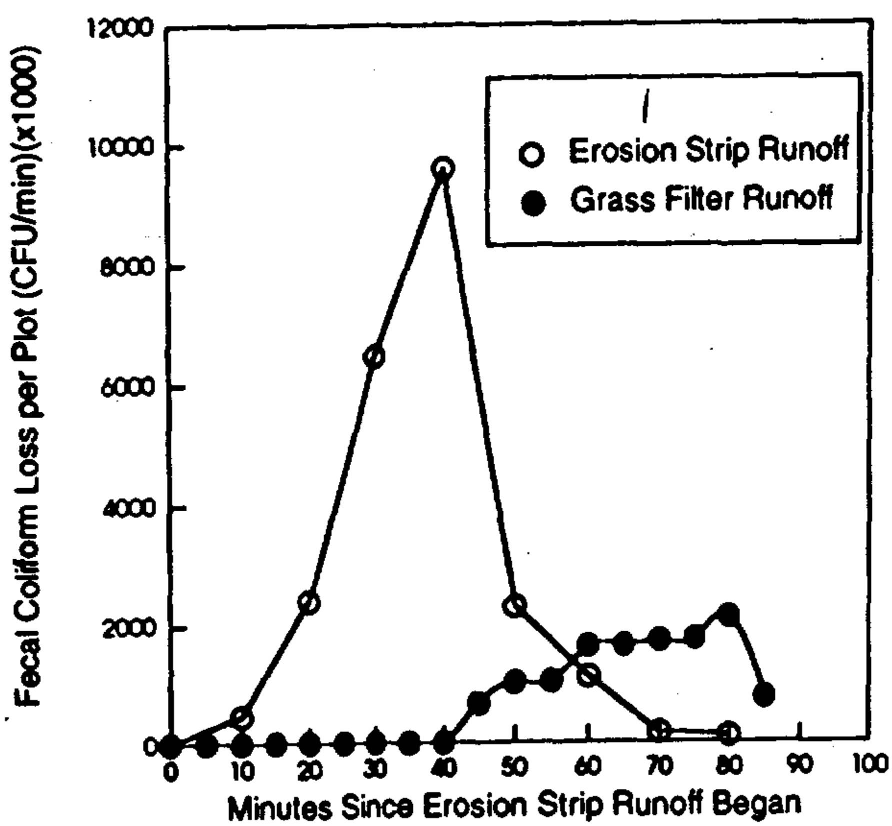

Figure 11. 1992 — Fecal coliform loss in surface runoff from erosion strips and grass filters — Plot 2

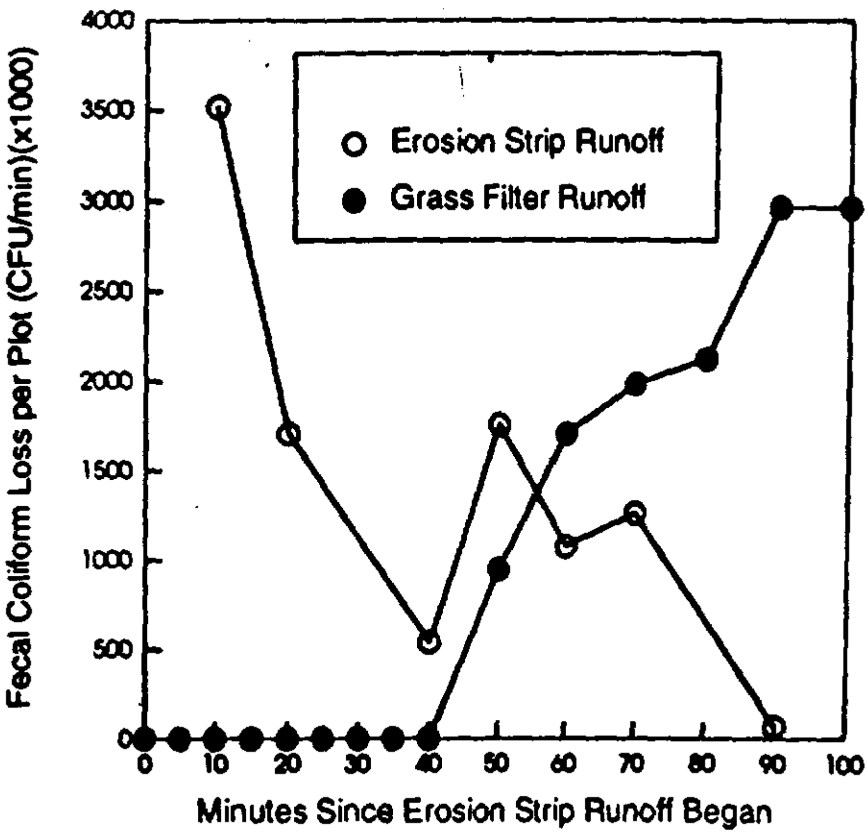

Figure 12. 1992 — Fecal coliform loss in surface runoff from erosion strips and grass filters — Plot 3

Table 3. Summary of fecal coliform trapping (per plot).

Plot	Peak Fecal Coliform Inflow Rate (CFU/min) †	Peak Fecal Coliform Outflow Rate (CFU/min)	Total Fecal Coliform Inflow (CFU)	Total Fecal Coliform Outflow (CFU)	Percent Fecal Coliforms Trapped
2	9,580,000	2,120,000	225,793,000	59,213,000	74
3	3,473,000	2,960,000	144,929 ,000	82,968 ,000	43

†CFU = Colony forming units

Fecal coliform runoff occurred more rapidly in Plot 3 than Plot 2. Consequently, the frequency of bacterial sampling in this plot probably did not reflect the period of maximum bacterial runoff and the apparent trapping efficiency of the grass filter for fecal coliforms in Plot 3 could be underestimated.

Dickey and Vanderholm (1981) found that filter strips did not greatly reduce bacteria levels in surface runoff. Young et al. (1980) found that grass filters reduced fecal coliforms by about 70% and that there was a linear relationship between filter strip length and bacteria removal. Compared to Young et al.'s study, the grass filter in our experiment had a greater slope (9% vs. 4%) and shorter length (9 m vs. 27.4 m) (1980). Consequently, runoff velocity was probably greater, and the area permitting infiltration and trapping reduced.

We observed channelized flow which meant that much of the grass filter did not participate in runoff filtration. Albrect and Barfield (1981) noted that while filter strips effectively trap fine particles, they are less effective at trapping clay-sized particles. Fecal bacteria are 1 to 2 mm or smaller in diameter and would behave much like clay particles in terms of solution transport.

Fecal coliform loss in erosion strip runoff peaked and then declined as one would expect from a finite source of manure (Fig. 11, 12). In contrast, the decline in fecal coliform loss in grass filter runoff was more gradual. We did not continue rain simulation long enough to determine if fecal coliform loss from the grass filters persisted. We suspect that the upper edge of the grass filter acted as a reservoir for fecal coliforms, which were steadily released as rain continued. We did not try to examine, however, whether fecal coliforms in the grass filters were unassociated or adsorbed to soil and vegetation .

The fecal coliform concentration in erosion strip soil prior to rain was higher in Plot 2 than Plot 3 (Table 4). By covering the erosion strips with plastic tarpaulins during a period when maximum daily temperatures consistently exceeded 27 °C we probably increased fecal coliform mortality.

Fecal coliforms were not detected in the grass filters before simulated rain . After simulated rain the number of fecal coliforms in the grass filters increased as expected (Tables 4). The same was true of fecal streptococci (Table 5). It is not surprising that the fecal coliform concentrations in soil from the grass filter strips after rainfall were higher than in the erosion plots. Since the upper edge of the filter strip trapped sediment from a much larger area (0.01 ha) this probably represents a concentration effect.

Table 4. 1992 - Fecal coliform concentrations in soil before and after simulated rain.

Sample	Depth (cm)	Plot 1	Plot 2	Plot 3	Plot 4	Plot 5	Plot 6
		---------- - log CFU†/g dry soil ----------					
Erosion Strip							
Before	0-15	nd‡	2.8	3.2	2.4	2.0	1.4
	15-30	nd	nd	1.1	0.8	nd	0.4
After	0-15	3.8	nd	nd	nd	3.6	4.0
	15-30	1.9	nd	nd	nd	1.4	1.0
Grass Filter							
After	0-15	nd	3.1	3.0	0.8	1.5	3.5
	15-30	nd	1.1	nd	nd	nd	2.0

†CFU = Colony forming units

‡nd = not detected, detection limit is 3 CFU g^{-1} soil

Table 5. 1992 - Fecal streptococci concentrations in soil before and after simulated rain.

