SEDIMENT AND NITROGEN TRANSPORT IN GRASS FILTER STRIPS

AIDA MENDEZ, THEO A. DILLAHA, AND SAIED MOSTAGHIMI

Made in United States of America
Reprinted from JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION
Vol. 35, No. 4, August 1999

JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION
VOL. 35, NO. 4 AMERICAN WATER RESOURCES ASSOCIATION AUGUST 1999

SEDIMENT AND NITROGEN TRANSPORT IN GRASS FILTER STRIPS[1]

Aida Mendez, Theo A. Dillaha, and Saied Mostaghimi[2]

ABSTRACT: An 18-month field experiment was conducted to evaluate the effectiveness of grass filter strips in removing sediment and various nitrogen species from runoff. Runoff was collected from six 3.7 m wide experimental plots with 24.7 m long runoff source areas. Two plots had 8.5 m filters, two plots had 4.3 m filters, and two plots had no filters. Runoff was analyzed for total suspended solids (TSS), total Kjeldahl nitrogen (TKN), filtered TKN (FTKN), NH_4^+-N, and NO_3^--N. The Mann-Kendall nonparametric test for trend (changes in filter effectiveness over time) indicated that there were no trends in the yields and concentrations of TSS, NO_3^--N, NH_4^+-N, TKN, and FTKN for the 8.5 m filter over time. For the shorter 4.3 m filters, there were significant upward trends in TKN yield and downward trends in TSS, NH_4^+-N, and FTKN concentrations, indicating that trapping efficiency may have started changing with time. The Kruskal-Wallis test indicated that the 8.5 m filters reduced median yields and concentrations of TSS and all N species, but the 4.3 m filters only reduced the median yields and concentrations of TSS, NH_4^+-N, TKN, and the median concentration of FTKN. The 8.5 and 4.3 m filters reduced contaminate yields and concentrations from 42 to 90 percent and from 20 to 83 percent, respectively.
KEYWORDS: nonpoint source pollution, water quality, surface runoff pollution, best management practices, vegetative filter strips, buffer strips, monitoring.

INTRODUCTION

Vegetative filter strips (VFSs) are bands of planted or indigenous vegetation situated between pollutant source areas and receiving waters. Although VFSs do not prevent soil erosion or chemical losses from cropland, they are presumed to trap contaminants and decrease sediment and nutrient loadings to adjacent waterbodies. Researchers have reported that VFSs are effective in removing sediment, total nitrogen, NO_3^--N and NH_4^+-N from agricultural runoff during short-term field experiments (Young *et al.*, 1980; Dillaha *et al.*, 1989; Magette *et al.*, 1989; Parsons *et al.*, 1994). Filter effectiveness appears to be highly correlated with how uniformly runoff is distributed across the filter and the resistance offered by the filter strip vegetation to overland flow. As the vegetation retards the overland flow, the flow velocity decreases in the area immediately upslope of and within the filter. A decrease in runoff velocity reduces sediment transport capacity and sediment-bound pollutants deposit in and upslope of the filter (Hayes *et al.*, 1979). The decrease in runoff velocity also promotes infiltration of runoff as well as dissolved and very fine sediment-bound pollutants.

Since VFSs are expected to be operative for their structural life, ideally 10 years (Hayes and Dillaha, 1992), it is necessary to understand the long-term fate of sediment and N species in the filter. Two major concerns are that VFSs trapping efficiency may decline with time and that VFSs may eventually become a source of mineral N. This could happen if organic N trapped in the filters is mineralized, not assimilated by vegetation, and is later lost in surface runoff (Young *et al.*, 1980; Dillaha *et al.*, 1989). The goal of this study was to evaluate the short and long-term effectiveness of grass filter strips of different lengths in removing sediment and N from cropland runoff.

[1]Paper No. 97154 of the *Journal of the American Water Resources Association*. **Discussions are open until April 1, 2000.**
[2]Respectively, Post Doctorial Researcher, USDA-ARS Southwest Watershed Research Center, 2000 East Allen Road, Tucson, Arizona 85719; Associate Professor, Biological Systems Engineering Department, Virginia Tech, Blacksburg, Virginia 24061-0303; and Professor, Biological Systems Engineering Department, Virginia Tech, Blacksburg, Virginia 24061-0303 (E-Mail/Dillaha: dillaha@vt.edu).

