ABSTRACT

BUTLER, DAVID MICHAEL. Runoff, sediment, and nutrient export from manured riparian pasture as affected by simulated rain and ground cover. (Under the direction of N.N. Ranells.)

The impact of livestock pastures on sediment and nutrient export to surface waters in North Carolina is not well documented. The objective of this work was to determine the effect of ground cover on sediment and nutrient export from pastured riparian areas. In one experiment, plots 0.75 m by 2.0 m were established on 10% and 20% slopes of existing stands of mixed tall fescue (*Festuca arundinacea* Schreb.) / dallisgrass (*Paspalum dilatatum* Poir.), with stands modified to represent a range of ground cover levels by establishing 100% bare soil area with 0% ground cover (bare ground), 55% bare soil area with 45% ground cover (low cover), 30% bare soil area with 70% ground cover (medium cover), and not altering full vegetative cover plots (high cover). The bare ground treatment was also compacted to simulate a cattle heavy use area. Measured forage canopy cover on vegetated plots was generally higher than the level of ground cover established at the soil surface. At 45% ground cover, canopy cover was 63 to 78%, at 70% ground cover canopy cover was 76 to 83%, and at full ground cover canopy cover was 83 to 98%. In a second experiment, plots 0.75 m by 2.0 m were established on stands of existing wetland vegetation that consisted of pickerelweed (*Pontederia cordata* L.), showy goldenrod (*Solidago erecta* (Pursh) MacM.), Japanese honeysuckle (*Lonicera japonica* Thunb.), arrowleaf tearthumb (*Polygonum sagittatum* L.), and leathery rush (*Juncus coriaceus* Mackenzie), with stands modified to establish either no ground cover (bare ground) or to allow full cover. Rainfall simulators were used to evaluate runoff,

total suspended sediment (TSS), total Kjeldahl nitrogen (TKN), nitrate-N (NO_3-N), ammonium-N (NH_4-N), total Kjeldahl phosphorus (TKP), and dissolved reactive P (DRP) export from plots with applied cattle (*Bos* spp.) feces and urine at a rate representing stocking of ~1.4 cows ha^{-1} yr^{-1}. Rainfall simulation events were conducted in April 2003 to evaluate baseline runoff conditions. Feces and urine were applied in May and Sept 2003 and immediately followed with rainfall simulation events. Additional rainfall simulations without additional application of feces and urine occurred in June and Oct 2003.

In the first experiment involving mixed tall fescue / dallisgrass, mean runoff volume from bare ground was generally greater than from low, medium, and high levels of cover, which had similar mean runoff volume. Export of TSS was greater from 20% than 10% slope on bare ground and low cover, but when cover was medium or high, TSS export was similar at both slopes. Export of DRP was elevated after application of feces and urine to plots, generally with the greatest export in Sept. However, there was not a consistent response of DRP export to cover at each rain event. Mean TKP export was similar at low, medium, and high cover levels over all levels of slope and rain event, suggesting that low cover may be sufficient to preventing excessive TKP export. Mean NO_3-N export was greatest from bare ground and few differences were observed at low, medium, and high cover levels. Mean NO_3-N export was also greatest during Oct, whereas there were no differences between May, June, and Sept rain events. Mean NH_4-N export was elevated (~1.37 kg N ha^{-1}) in months when feces and urine were applied and minimal (< 0.05 kg N ha^{-1}) in all other months. Mean export of both TKN and TN was greatest at bare ground and did not differ at low, medium, and high cover levels at

each rainfall event except June, where mean TKN and TN export at bare ground did not differ from high cover.

In the second experiment on existing wetland vegetation, mean runoff volume was generally greater from bare ground than full cover, and from the wetland plots compared to upland mixed tall fescue / dallisgrass plots on a similar 10% slope. Plots with full cover were remarkably effective at minimizing mass export of TSS during all simulated rain events. There was a greater mean export of DRP from plots with full cover compared to plots at bare ground, though mean export at both levels of cover was < 0.20 kg P ha^{-1}. Unlike DRP, mean TKP export was much greater from bare ground than full cover, which is not surprising given the large amount of TSS exported from bare ground plots. Greater mean NO_3-N export was observed from the bare ground treatment than from full cover, as well as one month following application of feces and urine. Conversely, rain events that included feces and urine application had a far greater export of NH_4-N, TN, and TKN than did rain events without feces and urine application.

Results indicate that livestock heavy use areas in riparian zones may export substantial sediment, N, and P, but when cover is maintained sediment and nutrient loss from cattle feces and urine may be minimal. Cover and time of rainfall following grazing are important determinants of sediment, N, and P export to surface waters.

RUNOFF, SEDIMENT, AND NUTRIENT EXPORT FROM MANURED RIPARIAN PASTURE AS AFFECTED BY SIMULATED RAIN AND GROUND COVER

by

DAVID MICHAEL BUTLER

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BIOGRAPHY

David Michael Butler is a native of rural Maryland. During his youth he developed a strong interest in agriculture and environmental quality. He received his B.S. in Biology from Frostburg State University, Frostburg, Maryland in 2002. He began graduate work in the department of crop science at North Carolina State University in June 2002, under the direction of Dr. Noah N. Ranells, with an interest in nutrient management, forages, and water quality. He hopes to continue his education so as to be better able to serve farmers, rural communities, and humankind in general through contributions to a more sustainable agriculture.

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1

REVIEW OF LITERATURE

INTRODUCTION

Nitrogen and phosphorus from agricultural nonpoint pollution can contribute to eutrophication of surface waters (Carpenter et al., 1998). Eutrophication can lead to algal blooms, reduced dissolved oxygen levels, fish kills, reduction in biodiversity, and other potential negative impacts which may reduce the suitability of surface waters for use by humans and wildlife. Increased sediment loads have been widely reported to reduce populations of benthic organisms and fish, as well as overall primary productivity of aquatic ecosystems (Cooper, 1993).

Grazing cattle can be a source of nutrients and sediment to surface waters in the eastern USA (White et al., 1980). However, the impact of grazing livestock on nutrient and sediment export to North Carolina surface waters has not been well quantified. In 1997, it was estimated that over 0.71 million hectares or ~16% of total land area in North Carolina was used as pastureland (USDA, 1997), which makes the impact of grazing on water quality critically important. Specifically, the importance of riparian area management on water quality underscores the need for scientific investigation of environmental impacts when riparian areas are grazed. For this work, riparian areas are considered to be streamside ecosystems that help maintain the integrity of surface water resources, essentially serving to "buffer" the stream from the impact of human activities such as agriculture (Wenger, 1999). Depending on site, riparian areas may contain a variety of vegetation that may be any combination of native or introduced wetland plants, trees, shrubs, summer annuals, and grasses. Well-established perennial forages, such as

tall fescue (*Festuca arundinacea* Schreb.), are common in and near riparian areas of the North Carolina Piedmont.

Riparian areas are often grazed because they are typically unsuitable for row crop production due to topography and seasonal flooding, but can have relatively high forage productivity because of favorable moisture conditions during drier periods of the year. However, poor grazing management can lead to variable stand density and forage ground cover (Alderfer and Robinson, 1947), which can negatively influence infiltration, runoff, erosion, and sediment deposition (McGinty et al., 1979; Self-Davis et al., 2003) thereby limiting the ecosystem services provided by the riparian area. However, while there are data relating to runoff volume, sediment, and nutrient export from grasslands in the Southeast, there are few data that report the level of ground cover necessary in order to protect water quality when livestock graze riparian areas.

PASTURE GROUND COVER

Research has shown that ground cover is an important determinant of runoff from grazing lands. Lang (1979) reported that when ground cover, as estimated by the point or area quadrat method, decreased from 100% to 10% the incidence of surface runoff events from plots grazed with sheep increased from 1 event to 13 events. Likewise, decreasing levels of ground cover led to an increase in maximum volumetric runoff yield from storm events from 30% of rainfall at 100% ground cover to 85% of rainfall at 10% cover. Average runoff from plots with > 95% ground cover was 2 mm yr^{-1} and 0.3% of rainfall, which averaged 642 mm yr^{-1}. When ground cover was less than 75%, runoff increased exponentially from ~10 mm yr^{-1} at 75% ground cover to over 200 mm yr^{-1} when ground

cover was less than 20%. The author suggested that when ground cover is below 75%, bare areas begin to connect with each other which allows for faster flow of runoff and less time for infiltration. Similarly, Costin (1980) reported that ground cover values less than 70%, as estimated using the quadrat method, resulted in significantly greater mean soil loss and runoff rate than ground cover levels greater than 70%. For ground cover levels below 70%, runoff as a proportion of rainfall increased to greater than 15% of rainfall with decreasing levels of ground cover, with a maximum value of over 30% of rainfall recorded as runoff. Soil loss increased above 5 g m^{-2} with decreasing ground cover, with a maximum value of 60 g m^{-2}. There was no difference between mean soil loss and runoff rate at ground cover levels above 70% or at levels below 70% ground cover. This experiment was conducted in Australia on mixed canarygrass (*Phalaris* spp.) and subterranean clover (*Trifolium subterraneum* L.) pasture that was moderately to heavily grazed (12 to 30 sheep ha^{-1}).

A ground cover level of 75% was also suggested as a threshold value by Ethiopian researchers (Mwendera and Saleem, 1997). Soil trampling and reduced vegetative cover, estimated using the point technique, were reported to have increased cumulative runoff from heavily and very heavily grazed plots to ~15 mm, when cumulative rainfall was 258 mm. At no grazing and light grazing conditions cumulative runoff was ~5 mm. For pasture, a 75% level of ground cover on a 0 to 4% slope and 85% ground cover on a 4 to 8% slope were suggested as critical values for preventing soil loss. Soil loss was generally less than 30 kg ha^{-1} at ground cover greater than 75% on 0 to 4% slope and 85% on 4 to 8% slope. Soil loss increased rapidly when ground cover was

below those threshold levels. Soil loss of ~100 kg ha^{-1} was observed at 65% ground cover on 4 to 8% slope.

Slightly different results were reported by Dadkah and Gifford (1980) in a Utah study of infiltration rate and sediment production with intermediate wheatgrass (*Agropygon intermedium* (Host) Beauv.) and crested wheatgrass (*Agropygon cristatum* (L.) Gaertn.) vegetation. Cover was established using a grid and uprooting vegetation in randomly selected squares. When rainfall was applied at a rate of 76 mm h^{-1}, the researchers reported no difference in infiltration rate on 0.5 m by 0.75 m plots with either 50 or 80% ground cover. This relationship did not change with the addition of trampling, as no difference in infiltration rate was reported for 50 to 80% ground cover. When the trampling rate was ≥ 40%, there was no difference in infiltration rate from 30 to 80% ground cover. Averaged over all trampling levels of 0 to 60%, plots with 30% ground cover generally had higher sediment yield than plots with 50 to 80% cover. Mean sediment yield for untrampled plots at 30% cover was 1356 kg ha^{-1}, while plots at 80% cover yielded significantly less sediment at 665 kg ha^{-1}, and plots with 50% cover yielded 787 kg ha^{-1}. The researchers suggested that ground cover levels of 50% or greater may be sufficient for adequate watershed protection. Relatively low plant stand density and seeding rates were also suggested as a threshold level for prevention of sediment loss in tall fescue turfgrass in Maryland (Gross et al., 1991). The researchers reported that at all three rainfall intensities examined, sediment loss did not differ between a low tall fescue seeding rate of 98 kg ha^{-1} that produced a visual estimate of ~30 to 40% cover and a high seeding rate of 488 kg ha^{-1} that produced ~60 to 80% cover. However, at the highest

rainfall intensity of 120 mm h^{-1}, only the highest seeding rate of 488 kg ha^{-1} reduced sediment loss compared to a seeding rate of 0 kg ha^{-1}.

Hofmann et al. (1983) reported that percent ground cover was highly correlated with soil loss, runoff, and soil loss: runoff ratio from range plots in North Dakota during a 1-yr study. Live cover value was not a good predictor of susceptibility of grazing lands to erosion. The authors further suggested that the point frame technique for determining total vegetative ground cover percentage was adequate for predicting susceptibility of rangelands to erosion.

In Pennsylvania, Alderfer and Robinson (1947) reported runoff rates of 33 to 80% of water applied from heavily grazed pasture plots, 1 to 50% from moderately to lightly grazed plots, and 0 to 2% from ungrazed plots. They attributed the differences in runoff rate to loss of soil cover as well as increased compaction in the 0- to 2.5-cm soil surface layer due to grazing pressure. Increased compaction was evident due to decreased noncapillary porosity, which ranged from 15 to 33% on ungrazed and lightly grazed plots, but was 3 to 10% on heavily grazed plots. Bulk density was also affected, showing increased compaction in the 0- to 2.5-cm layer. However, bulk density in the 2.5- to 7.6-cm and 7.6- to 15.2-cm layers was not affected by heavy grazing on the clay loam and sandy loam soils in the study.

In Arkansas, Self-Davis et al. (2003) examined the effects of canopy cover to runoff volume from plots 3 m by 6 m on a 5% slope. Forage species of switchgrass (*Panicum virgatum* L.), Caucasian bluestem (*Bothriochloa caucasia* (trin.) C.E. Hubb.), bermudagrass (*Cynodon dactylon* (L.) Pers.), eastern gamagrass (*Tripsacum dactyloides* (L.) L.), and tall fescue were either unclipped or clipped to 6 or 10 cm and subjected to

rainfall simulation. Although there was a trend towards greater runoff volume, clipping did not significantly increase volume from any forage species compared to unclipped plots, even though ground cover determined by the line transect method was reduced after harvest. Tall fescue generally had the least runoff of the forage species, and even though a cool season grass, showed the potential for relatively low runoff during the entire year. Dry matter produced was not found to be significantly related to runoff volume.

SEDIMENT EXPORT

Several studies have reported the effectiveness of vegetated buffer strips at controlling non-point sources of pollution from agriculture (Bingham et al., 1980; Chaubey et al., 1995; Daniels and Gilliam, 1996). Sediment loss from grazed riparian areas presents a slightly different research problem because significant amounts of sediment can be exported due to grazing disturbance within the buffer and adjacent to surface waters. However, pasture loss of sediment can be much lower than loss from other agricultural crops (Berg et al., 1988). Berg et al. reported sediment loss of 300 kg ha^{-1} yr^{-1} from rangeland watersheds, whereas loss from conservation tillage wheat was ten-fold greater at 3,000 kg ha^{-1} yr^{-1}, and loss from conventionally tilled wheat was 200-fold greater at 68,000 kg ha^{-1} yr^{-1}. Closely grazed tall fescue has been suggested for prevention of soil loss in the Southern Piedmont (Barnett et al., 1972). In this study, a simulated rain of 65 mm h^{-1} for 2 h resulted in a soil loss of 360 kg ha^{-1} on pastures grazed to a height of 2.5 to 5 cm.

In a study of sediment trapping in the riparian zone, Pearce et al. (1994) found that sediment yield and travel distance of applied sediments were not correlated to vegetation height, which was either clipped to 10 cm or unclipped. The study also noted that 1.2-m^2 micro-plots showed increased sediment yield over 30-m^2 macro-plots when overland flow was applied at the upper edge of the plots. This indicates the importance of width of riparian buffers to provide adequate trapping of sediments that may evolve from upland land uses as well as the variability in runoff parameters when comparing plots of different sizes.

In Ontario, Abu-Zreig et al. (2003) reported a 65% sediment trapping efficiency for 2-m buffer lengths, 81% for 5-m, 92% for 10-m, and 91% for 15-m. Buffer vegetation was a mixture of creeping red fescue (*Festuca rubra* L.) and birdsfoot trefoil (*Lotus corniculatus* L.). In Montana, Hook (2003) examined riparian buffer plots to evaluate the influence of several factors to sediment retention. Mean retention of applied sediment increased with buffer width: 83% at 1-m, 94% at 2-m, and 99% at 6-m buffer widths. Increasing slope from 2 to 20% reduced sediment retention from 96 to 91%. At a 1- and 2-m buffer widths, ~20% and ~10% more sediment was retained in the dense vegetation of the wetland and transitional plots than in the relatively sparse vegetation of upland plots. Wetland plots contained ~6 to 7% bare soil area, whereas upland plots had ~55 to 60% bare soil area. Generally, measurements of biomass, basal cover, and bare soil area explained sediment retention fairly well.