Sample	Depth (cm)	Plot 1	Plot 2	Plot 3	Plot 4	Plot 5	Plot 6
		- - - - - - - - - - - log CFU/g dry soil - - - - - - - - - -					
Erosion Strip							
Before	0-15	4.6	3.8	3.3	1.4	nd†	2.4
	15-30	nd	4.0	1.8	1.7	nd	0.6
After	0-15	3.1	nd	1.4	1.9	2.1	3.1
	15-30	0.4	nd	1.1	nd	1.1	0.6
Grass Filter							
After	0-15	nd	3.2	2.2	1.1	2.0	2.2
	15-30	nd	2.1	nd	nd	nd	1.2

†nd = not detected

The increased concentration of fecal coliforms in filter strip soil represented only a fraction of the total fecal coliforms which entered the grass filters. The balance of fecal coliforms which remained in the grass filters presumably infiltrated the filter strip soil.

Bacterial contamination of agricultural waters often exceeds the primary contact standard of 200 fecal coliforms per 100 mL water (Walker et al., 1990). The fecal coliform concentration exceeded the primary water contact standard in runoff from the grass filters in every sample we analyzed. It should be noted, however, that this study reflected very heavy storm conditions and relatively recent manure addition. In an actual storm, some dilution of grass filter runoff would also occur. Nevertheless, even with the grass filters in place, significant numbers of fecal coliforms were lost in surface runoff.

1993 - Fecal Contamination in Surface Runoff

Agricultural land will be the final destination for most of Kentucky's poultry waste in the near future. We decided to study grass filter strips as a management practice to control fecal bacteria runoff from soils. We used a rain simulator to cause surface runoff from chisel-tilled plots amended with poultry litter. We then measured sediment, fecal coliform, and fecal streptococci trapping by grass filter strips that were either 9.0 or 4.5 meters long. Our objective was to assess and compare the trapping efficiency of grass filter strips for these runoff contaminants.

Surface flow. The maximum surface runoff from 13.7 m erosion strips was 25 to 45 L min^{-1} (Fig. 13A). There was at least an hour difference between the earliest and latest onset of surface runoff. Surface runoff began earliest in plots with the highest initial soil moisture content at 0 to 5 cm (Table 6). This was 28.5% in Plot 3, 19.4% in Plot 1, and 14.4% in Plot 2.

Table 6. Gravimetric soil moisture before rainfall simulation.

Sample	Depth (cm)	Plot 1	Plot 2	Plot 3	Plot 5	Plot 6
		%				
Erosion strip	0 - 5	19.4	14.4	28.5	17.0	22.4
	5 - 15	17.6	16.1	23.0	19.4	nd†
Grass Filter	0 - 5	23.7	16.5	22.4	20.5	21.5
	5 - 15	16.6	15.3	19.8	19.2	nd

†nd = not determined

The average surface runoff from 18.2 m erosion strips was 58 L min^{-1} by the end of rain simulation (Fig. 13B). Since surface runoff began after approximately the same amount of rainfall was added in plots 5 and 6, the duration of simulated rain was similar (Table 7). This was not the case with plots 1, 2, and 3.

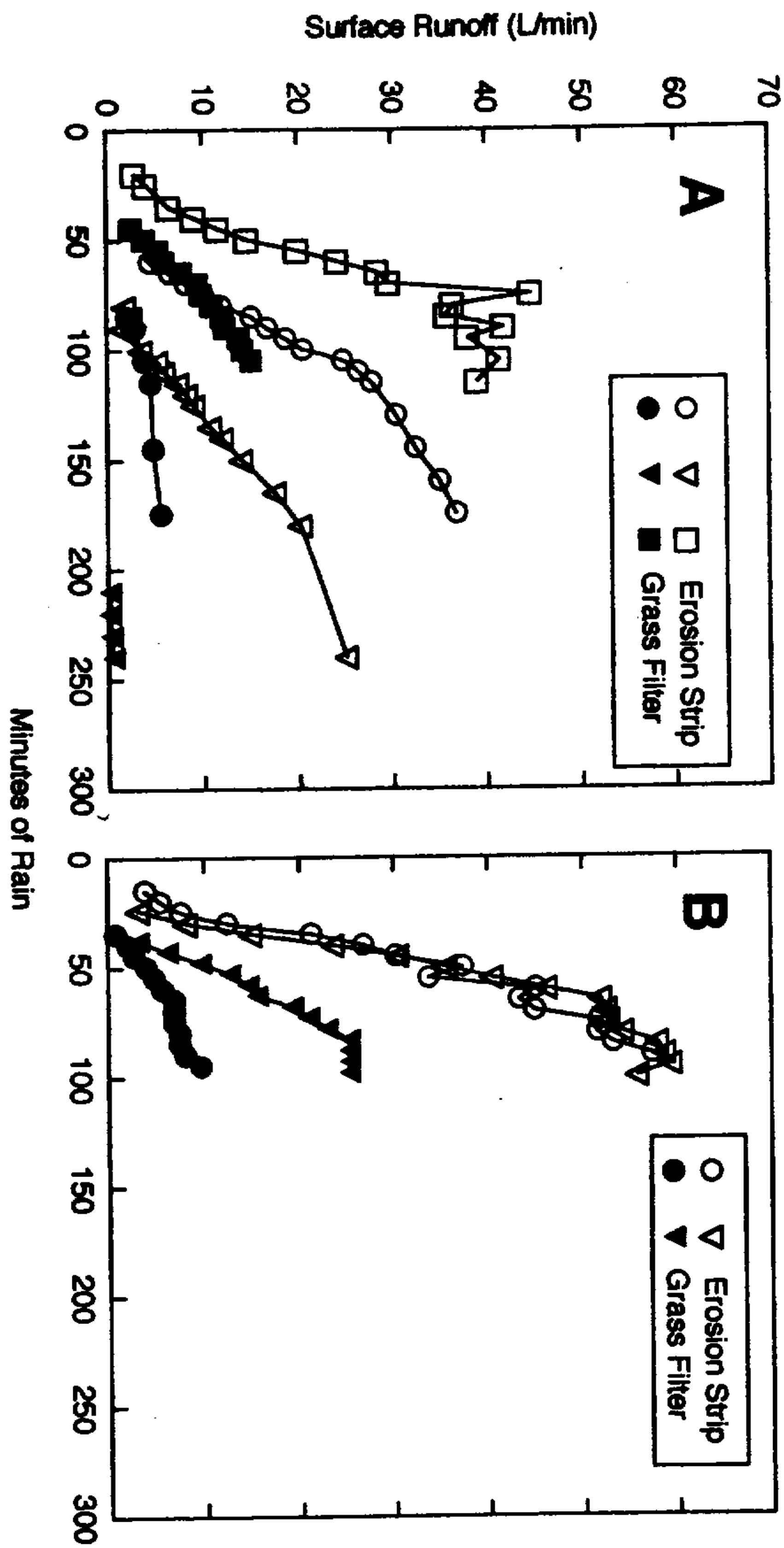

Figure 13. 1993 — Surface runoff from research plots during simulated rainfall (A) 9.0 m grass filter strip; (B) 4.5 m grass filter strip

Table 7. Surface runoff mass balance.

Filter strip length (m)	Plot	Minutes of rain	Total liters inflow (M_i)	Total liters outflow (M_o)	% Trapped
9.0	1	175	2830	199	93.0
	2	229	2378	10	99.6
	3	115	2618	569	78.3
4.5	5	96	2474	339	86.3
	6	108	3087	1057	65.8

In most cases, surface runoff from the grass filter strips began 15 to 25 minutes after we observed surface runoff from the erosion strips. In Plot 2 this period was 130 minutes. Plot 2 had the lowest rate of surface runoff (25 L min^{-1}) and was the only plot we studied in which the erosion strip was in no-tillage prior to 1992. Infiltration was also greater in Plot 2 than the other plots. Based on the amount of rain applied to each plot and the total surface runoff we observed, 84% of the rain applied to the erosion strip in Plot 2 infiltrated the soil. Infiltration in the other erosion strips was 76% in Plot 1, 66% in Plot 3, 71% in Plot 5, and 68% in Plot 6.