METHODS

Experimental Setup and Instrumentation

The study was conducted at the Virginia Tech Prices Fork Research Farm, 10 km west of Blacksburg, Virginia. Figure 1 shows the layout of the experimental plots. Each plot had a 3.7 m wide by 24.7 m long source area that was planted in conventionally tilled corn with up and down slope tillage. Details on management practices employed on the plots are reported by Mendez-Delgado (1996). The experiment had a completely randomized design, with three treatments and two replicates per treatment. The treatments were 3 VFS lengths: 0 m (plots C and F), 4.3 m (plots B and D), and 8.5 m (plots A and E) filters located at the down slope end of each source area. The experiment assumed that runoff and contaminant losses from the plot source areas were equal and could be characterized by the effluent from the plots without filters (C and F). Runoff entering the 8.5 and

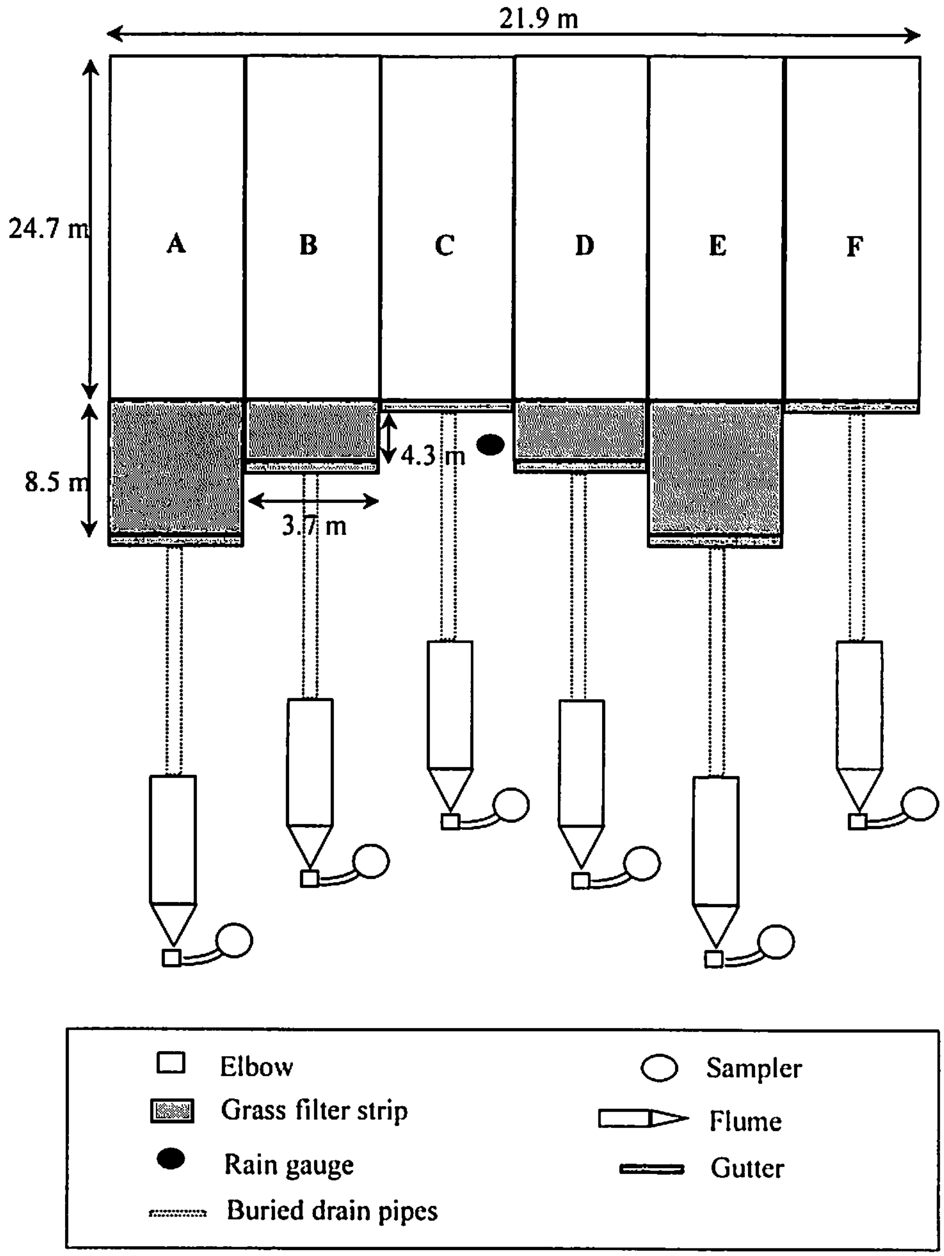

Figure 1. Experimental plot layout.

4.3 m filters was assumed equal to the effluent from the plots without filters (Dillaha *et al.*, 1989). The plots had an average slope of 18 percent and an average cross slope of 4 percent. Cross slope effects on runoff water movement were minimal, however, as the plots were tilled up and down slope and the resulting furrows prevented cross slope flow. Prior to the experiment the plot area had been a grass/shrub meadow for about five years. The grass filters were seeded to Kentucky 31 tall fescue in December 1991 and had a final grass surface cover of about 70 percent by July 1992.

Automatic equipment was used to measure temperature, rainfall amount and intensity, and to measure and sample surface runoff leaving the plots. Soil and air temperature were recorded every six hours using thermocouples. Rainfall amount and intensity were recorded using a tipping bucket type raingage (Model TE 525, Texas Electronics) interfaced with a CR10 Datalogger (Campbell Scientific, Inc.). Rainfall recording began on July 6, 1992.