The effect of aboveground biomass on sediment loss has been reported in several studies. However, because biomass may be strongly correlated to ground cover, discrete determination of the effects can be challenging. In an erosion study in New York, soil

loss from pastures was 183 kg ha^{-1} for heavily-grazed, non-fertilized, 75 kg ha^{-1} for heavily-grazed, fertilized, and 29 kg ha^{-1} for lightly-grazed, fertilized pastures during a period when total rainfall was 110 cm (Johnstone-Wallace et al., 1942). Pasture biomass also significantly affected total sediment loss from 1-m^2 runoff plots in Pakistan under wet and dry conditions (Bari et al., 1995). When biomass was reduced ~50%, sediment concentration in runoff increased by a factor of 1.5. When biomass was 35% of ungrazed conditions, sediment concentration was two-fold greater and when biomass was just 22%, sediment concentration was 2.5-fold greater. It is interesting to note that a relatively large decrease in biomass was needed to cause a 1.5-fold increase in sediment concentration.

In New Zealand, Elliott et al. (2002) reported a linear increase in runoff sediment concentration from 0.5-m^2 plots when compared with percentage of bare ground. Near 0% bare ground conditions (full ground cover) sediment concentration was typically less than 300 g m^{-3}, but concentration increased to about 1000 g m^{-3} with 50% or greater bare ground in 0.5-m^2 plots with slopes of 7 to 33%.

In New Mexico, Weltz and Wood (1986) examined sediment losses from two watersheds under moderate continuous, heavy continuous, and short duration grazing, as well as total exclusion from grazing livestock. Under a 76 mm h^{-1} simulated rainfall for 1 h, total plot losses of suspended sediment were the greatest under short duration grazing in both watersheds (~550 and 270 kg ha^{-1}), intermediate under moderate continuous grazing (~300 and 80 kg ha^{-1}), and lowest when grazing livestock were excluded (~80 and 20 kg ha^{-1}). Values for heavy continuous stocking were reported for a single watershed, where the values were similar to those from moderate continuous grazing. In

Texas, Warren et al. (1986) also reported higher sediment export in runoff following short-duration grazing. Simulated rainfall was applied at a rate of 203 mm h^{-1} for 30 min. During the dormant season, sediment export increased in the post-grazing period from 1,241 kg ha^{-1} before the 8-day grazing period to 2,017 kg ha^{-1} shortly afterwards. During the growing season there was an increase in sediment export of ~200 kg ha^{-1}, but the increase was not shown to be significant. The percentage of bare ground was ~30% for both the growing season and the dormant season.

NUTRIENT EXPORT

The filtering effect of vegetated riparian areas is important for preventing export of nutrients, notably nitrogen and phosphorus, to surface waters. The recycling of nutrients by grazing cattle leads to nutrient "hot spots" in pasture where excretions are deposited. These microsites have a high concentration of nutrients relative to neighboring areas. The ability of riparian systems to attenuate large exports of these nutrients is critical to maintaining quality of receiving waters.

Lim et al. (1998) examined length effects of filter strip removal of nutrients from cattle feces in runoff. Plots 30.5 m in length on a 30% slope were treated with cattle feces containing 60 kg N ha^{-1}, which represented a stocking rate of 9 animal units (AU) ha^{-1} for 7 days. The simulated pasture represented the upper 12.2 m of total plot length, below which was the vegetative filter strip (VFS). The feces were placed at the lower edge of the simulated pasture plots and runoff subject to flow through the VFS where sampled at 0-, 6.1-, 12.2-, and 18.3-m distances from the feces application. Reductions in flow-weighted concentrations of total Kjeldahl nitrogen (TKN), orthophosphate-

phosphorus (PO_4-P), total phosphorus (TP), and total suspended solids in runoff were observed with a vegetative filter strip of 6.1 m compared to no filter strip. Mean concentrations of TKN were reduced from 10.12 to 2.04 mg L^{-1}, PO_4-P from 1.28 to 0.31 mg L^{-1}, TP from 1.42 to 0.32 mg L^{-1}, and suspended solids from 134 to 37.5 mg L^{-1}. At filter strip lengths greater than 6.1 m, there were no further significant decreases in nutrient concentrations. However, mean mass transport of TKN was reduced from 2.44 to 0.452 g at a filter strip length of 18.3 m. In Ontario, Abu-Zreig (2003) reported TP trapping efficiencies at 32% for a 2-m buffer length, 54% for 5-m, 67% for 10-m, and 79% for 15-m. Buffer vegetation was a mixture of creeping red fescue (*Festuca rubra* L.) and birdsfoot trefoil (*Lotus corniculatus* L.). With a 5-m buffer and no ground cover, TP trapping efficiency averaged 35%. The authors suggested that buffers less than 5 m are very effective for removal of sediment from runoff, but not as effective for removal of TP because P tends to be bound to silt and clay particles smaller than 100 microns in diameter.

Trlica et al. (2000) examined nutrient runoff from 3 m by 10 m riparian plots grazed by cattle in northern Colorado. Under simulated rain events at a rate of 125 mm h^{-1} for 100 min, runoff rates in the grazed plots were ~70% greater than those of ungrazed control plots. On grazed plots, mean nitrate-N (NO_3-N) flux increased from 48 mg h^{-1} to 138 mg h^{-1}, ammonium-N (NH_4-N) increased from 156 mg h^{-1} to 2258 mg h^{-1}, and PO_4-P increased from 0.04 mg h^{-1} to 0.77 mg h^{-1} compared to ungrazed control plots.

In Kentucky, Edwards et al. (2000a) examined nutrient export from tall fescue plots with applied cattle feces. Simulated grazing treatments included an ungrazed control, continuous stocking, and rotational grazing at stocking densities comparable to

the recommended stocking density of cattle for Kentucky pastures of ~8 animal units (AU) ha^{-1}. The tall fescue in the plots was mowed to ~10 cm for all stocking treatments. Hoof traffic and urine addition were also not modified as part of the simulated grazing treatments. The researchers reported minimal effect of stocking treatment on N and P export. Changes in vegetation and soil characteristics resulting from grazing may be more important than the amount of manure associated with varying stocking treatments.

In a related study, Edwards et al. (2000b) examined relationships between tall fescue height and nutrient export from feces and urine applied to 2.4 m by 6.1 m plots. Feces was applied at a rate of 0.5 kg $plot^{-1}$ day^{-1} and urine was applied at a rate of 330 mL $plot^{-1}$ day^{-1} for 7 days, followed by a 21-day rest period. Plots with forages clipped to 20 cm or left unclipped were associated with the greatest flow-weighted NO_3-N concentrations in runoff; unclipped plots averaged 1.3 mg L^{-1} NO_3-N after feces and urine application, plots clipped to 20 cm averaged 1.0 mg L^{-1}, plots at 10 cm averaged 0.7 mg L^{-1}, and plots at 2.5 cm averaged 0.6 mg L^{-1}. The authors suggested that the slower growth of the forages managed at greater heights resulted in less NO_3-N uptake, leaving more NO_3-N available for transport in runoff. Generally, the addition of feces and urine did not affect NO_3-N concentrations at the 2.5- and 10-cm clipping heights. While results were similar for TKN concentrations, NH_3-N concentrations were not affected by forage management treatments. The authors suggested this was likely due to moisture conditions favorable to nitrification of NH_3-N. Following application of feces, runoff TP concentrations were increased by ~0.3 mg L^{-1} over the control (no application of feces or urine) for unclipped plots and plots clipped to a 2.5-cm height. The addition of both urine and feces did not increase TP concentrations over those from plots with feces

application alone. Results for PO_4-P concentrations were similar to TP concentrations, while PO_4-P accounted for 85% of TP. The authors concluded that forage management to promote growth and nutrient uptake may negate the impacts of applied feces and urine to N export, though the same effect is likely not to be seen with P. While this may have accounted for some of the reduction in nutrients, the low forage height may have also caused a greater tiller density at the soil surface, which in turn may have reduced nutrient export.

In South Carolina, McLeod and Hegg (1984) examined runoff water quality from tall fescue pasture with dairy manure, poultry litter, municipal sludge, or ammonium-nitrate (NH_4NO_3) applied at a target rate of 112 kg N ha^{-1} and simulated rainfall applied at a rate of 37 mm h^{-1}. Simulated rainfall was applied and runoff sampled at 1, 7, 14, and 21 days after manure or fertilizer applications. Flow-weighted concentration of TKN in total runoff was initially as high as 40 mg L^{-1} from pasture treated with poultry manure or NH_4NO_3, but was reduced below 10 mg L^{-1} when runoff was initiated 7 days after fertilizer application. Initial TKN concentration in runoff from dairy manure was between 15 and 20 mg L^{-1}, but was reduced to < 10 mg L^{-1} at 7 days. Total P concentration was also reduced when runoff occurred 7 days after manure application, but not as drastically. Initial TP concentration of runoff from poultry litter averaged ~12 mg L^{-1} and was reduced to ~5 mg L^{-1} at 7 days. TP concentration in runoff from dairy manure was initially ~8 mg L^{-1} and was reduced to ~7 mg L^{-1} at 7 days. Comparatively, concentration of NO_3-N in runoff was not as elevated initially as were other nutrients with the exception of runoff from applied NH_4NO_3. While NO_3-N concentration in

runoff from plots with applied NH_4NO_3 was initially 15 to 20 mg L^{-1}, at no time was NO_3-N concentration in runoff from dairy manure or poultry litter higher than 5 mg L^{-1}.

In Pennsylvania, Kleinman et al. (2002) examined runoff P losses from dairy manure, poultry litter, swine slurry, and diammonium phosphate applied to soil boxes. Among their results, they reported that DRP and TP losses from soils with applied dairy manure were often not significantly greater than P concentrations in runoff from soil boxes without applied manure. The researchers suggested that the applied dairy manure was shielding the soil from rain impact, as DRP accounted for 46 to 83% of TKP when dairy manure was applied compared to 5 to 16% when soils were unamended.

In Arkansas, Sauer et al. (1999) compared nutrient runoff from grazing animal depositions to that from application of poultry litter. Dairy feces and urine generally produced lower levels of nutrients in runoff than did poultry litter or a combination of poultry litter and dairy manure. This was likely due to a lower rate of application of dairy manure as the researchers attempted to simulate stocking densities of cattle that would be representative of northwest Arkansas pastures. Upon application of simulated rainfall at a rate of 75 mm h^{-1} one day after manure applications, 5.0% of TN, 29.5% of NH_4-N, and 21.9% of soluble reactive P (SRP) in applied poultry litter was transported in runoff as compared to 3.9% of TN, 5.0% of NH_4-N, and 15.3% of SRP from dairy feces and urine. When rainfall was applied two weeks afterwards, less than 1% of applied TN, NO_3-N, and NH_4-N was transported in runoff for all treatments, whereas 4.7% of applied SRP from dairy feces and urine and 2.0% of applied SRP from poultry litter was transported in runoff.

Bingham et al. (1980) found that TKN, TP, and total organic carbon (TOC) were reduced when runoff events occurred three days after poultry waste application, rather than directly afterwards. Similarly, Kleinman and Sharpley (2003) evaluated P losses in runoff from dairy, layer poultry, and swine manure over successive rainfall events in Pennsylvania. They reported that soluble P was related to runoff P loss at higher manure application rates. Below 75 kg TP ha^{-1}, no relationship was reported between water extractable P (WEP) and P concentration in runoff. At manure application rates containing greater than 50 kg TP ha^{-1}, dissolved reactive phosphorus (DRP) and TP concentrations decreased significantly with successive rainfall events (3, 10, and 24 days following application). The authors suggested that P indices could be changed to reflect that applied manure may influence runoff P concentration for a limited time. Pierson et al. (2001) also suggested that NH_4-N and DRP losses were limited if runoff did not occur soon after application of poultry litter. The implications of these studies to riparian grazing management are important, as feces and urine deposited directly before a rainfall would be likely to contribute more nutrients to surface runoff, representing a worst-case scenario in terms of potential environmental contamination.

Researchers in France studied the fate of ^{15}N-labeled cattle urine on Calciudoll, Hapludoll, and Hapludalf soils over spring, summer, and fall seasons for two years (Decau et al., 2003). Urinary-N uptake by the ryegrass (*Lolium perenne* L.) herbage averaged 37% for the Calciudoll, 46% for the Hapludoll, and 49% for the Hapludalf. Similar to other studies, N-uptake was greater in the spring and summer than in the fall, leaving fall applied urinary-N vulnerable to leaching. Mean urinary-N recovered in leachate was 9.0% for the Calciudoll, 15% for the Hapludoll, and 1.3% for the Hapludalf.

For all soils, mean spring recovery of urinary-N in leachate was 0.7%, summer recovery was 7.7%, while fall recovery was relatively large at 17%. For all soils and seasons, mean urinary-N recovered in soil organic matter ranged from 25% to 31% of initial applied N.

In New Zealand, a laboratory lysimeter study with ^{15}N-labeled synthetic urine reported N balance information for urine applied to a ryegrass-white clover pasture (Fraser et al., 1994). Forage plant N-uptake accounted for 43% of applied N, the soil N pool accounted for 20%, leachate-N, primarily NO_3-N, accounted for 8%, and 28% was lost in gaseous-N form due to denitrification. Ammonia volatilization was assumed to be minimal as simulated rainfall was applied directly after urine application. A greater proportion of urinary-N was recovered in leachate in a field study in central Pennsylvania. Stout et al. (1997) reported that 25% of urinary-N and 2% of fecal-N were recovered as NO_3-N in leachate. In addition, Stout et al. (1998) also reported a seasonal effect of urinary-N leaching, with greater NO_3-N leaching in the fall compared to spring and summer, likely due to limited plant growth after application of the urine. Urine was applied at an average rate of 99.4 g N m^{-2} in the spring, summer, and fall, whereas feces were applied at a rate of 26.2 g N m^{-2} in the summer only. Under applied feces, NO_3-N losses were not significantly different from that of control lysimeters with no applied feces or urine. The researchers suggested that this was due to the lower amount of N in feces compared to urine as well as a low mineralization rate of organic-N in the feces.

In France, Loiseau et al. (2001) examined NO_3-N leaching under pure perennial ryegrass or white clover (*Trifolium repens* L.) forages, a mixture of the two, and bare plots. Sheep feces and synthetic urine solution were applied to lysimeters four times

annually based on the amount of forage produced from the plots. Mean N loss was 129 kg N ha^{-1} for the bare plots, 72 kg N ha^{-1} for white clover, and less than 10 kg N ha^{-1} for plots containing ryegrass in pure stands or mixtures. While the legume reduced N-leaching compared to bare ground, the grass cover was most effective, likely due to a higher use of available soil N than the N-fixing legume.

CATTLE TREADING AND HEAVY USE AREAS

Cattle often do not evenly utilize all areas of available pasture. Rather, they typically create certain heavy use or "lounging" areas that can have increased levels of soil compaction, less vegetative cover, and greater concentrations of nutrients compared to other areas of the pasture (Greenwood and McKenzie, 2001; Haynes and Williams, 1999; Sheath and Carlson, 1998). Mathews et al. (1994) reported N and P accumulation in the third of pasture near shade, water, and supplemental feed, and in Venezuela, Buschbacher (1987) reported that the 30% of the pasture nearest the lounging areas accounted for over half of all cattle defecations. Livestock lounging areas in the riparian zone may be especially detrimental to water quality. In this case, close proximity to streams shortens the length of buffer vegetation between the lounging area and the stream to filter sediment and nutrients leaving the site in runoff.