The grass filters effectively trapped rainfall that did not infiltrate the erosion strips (Table 7). The 9.0 m grass filter strips trapped an average of 91% of the incoming water while the two plots with 4.5 m grass filter strips trapped 70% of the surface runoff. This difference was not significantly significant ($p<0.29$). With the exception of Plot 6 (in which 9% of the simulated rain became surface runoff) we recovered less than 5% of the water we applied during rain simulations as runoff from the grass filter strips.

Sediment loss. The grass filter strips effectively reduced sediment in surface runoff. The sediment concentrations in surface runoff leaving 18.2 m erosion strips and the 4.5 m grass filter strips abutting them were higher than the concentrations leaving 13.7 m erosion strips and 9.0 m grass filter strips (Fig. 14A, B). However, sediment trapping efficiency was effective in both filter strip lengths. The average sediment trapping efficiency in 9.0 m grass filter strips was

98.6% (Table 8). In 4.5 m filter strips it was 95.2%. This difference was not statistically significant ($p<0.18$).

Table 8. Sediment mass balance in surface runoff.

Filter strip length (m)	Plot	Total grams inflow (M_i)	Total grams outflow (M_o)	% Trapped
9.0	1	9779	82	99.2
	2	8445	3	100.0†
	3	11340	384	96.6
4.5	5	15955	455	97.2
	6	26500	1820	93.1

† Due to rounding

In Plot 6, sediment concentrations in runoff continually increased (Fig. 14B). In the remaining plots, once runoff started, sediment concentrations were relatively constant for the duration of simulated rain. Neither the duration of rain nor the total surface flow correlated well with sediment loss.

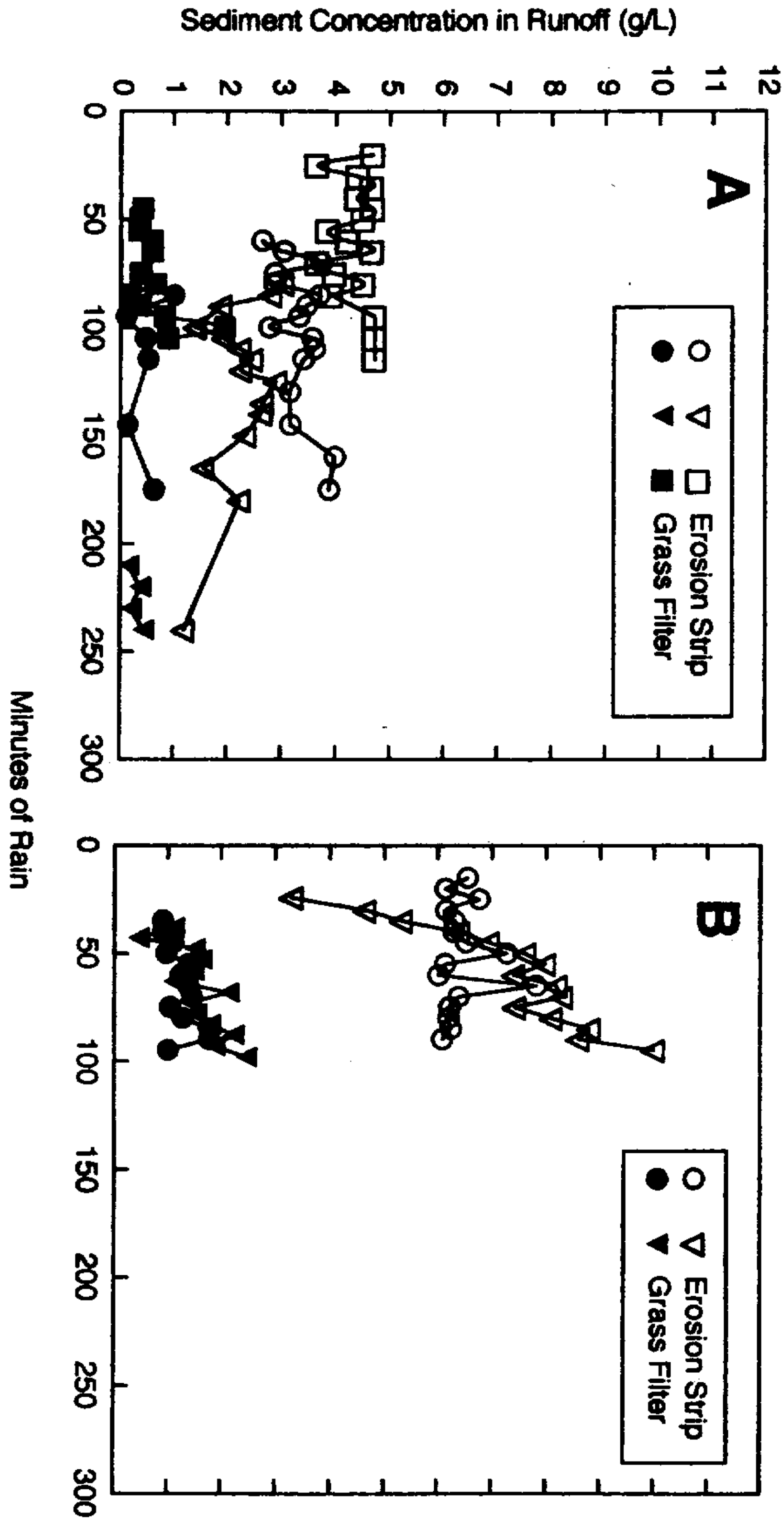

Figure 14. Sediment runoff from research plots during simulated rainfall
(A) 9.0 m grass filter strip; (B) 4.5 m grass filter strip

Fecal bacteria runoff. Our data for bacterial runoff from Plot 2 contained too few samples to assess trapping efficiency and are not included in these results. The remaining data are shown in Fig. 15 and Fig. 16. The pattern of fecal bacteria runoff was similar in all plots. We measured high fecal bacteria concentrations as soon as we began collecting runoff. Unlike the sediment runoff, these concentrations usually declined in the erosion strip runoff with time. In the grass filter runoff, there was little or no decline in fecal bacteria concentrations during the course of rain simulation relative to what we observed in erosion strip runoff.

By the end of rain simulation, the fecal coliform and fecal streptococci concentrations we measured in runoff from the grass filter strips of every plot exceeded the concentrations we measured in erosion strip runoff. Fecal bacteria concentrations in runoff from both locations always exceeded 10^5 colony forming units/100 mL and were generally much higher. These concentrations exceed the standards for fecal contamination of primary contact water (200 fecal coliforms/100 mL).

The total number of fecal bacteria in erosion strip runoff were reduced during passage through the grass filter strips (Table 9). Fecal coliform and fecal streptococci trapping in 4.5 m grass filter strips averaged 75% and 56%, respectively while in 9.0 m filter strips average fecal coliform trapping was 90% and average (Table 9) fecal streptococci trapping was 77% (Table 9). However, because of the limited replication and plot-to-plot variability, in neither case were these differences significant ($p<0.32$ for fecal coliforms and $p<0.39$ for fecal streptococci).

Fecal bacteria in soil. The fecal coliform concentration in the first 5 cm of soil immediately before rain simulation was between 10^4 and 10^5 colony forming units per gram of soil (Table 10). Initial fecal streptococci populations varied more and ranged from approximately 10^4 to 10^6 colony forming units per gram of soil. With the exception of Plot 6, the fecal bacteria concentrations in the grass filter strips were close to, or below, our detection limit of 3 colony forming units per gram of soil.

Table 9. Mass balance of fecal bacteria in surface runoff.

	Filter strip length (m)	Plot	Total inflow (M_i)	Total outflow (M_o)	% Trapped
			-- Colony Forming Units --		
Fecal coliforms					
	9.0	1	1.6×10^{10}	1.7×10^{9}	89.4
		2	3.5×10^{10}	nd†	nd
		3	5.8×10^{11}	5.0×10^{10}	91.4
	4.5	5	4.0×10^{11}	2.0×10^{10}	95.0
		6	9.8×10^{10}	4.5×10^{10}	54.1
Fecal streptococci					
	9.0	1	4.2×10^{10}	7.8×10^{9}	81.4
		2	6.4×10^{11}	nd	nd
		3	3.4×10^{11}	9.5×10^{10}	72.1
	4.5	5	1.5×10^{12}	1.3×10^{11}	91.3
		6	1.4×10^{11}	1.1×10^{11}	21.4

† nd = not determined (insufficient data to calculate mass balance)

In most plots, the fecal bacteria concentration in the erosion strips did not change much after simulated rain. The exception was Plot 6, where the fecal coliform concentration from the first 15 cm of soil decreased by almost 3 orders of magnitude. Plot 6 also had the greatest sediment loss of any erosion strip (Table 8). In Plots 1 to 5, fecal bacteria concentrations in soil from 5 to 15 cm deep increased between 0.2 and 2 orders of magnitude (data not shown).