The upper edge of the plots was bounded by a berm that prevented upslope runoff from entering the plots. The plots were bounded on the sides by plywood boards buried to a depth of 0.2 m depth to minimize lateral surface and subsurface flow across the plot boundaries. The lower edge of each VFS was bounded by a gutter which collected surface runoff. The gutter was attached to a 0.5 m deep plywood cut-off wall that minimized subsurface flow under the gutter. A plastic pipe connected the gutter to a 150 mm H-flume (with a 1:8 sloping approach box). The flow rate was measured with a FW-1 stage recorder equipped with a one-turn potentiometer (5 KΩ, +/- 5 percent tolerance, +/- 0.25 percent linearity, Newark Electronics). A FORTRAN program (Mendez-Delgado, 1996) was developed to transform the potentiometer voltage readings to stage heights and discharge rates. The discharge from the flume flowed into a plastic elbow that contained the automatic sampler intake. The elbow was located below the flume outlet to avoid backwater effects. The elbow shape enhanced flow turbulence and provided a more representative runoff sample. The datalogger, in response to changes in potentiometer readings, sent pulses to the ISCO 3700 automatic samplers (Isco, Inc.) when runoff samples were to be collected. Surface runoff samples were collected from July 1992 through December 1993 and were analyzed for total suspended solids (TSS), NO_3^--N, NH_4^+-N, total Kjeldahl nitrogen (TKN), and filtered total Kjeldahl nitrogen (FTKN). Yields and flow-weighted concentrations were computed for each storm event.

Soil Characteristics

The dominant soil at the site is Groseclose silt loam (*clayey, mixed, mesic Typic Hapludult*). This soil is deep and well-drained with slowly permeable subsoil. The surface A_p layer is brown loam with moderate fine granular structure and about 0.26 m thick. The subsoil, B_t horizon, is sticky and plastic clay with moderate, very fine, and fine subangular blocky structure and is about 0.74 m thick. The upper 0.46 m of the subsoil is yellowish brown, while the remaining 0.28 m is mottled in shades of brown, yellow, and red. The substratum is 0.84 m thick with brown, yellow, and red mottling. The upper 0.3 m of the substratum is sticky and plastic clay and the remaining 0.54 m is clay loam (Creggar *et al.*, 1985; Kool *et al.*, 1986). The soil organic matter content is about 3.2 percent. The average bulk density in the filters was 1.26 g/cm^3. The soil has 44.4, 48.9, and 6.7 percent sand, silt, and clay, respectively (Lee, 1987). Two undisturbed soil core samples were collected from each filter and standard procedures (Mendez-Delgado, 1996) were used to determine porosity, dry bulk density, vertical saturated hydraulic conductivity, moisture content at saturation, field capacity, and wilting point. The steady-state infiltration rates for each plot (source area plus VFS) were obtained from the second and third simulated rainfalls applied on June 7, 1993. The first simulated rainfall event had a one-hour duration with a 35 mm/h intensity and was followed by two 0.5-hour duration events 30 minutes apart with the same intensity 24 hours later. The means of the final infiltration rates were 3.81, 2.65, and 3.28 cm/h for 8.5 m, 4.3 m, and 0 m filters, respectively.

Data Collection and Analyses

Rainfall and Runoff. Rainfall recording started on July 6, 1992, and ended on December 8, 1993. No data were recorded from December 8, 1992, to April 1 due to equipment failure and freezing conditions. In 1992 and 1993 there were 62 and 67 natural rainfall events, and 15 and 20 natural runoff events were monitored, respectively. In 1993, three runoff events were monitored after simulated rainfall was applied to the plots. Runoff samples collected during natural storm events were retrieved either the day the runoff event occurred or the following morning and stored at 4°C until they were analyzed for TSS, NH_4^+-N, NO_3^--N, TKN, and FTKN according to standard procedures (USEPA, 1993). Ammonium-N and FTKN were determined on samples filtered through a 0.21 µm filter.

The pollutant yield, L (mg), and flow-weighted concentration for each runoff event, $\overline{C}$ (mg/L), were computed as:

$$L = \sum_{i}^{n} C_i V_i \qquad (1)$$

$$\overline{C} = \frac{L}{\sum_{i}^{n} V_i} \qquad (2)$$

where C_i is runoff sample concentration during time interval i (mg/L), V_i is runoff volume during time interval i (L), and n is number of time intervals.