Cattle hooves can exert a force of over 800 kN m^{-2} (Webb and Clark, 1981) and a static pressure of ~100 to 200 kPa on the soil (Greenwood and McKenzie, 2001). Saturated soils are more susceptible to treading damage, whereas protective ground cover and well-drained soils limit susceptibility to damage from cattle hoof action (Climo and Richardson, 1984). Generally, cattle will congregate in moist areas for available forage

and water, cooler temperatures, and shade (Belsky et al., 1999). These factors combine to increase the likelihood of heavy use areas in grazed riparian zones.

While there has not been a great deal of research on heavy use area conditions in pastures, research on soil conditions in cattle feedlots may provide useful information in many instances. Mielke et al. (1974) reported that three layers develop on top of the soil profile in a confinement cattle feedlot: a layer of manure on the surface, an interface layer with an organic-mineral soil mixture, and the third layer representing the top of the soil profile which has been affected by cattle compaction and nutrients from the feces and urine. While the absorption of water into the surface layer may be significant, overall soil infiltration is minimal. In addition to compaction, products of microbial decomposition further reduce soil infiltration by plugging soil pores. This is analogous to findings of Nguyen et al. (1998) who reported decreased infiltration as a result of cattle treading, as well as increased load of sediment and nutrients from small hillside pasture plots. Warren et al. (1986) also reported consistently lower mean infiltration rate following grazing, while Liacos (1962) reported increased compaction and lower water storage under heavy grazing conditions.

Clary (1995) studied riparian responses to grazing and compaction in Oregon and Idaho. To impose a compacted treatment, a 14-kg steel "impactor" with a surface area of 100 cm^2 was dropped from a 75-cm height. The procedure was selected to produce a similarly deep impression as a hoof print from a mature cow. On average, this compaction increased the compressive strength of the soil 59%, from 1.57 kg cm^{-2} to 2.50 kg cm^{-2}, at an Oregon site with a sandy loam to loamy sand surface. Treatment responses to compaction were not as pronounced on the Idaho site with high surface organic

content, where the spring compaction did not increase soil strength and the late summer compaction increased soil strength from 1.77 kg cm^{-2} to 2.11 kg cm^{-2}. The response of soils and vegetation to treading seems to be in large part a function of site conditions.

CATTLE FECES AND URINE

Cattle feces and urine output are important aspects of nutrient export from riparian pastures. Larsen et al. (1994) estimated average fecal output of cattle at 2 to 3 kg $defecation^{-1}$ for a 450-kg animal. In two wheatgrass pastures in central Utah, Julander (1955) determined defecation rates to be 11.2 and 11.6 defecations day^{-1}. When related to stocking rates, these values can provide the basis for estimates of the total amount of nutrients applied to a riparian area during a grazing period.

Of the total N excreted by cattle, 70 to 75% is excreted in urine and 25 to 30% is excreted in the feces (Doak, 1952). The urine contains between 2.5 and 8.3 g N L^{-1}, of which about 50 to 75% is in the form of urea. Nitrogen deposited by cattle urine can be taken up by plants, remain in the soil, volatilize to the atmosphere as NH_3^+, or it can leave the site as runoff or leachate of urea, nitrite (NO_2^-), NO_3^-, or NH_4^+. Ball and Ryden (1984) reported that urine has the greatest potential for N losses, through NH_3 volatilization or leaching of NO_3^- and NH_4^+. The rate of N application is likely to affect the amount of N susceptible to leaching or runoff. Ball et al. (1979) reported that when urine was applied to ryegrass-white clover plots at a rate of 300 kg N ha^{-1}, 37% of the N was recovered by the forages, whereas just 22% was recovered when urine was applied at a rate of 600 kg N ha^{-1}. Matching the amount of feces and urine deposited on a pasture to

the nutrient requirements of forages will likely reduce the potential of nutrient loss in leachate or runoff from the pasture.

In a broad survey of P in animal manures, Barnett (1994) reported TP values for beef feeder cattle feces at 6.7 g kg^{-1} on a dry matter basis. About 45% of this P was in an inorganic form, but the author suggested that values are in large part a function of diet. Of the inorganic fraction of P in dairy manures, Sharpley and Moyer (2000) reported that 81% was water extractable and vulnerable to transport as a dissolved form in runoff or leachate. In a study of pasture fertilization, During and Weeda (1973) reported a mean annual recovery of 39% of P from cattle feces by mixed grass-clover forages. They also reported lower sorption of P from cattle feces in the soil compared to P from applied super-phosphate. While manure from grazing cattle generally has a lower concentration of P than other livestock manures, low soil sorption and high percentage of water extractable P require that P be well managed in cattle grazing systems to prevent environmental contamination.

CONCLUSION

The riparian zone acts as a critical filter to protect water quality from human impacts, including agriculture. However, continuous grazing of livestock in riparian areas can reduce effective filter length of the buffer area. Consequently, it is critical that livestock are not permitted to degrade riparian areas to a degree that may negatively impact water quality. Vegetative cover has been shown to be important in filtering nutrients and sediments. Data are lacking, though, as to the impact of varying levels of

ground cover to sediment and nutrient export from cattle feces and urine in the riparian zone, specifically in North Carolina and the southeastern USA.

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CHAPTER 2

RUNOFF, SEDIMENT, AND PHOSPHORUS EXPORT FROM MANURED RIPARIAN PASTURE AS AFFECTED BY SIMULATED RAIN AND GROUND COVER

D.M. Butler, D.H. Franklin, N.N. Ranells, M.H. Poore, and J.T. Green, Jr.

ABSTRACT

The impact of livestock pastures on sediment and phosphorus export to surface waters in North Carolina is not well documented. The objective of this work was to determine the effect of ground cover on sediment and phosphorus export from pastured riparian areas. Plots 0.75 m by 2.0 m were established on 10% and 20% slopes of existing stands of mixed tall fescue (*Festuca arundinacea* Schreb.) / dallisgrass (*Paspalum dilatatum* Poir.), with stands modified to represent a range of ground cover levels by establishing 100% bare soil area with 0% ground cover (bare ground), 55% bare soil area with 45% ground cover (low cover), 30% bare soil area with 70% ground cover (medium cover), and not altering full vegetative cover plots (high cover). The bare ground treatment was also compacted to simulate a cattle heavy use area. Measured forage canopy cover on vegetated plots was generally higher than the level of ground cover established at the soil surface. At 45% ground cover, canopy cover was 63 to 78%, at 70% ground cover canopy cover was 76 to 83%, and at full ground cover canopy cover was 83 to 98%. Rainfall simulators were used to evaluate runoff volume, total suspended sediment (TSS), total Kjeldahl phosphorus (TKP), and dissolved reactive P (DRP) export from plots receiving cattle feces and urine at a rate representing stocking of ~1.4 cows ha^{-1} yr^{-1}. Rainfall simulation events were conducted in April 2003 to evaluate baseline runoff conditions. Feces and urine were applied in May and Sept 2003 and immediately

followed with rainfall simulations. Additional rainfall simulations without additional application of feces and urine occurred in June and Oct 2003. Mean runoff volume from bare ground was generally greater than from low, medium, and high levels of cover, which had similar mean runoff volume. Export of TSS was greater from 20% than 10% slope on bare ground and low cover, but when cover levels were medium or high, TSS export was similar at both slopes. With the exception of plots at bare ground, TSS export was generally greater during fall rainfall simulations than during spring simulations. Export of DRP was elevated after application of feces and urine to plots, generally with the greatest export in Sept. However, there was no consistent response of DRP export to cover at each rain event. Mean TKP export was similar at low, medium, and high cover over all levels of slope and rain event, suggesting that the low level of cover may be sufficient to preventing excessive TKP export. These results indicate that heavy use areas in the riparian zone have the potential to contribute a substantial amount of sediment and phosphorus to surface waters. Canopy cover of ~70% may be equally effective as full ground cover at reducing sediment and nutrient export.

INTRODUCTION

Nonpoint pollution of agricultural phosphorus can contribute to eutrophication of surface waters (Carpenter et al., 1998). Eutrophication can lead to algal blooms, reduced dissolved oxygen levels, fish kills, reduction in biodiversity, and other potential negative impacts which may reduce the suitability of surface waters for beneficial ecosystem functions and use by humans. Increased sediment loads have been widely reported to

reduce populations of benthic organisms and fish, as well as overall primary productivity of aquatic ecosystems (Cooper, 1993).

Grazing cattle (*Bos* spp.) can be a source of nutrients and sediment to surface waters in the eastern USA (White et al., 1980). However, the impact of grazing livestock on nutrient and sediment export to North Carolina surface waters has not been well quantified. In 1997, it was estimated that over 710 thousand hectares or ~16% of North Carolina agricultural land area was used for grazing (USDA, 1997), which makes the impact of grazing on water quality critically important. Specifically, the importance of riparian area management to quality of receiving surface waters underscores the need for scientific investigation of environmental impacts when riparian areas are grazed.

Riparian areas are often grazed because they are typically unsuitable for row crop production due to topography and seasonal flooding, but can have relatively high forage productivity because of favorable moisture conditions, even during drier periods of the year. However, poor grazing management can lead to variable stand density and forage ground cover (Alderfer and Robinson, 1947), which can negatively influence infiltration, runoff, erosion, and sediment deposition (McGinty et al., 1979; Self-Davis et al., 2003) and limit the ecosystem services provided by the riparian area. However, while there are data relating to runoff volume, sediment, and nutrient export from grasslands in the Southeast, there are few data that report the level of ground cover necessary to protect water quality when livestock graze in or near riparian areas.

Several studies in diverse environments have suggested threshold levels of 70% to 75% ground cover, below which significantly greater runoff and sediment loss can occur (Costin, 1980; Lang, 1979; Mwendera and Saleem, 1997). Lang (1979) suggested that

when ground cover, as estimated by the point or area quadrat method, dropped below 75% bare areas began to connect with each other which allowed for faster flow of runoff and less time for infiltration. Similarly, Costin (1980) reported that ground cover values less than 70%, also estimated using the quadrat method, resulted in significantly greater mean soil loss and runoff rate than at ground cover levels greater than 70%. For ground cover levels below 70%, runoff as a proportion of rainfall increased to above 15% of rainfall with decreasing levels of ground cover, with a maximum value of over 30% of rainfall seen as runoff. With decreasing ground cover, soil loss increased above 5 g m^{-2} up to a maximum value of 60 g m^{-2}.

Slightly different results were reported by Dadkah and Gifford (1980) in a Utah study on infiltration rate and sediment production. Cover was established by uprooting vegetation in randomly selected squares of a grid. When rainfall was applied at a rate of 76 mm h^{-1}, the researchers reported no difference in infiltration rate on 0.5 m by 0.75 m plots with either 50 or 80% ground cover. Mean sediment yield for plots at 30% cover and no trampling was 1356 kg ha^{-1}, while plots at 80% cover yielded significantly less sediment of 665 kg ha^{-1}, and plots with 50% cover yielded 787 kg ha^{-1}. The researchers suggested that ground cover levels of 50% or greater may be sufficient for adequate watershed protection.

In Arkansas, Self-Davis et al. (2003) examined the effects of canopy cover on runoff volume from plots 3 m by 6 m on a 5% slope. Runoff from several warm season forage species and tall fescue was examined. Plots were either unclipped or clipped to 6 or 10 cm and subjected to rainfall simulation at a rate of 50 mm h^{-1}. Though there was a trend towards greater runoff volume, clipping did not significantly increase volume from

any forage species compared to unclipped plots, even though canopy cover was reduced after harvest. Tall fescue generally had the least runoff of the forage species, and even though a cool season grass, showed the potential for relatively low runoff during the entire year.

Edwards et al. (2000b) examined relationships between tall fescue clipping height and nutrient export from feces and urine applied to 2.4 m by 6.1 m plots. Feces was applied at a rate of 0.5 kg plot^{-1} day^{-1} (525 mg P plot^{-1} day^{-1}) and urine applied at a rate of 330 mL plot^{-1} day^{-1} (~33 mg P plot^{-1} day^{-1}) for 7 days, followed by a rest period of 21 days. There were few differences between total phosphorus (TP) concentrations in runoff between treatments of forage unclipped or clipped to heights of 2.5, 10, or 20 cm. Application of urine to plots in addition to feces did not increase TP concentrations over those from plots with feces application alone. As expected, the lowest TP concentrations in runoff were observed from plots with no application of feces and urine, and during rainfall events 21 days after feces and urine application, rather than immediately after application of manure. Results for orthophosphate-P (PO_4-P) concentrations were similar, with PO_4-P accounting for 85% of TP.

In South Carolina, McLeod and Hegg (1984) examined runoff water quality from tall fescue pasture with dairy manure, poultry litter, or municipal sludge applied at a target rate of 112 kg N ha^{-1}. Simulated rainfall at a rate of 37 mm h^{-1} was applied and runoff sampled at 1, 7, 14, and 21 days after manure or fertilizer applications. Concentration of TP in runoff from poultry litter averaged ~12 mg L^{-1} and reduced to ~5 mg L^{-1} at 7 days. Concentration of TP in runoff from dairy manure was initially ~8 mg

L^{-1} and reduced to ~7 mg L^{-1} at 7 days. Pierson et al. (2001) also suggested that DRP losses were limited if runoff did not occur soon after application of poultry litter.

This paper is the first of three companion papers from a study that evaluated the impact of ground cover in riparian pasture plots to the export of sediment and nutrients. This paper examines the impact of bare ground and low, medium, and high ground cover levels to runoff volume and export of total suspended sediment (TSS), dissolved reactive phosphorus (DRP), and total Kjeldahl P (TKP) on two sites of 10% and 20% slope. The second paper examines nitrogen constituents of nitrate-nitrogen (NO_3-N), ammonium-nitrogen (NH_4-N), total Kjeldahl nitrogen (TKN) and total nitrogen (TN) in runoff from those same plots (Butler et al., 2004a). A third paper examines TSS, DRP, TKP, NO_3-N, NH_4-N, TKN, and TN in runoff from wetland plots of bare ground or full cover adjacent to the plots examined in the first two papers (Butler et al., 2004b).

MATERIALS AND METHODS

In February 2003, research plots were established on existing mixed tall fescue (*Festuca arundinacea* Schreb.) / dallisgrass (*Paspalum dilatatum* Poir.) pasture which had been established for over 20 years at North Carolina State University's Lake Wheeler Road Field Laboratory, Raleigh, NC (35°43'N; 78°41'W; elevation = 100 m). Randomized complete block experiments were established on two slopes (~10% and ~20%), 5 to 10 m landward from a stream. Soils were Appling sandy loam (fine, kaolinitic, thermic Typic Kanhapludults) on the 10% slope site and Wedowee sandy loam (fine, kaolinitic, thermic Typic Kanhapludults) on the 20% slope site. Upon initiation of this study, ground cover treatments were established at 95%, 70%, 45%, and 0%, with

four replications of each treatment, for a total of sixteen plots on each site. The 0% cover treatment was established in each replication to simulate a compacted, cattle heavy use area. Soil cores were examined prior to plot establishment to ensure consistency of soils within replication and within slope.

Experimental plots were 0.75 m by 2.0 m and delineated with flashing 23 cm wide, placed into the soil to a depth of 18 cm to isolate the surface hydrology of the plots. A runoff collection gutter was placed at the downslope edge of each plot.

Ground cover levels were created by establishing 100% bare soil area with 0% ground cover, 55% bare soil area with 45% ground cover, 30% bare soil area with 70% ground cover, and not altering full vegetative cover plots. Bare soil areas were created using a grid of 5 cm by 10 cm rectangles and a small hand torch to quickly heat and kill statistically random blocks of plants. The random blocks created a pattern of varying sizes of bare soil areas as selected blocks connected with each other. After using the torch, any herbage residue was then raked from the plots. The plots were given time to recover and any remaining residue to be removed during subsequent natural rainfall events. Small rectangles of black plastic were placed on the plots in the same random grid pattern to maintain desired ground cover between rain events once the treatments of feces and urine were applied to the plots. All plastic was removed before simulated rain events.