Fecal bacteria concentrations, as expected, typically increased in the grass filter strip soil after rain simulations (Table 10). We did not find a consistent increase or decrease in the fecal bacteria populations from grass filter strip soil at depths of 5 to 15 cm.

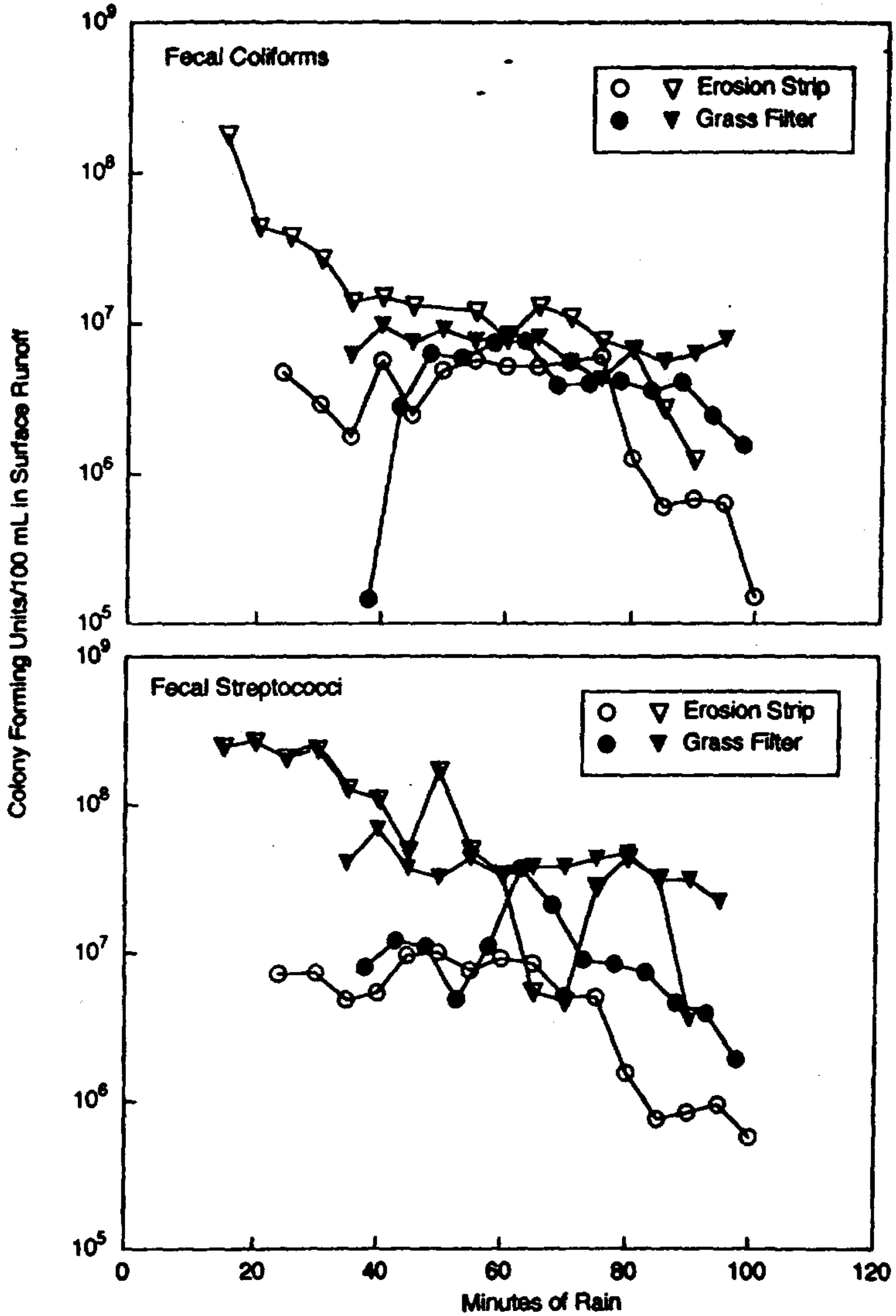

Figure 15. Fecal bacteria runoff from research plots with 4.5 m grass filter strips during simulated rain

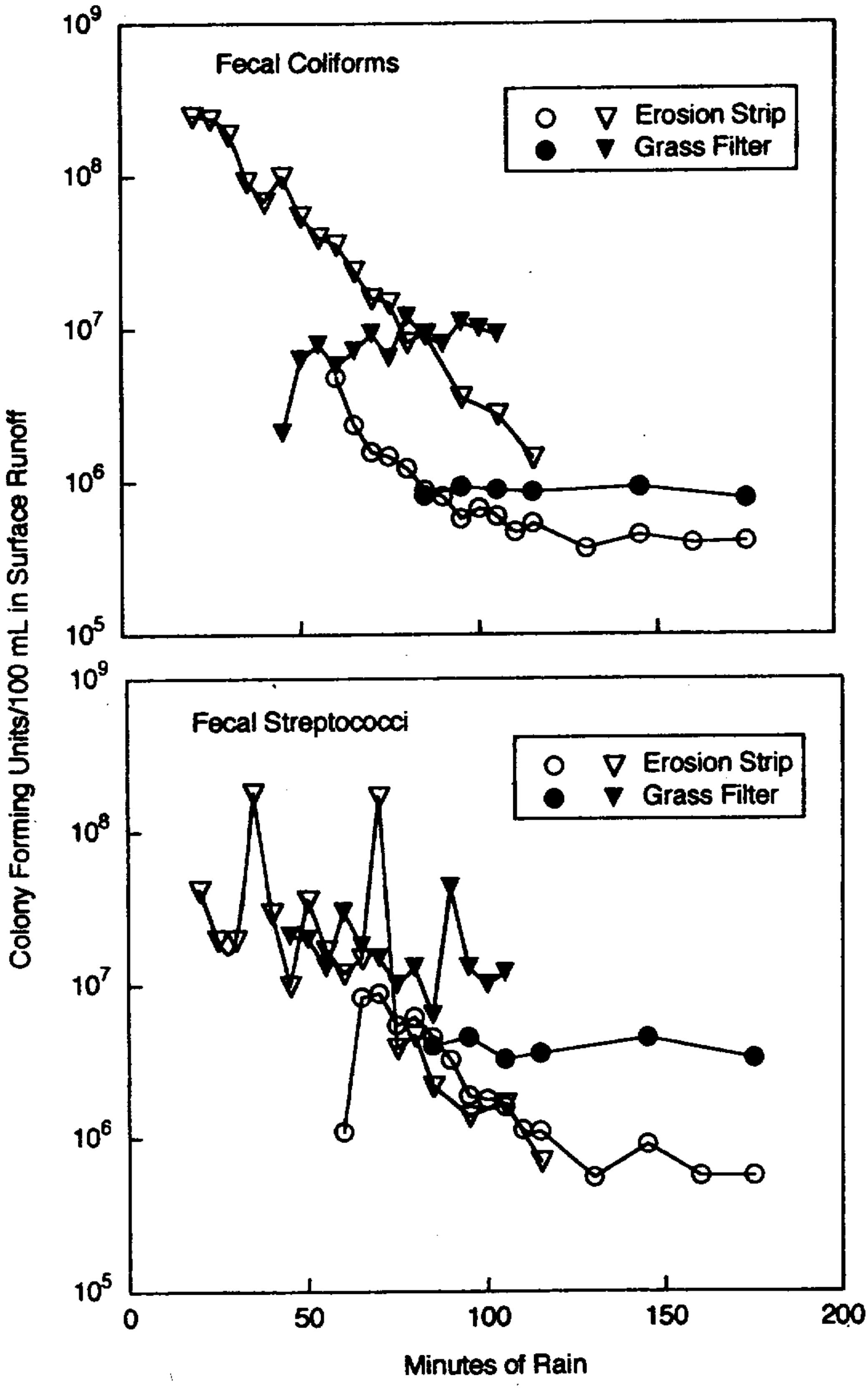

Figure 16. Fecal bacteria runoff from research plots with 9.0 m grass filter strips during simulated rain

Table 10. Fecal bacteria in soil at 0 to 5 cm depth before and after rain simulations.