Statistical Analysis. Two hypotheses were tested. The first was that the loads and flow-weighted concentrations of TSS, TKN, NO_3^--N, and NH_4^+-N were not affected by VFS length. The second hypothesis was that the trapping efficiency of the filters did not decrease with time. The loads and flow-weighted concentrations of FTKN of runoff events in 1992 were not used in the analysis, because of suspected problems with the FTKN analyses during this period (Mendez-Delgado, 1996). The Kruskal-Wallis large sample approximation distribution-free procedure (Hollander and Wolfe, 1973) was used to test the first hypothesis: no differences among the medians of the plot effluent loads and concentrations for the two filter lengths and for the plots without filters. This test compares the median rather than the mean values of data sets. The median reflects the most frequent values in a data set giving lesser weight to extreme high or low values, while the mean is biased toward the extreme values. A distribution-free test was utilized because the potential pollutant's flow-weighed concentrations and yields were not normally distributed. If there was a significant difference among medians, the distribution-free large sample approximation multiple comparisons test, based on Kruskal-Wallis rank sums test, was used to determine if two medians were different (Hollander and Wolfe, 1973). A probability level of $\alpha = 0.05$ was selected based on previous studies (Dillaha *et al.*, 1989; Schellinger and Clausen, 1992).

The second hypothesis was tested using the Mann-Kendall distribution-free test (Gilbert, 1987). Specifically, the series formed by the filter pollutant yields and flow-weighted concentrations for storms occurring from 1992 to 1993 were tested for trends at $\alpha = 0.05$. The trapping efficiency was defined as the ratio of the difference between pollutant mass entering and leaving a filter to pollutant mass entering the filter.

RESULTS AND DISCUSSION

The effectiveness of grass filters is usually measured as a function of their capacity to reduce pollutant concentrations and loads (Schellinger and Clausen, 1992). This is normally done by comparing runoff loads and concentrations measured at the entrance of the filters with runoff loads and concentrations measured at the outlet of the filters (Dillaha *et al.*, 1989; Schellinger and Clausen, 1992). In this study, the effectiveness of the VFS was also estimated as a function of the pollutant concentrations and loads. The filter effectiveness was defined as the mean percentage reduction in runoff volume or pollutant yield from the 4.3 or 8.5 m filters compared to those from the plots without filters (0 m filter).

Since VFS are structural practices that are usually assumed to last for 10 years, their effectiveness for pollutant reduction should be evaluated over the same period. Unfortunately, no long-term studies of this length have ever been conducted on VFS and there is little information on their long-term effectiveness (Dillaha *et al.*, 1989). The present field study lasted for a period of 18 months. It is longer than most VFS field studies, but it probably was not of sufficient duration to detect significant trends in pollutant trapping efficiencies. Usually, when BMP practices are evaluated, data must be gathered for at least five to ten years, to detect significant effects on runoff quality because of the high degree of variability of hydrologic data. However, trends may be detectable in short-term studies, such as the present study, if loads and concentrations are continuously monitored. In any case, this study should provide a better indication of long-term VFS effectiveness than many of the short-term studies reported in the literature.

Rainfall and Runoff

Precipitation recorded at the experimental site in 1992 and 1993 was approximately one third lower than normal. Cumulative precipitation recorded on the plots from August to October 1992 and May to October 1993 were 187 and 371 mm, respectively. Table 1 displays the mean volume of runoff from the plots, the reduction in runoff volume from plots with filters as a percentage of the runoff volume from plots with no filters, and the standard deviation and median of the ratio of the volume of runoff leaving a VFS to the volume of rainfall. The volume of runoff leaving a VFS is a function of the volume of runoff entering the VFS, rainfall falling on the VFS, VFS length and slope, initial soil moisture conditions, infiltration rate,

TABLE 1. Effect of Vegetative Filter Strip Length on Runoff Volume.

VFS Length (m)	Number of Runoff Events	Mean Runoff Volume (mm)	Reduction in Runoff (percent)	Standard Deviation of Ratio of Runoff to Precipitation Volume	Median Value of Ratio of Runoff to Precipitation Volume
8.5	72	0.070a[1]	71	0.88	0.010a
4.3	71	0.144a	40	1.27	0.080ab
0.0	63	0.240a	NA	2.03	0.310b

[1]Means and medians followed by the same letter are not significantly different at $\alpha = 0.05$, based on the Kruskal-Wallis test.
NA = Not applicable.

and surface roughness. Infiltration rates are higher in the filters than in the source areas because the vegetation cover reduces surface sealing and the vegetation roots and the fauna in the soil make the filter soil more porous. Runoff also has more time to infiltrate into the filter than in the source area since it moves slower, due to vegetation-induced flow retardance, and because of increased surface ponding which permits maximum infiltration over a larger area. Consequently, the volume of runoff that exits a VFS is usually smaller than the volume of runoff that enters it, unless the additional rainfall falling directly in the filter exceeds the additional infiltration volume.

Runoff volume was reduced by 71 and 40 percent for the 8.5 m and the 4.3 m filters, respectively, but the differences in the means were not significant (Table 1). The median value of the ratios of runoff to precipitation volume from the 8.5 m filter was significantly less than the ratios of the plots without filters (Table 1). However, there were no significant differences in the runoff to precipitation volume ratios between the 4.3 m and 8.5 m filters or between the 4.3 m filters and the plots without filters (Table 1). Apparently, the effect of the 4.3 m filter on runoff volume was masked by variability in the data, while the effect of the 8.5 m filter on runoff volume was large enough to offset the data variability. The Mann-Kendall test (Table 2) indicated that there was no trend in the ratio of the volume of runoff leaving a VFS to the volume of rainfall from 1992 to 1993 for the 8.5 m filters and the 4.3 m filter of plot D. For these plots, the effectiveness of the filters in reducing runoff did not decrease with time. The 4.3 m filter of plot B, however, experienced a significant upward trend in the ratio of runoff to rainfall, indicating a decrease in infiltration over time.