Simulated lounging treatments were established by using black plastic to cover the whole plot and solarize all vegetation. A steel compaction device with an impact surface area of ~100 cm^2 was used to simulate cattle hoof compaction over the entire plot, using methods described by Clary (1995). Plots were compacted before the first

simulated rain event in the spring and again before the first fall rain event. There was no attempt to simulate compaction from cattle hoof action in vegetated plots.

To standardize canopy height at each rain event, plots were harvested to a 10-cm stubble height prior to each rainfall simulation. Plots were also harvested in July 2003 for plot maintenance as well as in November 2003 to obtain a seasonal estimate of forage production. Forage samples were weighed and dried, then analyzed by the North Carolina Dept. of Agriculture and Consumer Services (NCDA&CS) Agronomic Division for total P (TP) by the photometric method described in AOAC method 965.17 (Cunniff, 1995). Percent canopy cover was determined using the line transect method (Laflen et al., 1981) with 40 points after the vegetation was harvested to 10 cm and just prior to the rainfall simulations.

Before each simulated rainfall, three soil cores of 0- to 5-cm depth (1.75-cm inside diameter) were obtained from each plot. The cores from individual plots were combined to form a composite sample, which was divided into two subsamples. One sample was air-dried and ground, while the second was placed in a soil tin and dried at 105ºC to determine gravimetric soil moisture content. Mehlich-3 soil P was determined on the dried and ground sample using methods described by Mehlich (1984).

Three rainfall simulators (Tlaloc 3000, Joern's Inc., West Lafayette, IN) were used to simulate rainfall at an intensity of 70 mm h^{-1} for a 1-h duration. This is just above a 10-yr, 1-h rain event of ~65 mm h^{-1} for Raleigh, NC, whereas a 25-yr, 1-h rain event for Raleigh is ~85 mm h^{-1} (National Weather Service: Eastern Region Headquarters (NOAA-ERH), 2004). Each simulator rained on two plots simultaneously. In April 2003, an initial rainfall simulation was conducted to determine baseline runoff conditions from

each plot prior to application of feces and urine. Deionized water was used as source water for the rainfall simulators. Great care was taken to ensure equal volume and distribution of rainfall from each simulator each time used. Simulators were calibrated by measuring volume and distribution of rain and adjusting the pressure valve at each simulator accordingly before moving simulators onto plots. Rain gauges were placed in plots to verify rainfall rate for each rain event.

Time was recorded as runoff began to drip from the gutter and the first 125-mL of runoff was collected. When steady flow began, time was again recorded and a timer started so that runoff could be sampled beginning at 5, 10, 15, 20, 25, and 30 min after initiation of steady runoff flow. At each sampling time interval, 500-mL of runoff was obtained. Runoff between samplings was collected into large bins and weighed every 5 min after the initiation of steady runoff state until the end of the 1-h simulated rain event. This runoff was sampled at 30 min and the bin emptied, then sampled again at the end of the rain event. The same methods of rainfall application and runoff collection were used for all rainfall simulations.

On May 20, 21, 27, or 28, 2003, feces and urine were applied to plots immediately before rainfall simulations at a rate that approximated 10% of the average daily output for mature cattle. This equated to a 2.4-kg deposit of feces (85% moisture) applied on a 550-cm^2 area and a 1-L urine deposit applied over an equal area. The center of the fecal deposit was placed 30.5 cm from the top of the plot and centered between the plot sides. The 1-L urine deposit was placed directly down slope of the fecal deposit.

The feces and urine were collected from four beef steers fed switchgrass (*Panicum virgatum* L.) and gamagrass (*Tripsacum dactyloides* (L.) L.) hay as part of an

unrelated study. Feces were mixed and formed into 2.4-kg portions (~0.33 kg dry matter) 25 cm by 14 cm wide and 6 cm thick before being frozen until thawed for plot application. Urine was collected in buckets placed under the steers while in metabolism crates. All collected urine was mixed, adjusted to a pH level between 5 and 6 with hydrochloric acid to prevent precipitation of solids and nitrogen losses to ammonia (NH_3), and then frozen until thawed for plot application. Feces and urine were analyzed by the NCDA&CS Agronomic Division for TP by inductively coupled plasma-atomic emission spectrometry (ICP-AES) as described in US-EPA (United States Environmental Protection Agency) method 200.7 (US-EPA, 1992). Water extractable P (WEP) was determined by methods described by Kleinman et al. (2002b). Fresh manure samples (1-g dry weight equivalent) were shaken with 200 mL of deionized water for 1 h, centrifuged, filtered through 0.45-µm cellulose nitrate membranes, and analyzed for dissolved reactive P (DRP) by the molybdate blue method (Murphy and Riley, 1962).

Following rainfall simulations in May 2003, plots were covered with translucent plastic attached to a hoop frame ~1 m above the plot surface to prevent precipitation from reaching the plots during natural rain events. Plots remained covered until the final forage harvest of the experiment in November 2003.

Rainfall simulations were again conducted June 9 to 11, 2003 to determine runoff constituents from plots, but without additional application of feces and urine. Plots were also treated with feces and urine before rainfall simulations on September 9 to 11, 2003, which were followed by rainfall simulations October 6 to 8, 2003 without additional plot treatment with feces and urine. Feces and urine application in September was on the same location on the plot, but the fecal material remaining from the spring was manually

removed from the plot, weighed, and a 30-g sample removed for TP determination. The remaining fecal deposit was manually broken up into ~25 pieces of similar size and distributed in the 2500-cm^2 area surrounding the original fecal deposit immediately prior to the new application of feces and urine in Sept. Total feces and urine application to plots represented a stocking rate of ~1.4 cows ha^{-1} yr^{-1}.

Runoff sample vials were placed in ice in the field until transport to the lab. Vacuum filtration of 100 mL of runoff sample through 0.45-µm cellulose nitrate membranes was used to determine concentration of TSS. Filters were dried at 105ºC and weighed before and after filtration. DRP was determined by analyzing the filtered sample by the molybdate blue method (Murphy and Riley, 1962). TKP was determined similarly following Kjeldahl digestion of an unfiltered sample according to US-EPA method 365.4 (US-EPA, 1979).

Samples collected at 5-min intervals represented point estimates of concentrations and were plotted versus cumulative runoff volume. The points were joined with straight lines and the area under the straight lines was integrated using the PROC EXPAND procedure to determine cumulative mass of sediment and nutrients lost at each collection time (SAS Institute Inc., 1994). For an estimate of total export for the 1-h rain event, cumulative mass export was added to export estimates obtained from the post 30-min composite sample.

The effect of cover, slope, and month of rain event to runoff volume, TSS, DRP, and TKP mass export and concentrations was determined by using the PROC GLM procedure (SAS Institute Inc., 1994). Means were separated using Fisher's least

significant difference. Unless otherwise noted, all differences were considered to be significant at $P < 0.05$.

RESULTS AND DISCUSSION

Cover

Target ground cover levels of 0%, 45%, 70%, and 95% were established at the soil surface. To facilitate discussion of ground cover treatments, 45% ground cover will be considered as low cover, 70% ground cover will be considered as medium cover, and 95% ground cover will be considered as high cover, with the 0%, compacted treatment referred to as bare ground. Line transect measurements of forage canopy cover were generally not closely matched to the established level of ground cover (Table 1). Given the data from all rain events, canopy cover at low cover treatments ranged from 63 to 78%, canopy cover at medium cover ranged from 76 to 83%, and canopy cover at high cover ranged from 83 to 98%. Mean canopy cover differed at each level of ground cover for every rainfall event, except during Oct. In Oct, no difference in mean canopy cover was observed between low, medium, and high ground cover levels.

In this study, thicker forage growth in grid rectangles that were selected to include live forage may have accounted for increased canopy cover measurements at 45% and 70% ground cover levels. As the forage in these areas of the plot grid grew thicker as the season progressed, they were likely able to increase canopy cover values through some shielding of adjacent bare grid rectangles. This is important to note, as this shielding effect could also shield soil in adjacent blank grid rectangles from raindrop impact.

The forage in the plots was a mixture of tall fescue and dallisgrass, but the proportion of each was not consistent over the growing season. In Apr and May, nearly all forage was tall fescue (> 95%). As the season progressed, dallisgrass represented a greater percentage of the stand, reaching > 50% in Sept. This proportion is only an estimate, as forage species composition was not quantified at each rain event. However, such forage stands are typical of established pastures in North Carolina as warm season species provide a much greater percentage of forage during the summer months.

The amount of P removed at forage harvests can be somewhat related to cover level as more dense cover is likely to produce more forage dry matter. However, in this study there was not great variation in total forage P uptake by low, medium, or high cover treatment levels (Table 2). There was a far greater amount of P represented as removed at the Sept event, as this included total P uptake for the entire summer, after the June rain event.

Rainfall Rate and Distribution

At simulated rain events, a rainfall rate of 70 mm h^{-1} was the output target for each rainfall simulator. However, rainfall rate determined from rain gauges at the plots recorded a higher mean rainfall rate of 102 mm h^{-1}. There was some variation among rain event and cover (Table 3), though the numbers should be examined with caution, as rain gauges were of a different design and somewhat less reliable at the April and May events compared to the June, Sept, and Oct events. Mean rainfall rate and rainfall rate recorded adjacent to feces and urine deposit were included as covariates in initial statistical models to account for any variation, and were significantly related to export of some runoff constituents that will be noted.

Feces and Urine

Relatively low agronomic rates of P (< 10 kg P ha^{-1}) were applied to plots at each application in feces and urine (Table 4). Though effort was made to apply an equal amount of P in May and Sept, slightly less P was present in samples of Sept applied feces and urine than in May samples. This may be due to inherent variability within the feces and urine, considering feces and urine for both May and Sept applications were collected, mixed, and packaged for storage at the same time. The amount of WEP in manures has been reported to be highly correlated to runoff P (Kleinman et al., 2002a) and in this study, WEP in feces was 2.12 kg P ha^{-1} in May and 2.43 kg P ha^{-1} in Sept.

Soil Phosphorus

Mehlich-3 soil P varied slightly on the two sites in this study. On the site of 10% slope, soil P averaged 58.8 mg P kg^{-1} (44.1 kg P ha^{-1}, 0 to 5 cm soil depth), while the site of 20% slope soil P was 47.5 mg P kg^{-1} (35.6 kg P ha^{-1}). There was also some variation in soil P at the levels of cover (Appendix 5.2). This was expected, as the greater biomass of forage in the plots of greater cover utilized more P for growth, leaving less extractable P in the soil. Soil P was included as a covariate in statistical analysis of runoff P constituents if found to be significant in initial analysis.

Runoff Volume

Initial analysis indicated that soil moisture and rainfall rate were not significantly related to total runoff volume. Further analysis indicated that cumulative runoff volume 30 min after runoff initiation and total runoff volume for the 1-h rain event were significantly related to cover ($P < 0.0001$) and month of rain event ($P < 0.05$), but not to slope (Table 5). An interaction between cover and month of rain event was also evident

($P < 0.0001$). Therefore, runoff volume means are reported separately by event and by level of cover, but averaged over both slopes.

Total runoff volume from the 1-h rain event did not differ between low, medium, and high cover for any rainfall simulation event (Fig.1). The bare ground treatment differed from all other cover levels in every month except June, when the bare ground and low cover treatments were similar. Cumulative volumes of runoff calculated every 5 min displayed similar patterns of differences between cover treatments as did total runoff volume (data not shown). It was expected that bare ground plots produced the greatest volume of runoff, but it was somewhat unexpected that there were no differences between plots at low, medium, and high cover. Several researchers have reported a 70% to 75% range of cover that below which greater volume of runoff is observed (Costin, 1980; Lang, 1979; Mwendera and Saleem, 1997), which was similar to the canopy cover value of the low cover treatment in this study. At low cover, the established ground cover of 45% was also similar to the ~50% threshold level of cover suggested by Dadkhah and Gifford (1980), which was also established at the soil surface.

Differences in mean runoff volume at rain events were also examined at each level of cover. On bare ground, the June rainfall event resulted in less mean total runoff than all other events (Fig. 2). This pattern at bare ground was not consistent across cover treatments as runoff was generally less during fall simulations than during spring rain simulations from plots at low, medium, and high cover. Given that rainfall rate and soil moisture were not significantly related to runoff volume, lower mean runoff during fall rain events may instead have been related to vegetation changes in the pasture plots as plants matured and warm season species such as dallisgrass predominated during the

warmer months. Self-Davis et al. (2003) reported a similar trend of greater runoff volume in May than in July or Oct in a study of canopy height of tall fescue and several warm-season forage species in Arkansas.

Total Suspended Sediment

Initial analysis indicated that soil moisture and rainfall rates were not significantly related to TSS export, and so were not included in statistical models as covariates. Further analysis indicated that cover ($P < 0.0001$), slope ($P < 0.01$), and rain event ($P < 0.05$) were all significantly related to cumulative TSS export 30 min after runoff initiation (Table 6). In addition, there were significant interactions between cover and slope ($P < 0.0001$) and between cover and rain event ($P < 0.0001$), but not between slope and rain event. At each rain event, mean total TSS export for the 1-h rain event and cumulative TSS export 30 min after runoff initiation were greater from the bare ground treatment than at other cover levels. This relationship was generally consistent for each cumulative TSS mass export 5 to 30 min after runoff initiation.

Due to interactions, mean TSS export by rain event and slope was analyzed at each level of cover. From bare ground and low cover treatments, mean cumulative TSS export 30 min after runoff initiation at 10% slope was less than one-half the export at 20% slope (Table 7). This was expected as other researchers have reported greater soil loss (Mwendera and Saleem, 1997) and reduced trapping of sediment (Jin and Romkens, 2001) with increased slope. However, at medium and high cover, there were no significant differences in mean export between the two slopes. This indicates that the medium cover level (~80% canopy cover) protected sites on slopes of 20% and 10%

equally well. In contrast, the low cover (~70% canopy cover) increased TSS export from plots at 20% slope.

Variation in TSS export was also noted between rain events (Fig. 3). This was unexpected as TSS export would likely be unaffected by application of feces and urine over a relatively small area of the plot. With the exception of bare ground, TSS export generally decreased as the season progressed with the least TSS export during Sept and Oct rain events, consistent with the trend of lower runoff volume during the fall months. At bare ground, the June rain event had the lowest TSS export, which differed from that at all other rain events except May. This is likely related to the low volume of runoff from bare ground plots during the June event, which resulted in less TSS exported from the plots.

Dissolved Reactive Phosphorus

Export of DRP was generally less related to cover, rain event, and slope than other runoff constituents. Initial analysis indicated that soil moisture, Mehlich-3 soil P, uptake of P by forage, and rainfall rates did not affect DRP export. At 30 min after runoff initiation, rain event had the greatest impact ($P < 0.01$) on cumulative DRP export, but cover was also significantly related ($P < 0.1$) (Table 8). There were interactions between cover and rain event ($P < 0.001$) and slope and rain event ($P < 0.05$), though slope alone was not significantly related to DRP export. Concentration of DRP in runoff was also examined, and while rain event was highly significant ($P < 0.0001$), slope and cover were rarely significant at any time of runoff collection (Table 9).

Due to interaction, cumulative DRP export means were examined by rain event at each level of cover (Fig. 4). Mean DRP export from vegetated plots exhibited a similar

relationship among rain event. At low, medium, and high cover, mean DRP export was generally greatest during the Sept rain event and least during the Apr rain event. Low export during the Apr rain event was expected, as there was no application of feces or urine to plots. High export during the Sept event was also expected due to application of feces and urine at that rain event and four months earlier in May. However, DRP export in May did not respond as expected to feces and urine application. While export in May was higher than during the baseline event, export in May did not differ from export during June, except from the bare ground treatment. Mean DRP export in May and June was rather evenly distributed between the two events, whereas in the fall, there was a large export of DRP in Sept and a much smaller export in Oct at each level of cover. At bare ground, there was a trend of lesser export of DRP in months following application of feces and urine.