	Plot	Erosion Strip Before	Erosion Strip After	Grass Filter Before	Grass Filter After
		------------- Log_{10} Colony Forming Units	---------------		
Fecal coliforms	1	4.4	4.8	0.9	0.6
	2	4.8	6.6	nd†	2.8
	3	4.6	4.6	1.0	2.2
	5	4.0	3.9	nd	2.6
	6‡	4.2	1.6	1.5	2.8
Fecal streptococci	1	3.9	4.4	1.0	nd
	2	6.1	5.6	nd	2.1
	3	4.5	4.1	1.4	2.1
	5	4.7	4.8	nd	2.4
	6‡	4.0	1.7	3.5	1.2

† nd= not detected
‡ 0 - 15 cm used for all measurements

1992 - Denitrification in Filter Strips

Our goal was to assess the N_2O flux in poultry manured soil immediately after rain in comparison to reported N_2O flux measurements from similar agricultural settings. We also wanted to assess N_2O flux, as a measure of denitrification, from grass filters receiving the runoff from poultry-manured fields.

Nitrous oxide flux rates immediately after simulated rain were greater in the erosion plots than they were in the grass filters (a = 0.05; Table 11). The average coefficient of variation in flux rates between plots was 56% in erosion plots (range 25 to 100%) and 91% in grass filters (range 36 to 131%). This spatial variability is not unique for N_2O field measurements (Goodroad et al., 1984; Mosier et al., 1986). The average N_2O flux rate in Plots 1, 2 and 3 (775 mg N_2O-N m^{-2} h^{-1}), the first plots treated, was greater than the average N_2O flux rate in Plots 4, 5, and 6 (134 mg N_2O-N m^{-2} h^{-1}; a = 0.05). We did not find this difference in the grass filters.

Table 11. Mean N_2O evolution immediately after simulated rainfall in poultry-manured erosion plots and grass filters receiving their runoff.

Plot	Days after litter applied	Erosion plot †	Grass filter
		------------- mg N_2O-N m^{-2} h^{-1}	-----------
1	2	1000 ± 527	148 ± 177
2	8	559 ± 191	58 ± 76
3	10	763 ± 581	85 ± 44
4	15	149 ± 110	79 ± 57
5	17	51 ± 51	76 ± 83
6	22	201 ± 51	76 ± 83

† Mean of 5 soil covers ± 1 standard deviation.

Nitrous oxide fluxes have been measured in numerous environments. Based on an evaluation of various field experiments Eichner (1990) estimated that 2% of N fertilizer is lost as N_2O over a one year period in fertilized and manured soils. Average daily flux was 24 mg N_2O-N m^{-2} h^{-1} from ammonium-fertilized grass and 7 mg N_2O-N m^{-2} h^{-1} from soil (Eichner, 1990). The N_2O flux immediately after rain in poultry-manured soils suggests that this flux could be much greater, albeit, for a short period. The flux rates we measured may underestimate the true N_2O flux rate due to the solubility of nitrous oxide in water. Nitrous oxide has an adsorption coefficient between 0.544 and 0.472 mL N_2O per mL of water from 25 to 30 °C (Tiedje, 1982). This spans the soil temperatures observed during flux measurements.

Goodroad et al. (1984) found mean N_2O flux from a manure-amended, no-till corn experiment in Wisconsin was about 100 mg N_2O-N m^{-2} h^{-1}. However, during spring thaw, when soils were presumably saturated, N_2O flux reached 634 mg N_2O-N m^{-2} h^{-1}. A period in their study, comparable to conditions we created with our rainfall simulation, occurred during the first rainfall after manure addition to soil. Goodroad et al. (1984) measured a N_2O flux of about 400 mg N_2O-N m^{-2} h^{-1}. This is comparable to the average N_2O flux rates we observed in our first three plots after simulated rain. Mosier et al. (1986) observed that in irrigated corn, fertilized with 200 kg N ha^{-1} as NH_4SO_4, N_2O emissions peaked at 565 and 504 mg N_2O-N m^{-2} h^{-1} after a 7.8 cm and 7.0 cm rain, respectively.

Nitrous oxide may be evolved during autotrophic and heterotrophic nitrification (Robertson and Tiedje, 1987) and NO_3^- respiration (Smith and Zimmerman, 1981) as well as during denitrification. We cannot rule out the first three processes as the source of N_2O because we did not selectively inhibit them (Robertson and Tiedje, 1987). However, the intensity and duration of rainfall would have created soil conditions favorable for denitrification.

The NO:N_2O ratio may also be a useful indicator of whether N-oxide flux comes from denitrification or some other process (Davidson, 1991). Anderson and Levine (1986) found that the NO:N_2O ratio was 0.01 to 0.3 for denitrifiers and 0.9 to 5.6 for nitrifiers and NO_3^- respirers. The Vacutainers we used for gas sampling were contaminated with NO. However, even with this background NO, the NO:N_2O ratio was < 0.01 (data not shown) which suggests that, initially, denitrification was the principle source of N_2O from these plots.

What could account for the different flux rates between the erosion plots and grass filters, and the dramatic difference in N_2O flux rates between the first three erosion plots and the last three? We suspected that if denitrification were a major N_2O source, different denitrifier population size might be responsible. When examined, Plots 1, 2 and 3 had 10 fold greater denitrifier MPN than Plots 4, 5 and 6 (Table 12). We believe this difference may be because erosion Plots 4, 5, and 6 were covered for an extended period by a plastic tarp when maximum daily temperatures exceeded 27 °C (Fig. 5). However, we did not test this hypothesis by comparing pre- and post-experiment denitrifier MPN.

The grass filters were not covered during the experiment, yet we also found a difference in the average denitrifier MPN of Plots 1, 2, and 3 and Plots 4, 5, and 6. If intrinsic soil properties varied across the experiment site from Plot 1 to Plot 6, they were not among the parameters we measured. Neither pH nor %C differed significantly across erosion plots or across grass filters (Table 12).

Table 12. Chemical and biological differences between erosion plots and grass filters.

Soil Property	Average of Plots 1, 2, and 3		Average of Plots 4, 5, and 6	
	Erosion plot	Grass filter	Erosion plot	Grass filter
pH	5.5	6.7 *	5.7	6.6*
% C	2.08	1.93	1.98	1.97
MPN † Denitrifiers	2.8×10^4	4.0×10^6	1.2×10^3	2.6×10^5

* Indicates a difference between the erosion plot and grass filter in each group of plots (t-test, $\alpha = .05$).

† MPN (Most Probable Number) of denitrifiers g^{-1} oven dry soil. The 95% confidence interval is ± 3.3 x MPN.

Grass filters had 100 fold greater denitrifier MPN and greater pH than erosion plots (α = 0.05; Table 12). So, denitrifier population size is an unsatisfactory explanation for the differences in N_2O flux rates between erosion plots and grass filters. A better explanation is that more NO_3^- or NH_4^+ was available in the erosion plots than the grass filters. In 1991, all erosion plots were fertilized with 170 kg N ha^{-1} as NH_4NO_3 and subjected to at least two separate rain simulations. The poultry manure we added to each erosion plot in 1992 contained about 304 kg N ha^{-1}. Fresh poultry manure typically consists of 25 to 30% urea and ammonium forms (Rasnake et al., 1991) some of which would have been lost to volatilization under the plastic tarps. The remaining ammonium and mineralizable N forms would nitrify over time.

We did not measure soil NO_3^- and NH_4^+ concentrations either before or after rain simulation. However, we measured NO_3^- concentrations in surface runoff. The maximum NO_3^- concentration in surface runoff from erosion plots increased from Plot 1 to Plot 6. This corresponds with the order in which simulations were done (Table 13).

Table 13. Concentration of NO_3^- -N in surface runoff.

Plot	Days After Litter Applied	Maximal		Final	
		Erosion plot	Grass filter	Erosion plot	Grass filter
		---------------------- mg NO_3^--N L^{-1}	----------------		
1	2	0.53	0.56	0.52	0.42
2	8	0.73	0.57	0.57	0.50
3	10	1.02	0.45	0.39	0.38
4	15	1.12	1.15	0.43	0.43
5	17	0.69	0.64	0.49	0.43
6	22	1.50	0.55	0.59	0.36

The only source of N added to the grass filters was runoff from the erosion plots. The final NO_3^- concentration in surface runoff from the erosion plots immediately before N_2O flux measurements began was greater than in grass filters ($\alpha = 0.05$; Table 13). This evidence, along with the fertilization history of the erosion plots, suggests that there was more available NO_3^- in the erosion plots than the grass filters at the start of N_2O flux measurements.