Pollutant Concentrations and Loads

Sediment (Total Suspended Solids). As runoff enters a VFS, it is retarded by the vegetation, its velocity and transport capacity decrease, and a portion of the sediment load, particularly large sediment particles, are trapped. Deposition first occurs just upslope of the upper edge of the filter. Over time, sediment from successive storms enters the filter, fills depressional areas, buries the vegetation, and moves downslope. During our experiment, sediment deposition was observed primarily upslope of and within the first meter of the filters. Thus, the 4.3 m filters were expected to be almost as effective as the 8.5 m filters in retaining sediment. Small differences between the effectiveness of the 4.3 m and the 8.5 m filters were anticipated because the longer filters had more opportunity to remove fine sediment particles due to increased infiltration and time for deposition.

The experimental results (Table 3) indicated that both filter lengths were effective in reducing sediment concentrations and yields. The median flow-weighted sediment concentrations from the 8.5 and 4.3 m filters were significantly less than the concentrations from the plots with no filters. Relative to the plots with no filters, the 8.5 and 4.3 m filters decreased mean sediment concentrations by 87 and 83 percent, respectively. The median sediment yields from 8.5 and 4.3 m filters were also significantly less than their influent sediment loads. The mean sediment yield reductions from the 8.5 and 4.3 m filters were 90 and 82%, respectively. The median sediment concentrations in runoff and sediment yield from the 8.5 m filters were not significantly different from the concentrations of sediment leaving the 4.3 m filters, indicating that most of the filtering action and deposition occurred in the upslope portions of the filters as originally hypothesized.

TABLE 2. Mann-Kendall Test Results for Determination of Trends in Yields and Concentrations Over Time.

PLOT		V/R	TSS		NO_3^--N		NH_4^+-N		TKN		FTKN[1]	
			kg/ha	mg/L	kg/ha	mg/L	kg/ha	mg/L	kg/ha	mg/L	kg/ha	mg/L
A	N	36	26	26	26	26	26	26	26	26	13	13
8.5 m	$S_{0.05}$	122	74	74	74	74	74	74	74	74	28	28
	S	69	52	-2	50	40	48	16	62	16	-19	-17
	T	N	N	N	N	N	N	N	N	N	N	N
B	N	35	31	31	30	30	30	30	28	28	18	18
4.3 m	$S_{0.05}$	117	95	95	93	93	93	93	86	86	55	55
	S	201	88	-7	84	63	-11	-32	87	3	-9	-45
	T	U	N	N	N	N	N	N	N	N	N	N
C	N	29	29	29	29	29	29	29	28	28	15	15
0 m	$S_{0.05}$	89	89	89	89	89	89	89	86	86	35	35
	S	0	6	-86	36	-4	-13	-67	46	-16	-33	-55
	T	N	N	N	N	N	N	N	N	N	N	D
E	N	36	28	28	28	28	28	28	28	28	16	16
8.5 m	$S_{0.05}$	122	85	85	85	85	85	85	85	85	37	37
	S	76	42	24	58	15	25	8	56	-3	-5	-3
	T	N	N	N	N	N	N	N	N	N	N	N
D	N	36	33	33	33	33	33	33	33	33	21	21
4.3 m	$S_{0.05}$	122	107	107	107	107	107	107	107	107	55	55
	S	73	8	-111	64	7	-52	-115	20	-89	-42	-74
	T	N	N	D	N	N	N	D	N	N	N	D
F	N	34	26	26	26	26	26	26	26	26	15	15
0 m	$S_{0.05}$	112	74	74	74	74	74	74	74	74	35	35
	S	103	-24	-126	18	-20	-12	-24	4	-94	-25	-51
	T	N	N	D	N	N	N	N	N	D	N	D

Notes: V/R = ratio of volume of runoff to volume of rainfall; N = sample size; $S_{0.05}$ = Mann-Kendall critical value from Hollander and Wolfe (1973) for sample size N and $\alpha = 0.05$; S = calculated Mann-Kendall statistic; T = trend; U = upward trend; D = downward trend; N = no trend.

[1]FTKN 1993 data only.

The Mann-Kendall test (Table 2) indicated that there were no significant trends over time in the sediment yields from any plots. There were no trends in sediment concentrations from the 8.5 m filters, the 4.3 m filter of plot D, and the source area plot C, but there was a significant downward trend in sediment concentration from the 4.3 m filter of plot D and the source area of plot F. The downward trend in sediment concentrations from the 4.3 m filter of plot D is unexpected, but could be the result of a decrease in influent sediment concentrations as was observed with plot F. Since the influent concentrations to the plots were not measured directly, this hypothesis cannot be proved.