The pattern of reduced DRP export in months following application of feces and urine was similar to results reported by Franklin et al. (2004) who reported on DRP losses from applied poultry litter. Export of DRP from plots with rainfall applied one month after litter application was reduced to 27% of the export observed when rain was applied to plots immediately after litter application. Working with applied cattle feces and urine, Edwards et al. (2000b) reported a similar relationship in DRP concentration in runoff at some rain events. When rainfall was applied 3 weeks after manure application, DRP concentration in runoff decreased ~0.1 mg L^{-1} in July, ~0.3 mg L^{-1} in Aug, and did not change in Sept compared to DRP concentration from rainfall applied immediately after a manure application period. Similarly, Sauer et al. (1999) reported that soluble reactive P (SRP) export from applied dairy feces and urine was reduced when rain was applied two

weeks after feces and urine application to 31% of the export observed when rain was applied 1 day after feces and urine application.

In this study, means of cumulative DRP export were also examined by level of cover at each rain event (Fig. 5). In Apr and Oct, DRP export from bare ground treatment was greater than that from low, medium, or high cover levels, which did not differ. In May, there was no difference between DRP export at bare ground or low cover and in June, no differences were observed at any level of cover. In Sept, DRP export was lower at medium cover than at high cover, which was the only rain event at which medium and high cover differed. However, neither differed from bare ground or low cover. There are few data that report DRP export as compared to percent ground cover, though Edwards et al. (2000b) reported a trend of increased DRP concentration in runoff with simulated "overgrazed" treatments clipped to 2.5 cm compared to higher clipping heights or no clipping.

Export of DRP was also plotted against actual canopy cover measurements to determine if any differences were evident that were not noted when evaluating DRP export in relation to treatment cover levels. The bare ground treatment was not used in this analysis, which indicated that canopy cover was not significantly related to export of DRP.

Total Kjeldahl Phosphorus

Soluble-P, sediment bound-P, and P in eroded organic matter are constituents of TKP export (Daniel et al., 1994). Initial analysis indicated that soil moisture, Mehlich-3 soil P, uptake of P by the forage, and rainfall rate at the fecal deposit were not significantly related to TKP export. Mean rainfall rate was somewhat related to TKP

export and was included in the model as a covariate. Subsequent analysis indicated that cover had a substantial impact ($P < 0.0001$), rain event had a lesser impact ($P < 0.05$), and slope had no impact on cumulative TKP export 30 min after runoff initiation (Table 10). Interactions were observed between cover and rain event ($P < 0.0001$) and between cover and slope ($P < 0.01$). Therefore, means of TKP export were examined by cover level at each rain event (Fig. 6) and by rain event at each level of cover (Table 12). Mean TKP concentration in runoff was similarly related to cover, slope, and rain event, but only cover was significant at each time of runoff sample collection (Table 11).

Regardless of slope, at each rain event with the exception of June, bare ground plots had greater mean TKP export than plots at low, medium, or high cover, which were similar (Fig. 6). In June, no differences were observed between mean TKP export at any level of cover. This may be due to the low runoff volume from 0% cover plots during the June rain event (Fig. 1 & 2) and lower associated TSS export from those same plots (Fig. 3). Considering the 0% cover plots were compacted in Apr and Sept, the soil surface on these plots during the June rain event was likely less friable at the surface during the June event and may have prevented greater TSS and TKP losses. Thus, TKP at the June event was largely DRP, as is usual when TSS is low (Edwards et al., 2000a). When the bare ground treatment was removed from the analysis, cover was not significantly related to TKP export.

Within each cover level, mean TKP export was examined by month of rain event (Table 12). An expected pattern of increased TKP mass export at rain events with applied feces and urine was not observed. At low, medium, and high cover levels, the Apr and Oct rain events generally had the lowest mean TKP export 30 min after runoff

initiation, though not significantly less than all other treatments. As the baseline event, Apr was expected to have the lowest TKP export, whereas lower export in Oct may be due to increased canopy cover during that rain event for low and medium cover levels (Table 1). At bare ground, only June had a significantly lower TKP export than all other rain events, which is likely related to the low volume in June, as mentioned previously. With the exception of bare ground, the trend of lower mass export of TKP in Apr and Oct was observed for all cover levels, at each time interval after runoff initiation (Fig. 7). Also with exception of 0% cover, a greater mean export of TKP from June and Sept rain events was observed.

CONCLUSION

The greatest mean runoff volume, averaged across slopes and rain events, was observed from plots with bare ground, whereas low, medium, and high cover produced similar mean runoff volume. There was also an indication of a seasonal effect on runoff volume, with less volume from fall rain events. However, replication of this experiment in subsequent years would help verify the seasonal response indicated during this study and possibly suggest seasonal grazing management for sensitive riparian areas that would minimize environmental contamination. Mean TSS export was ~65% less at 10% slope than at 20% slope for bare ground and low cover treatments, while no difference between slopes was observed at medium and high cover. A level of canopy cover between 70% and 80% may be important for protecting higher slope sites from increased TSS loss. Considering canopy cover values were somewhat higher than established ground cover levels in this study, results were similar to those of Mwendera and Saleem (1997) who

suggested critical values for preventing soil losses in Ethiopia at 70% canopy cover at 0 to 4% slope and 85% cover for pasture with 4 to 8% slope.

Export of DRP was elevated after application of feces and urine to plots, and the greatest DRP export was generally observed in Sept. Generally, DRP export was much lower than that seen in other studies in the Southeast, which may in part be a function of moderate Mehlich-3 P values and a relatively low rate of P application in the cattle feces and urine. Many other studies of runoff from grasslands in the Southeast involve application of poultry litter because it is a source for improved pasture productivity. In Georgia, Kuykendall et al. (1999) reported annual losses of 8.60 to 11.5 kg P ha^{-1} yr^{-1} from pasture with applied poultry litter, compared to the mean total DRP export from this study of 0.68 kg P ha^{-1} yr^{-1}.

Export of TKP was closely related to DRP and TSS export, which was somewhat expected as TKP is comprised of sediment-bound P as well as soluble-P. The low level of cover was as effective at reducing TKP export as medium and high cover levels. Generally the proportion of DRP accounting for TKP was 55 to 60%, which is similar to the 68% reported by Kuykendall et al. (1999) from pastures with applied poultry litter, but less than the 85% reported by Edwards et al. (2000b) from applied cattle feces and urine. In New Zealand, Gillingham and Thorrold (2000) reviewed several sources and reported TP losses from pasture ranged from 0.1 to 1.67 kg P ha^{-1} yr^{-1}, which was similar to the mean annual export of 2.34 kg P ha^{-1} yr^{-1}observed in this study. However, under typical precipitation patterns actual annual export would likely be much less than in this study, as the 5 heavy simulated rain events represented very much a worse case scenario. There are also some limitations of converting export from small plot experiments to the

field scale, though processes controlling P transport in runoff are likely similar (Sharpley and Kleinman, 2003).

Ground cover is an important component of runoff, TSS, DRP, and TKP export from pastured riparian areas. Management for cover maintenance and elimination of lounging areas would likely minimize sediment and P export. Results from this study simulating a stocking rate of 1.4 cows ha^{-1} yr^{-1} suggest that full ground cover is not necessary to reduce sediment and P export from riparian pasture, but that forages in riparian areas could be utilized as part of a rotational grazing system where canopy cover of 70% to 80% are maintained. While lower cover than these levels may also be sufficient, such levels were not examined in this study. Future work examining levels of cover lower than those used in this study would likely provide useful information to those managing livestock in and near riparian areas. A range of forage species could also be examined to determine sediment and phosphorus export from forages with differing growth habits and seasonal distribution of growth.

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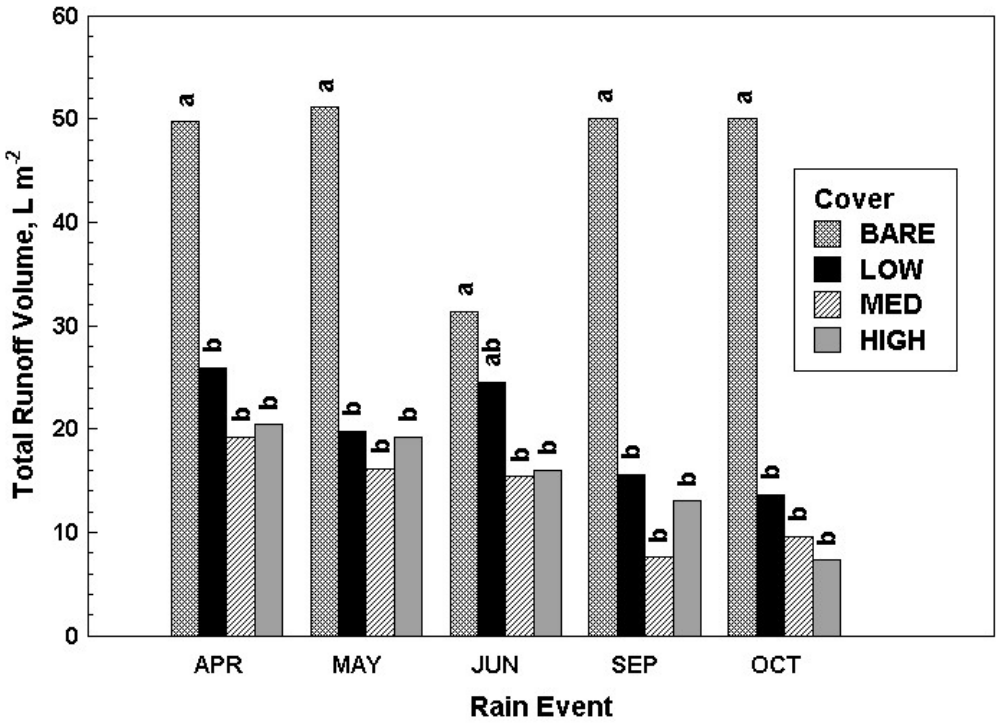


Fig. 1. Mean total runoff volume from 1-h rain event as affected by cover level at each rain event (means within the same rain event group followed by the same letter are not significantly different, P < 0.05)

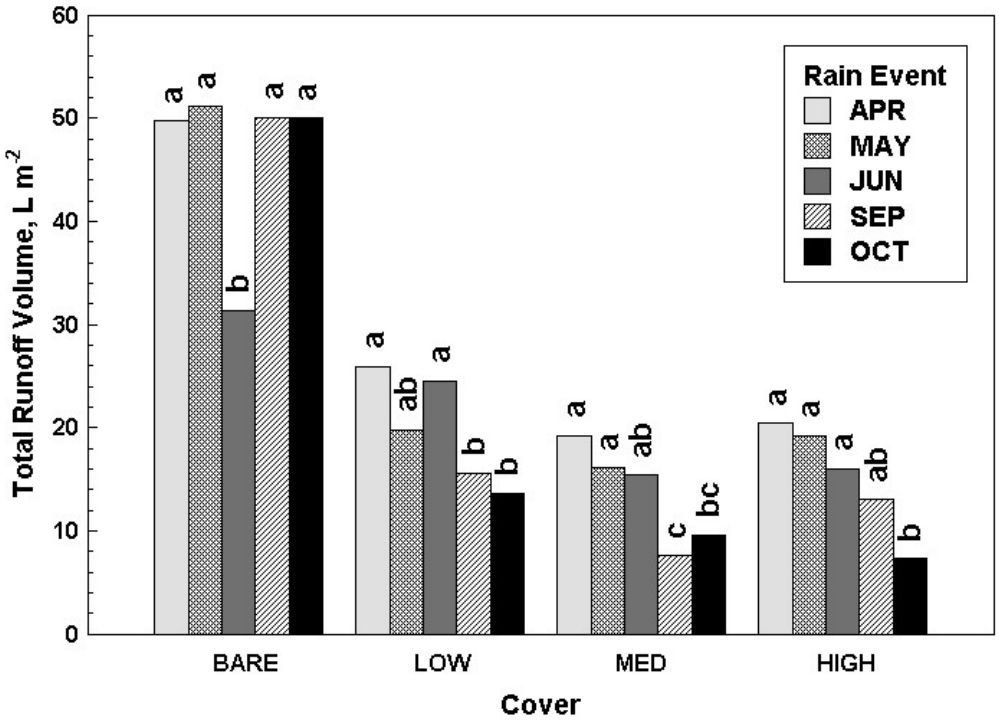
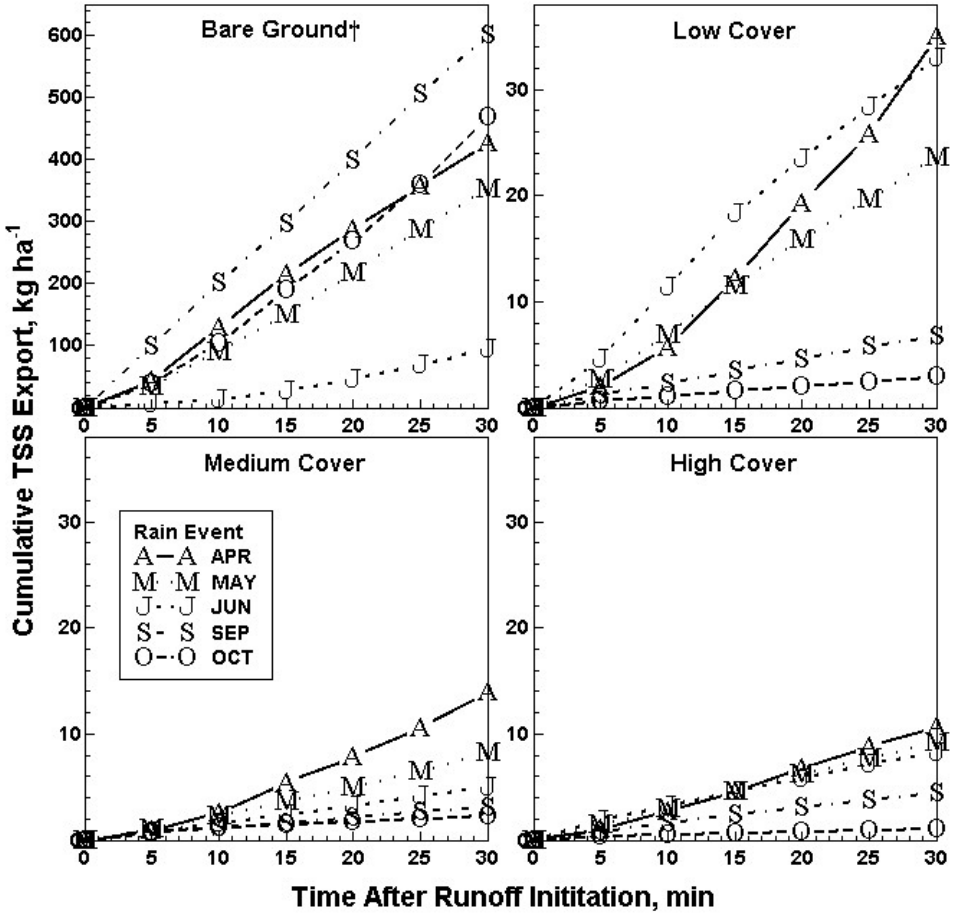


Fig. 2. Mean total runoff volume from 1-h rain event as affected by month of rain event at each cover level (means within the cover level followed by the same letter are not significantly different, P < 0.05)



†larger y-axis range on bare ground plot

Fig. 3. Mean cumulative total suspended sediment (TSS) export as affected by rain event at each cover level