Was the potential N_2O flux from an erosion plot simply greater than in a grass filter? If part of the N_2O flux were associated with denitrification activity, denitrification potential in erosion plots and grass filters should reflect the different N_2O flux rates observed. Intact soil cores were removed from erosion plots and grass filters to test this hypothesis.

The coefficients of variation were considerably greater in our intact core studies than they were in our field measurements. The average coefficient of variation was 147% for erosion plot cores (range 102 to 205%) and 131% for grass filter cores (range 47 to 224%). There was no significant difference between the denitrification potential of the two sets of cores ($\alpha = 0.05$; Table 14).

Table 14. Mean N_2O evolution in acetylene-blocked, NO_3^--amended, intact soil cores from erosion plots and grass filters.

Plot	Erosion plot	Grass filter
	------------------- mg N_2O-N m^{-2} h^{-1}†	-------------------
1	393 ± 596	297 ± 483
2	2590 ± 3580	4810 ± 4310
3	2750 ± 2810	2710 ± 1260
4	3840 ± 7850	557 ± 1250
5	190 ± 279	629 ± 818
6	112 ± 155	nd‡

† Mean of 5 cores ± 1 standard deviation
‡nd - No Data

The average N_2O flux from erosion Plots 1, 2, and 3, 755 mg N_2O-N $m^{-2}h^{-1}$, was equivalent to 39% of the average total N-flux we observed in NO_3^--amended, acetylene blocked, intact cores from Plots 1, 2, and 3. (Table 11, 14). In contrast, the average N_2O flux we observed from the grass filters in Plots 1, 2, and 3, 97 mg N_2O-N m^{-2} h^{-1}, was only 4% of the average total N gas flux under acetylene blocked conditions. Groffman et al. (1991) suggested that surface runoff from manured soils might carry sufficient C to enhance denitrification in adjacent grass filters, thus removing NO_3^- before it reaches ground water. If denitrifying conditions were created in the grass filters, the smaller N_2O flux rates may simply be due to more N_2 production than in the corresponding erosion plots.

1993 - Denitrification in Filter Strips

Our objective was to quantify N_2O loss immediately after simulated rain from agricultural soil recently amended with poultry litter. A second objective was to compare that loss to the N_2O loss from grass filter strips receiving surface runoff from the litter-amended plots, and relate this to the potential denitrification activity in both the litter-amended plots and grass filter strips.

Six individual erosion strips were prepared but only 5 were used in this study. Plot 4 was washed out by a storm shortly after manure application and was not further examined. Nitrous oxide loss ranged from 2050 to 11120 mg N_2O-N $m^{-2}h^{-1}$ in the erosion strips and 160 to 1060 mg N_2O-N $m^{-2}h^{-1}$ in the filter strips (Table 15).

Table 15. Conditions during rainfall simulation experiments.

Plot	Date	Air temperature at time of run (°C)	Soil temperature † (°C)		Duration of rain (min)
			Erosion strip	Filter strip	
1	July 15	29.2	25.5 ± 0.3	24.3 ± 0.2	175
2	July 8	33.9	28.6 ± 0.4	26.7 ± 0.7	205
3	July 1	29.8	25.7 ± 0.5	24.1 ± 0.2	115
5	June 23	28.2	24.2 ± 0.3	22.5 ± 0.3	92
6	June 6	27.7	22.5 ± 0.2	22.3 ± 0.3	110

†Mean of 10 measurements ± one standard deviation at 0 to 15 cm soil depth.

The average N_2O loss from the erosion plots (8640 mg N_2O-N $m^{-2}h^{-1}$) was significantly greater ($P < 0.05$) than the average N_2O loss from the filter strips (760 mg N_2O-N $m^{-2}h^{-1}$). Nitrous oxide loss was spatially variable in each plot. The coefficient of variation (CV) for N_2O loss measurements in the erosion strips was 59% while in the filter strips it was 75%.

The N_2O loss we measured (average loss 8640 mg N_2O-N $m^{-2}h^{-1}$) greatly exceeded the N_2O loss we measured in 1992 from the same litter- amended soils. Coyne et al. (1994) applied all of their poultry litter at one time and covered their litter-amended soil with plastic tarps. The highest N_2O loss they observed was on the first plot treated. It is possible that much of the available N was volatilized with time. Rasnake et al. (1991) note that fresh poultry manure typically consists of 25 to 30% urea and ammonium forms. A contributing effect was the possibility that Coyne et al (1993), by covering the soil with plastic tarps in summer, effectively pasteurized the soil and limited N_2O loss by reducing the microbial community.

Nitrous oxide flux has been measured in many environments. Eichner (1990) estimated that 2% of N fertilizer is lost as N_2O over a one year period in fertilized and manured soils. Average daily N_2O flux ranged from 7 mg N_2O-N m^{-2} h^{-1} in soil to 24 mg N_2O-N m^{-2} h^{-1} in ammonium fertilized grass. Mosier et al. (1986) observed that in irrigated corn fertilized with 200 kg N ha^{-1} as NH_4SO_4, N_2O flux peaked at 565 and 504 mg N_2O-N m^{-2} h^{-1} after 7.8 cm and 7.0 cm rains, respectively. The N_2O loss immediately after rain in poultry litter-amended soil suggests that this flux could be much greater, although for a short period.

Nitrous oxide loss was high in some controls, particularly in the erosion strips (Table 16). We attribute this to placing the soil covers in areas just outside the plot border which may have been inadvertently amended with some poultry litter during the plowing operation. Nevertheless, in the erosion strips there was still a significant difference ($P < 0.05$) between the average N_2O loss in amended and control locations. Likewise, grass filters receiving runoff from the erosion strips had significantly greater N_2O loss ($P < 0.05$) than parts of the filters which received no runoff (average loss 760 vs. 170 mg N_2O-N $m^{-2}h^{-1}$) (Table 16).

We observed that N_2O loss was greater in erosion strips than grass filter strips. Both locations were rained on to a similar extent. Assuming that most of the N_2O loss was due to denitrification, one likely explanation is that more NO_3^- was available for denitrification in the erosion strips than the filter strips. This conclusion is supported by the significant difference between initial soil NO_3^- concentration between the soils in both locations. Since we did not measure the total gaseous N loss from these plots, a second explanation could be that more N_2O was reduced to N_2 in the filter strips than in the erosion strips. This would result in lower N_2O loss. The potential denitrification assays we conducted support this explanation. Under ideal conditions, in which N gas flux was as N_2O because of acetylene inhibition, N_2O production by the filter strip soil was greater than N_2O production by the erosion strip soil, even though MPN denitrifiers were slightly lower. Further study will be needed to resolve these alternative explanations. The N_2O production rates we observed for filter strip control and glucose-amended soil were about half the rate reported by Smith and Tiedje for similar assays on a loam soil (1979).

Table 16. N_2O evolution after simulated rainfall.

	Erosion strip		Filter strip	
Plot	Treated†	Control ‡	Treated	Control
	mg N_2O-N m^{-2} h^{-1}			
1	8120 ± 5610	376 ± 130	1060 ± 470	70 ± 5
2	2050 ± 1370	2318 ± 890	750 ± 470	340 ± 250
3	6990 ± 6350	1132 ± 850	430 ± 610	60 ± 20
5	11120 ± 2530	1127 ± 1250	620 ± 430	270 ± 210
6	6290 ± 2530	6830 ± 2920	160 ± 100	110 ± 80

† Mean of five replicates ± one standard deviation.

‡ Mean of three replicates ± one standard deviation.

The longer it rained the smaller the N_2O loss became (Figure 17). We also observed that the greater the initial soil NO_3^- (Fig. 18) or NH_4^+ (Fig. 19) concentration, the greater the N_2O loss from those strips. These trends were not as evident in the grass filters. Nitrate and NH_4^+ concentrations in soil before rainfall were both significantly higher ($P < 0.1$) in the erosion strips than the filter strips (Table 17).

Table 17. Soil NO_3^- and NH_4^+ concentration Of the 0 to 5 cm soil depth immediately before simulated rainfall.