Nitrogen. Most N in soil is in a particulate organic N form, and as such, N transport is highly correlated with sediment detachment and transport (Novotny and Olem, 1994). The transport of dissolved N from an area is more dependent on the influent dissolved N concentrations, the soluble N available in the upper layer of the soil profile that can interact with surface runoff, and the volume of water that infiltrates and leaches soluble forms of N from the upper soil layer before runoff. The availability of soluble N depends on chemical, physical, and biological soil processes (Reddy, 1981). The biological processes (mineralization, denitrification, etc.) depend on soil moisture, temperature, and carbon content, among other factors. Due to the mineralization of particulate organic N that is trapped and accumulates in the filter, soluble N levels in the filter may increase over time. If soluble N is lost with surface runoff, effluent dissolved N concentrations and yields may exceed influent concentrations and loads (Dillaha *et al.*, 1989).

TABLE 3. Effect of Vegetative Filter Length on Sediment and Nitrogen Yields and Concentrations.

Length (m)	Samples	Mean	Percent Reduction	Standard Deviation	Median	Mean	Percent Reduction	Standard Deviation	Median
		Total Suspended Solids (g/L)				Total Suspended Solids Yield (kg/ha)			
8.5	54	1.01	87.3	1.92	0.20a[1]	14.8	90.2	31.6	0.1a
4.3	64	1.34	83.0	2.02	0.52a	27.4	81.9	53.1	1.3a
0.0	55	7.89	NA	9.34	4.04b	151.4	NA	327.0	26.3b
		NO_3^--N Concentration (mg/L)				NO_3^--N Yield (g/ha)			
8.5	54	2.40	52.4	5.09	0.29a	19.2	76.9	35.7	0.4a
4.3	63	2.48	50.8	4.19	0.97ab	46.9	43.6	88.1	1.8ab
0.0	55	5.04	NA	8.41	1.83b	83.1	NA	276.8	20.2b
		NH_4^+-N Concentration (mg/L)				NH_4^+-N Yield (g/ha)			
8.5	54	1.52	64.7	3.09	0.36a	8.1	84.5	11.9	0.2a
4.3	63	1.79	58.4	3.77	0.37a	22.9	56.2	50.4	0.3a
0.0	55	4.30	NA	8.86	1.46b	52.3	NA	108.7	7.8b
		TKN Concentration (mg/L)				TKN Yield (g/ha)			
8.5	54	7.04	74.8	11.35	3.69a	88.6	81.5	159.2	4.1a
4.3	61	12.27	56.0	23.57	5.34a	212.9	55.6	327.1	8.1a
0.0	54	27.89	NA	31.16	15.95b	479.0	NA	809.0	89.0b
		Filtered TKN Concentration (mg/L)				Filtered TKN Yield (g/ha)			
8.5	29	3.29	41.6	6.52	1.77a	46.9	47.1	76.7	3.8a
4.3	39	2.94	47.8	3.91	1.90a	71.3	19.6	118.5	1.9a
0.0	30	5.63	NA	5.31	4.17b	88.7	NA	152.1	25.8a

[1]Medians followed by the same letter are not significantly different at $\alpha = 0.05$, based on Kruskal-Wallis test.
NA = Not applicable.

Nitrate. As expected, the longer 8.5 m filters provided more opportunity for infiltration than the shorter filters and were more effective in removing NO_3^--N. The median flow-weighted concentrations and the yields of NO_3^--N leaving the 8.5 m filters were significantly less than the plot influents, but there were no significant differences between the 8.5 m and 4.3 m filters and between the 4.3 m filters and plots with no filters (Table 3). Relative to the plots with no filters, the 8.5 m and 4.3 m filters decreased mean NO_3^--N concentrations by 52 percent and 51 percent, respectively. At first glance, one would not expect NO_3^--N concentrations to decrease since NO_3^--N is subject to neither deposition nor adsorption. The decrease in NO_3^--N concentrations is most likely due to the dilution of NO_3^--N in runoff entering the filter from the source areas with rainfall of lower NO_3^--N concentration that falls directly into the filters. According to the USEPA (1986), NO_3^--N concentrations below 10 mg/L have no adverse effects on human health and warm water fish. The mean and median NO_3^--N concentrations from the filters were less than 10 mg/L (Table 3), but levels lower than 1.0 mg/L in receiving waters can still contribute to objectionable alga blooms. The 8.5 and 4.3 m filters reduced the mean NO_3^--N yields from the source areas by 77 and 44 percent, respectively. The Mann-Kendall test indicated that the effluent NO_3^--N concentrations and yields did not change significantly with time (Table 2).