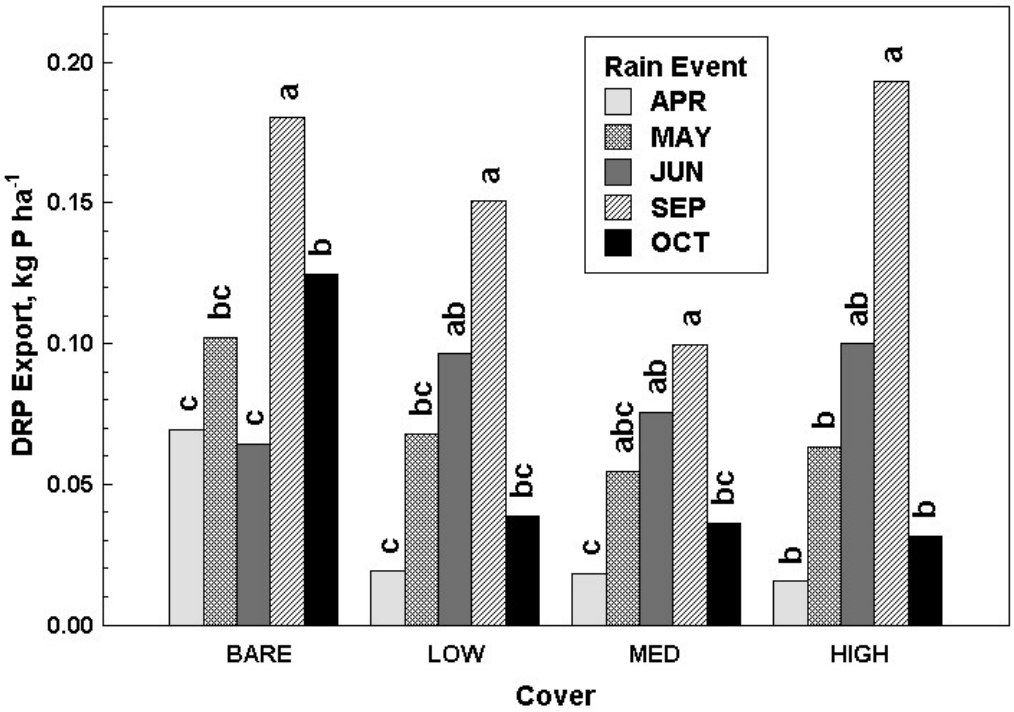


Fig. 4. Mean cumulative dissolved reactive phosphorus (DRP) export at 30 min after runoff initiation, as affected by rain event at each cover level (means within the same cover level followed by the same letter are not significantly different, P < 0.05)

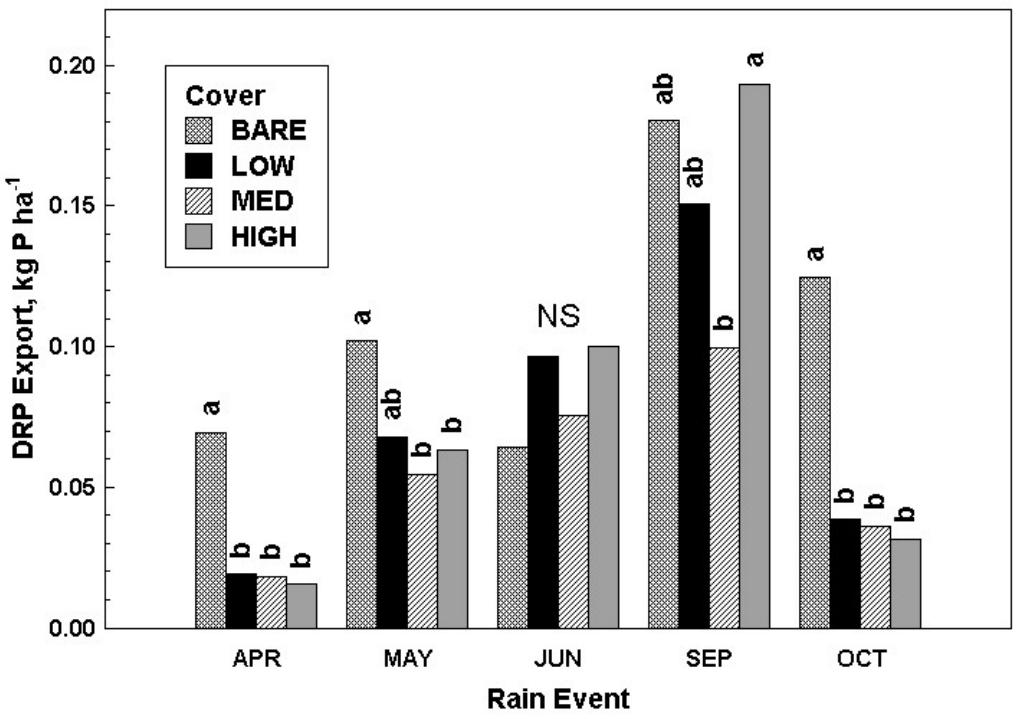


Fig. 5. Mean cumulative dissolved reactive phosphorus (DRP) export at 30 min after runoff initiation, as affected by cover at each rain event (means within the same rain event followed by the same letter are not significantly different, P < 0.05)

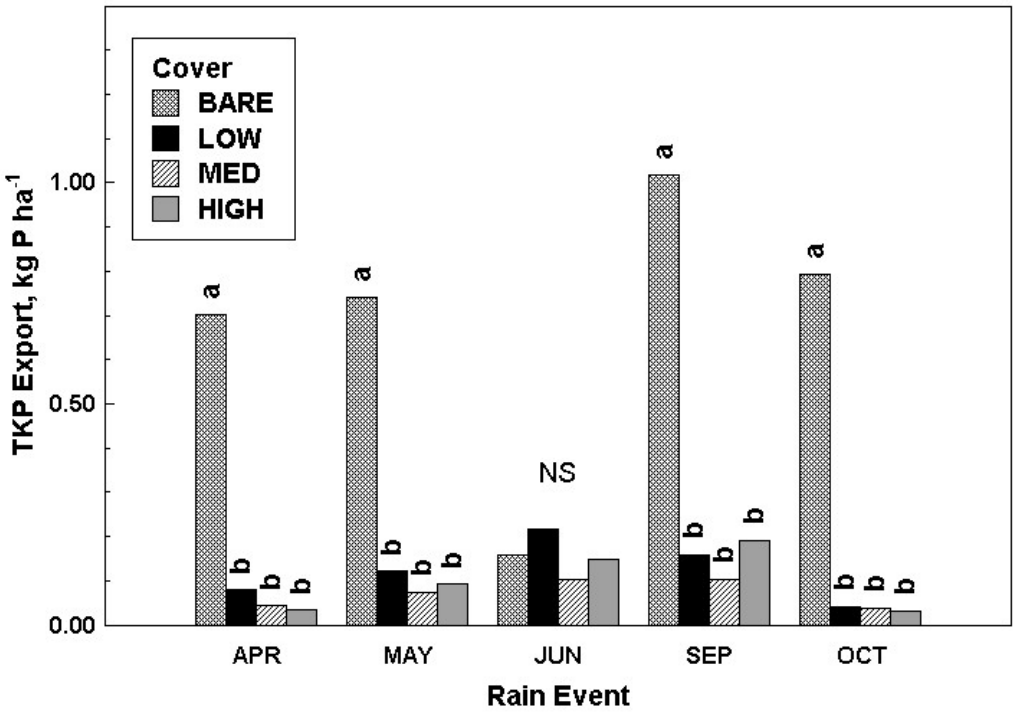
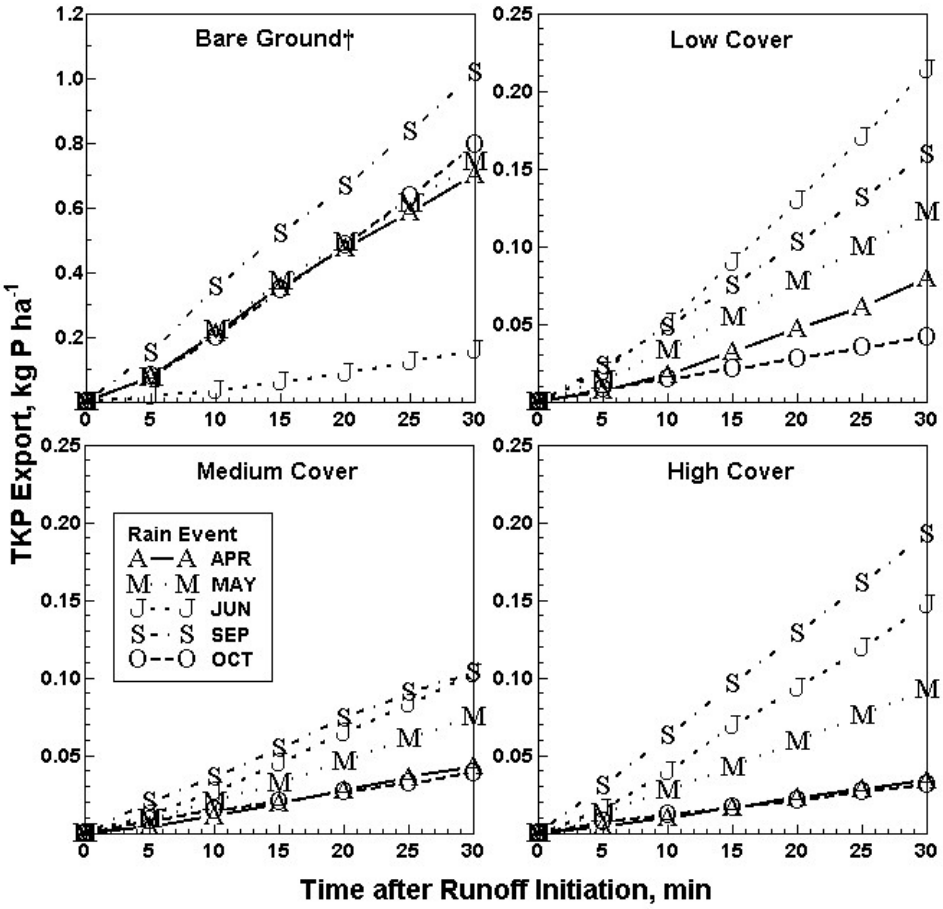


Fig. 6. Mean cumulative total Kjeldahl phosphorus (TKP) export at 30 min after runoff initiation, as affected by cover at each rain event (means within the same rain event followed by the same letter are not significantly different, P < 0.05)



†larger y-axis range on bare ground plot

Fig. 7. Mean cumulative total Kjeldahl phosphorus (TKP) export as affected by rain event at each cover level

Table 1. Mean canopy cover as measured by line transect method at each rain event and ground cover level

Established Ground Cover	Month of Rain Event				
	APR	MAY	JUN	SEP	OCT
	Canopy Cover (%)				
0% (Bare)	0a†	0a	0a	0a	0a
45% (Low)	63b	65b	78b	75b	78b
70% (Medium)	76c	78c	83c	80c	80b
95% (High)	94d	95d	98d	92d	83b

†means within the same rain event with the same letter are not significantly different ($P < 0.05$)

Table 2. Mean forage phosphorus removed from plots at harvest, at each cover level, rain event, and slope

Cover			
BARE	LOW	MED	HIGH
Forage P removed (kg P ha^{-1})			
0.688	2.34	2.79	2.53

Month of Harvest					
APR	MAY	JUN	SEP	OCT	NOV
Forage P removed (kg P ha^{-1})					
0.697	0.540	0.638	8.19†	0.363	0.552‡

Slope	
10%	20%
Forage P removed (kg P ha^{-1})	
2.56	1.61

†includes forage harvested in July
‡not included in calculation of means by ground cover or slope

Table 3. Mean rainfall rate at each rain event and cover level

	Cover	Rain Event				
		APR	MAY	JUN	SEP	OCT
		Rainfall rate, mm h^{-1}				
10% slope	BARE	86.0	129	96.3	104	99.9
	LOW	91.1	114	97.9	107	112
	MED	103	128	110	106	111
	HIGH	85.3	113	96.0	98.7	111
20% slope	BARE	103	86.3	95.1	97.2	108
	LOW	91.3	106	99.3	107	110
	MED	101	97.6	104	99.4	117
	HIGH	65.0	95.6	92.6	110	109

Table 4. Phosphorus (P) concentration in cattle feces and urine and application rate

Date	Manure Component	Total P	WEP†	Total P	WEP†	Total P	WEP†
		____g P kg^{-1}‡____		____g P $plot^{-1}$____		____kg P ha^{-1}____	
21, 22, 27, 28 May	Feces	0.478	0.133	1.15	0.318	7.65	2.12
	Urine	0.165	N/A	0.165	N/A	1.10	N/A
9, 10, 11 Sept	Feces	0.349	0.152	0.835	0.364	5.57	2.43
	Urine	0.118	N/A	0.118	N/A	0.784	N/A

† water extractable phosphorus
‡ wet basis

Table 5. ANOVA: Cumulative runoff volume

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	87	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	NS	NS	NS	NS	NS	NS	NS
Cover	3	****	****	****	****	****	****	****
Cover * slope	3	NS	NS	NS	NS	NS	NS	NS
Rain event	4	NS	NS	NS	+	*	*	*
Cover * event	12	****	****	****	****	****	****	****
Slope * event	4	NS	NS	NS	NS	NS	NS	NS
Cover * event * slope	12	NS	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 6. ANOVA: Cumulative total suspended sediment (TSS) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	87	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	+	*	**	**	**	**	**
Cover	3	****	****	****	****	****	****	****
Cover * slope	3	**	****	****	****	****	****	***
Rain event	4	**	*	*	*	*	*	+
Cover * event	12	****	****	****	****	****	****	***
Slope * event	4	NS	NS	NS	NS	NS	NS	NS
Cover * event * slope	12	NS	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 7. Mean cumulative total suspended sediment (TSS) export 30 min after runoff initiation as affected by slope and cover level

	Cover			
	BARE	LOW	MED	HIGH
	TSS export (kg ha^{-1})			
10% slope	215a†	10.5a	5.88a	4.68a
20% slope	562b	30.0b	7.11a	8.73a

†Means within columns followed by the same letter are not significantly different (P < 0.05)

Table 8. ANOVA: Cumulative dissolved reactive phosphorus (DRP) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	87	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	NS	NS	NS	NS	NS	NS	NS
Cover	3	**	*	*	+	+	+	**
Cover * slope	3	+	NS	NS	NS	NS	NS	NS
Rain event	4	***	****	***	***	**	**	**
Cover * event	12	***	****	****	***	***	***	***
Slope * event	4	NS	+	*	*	*	*	*
Cover * event * slope	12	****	**	*	+	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 9. ANOVA: Dissolved reactive phosphorus (DRP) concentration in runoff

		Time after runoff initiation, min						
		0	5	10	15	20	25	30
Source	DF	P-values†						
Model	88	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	NS	NS	NS	NS	NS	NS	NS
Cover	3	*	NS	NS	*	*	NS	NS
Cover * slope	3	NS	NS	NS	NS	NS	NS	NS
Rain event	4	****	****	****	****	****	****	****
Cover * event	12	****	***	****	****	****	****	NS
Slope * event	4	NS	*	*	NS	*	*	*
Cover * event * slope	12	NS	**	*	*	+	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 10. ANOVA: Cumulative total Kjeldahl phosphorus (TKP) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	88	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	NS	NS	NS	NS	NS	NS	NS
Cover	3	****	****	****	****	****	****	****
Cover * slope	3	**	***	***	**	**	**	***
Rain event	4	****	***	**	*	*	*	*
Cover * event	12	****	****	****	****	****	****	****
Slope * event	4	NS	NS	NS	NS	NS	NS	NS
Cover * event * slope	12	NS	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 11. ANOVA: Total Kjeldahl phosphorus (TKP) concentration in runoff

		Time after runoff initiation, min						
		0	5	10	15	20	25	30
Source	DF	P-values†						
Model	88	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	NS	*	*	+	+	NS	NS
Cover	3	****	****	****	***	****	****	***
Cover * slope	3	*	***	***	NS	NS	**	*
Rain event	4	****	***	**	NS	*	**	NS
Cover * event	12	**	****	****	****	****	****	**
Slope * event	4	+	NS	NS	NS	NS	NS	NS
Cover * event * slope	12	NS	+	+	NS	*	**	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 12. Mean cumulative total Kjeldahl phosphorus (TKP) export 30 min after runoff initiation

Month of Rain Event	Cover			
	BARE	LOW	MED	HIGH
	TKP export (kg P ha^{-1})			
APR	0.70a†	0.079b	0.043b	0.034b
MAY	0.74a	0.12ab	0.075ab	0.093ab
JUN	0.16b	0.22a	0.10a	0.15ab
SEP	1.0a	0.16ab	0.10a	0.19a
OCT	0.79a	0.041b	0.038b	0.031b

†Means in the same column followed by the same letter are not significantly different ($P < 0.05$)

CHAPTER 3

NITROGEN EXPORT FROM MANURED RIPARIAN PASTURE AS AFFECTED BY SIMULATED RAIN AND GROUND COVER

D.M. Butler, N.N. Ranells, D.H. Franklin, M.H. Poore, and J.T. Green, Jr.