Plot	NO_3^- † Erosion strip	NO_3^- † Filter strip	NH_4^+ Erosion strip	NH_4^+ Filter strip
	- - - - mg kg^{-1} NO_3^- -N - - - -		- - - - - mg kg^{-1} NH_4^+-N - - - - -	
1	20.6	9.9	98.9	8.7
2	18.5	4.4	66.7	8.2
3	53.9	4.4	125.7	8.1
5	54.8	3.6	176.6	5.0
6	246.7‡	2.7‡	165.6‡	2.7‡

† Sampled at 0 to 15 cm

We studied the potential denitrification rate in these soils by the denitrifying enzyme activity assay (DEA) using acetylene to block NO_3^- reduction at the level of N_2O (Smith and Tiedje, 1979). Potential denitrification was significantly greater ($P < 0.05$) in soil from the filter strips than from the erosion strips (Table 18). The greatest difference between these soils was in the glucose-amended samples. This result did not correspond with the MPN denitrifiers enumerated for these soil samples. In general, MPN denitrifiers were higher in the erosion strip soil than the filter strip soil (Table 19).

The greatest N_2O loss we observed was equivalent to about 2.7 kg N ha^{-1} d^{-1}. This only accounts for part of the N formed under these conditions. Nitrous oxide is relatively soluble in water (0.544 to 0.472 mL N_2O per mL H_2O between 25 and 30 °C) (Tiedje, 1982). Some N_2O may have also been further reduced to N_2. Firestone and Tiedje (1979) observed that N_2O was 40-90% of the major denitrification product between 3 and 33 hours after anaerobically incubating soils. Although the intensity and duration of rain would create conditions favorable for denitrification in our study, we cannot rule out other microbial sources of N_2O. Nitrous oxide may be evolved during autotrophic and heterotrophic nitrification (Robertson and Tiedje, 1987) and NO_3^- respiration (Smith and Zimmerman, 1981). We did not selectively inhibit either of these other two process.

We assume the N_2O formed primarily from denitrification for the following reasons. The K_m for NO_2^- reduction to N_2O by NO_3^- respirers is 0.9 mM, which is an unlikely concentration in soil (Smith, 1982). Davidson (1991) noted that heterotrophic and autotrophic N_2O evolution are primarily aerobic processes which decline significantly as water-filled pored space exceeds 60%. The soils in our experiment were near saturation immediately after rain. Nitrous oxide loss increased as the initial concentration of soil NO_3^- increased. This is consistent with denitrification as the route of loss. This relationship was also true with respect to N_2O loss and the initial NH_4^+ concentration. However, N_2O loss decreased the longer it rained, which suggests that the decline was due to leaching of a soluble anion like NO_3^- rather than a cation like NH_4^+. This supports denitrification as the dominant N_2O source.

Table 18. Potential denitrifying activity.

	Treatment †			
Sample site	Control	NO_3^-	Glucose	Glucose + NO_3^-
	----------- ng N_2O-N min^{-1} g^{-1} -----------------			
Erosion strip	1.3 ± 0.2‡	0.9 ± 0.2	1.4 ± 0.3	1.9 ± 0.3
Filter strip	2.0 ± 0.5	2.0 ± 0.5	4.2 ± 0.5	3.6 ± 0.6

† Control - no amendments
NO_3^- - 0.5 mg KNO_3 $flask^{-1}$
Glucose - 50 mg glucose $flask^{-1}$
Glucose + NO_3^- - 50 mg glucose $flask^{-1}$ + 0.5 mg KNO_3 $flask^{-1}$

‡ Average of five plots ± one standard error of the mean.

Table 19. MPN denitrifiers.

	MPN denitrifiers (g^{-1})	
Plot	Erosion strip	Filter strip
1	1.1×10^6	8.7×10^5
2	2.0×10^6	1.6×10^5
3	4.8×10^5	1.6×10^5
5	3.8×10^4	3.8×10^4
6	2.2×10^4	1.5×10^6

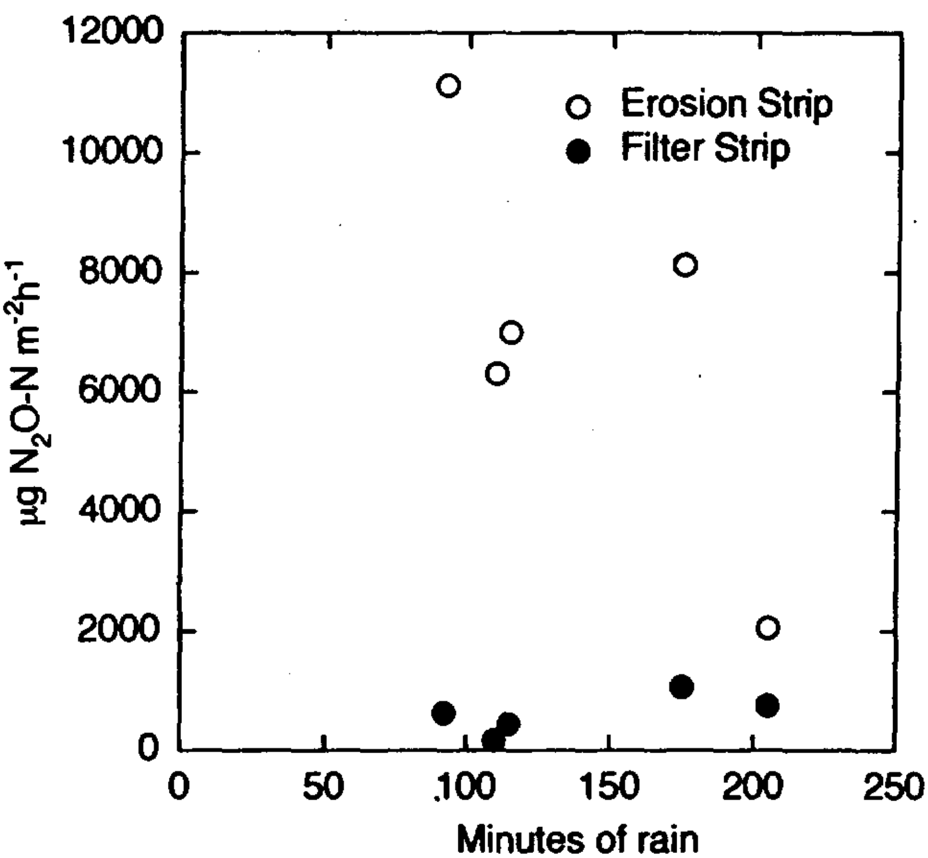

Figure 17. Relationship between N2O loss and rain duration

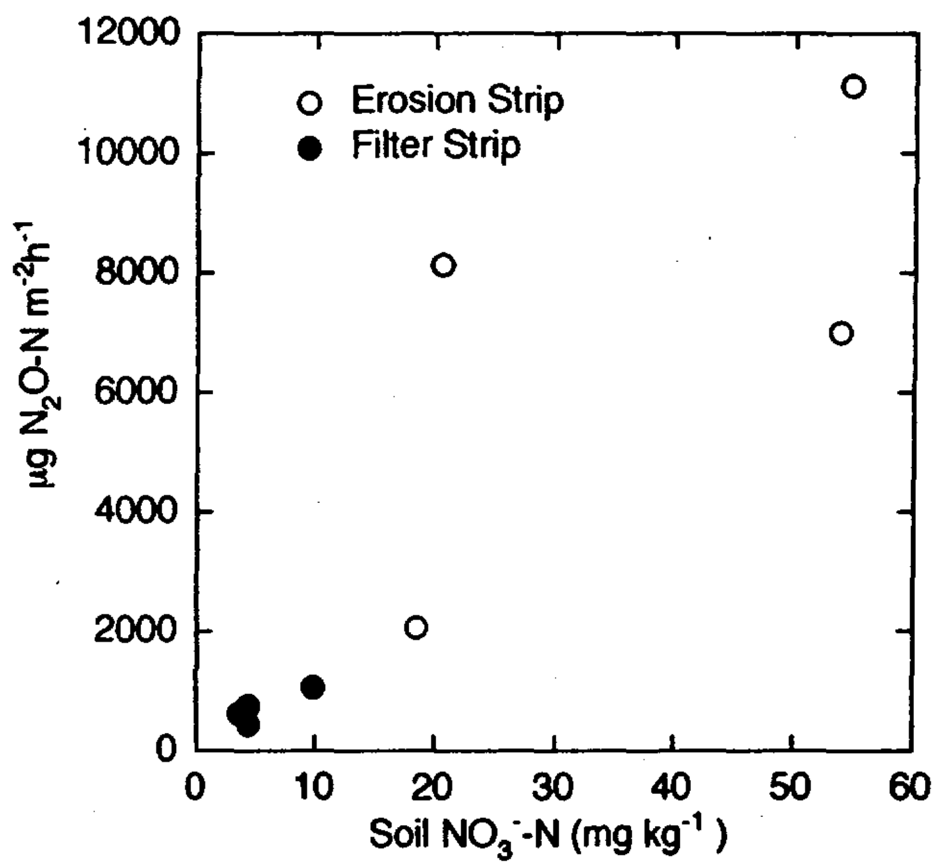

Figure 18. Relationship between N_2O loss and initial soil NO_3^--N concentration