Dissolved Ammonium. Ammonium is removed from runoff by infiltration, adsorption to the filter soil surface, and adsorption to sediment particles and organic matter that deposit in the filter. The longer VFS provides more surface area for infiltration and adsorption. In addition, the travel time in the 8.5 m filter is longer than in the 4.3 m VFS, and dissolved NH_4^+-N has more time to adsorb to the soil surface or to suspended particles. Therefore, it was expected that longer filters would be more effective in removing NH_4^+-N from runoff than shorter filters. The 8.5 m and 4.3 m filters reduced mean flow-weighted concentrations of NH_4^+-N by 65 and 58 percent, respectively, and mean NH_4^+-N yields by 85 and 56 percent, respectively (Table 3). Differences in median NH_4^+-N concentrations and yields between the 8.5 m and 4.3 m filters and the plots without filters were significant, but differences between the 8.5 m and 4.4 m filters themselves were not significant (Table 3). There were no significant trends in NH_4^+-N concentrations or yields over time in any of the plots except plot D (4.3 m filter), where concentrations decreased with time (Table 2).

Total Kjeldahl Nitrogen (nonfiltered). Total Kjeldahl N is mainly organic N but it also includes NH_4^+-N. Organic N in runoff can be dissolved or associated with particulate organic matter and sediment particles. When organic N is dissolved, it can infiltrate with infiltrating water. If organic matter is transported in suspension, its fate is similar to suspended sediment, that is, it may be filtered by vegetation, the smallest particles may infiltrate, or it may be trapped with settling sediment particles. When organic N is transported as bedload, its fate will be similar to the fate of bedload sediment, it will be filtered and/or deposited. There were significant differences in the median flow-weighted concentrations and yields of TKN for both the 8.5 m and 4.3 m filters compared to the plots without filters, however, there were no significant differences in the concentrations and yields of the two filters (Table 3). As with sediment, NO_3^--N, and NH_4^+-N, this suggests that most of the trapping was occurring in the shorter filter. The 8.5 and 4.3 m filters had mean TKN concentration reductions of 75 and 56 percent, respectively, and mean TKN yield reductions of 82 and 56 percent, respectively. The Mann-Kendall test indicated that, in general, the TKN concentrations and yields from the filters did not change significantly over time (Table 2). The 4.3 m filter of plot B was the only exception, and it had an increase in TKN yield over time.

Filtered Total Kjeldahl Nitrogen. Flow-weighted concentrations and yields of FTKN are given in Table 3 for 1993. As mentioned previously, 1992 FTKN data were excluded from the analyses because of suspected problems in the FTKN laboratory procedure. The median flow-weighted concentrations of FTKN in runoff from the 8.5 m and 4.3 m filters were significantly different from the plots without filters, but there were no significant differences between the two plots with filters (Table 3). There were no significant differences in median FTKN yields between any of the plots. Mean flow-weighted TKNF concentrations from the 8.5 m and 4.3 m filters were reduced by 42 and 48 percent, respectively, and FTKN yields were reduced by 47 and 20 percent, respectively (Table 3). The Mann-Kendall test indicated that there were no trends in the FTKN concentrations over time. However, there were significant decreases in FTKN yields for the plots without filters (C and F) and for the 4.3 m filter of plot D over time.

SUMMARY AND CONCLUSIONS

An 18-month field experiment was initiated in July 1992 to evaluate the effectiveness of grass filter strips in removing sediment and nitrogen from cropland runoff. Runoff from natural occurring rainfall events was collected from six experimental plots with 8.5 m, 4.3 m, and no grass filters. The 8.5 m filters reduced sediment, NO_3^--N, NH_4^+-N, and TKN yields significantly ($\alpha = 0.05$) with mean reductions of 90, 77, 85, and 82 percent, respectively. The concentrations of sediment, NO_3^--N, NH_4^+-N, and TKN and FTKN from the 8.5 m filters were also significantly less than the filter influent, 87, 52, 65, 75, and 42 percent, respectively.

The 4.3 m filters reduced median sediment, NH_4^+-N, and TKN yields significantly, with mean reductions of 82, 56, and 56 percent, respectively. Mean FTKN yield was reduced by 20 percent, but the difference was not significant. Mean yields and concentrations of NO_3^--N were reduced by 51 and 44 percent, respectively, but the reductions also were not significant. Median concentrations of sediment, NH_4^+-N, TKN, and filtered TKN from the 4.3 m filters were significantly less than the influent concentrations, with average mean reductions of 83, 58, 56, and 48 percent, respectively.

There were no significant differences in pollutant trapping efficiencies of the 8.5 and 4.3 m filters. The shorter filters were almost as effective as the longer filters in trapping most pollutants. This was expected for particulate contaminants since they are trapped just upslope of or within the first meter or so of the filter until the upper portion of the filter is buried with sediment. Since the 4.3 m filters were never totally inundated with sediment, they removed most of the particulate contaminants and the additional length of the 8.5 m filters did not trap significantly more sediment or sediment-bound pollutants.

The Mann-Kendall test was used to determine if the effectiveness of the filters decreased with time. The effectiveness of the 8.5 m filters and all but one of the 4.3 m filters (plot B for TKN) in reducing pollutant yield did not decrease significantly over the

18-month study period. However, a decreasing trend in filter efficiency in trapping pollutants might be expected over time if filters are not maintained, become inundated with sediment, or if concentrations of soluble pollutants increase on the filter surface.