ABSTRACT

The impact of livestock pastures on nitrogen export to surface waters in North Carolina is not well documented. The objective of this work was to determine the effect of ground cover on nitrogen export from pastured riparian areas. Plots 0.75 m by 2.0 m were established on 10% and 20% slopes of existing stands of mixed tall fescue (*Festuca arundinacea* Schreb.) / dallisgrass (*Paspalum dilatatum* Poir.), with stands modified to represent a range of ground cover levels by establishing 100% bare soil area with 0% ground cover (bare ground), 55% bare soil area with 45% ground cover (low cover), 30% bare soil area with 70% ground cover (medium cover), and not altering full vegetative cover plots (high cover). The bare ground treatment was also compacted to simulate a cattle heavy use area. Measured forage canopy cover on vegetated plots was generally higher than the level of ground cover established at the soil surface. At 45% ground cover, canopy cover was 63 to 78%, at 70% ground cover canopy cover was 76 to 83%, and at full ground cover canopy cover was 83 to 98%. Rainfall simulators were used to evaluate total nitrogen (TN), total Kjeldahl N (TKN), nitrate-N (NO_3-N), and ammonium-N (NH_4-N) export from plots with applied cattle feces and urine representing a stocking rate of ~1.4 cows ha^{-1} yr^{-1}. Rainfall simulation events were conducted in April 2003 to evaluate baseline runoff conditions. Feces and urine were applied in May and Sept 2003 and immediately followed with rainfall simulations. Additional rainfall

simulations without additional application of feces and urine occurred in June and Oct 2003. Mean NO_3-N export was greatest from bare ground and few differences were observed at low, medium, and high cover levels. Mean NO_3-N export was also greatest during Oct, while there were no differences between May, June, and Sept rain events. Mean NH_4-N export was elevated (~1.37 kg N ha^{-1}) in months when feces and urine were applied and negligible (< 0.05 kg N ha^{-1}) in all other months. Mean export of both TKN and TN was greatest from bare ground plots and did not differ at low, medium, and high cover at each rainfall event except June, where mean TKN and TN export from bare ground did not differ from high cover. Results indicate that livestock heavy use areas in riparian zones may contribute substantial export of N and that cover and time of rainfall following grazing are important determinants of the impact of riparian grazing to N export to surface waters. With good pasture cover, infrequent periodic grazing may have little impact on nitrogen loss.

INTRODUCTION

Nitrogen from agricultural nonpoint pollution can contribute to eutrophication of surface waters (Carpenter et al., 1998). Eutrophication can lead to algal blooms, reduced dissolved oxygen levels, fish kills, reduction in biodiversity, and other potential negative impacts which may reduce the suitability of surface waters for beneficial ecosystem functions and use by humans.

Grazing cattle (*Bos* spp.) can be a source of nitrogen export to surface waters in the eastern USA (White et al., 1980). However, the impact of grazing livestock on nitrogen export to North Carolina surface waters has not been well quantified. In 1997, it

was estimated that over 710 thousand hectares or ~16% of agricultural land area was used for grazing in North Carolina (USDA, 1997), which makes the impact of grazing to water quality critically important. Specifically, the importance of riparian area management to quality of receiving surface waters underscores the need for thorough scientific investigation of environmental impacts when riparian areas are grazed.

Areas in and near the riparian zone are often grazed because they are typically unsuitable for row crop production due to topography and seasonal flooding, but can have relatively high forage productivity because of favorable moisture conditions during drier periods of the year. However, poor grazing management can lead to variable stand density and forage ground cover (Alderfer and Robinson, 1947), which can negatively influence infiltration, runoff, erosion, and sediment deposition (McGinty et al., 1979; Self-Davis et al., 2003) and limit the ecosystem services provided by the riparian area. However, while there are data relating to runoff volume, sediment, and nutrient export from grasslands in the Southeast, there are few data that report the level of ground cover necessary in order to protect water quality when livestock graze in or near riparian areas.

Controlling runoff volume as well as associated sediment export is important to reducing the export of nitrogen from pastures. Several studies in diverse environments have suggested threshold levels of 70% to 75% ground cover, below which significantly greater runoff and sediment loss can occur (Costin, 1980; Lang, 1979; Mwendera and Saleem, 1997). Lang (1979) suggested that when ground cover, as estimated by the point or area quadrat method, dropped below 75% bare areas began to connect with each other which allowed for faster flow of runoff and less time for infiltration. Similarly, Costin (1980) reported that ground cover values less than 70%, also estimated using the quadrat

method, resulted in significantly greater mean soil loss and runoff rate than at ground cover levels greater than 70%.

Slightly different results were reported by Dadkah and Gifford (1980) in a Utah study of infiltration rate and sediment production. Cover was established by uprooting vegetation in randomly selected squares of a grid. When rainfall was applied at a rate of 76 mm h^{-1}, the researchers reported no difference in infiltration rate on 0.5 m by 0.75 m plots with either 50% or 80% ground cover. Mean sediment yield for plots at 30% cover and no trampling was 1356 kg ha^{-1}, while plots at 80% cover yielded significantly less sediment at 665 kg ha^{-1}, and plots with 50% cover yielded 787 kg ha^{-1}. The researchers suggested that ground cover levels of 50% or greater may be sufficient for adequate watershed protection.

In Kentucky, Edwards et al. (2000) examined relationships between height of tall fescue and nutrient export from feces and urine applied to 2.4 m by 6.1 m plots. Over a 7 day period, feces was applied at a rate of 0.5 kg $plot^{-1}$ day^{-1} and urine was applied at a rate of 330 mL $plot^{-1}$ day^{-1}, followed by a rest period of 21 days. Plots with forages clipped to 20 cm or left unclipped were associated with the greatest flow-weighted NO_3-N concentrations in runoff; unclipped plots averaged 1.3 mg L^{-1} of NO_3-N after feces and urine application, plots clipped to 20 cm averaged 1.0 mg L^{-1}, plots at 10 cm averaged 0.7 mg L^{-1}, and plots at 2.5 cm averaged 0.6 mg L^{-1}. The authors suggested that the slower growth of the forages managed at greater heights resulted in less NO_3-N uptake, leaving more NO_3-N available for transport in runoff. Generally, the addition of feces and urine did not affect NO_3-N concentrations from plots at the 2.5-cm and 10-cm clipping heights. While results were similar for TKN concentrations, NH_3-N concentrations were not

affected by forage management treatments. The authors suggested this was a product of moisture conditions favorable to nitrification of NH_3-N.

In Arkansas, Sauer et al. (1999) compared nutrient runoff from grazing animal depositions to that from application of poultry litter. Dairy feces and urine generally produced lower levels of nutrients in runoff than did poultry litter or a combination of poultry litter and dairy manure. This was likely due to a lower rate of application of dairy manure as the researchers attempted to simulate stocking densities of cattle that would be likely in northwest Arkansas pastures. Upon application of simulated rainfall at a rate of 75 mm h^{-1}, 1 day after manure applications, 5.0% of TN and 29.5% of NH_4-N in applied poultry litter was transported in runoff as compared to 3.9% of TN and 5.0% of NH_4-N from dairy feces and urine. When rainfall was applied two weeks afterwards, nutrient exports were much lower. Less than 1% of applied TN, NO_3-N, and NH_4-N was transported in runoff for all treatments.

In South Carolina, McLeod and Hegg (1984) examined runoff water quality from tall fescue pasture with dairy manure, poultry litter, municipal sludge, or ammonium-nitrate (NH_4NO_3) applied at a target rate of 112 kg N ha^{-1} and simulated rainfall applied at a rate of 37 mm h^{-1}. Concentration of TKN in total runoff was initially as high as 40 mg L^{-1} from pasture treated with poultry manure or NH_4NO_3, but was reduced below 10 mg L^{-1} when runoff was initiated 7 days after fertilizer application. Initial TKN concentration in runoff from dairy manure was between 15 and 20 mg L^{-1}, but was reduced to < 10 mg L^{-1} at 7 days. Comparatively, concentration of NO_3-N in runoff was not as elevated initially as were other nutrients with the exception of runoff from applied NH_4NO_3. While NO_3-N concentration in runoff from NH_4NO_3 was initially 15 to 20 mg

L^{-1}, at no time was NO_3-N concentration in runoff from dairy manure or poultry litter higher than 5 mg L^{-1}.

Researchers in France studied the fate of ^{15}N-labeled cattle urine and found that averaged over all soils, mean spring recovery of urinary-N in leachate was 0.7%, summer recovery was 7.7%, while fall recovery was relatively large at 17%. For all soils and seasons, mean urinary-N recovered in soil organic matter ranged from 25% to 31% of initial applied N. In New Zealand, a laboratory lysimeter study with ^{15}N-labeled synthetic urine reported N balance information for urine applied to a ryegrass-white clover pasture (Fraser et al., 1994). Forage plant N-uptake accounted for 43% of applied N, the soil-N pool accounted for 20%, leachate-N (primarily as NO_3^-) accounted for 8%, and 28% was lost in gaseous-N form due to denitrification. Ammonia volatilization was assumed to be low because simulated rainfall was applied directly after urine application.

A greater proportion of urinary-N was recovered in leachate in a field study in central Pennsylvania. Stout et al. (1997) reported that 25% of urinary-N and 2% of fecal-N were recovered as NO_3-N in leachate. In addition, Stout et al. (1998) reported a seasonal effect of urinary-N leaching, with greater NO_3-N leaching in the fall compared to spring and summer, likely due to limited plant growth after application of the urine. Urine was applied at an average rate of 99.4 g N m^{-2} in the spring, summer, and fall, whereas feces were applied at a rate of 26.2 g N m^{-2} in the summer only. Under applied feces, NO_3-N losses were not significantly different from that of control lysimeters with no applied feces or urine. The researchers suggested that this was due to the lower amount of N in feces compared to urine as well as a low mineralization rate of organic-N in the feces.

This paper is the second of three companion papers from a study that evaluated the impact of ground cover in manured riparian pasture plots to the export of sediment and nutrients. This paper examines the impact of bare ground and low, medium, and high ground cover levels to nitrate-nitrogen (NO_3-N), ammonium-nitrogen (NH_4-N), total Kjeldahl nitrogen (TKN), and total nitrogen (TN) export on two sites of 10% and 20% slope. On 10% slope, NH_4-N and NO_3-N leaching were also examined. The first paper examines runoff volume, total suspended sediment, and phosphorus export from the same research plots (Butler et al., 2004a). A third paper examines TSS, DRP, TKP, NO_3-N, NH_4-N, TKN, and TN in runoff from wetland plots of bare ground or full cover adjacent to the plots evaluated in the first two papers (Butler et al., 2004b).

MATERIALS AND METHODS

In February 2003, research plots were established on existing mixed tall fescue (*Festuca arundinacea* Schreb.) / dallisgrass (*Paspalum dilatatum* Poir.) pasture which had been established for over 20 years at North Carolina State University's Lake Wheeler Road Field Laboratory, Raleigh, NC (35°43'N; 78°41'W; elevation = 100 m). Randomized complete block experiments were established on two slopes (~10% and ~20%), 5 to 10 m landward from a stream. Soils were Appling sandy loam (fine, kaolinitic, thermic Typic Kanhapludults) on the 10% slope site and Wedowee sandy loam (fine, kaolinitic, thermic Typic Kanhapludults) on the 20% slope site. Upon initiation of this study, ground cover treatments were established at 95%, 70%, 45%, and 0%, with four replications of each treatment, for a total of sixteen plots on each site. The 0% cover treatment was established in each replication to simulate a compacted, cattle heavy use

area. Soil cores were examined prior to plot establishment to ensure consistency of soils within replication and within slope.

Experimental plots were 0.75 m by 2.0 m and delineated with flashing 23 cm wide, placed into the soil to a depth of 18 cm to isolate the surface hydrology of the plots. A runoff collection gutter was placed at the downslope edge of each plot.

Ground cover levels were created by establishing 100% bare soil area with 0% ground cover, 55% bare soil area with 45% ground cover, 30% bare soil area with 70% ground cover, and not altering full vegetative cover plots. Bare soil areas were created using a grid of 5 cm by 10 cm rectangles and a small hand torch to quickly heat and kill statistically random blocks of plants. The random blocks created a pattern of varying sizes of bare soil areas as selected blocks connected with each other. After using the torch, any herbage residue was then raked from the plots. The plots were given time to recover and any remaining residue to be removed during subsequent natural rainfall events. Small rectangles of black plastic were placed on the plots in the same random grid pattern to maintain desired ground cover between rain events once the treatments of feces and urine were applied to the plots. All plastic was removed before simulated rain events.

Simulated lounging treatments were established by using black plastic to cover the whole plot and solarize all vegetation. A steel compaction device with an impact surface area of ~100 cm^2 was used to simulate cattle hoof compaction over the entire plot, using methods described by Clary (1995). Plots were compacted before the first simulated rain event in the spring and again before the first fall rain event. There was no attempt to simulate compaction from cattle hoof action in vegetated plots.

To standardize canopy height at each rain event, plots were harvested to a 10-cm stubble height prior to each rainfall simulation. Plots were also harvested in July 2003 for plot maintenance as well as in November 2003 to obtain a seasonal estimate of forage production. Forage samples were weighed and dried, then analyzed by the North Carolina Dept. of Agriculture and Consumer Services (NCDA&CS) Agronomic Division for total nitrogen (TN) by combustion as described in AOAC method 990.03 (Cunniff, 1995). Percent canopy cover was determined using the line transect method (Laflen et al., 1981) with 40 points after the vegetation was harvested and just prior to the rainfall simulations.

Before each simulated rainfall, three soil cores of 0- to 5-cm depth (1.75-cm inside diameter) were obtained from each plot. The cores from individual plots were combined to form a composite sample, which was divided into two subsamples. One sample was air-dried and ground, while the second was placed in a soil tin and dried at 105ºC to determine gravimetric soil moisture content. The air-dried and ground sample was extracted by shaking 10-g soil samples in 25 mL of 1M potassium chloride (KCl) solution. The resulting filtrate was analyzed by the salicylate-hypochlorite method for NH_4-N (Crooke and Simpson, 1971) and by the Griess-Ilosvay method (Keeney and Nelson, 1982) for NO_3-N and nitrite-N (NO_2-N). Concentrations of NH_4-N, NO_3-N, and NO_2-N were then added to obtain a measure of soil inorganic N.