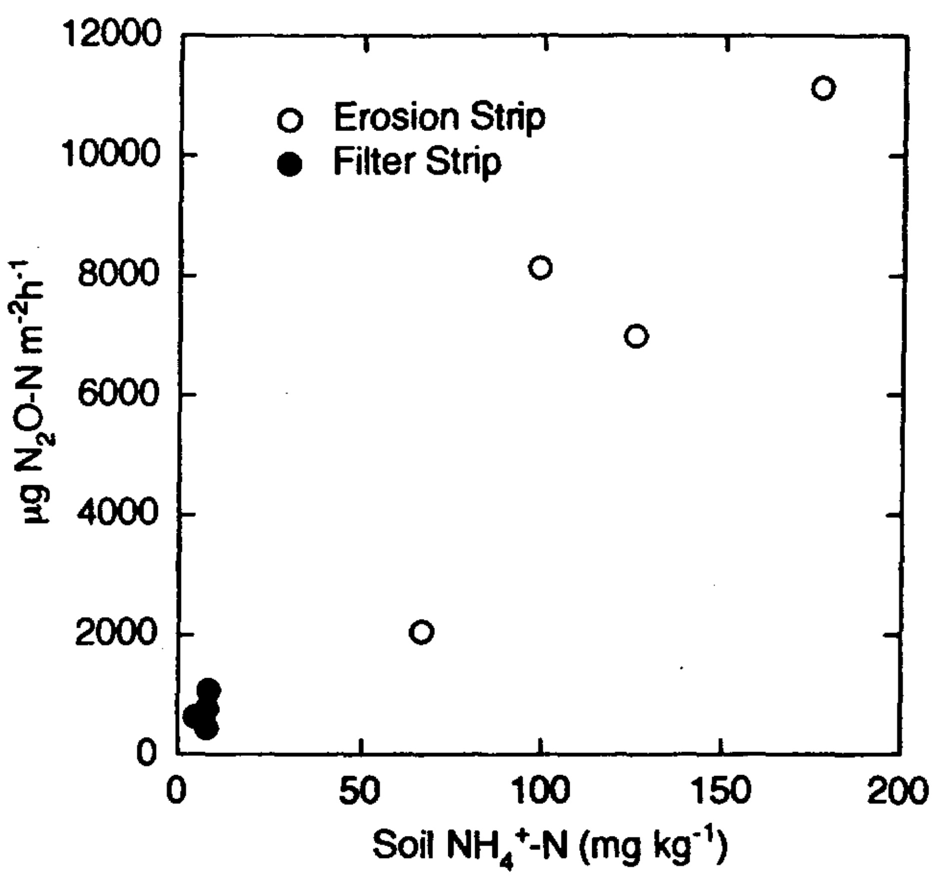

Figure 19. Relationship between N_2O loss and initial soil NH_4^+-N concentration

CHAPTER IV – Conclusions

1992 - Fecal Contamination in Surface Runoff

Grass filter strips are an effective management practice for controlling soil erosion. The 9 m grass filter strips we employed in this study trapped 99% of the soil from erosion strips subjected to simulated rain. These results cannot be uniformly extended to fecal coliforms in the same runoff. Under similar conditions, the grass filters only trapped up to 74% of the fecal coliforms. Heavy rain and rapid surface flow may keep fecal coliforms in solution while denser soil particles are trapped. Our field data support Walker et al.'s conclusion (1990) that grass filter strips will not reduce bacterial concentrations sufficiently to meet water quality goals for control of fecal coliforms, and by association, other bacterial contaminants in runoff from manured soils.

Grass filters trapped many, but not all, of the fecal coliforms in runoff. In conditions which maximized soil trapping, fecal coliforms were still found in surface runoff at concentrations in excess of 200/100 mL which would exceed minimum contamination standards for primary contact water. As long as runoff from grass filters occurs shortly after poultry wastes are deposited on soil, our data suggests that inadequate bacterial removal could contribute to groundwater contamination even while adequate best management practices for soil erosion are in place.

1993 - Fecal Contamination in Surface Runoff

The buffer strip length needed to protect water resources from contaminants in surface runoff is a relevant issue in waste management. Our data suggests that grass filter strips at least 4.5 m long will trap most of the sediment in surface runoff from agricultural fields. Grass filter strips this length will also trap most of the fecal bacteria that erode from waste-amended soil. However, the criterion for assessing fecal contamination of water is numerical. Although fecal bacteria mass is reduced, their concentrations remain high and can exceed primary water contact standards. By these standards, the runoff from freshly manured soil that passes through a grass filter strip will still reduce water quality.

Our study used atypically intense rainfall to cause runoff; in most natural storms, the intensity and duration of rainfall would be much less. Grass filters as short as 4.5 m would probably trap runoff from fields if it occurred. So, on most occasions, grass filters should deter surface water contamination by fecal bacteria in runoff from manured fields. However, runoff escaping grass filter strips can exceed water quality limits. Grass filter strips longer than we studied, or management practices in addition to grass filter strips will be required to prevent fecal contamination with absolute certainty.

1992 - Denitrification in Filter Strips

If an accurate estimate of agricultural N_2O input to global N_2O flux is to be made, models of global atmospheric N_2O flux from agricultural soils must account for N_2O flux in soils undergoing wetting and drying cycles. Nitrous oxide flux immediately after rainfall can exceed 1 mg m^{-2} h^{-1} for an indeterminate period in poultry manure amended fields. More refined field studies are needed to demonstrate the source of N_2O evolved in these settings. If the major N_2O source is denitrification, further research must demonstrate whether denitrification in grass filter strips is enhanced by the C contained in surface runoff from adjacent manured fields.

1993 - Denitrification in Filter Strips

The potential denitrification assay indicated that filter strip soil had greater N_2O production when amended with glucose, or NO_3^- and glucose. Nitrate addition alone had little or no effect. Groffman et al. (1991) indicated that potential denitrification in an undisturbed filter strip soil was greater than control soil, as we found. They further indicated that the soils had greater denitrification rates when amended with C and N, as we also found. They proposed that addition of a C and N source, like manure, to a filter strip, might increase the inherently greater denitrification potential of grass filters. Our results support that conclusion, at least with respect to

N_2O evolution. Filter strips which received runoff from the poultry litter amended erosion strips had greater N_2O loss than corresponding portions of the grass filter which clearly received no runoff. The practical use of this observation to manage NO_3^- in runoff remains an open question.

Average daily fluctuations in N_2O emission scarcely reflect the dynamic nature of this microbially mediated process. For an accurate estimate of agricultural N_2O input to global N_2O flux, models of global atmospheric N_2O flux from agricultural soils must account for those periods when N_2O flux in soils is most dynamic — during wetting and drying periods. Nitrous oxide loss immediately after rain can exceed 11000 mg N_2O-N m^{-2} h^{-1} in soils amended with a readily available C and N source like poultry litter. This not only represents a net nutrient loss, but a contribution of radiatively important trace gas to the atmosphere. Since the application of poultry litter to Kentucky soils is likely to increase in the near future, it is clear that this contribution is likely to increase rather than diminish. As in the previous year, there did not seem to be enhanced denitrification in filter strips do to manure runoff.

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PUBLICATIONS

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Coyne, M.S., R.A. Gillfillen, R. Rhodes, and R.L. Blevins, "Soil and Fecal Bacteria Trapping By Grass Filter Strips," Journal of Soil and Water Conservation. Accepted for publication 6-21-94 (No. 94-15).

Coyne, M.S., R.A. Gilfillen, and R. L. Blevins. 1994. Nitrous oxide flux from poultry-manured erosion plots and grass filters after simulated rain. J. Environ. Qual. 23:831-834.

Coyne, M.S., and R.L. Blevins, "Fecal Bacteria in Surface Runoff From Poultry Manured Fields," Proceedings of the Interdisciplinary Conference, "Animal Waste and the Land-Water Interface," Arkansas Water Resources Center, July 16-19, 1995, Fayetteville, AR. Received 08/01/94. Publication date 1995.

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