In summary, the 8.5 m and 4.3 m grass filters were found to be effective in trapping sediment and various species of N. Most of the trapping occurred in the first 4.3 m of the filters and the remaining length of the 8.5 m filter appeared to be more of a polishing filter. Pollutant removal efficiency did not appear to decrease over the 18-month study period. Additional longer-term studies are needed to further evaluate long-term trapping efficiencies.

ACKNOWLEDGMENTS

The authors acknowledge the Instituto Nacional de Investigaciones Agrarias of Spain for the doctoral fellowship that supported Dr. Mendez and the USDA-CSREES for supporting the field research. Additional support was provided by the Virginia Agricultural Experiment Station and USDA-CSREES Southern Regional Research Projects S-249 and S-273. Special acknowledgement is made to Phil McClellan, Carol Newell, Julie Jordan, Dexter Davis, Clyde Adkins, Faycal Bourauoui, and Shreeram Inamdar for their advice and assistance.

LITERATURE CITED

Creggar, W. H., H. C. Hudson, H. C. Porter, H. L. Mathews, W. J. Edmonds, W. G. Harris, and K. W. Molten, 1985. Soil Survey of Montgomery County, Virginia. USDA, Soil Conservation Service, Washington, D.C.

Dillaha, T. A., R. B. Reneau, S. Mostaghimi, and D. Lee, 1989. Vegetative Filter Strips for Agricultural Nonpoint Source Pollution Control. Transactions of the ASAE 32(2):513-519.

Gilbert, R. O., 1987. Statistical Methods for Environmental Pollution Monitoring. Van Nostrand Reinhold, New York, New York.

Hayes, J. C., B. J. Barfield, and R. I. Barnhisel, 1979. Filtration of Sediment by Simulated Vegetation. II. Unsteady Flow with Nonhomogeneous Sediment. Transactions of the ASAE 22(5):1063-1067.

Hayes, J. C. and T. A. Dillaha, 1992. Vegetative Filter Strips: I. Site Suitability and Design. ASAE Paper No. 92-2102, American Society of Agricultural Engineers. St. Joseph, Michigan.

Hollander, M. and D. A. Wolfe, 1973. Nonparametric Statistical Methods. John Wiley and Sons. Inc., New York, New York.

Kool, J. B., K. A. Albrecht, J. C. Parker, and J. C. Baker, 1986. Physical and Chemical Characterization of the Groseclose Soil Mapping Unit. Bulletin 86-4, Virginia Agricultural Experiment Station Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

Lee, D., 1987. Simulation of Phosphorus Transport in Vegetative Filter Strips. Dissertation presented in partial fulfillment of the Ph.D. degree in Agricultural Engineering, Virginia Polytechnic Institute and State University, Blacksburg, Virginia, 207 pp.

Magette, W. L., R. B. Brinsfield, R. E. Palmer, and J. D. Wood, 1989. Nutrient and Sediment Removal by Vegetative Filter Strips. Transactions of the ASAE 32(2):663-667.

Mendez-Delgado. Aida, 1996. Nitrogen Transport and Dynamics in Grass Filter Strips. Dissertation presented in partial fulfillment of the Ph.D. degree in Agricultural Engineering, Virginia Polytechnic Institute and State University, Blacksburg, 413 pp.

Novotny, V. and H. Olem, 1994. Water Quality Prevention Identification and Management of Diffuse Pollution. Van Nostrand Reinhold, New York, New York.

Parsons, J. E., R. D. Daniels, J. W. Gilliam, and T. A. Dillaha, 1994. Reduction in Sediment and Chemical Load Agricultural Field Runoff by Vegetative Filter Strips. UNC-WRRI-94-286, North Carolina State University, Raleigh, North Carolina, 202 pp.

Reddy, K. R., 1981. Land Areas Receiving Organic Wastes: Transformations and Transport in Relation to Nonpoint Source Pollution. *In:* Environmental Impact of Nonpoint Source Pollution, M. R. Overcash and J. M. Davidson (Editors). Ann Arbor Science Publishers, Inc., Ann Arbor, Michigan, pp. 243-274.

Schellinger, G. R. and J. C. Clausen, 1992. Vegetative Filter Treatment of Dairy Barnyard Runoff in Cold Regions. J. Environ. Qual. 21:40-45.

USEPA, 1986. Quality Criteria for Water 1986. Report No. EPA 440/5-86-001, U.S. Environmental Protection Agency, Washington, D.C.

USEPA, 1993. Methods for Chemical Analysis of Water and Wastes. Report No. EPA-600/4-79-020, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Young, R. A., T. Huntrods, and W. Anderson, 1980. Effectiveness of Vegetated Filter Strips in Controlling Pollution from Feedlot Runoff. J. Environ. Qual. 9(3):483-487.