Three rainfall simulators (Tlaloc 3000, Joern's Inc., West Lafayette, IN) were used to simulate rainfall at an intensity of 70 mm h^{-1} for a 1-h duration. This is just above a 10-yr, 1-h rain event of ~65 mm h^{-1} for Raleigh, NC, whereas a 25-yr, 1-h rain event for Raleigh is ~85 mm h^{-1} (National Weather Service: Eastern Region Headquarters (NOAA-

ERH), 2004). Each simulator rained on two plots simultaneously. In April 2003, an initial rainfall simulation was conducted to determine baseline runoff conditions from each plot prior to application of feces and urine. Deionized water was used as source water for the rainfall simulators. Great care was taken to ensure equal volume and distribution of rainfall from each simulator each time used. Simulators were calibrated by measuring volume and distribution of rain and adjusting the pressure valve at each simulator accordingly before moving simulators onto plots. Rain gauges were placed in plots to verify rainfall rate for each rain event.

Time was recorded as runoff began to drip from the gutter and the first 125 mL of runoff was collected. When steady flow began, time was again recorded and a timer started so that runoff could be sampled beginning at 5, 10, 15, 20, 25, and 30 min after initiation of steady runoff flow. At each sampling time interval, 500 mL of runoff was obtained. Runoff between samplings was collected into large bins and weighed every 5 min after the initiation of steady runoff state until the end of the 1-h simulated rain event. This runoff was sampled 30 min after initiation of runoff and the bin emptied, then sampled at the end of the rain event. The same methods of rainfall application and runoff collection were used for all rainfall simulations.

On May 20, 21, 27, or 28, 2003, feces and urine were applied to plots immediately before rainfall simulations at a rate that approximated 10% of the average daily output for mature cattle. This equated to a 2.4-kg deposit of feces (85% moisture) applied on a 550-cm^2 area and a 1-L urine deposit applied over an equal area. The center of the fecal deposit was placed 30.5 cm from the top of the plot and centered between the plot sides. The 1-L urine deposit was placed directly down slope of the fecal deposit.

The feces and urine applied to plots were collected from four beef steers fed switchgrass (*Panicum virgatum* L.) and gamagrass (*Tripsacum dactyloides* (L.) L.) hay as part of an unrelated study. Feces were mixed and formed into 2.4-kg portions (0.33-kg dry matter) before being frozen until thawed for plot application. Urine was collected in buckets placed under the steers while in metabolism crates. All collected urine was mixed, adjusted to a pH level between 5 and 6 with hydrochloric acid to prevent precipitation of solids and nitrogen losses to ammonia (NH_3), and then frozen until thawed for plot application. Feed grade urea was used to adjust to 1% N content directly before application to plots. Fecal samples were analyzed by the NCDA&CS Agronomic Division for TN by the combustion method as described in AOAC method 990.03 (Cunniff, 1995). A sample of feces that remained on each plot until the Sept rain event was similarly analyzed for TN. Urine-N was determined by Kjeldahl digestion by methods described in US-EPA (United States Environmental Protection Agency) method 351.2 (US-EPA, 1993). A measure of water soluble N was determined using methods described by Kleinman et al. (2002) for measures of water extractable phosphorus. Fresh manure samples (1-g dry weight equivalent) were shaken with 200 mL of deionized water for 1 h, centrifuged, filtered through 0.45-µm cellulose nitrate membranes, and analyzed for NH_4-N, NO_3-N, and NO_2-N by methods similar to determination of soil inorganic N. The resulting values were summed to obtain a measure of soluble N, which would be most available for immediate transport in runoff during rain events.

Following rainfall simulations in May 2003, plots were covered with translucent plastic attached to a hoop frame ~1 m above the plot surface to prevent precipitation from

reaching the plots during natural rain events. Plots remained covered until the final forage harvest of the experiment in November 2003.

Rainfall simulations were again conducted June 9 to 11, 2003 to determine runoff constituents from experimental plots without additional application of feces and urine. Plots were also treated with feces and urine before rainfall simulations on September 9 to 11, 2003, which were followed by rainfall simulations October 6 to 8, 2003 without additional plot treatment with feces and urine. (Butler et al., 2004a)

Before fall rain events, round bottom ceramic suction cup lysimeters attached to a 90-cm length of poly-vinyl chloride piping were installed under the plots on the site of 10% slope. Ceramic suction cups were 7.0 cm long with 2.2 cm diameter and a pore size of 2.5 µm and a 1 bar, high flow air entry value. At installation, an auger was used to drill a hole from outside the plot area at a 40º angle, so as to place the suction cup 80 cm below the center of the feces and urine application area. Soil removed with the auger was mixed with deionized water to form slurry, which was poured into the hole before insertion of the lysimeter. Samples were collected by vacuum at Sept and Oct rain events immediately before and after rainfall simulations, as well as on the following day.

Runoff sample vials were placed in ice in the field until transport to the lab. Following vacuum filtration of 100 mL of runoff sample through 0.45-µm cellulose nitrate membranes, filtrate was analyzed for NH_4-N and NO_3-N, by similar methods to soil inorganic N. The NO_2-N constituent in runoff was assumed to be negligible. TKN was determined similarly following Kjeldahl digestion of an unfiltered sample according to EPA method 351.2 (US-EPA, 1993). Soil water extracts from lysimeters were analyzed for NO_3-N and NH_4-N, without filtration.

Samples collected at 5-min intervals represented point estimates of concentrations and were plotted versus cumulative runoff volume. The points were joined with straight lines and the area under the straight lines was integrated using the PROC EXPAND procedure to determine cumulative mass of nitrogen lost at each collection time (SAS Institute Inc., 1994). For an estimate of total export for the 1-h rain event, cumulative mass export at 30 min was added to export estimates obtained from the post 30-min composite sample.

The effect of cover, slope, and month of rain event to NO_3-N, NH_4-N, TKN, and TN mass export and concentrations was determined using the PROC GLM procedure (SAS Institute Inc., 1994). Means were separated using Fisher's least significant difference. Unless otherwise noted, all differences were considered to be significant at $P < 0.05$.

RESULTS AND DISCUSSION

Cover

Target ground cover levels of 0%, 45%, 70%, and 95% were established at the soil surface. To facilitate discussion of ground cover treatments, 45% ground cover will be considered as low cover, 70% ground cover will be considered as medium cover, and 95% ground cover will be considered as high cover, with the 0%, compacted treatment referred to as bare ground. Line transect measurements of forage canopy cover were generally not closely matched to the established level of ground cover (Table 1). Given the data from all rain events, canopy cover at low cover treatments ranged from 63 to 78%, canopy cover at medium cover ranged from 76 to 83%, and canopy cover at high

cover ranged from 83 to 98%. Mean canopy cover differed at each level of ground cover for every rainfall event, except during Oct. In Oct, no difference in mean canopy cover was observed between ground cover levels of 45%, 70%, and 95%.

In this study, thicker forage growth in grid rectangles that were selected to include live forage may have accounted for increased canopy cover measurements at 45% and 70% ground cover levels. As the forage in these areas of the plot grid grew thicker as the season progressed, they were likely able to increase canopy cover values through some shielding of adjacent bare grid rectangles. This is important to note, as this shielding effect could also shield soil in adjacent blank grid rectangles from raindrop impact.

The forage in the plots was a mixture of tall fescue and dallisgrass, but the proportion of each was not consistent over the growing season. In Apr and May, nearly all forage was tall fescue (> 95%). As the season progressed, dallisgrass represented a greater percentage of the stand, reaching > 50% in Sept. This proportion is only an estimate, as forage species composition was not quantified at each rain event. However, such forage stands are typical of established pastures in North Carolina as warm season species provide a much greater percentage of forage during the summer months.

The amount of N removed at forage harvests can be somewhat related to cover level as denser cover is likely to produce more forage dry matter. Generally, there was not great variation in total N uptake by forages at low, medium, and high cover levels (Table 14). A greater amount of N w as removed by the forage at the Sept event compared to other months, as Sept includes total N uptake for the summer, after the June rain event.

Rainfall Rate and Distribution

At simulated rain events, a rainfall rate of 70 mm h^{-1} was the output target for each rainfall simulator. However, rainfall rate determined from rain gauges at the plots recorded a higher mean rainfall rate of 102 mm h^{-1}. There was some variation among rain event and cover (Table 15), though the numbers should be examined with caution, as rain gauges were a different design and somewhat less reliable at the Apr and May events compared to the June, Sept, and Oct events. Mean rainfall rate and rainfall rate recorded adjacent to the feces and urine deposit were included as covariates in initial statistical models to account for any variation, and were significantly related to export of some runoff constituents that will be noted.

Feces and Urine

The rate of N application in feces and urine was ~50 kg N ha^{-1}, at both the May and Sept rain events (Table 16). Of this amount, ~20 kg N ha^{-1} was applied in urine, which, because in a liquid phase, is very susceptible to runoff or leaching. The amount of inorganic N extracted by water from the feces was 0.416 kg N ha^{-1} in May and 0.494 kg N ha^{-1} in Sept.

Soil Inorganic Nitrogen

Soil inorganic N averaged ~16 mg N kg^{-1} (12 kg N ha^{-1}, 0 to 5 cm soil depth) on plots at low, medium, and high cover. However, at bare ground, mean soil inorganic N was ~ 43 mg N kg^{-1} (32 kg N ha^{-1}). This was expected, as greater biomass of forage in plots with greater cover utilized more N for growth, leaving less extractable N in the soil. Soil N was included as a covariate in analysis of runoff N constituents if found to be significant.

Nitrate-Nitrogen

Both cover ($P < 0.05$) and rain event ($P < 0.05$) were significantly related to cumulative NO_3-N mass export at 30 min after runoff initiation (Table 17). There were no interactions between cover, slope, or month of rain event. The model yielded similar results at each time of cumulative export, except for NO_3-N export 5 and 10 min after runoff initiation. At these times, there was an interaction between cover and rain event. Nevertheless, means of cumulative NO_3-N export were examined in relation to both cover and rain event, each averaged over all levels of the other variable. Amount of N removed at forage harvest, soil inorganic N, soil moisture, and rainfall rate were all included as covariates for analyses of NO_3-N export, but only amount of N removed at forage harvest improved the analysis.

At each 5-min interval following initiation of runoff up to 25 min, mean NO_3-N export was greater at bare ground than at low, medium, and high cover levels, which did not differ from each other (Fig. 8). The relationship between cover levels of low, medium, and high was slightly different at 30 min, where the highest mean NO_3-N export was also observed at bare ground, but mean export was less at medium cover than at low cover. However, mean export associated with low and medium cover did not differ from that at high cover. These data suggest that when runoff persists for 30 min or longer, mean NO_3-N export may begin to differ among 70%, 80%, and 90% canopy cover. On the other hand, 70% canopy cover may provide protection of surface waters that is equivalent to full ground cover, at least for storm events that produce runoff for shorter than a 30-min period (and is likely to produce less NO_3-N than heavy use areas).

Mean cumulative NO_3-N export varied unexpectedly over the five simulated rain events (Fig. 9). The baseline rain event in Apr had the lowest mean export at each time interval, though this relationship was not significant until 15 min after runoff initiation. Following application of feces and urine at the May event, mean NO_3-N export increased over the baseline level, beginning at 15 min, whereas mean export did not differ for May, June, and Sept rain events. Mean export did not decrease in June even though there was no additional application of feces and urine, nor did mean export increase in Sept following application of feces and urine. The Oct rain event had the highest mean export at each time interval, even though canopy cover measurements at low and medium cover levels were highest during this month (Table 13). The higher export may have been due to slower growth of forages during the fall, leaving more NO_3-N available for transport in runoff, similar to findings in leaching studies by Decau et al. (2003) and Stout et al. (1998). This finding provides contrasting results to Edwards et al. (2000) who reported the lowest flow-weighted NO_3-N concentrations during Sept and Oct rain events in a July to Oct study in Kentucky.

Concentrations of NO_3-N in runoff were generally only affected greatly by cover at initial collection of runoff from the plot (Table 18). Because of interaction with rain event and slope, this relationship was only evident at rain events that did not include application of feces and urine (data not shown). At rain events that did include feces and urine application, analyses of NO_3-N concentration in runoff were rarely significant. Compared to cover, concentrations of NO_3-N were most significant at the initial collection of runoff due to greater volume of runoff from bare ground plots as reported in Butler et al. (2004a), with additional runoff causing dilution of concentrations of runoff

constituents. Concentrations of NO_3-N from the Oct rain event were examined closely due to the large amount of nitrate exported during that rain event (Fig. 10). As with other concentration data, there was generally a very high concentration of NO_3-N in initial runoff from the bare ground treatment, but as runoff continued there were few differences in concentrations among cover levels. The relationship between NO_3-N concentration and mean cumulative runoff volume provides an estimate of mass export and captured differences from cover levels that were not apparent using only concentration data at various sampling times. While concentration of NO_3-N in runoff from bare ground may be similar to other treatments, a much greater volume of runoff was typically associated with bare ground plots, which leads to a greater export of NO_3-N from the pasture. For this reason, mass export data is the main focus of this paper.

In addition to nitrification of NH_4-N and mineralization of organic N, the build up of fecal material on the plots may be partly responsible for increased NO_3-N export later in the season in Oct (Table 19). The amount of N remaining in above-ground fecal material before fresh application at the Sept event was generally between 15 and 25 kg N ha^{-1}, compared to ~30 kg N ha^{-1} added in feces at May and Sept rain events.

Ammonium-Nitrogen

Analysis of cumulative NH_4-N export 30 min after runoff initiation indicated significance of cover ($P < 0.05$) and rain event ($P < 0.05$), but only slight significance of slope ($P < 0.1$) (Table 20). Interactions were observed between cover and rain event ($P < 0.0001$) and between slope and rain event ($P < 0.05$). Relationships among these variables were similar for all times of cumulative export. Total forage N removed was

included as a covariate in the model, whereas soil inorganic N, soil moisture, and rainfall rates were not significantly related to NH_4-N export.

Due to the significant interaction effect, NH_4-N export means were separated by level of cover for each rain event. Mean cumulative export 30 min after runoff initiation was highest at bare ground during rain events in May and Sept (Fig. 11). This is consistent with the application of feces and urine at those rain events. However, no differences in mean mass export of NH_4-N were detected between low, medium, and high cover levels at any rain event. Mass export of NH_4-N in both June and Oct did not differ between cover, as all levels of cover showed minimal levels of NH_4-N export (< 0.05 kg N ha^{-1}) during those rain events without application of feces and urine.

Means of cumulative NH_4-N export were also examined at all eight combinations of slope and cover and means were examined by rain event (Table 21). While mean NH_4-N export 30 min after runoff initiation did not differ by rain event at medium and high cover on either slope, the Sept rain event showed increased export of NH_4-N over all other rain events for bare ground and low cover treatments on both slopes. Although not significant, slightly greater export of NH_4-N was also evident during the May rain event compared to rain events that did not include feces and urine application. Greater NH_4-N export following feces and urine application is consistent with the findings Trlica et al. (2000), who reported higher NH_4-N concentrations of runoff following grazing. The pattern of lower export of NH_4-N during rain events one month after manure application was similar to the findings of Franklin et al. (2004) who reported NH_4-N losses one month after poultry litter application to be only 1.4% of losses seen immediately after litter application. Pierson et al. (2001) also reported a rapid decrease in NH_4-N

concentrations following application of poultry litter. Nitrification of NH_4-N to NO_3-N may be partly responsible for the response of NO_3-N mass export over rain events, as export of NO_3-N did not decrease in the months following application of feces and urine.

Of all 5-min sampling intervals up to 30 min after runoff initiation, concentration of NH_4-N in runoff was significantly related to cover only at 0 (initial concentration), 5, and 10 min (Table 22). As runoff continued, mean concentrations at each level of cover did not differ. While this points to the importance of nutrients evolving from initial runoff, it should also be noted that runoff volume at initiation of runoff was relatively low and thus the mass export of NH_4-N may not necessarily be greatly affected.

Total Kjeldahl Nitrogen and Total Nitrogen

The different responses of NH_4-N and NO_3-N with cover, slope, and rain event suggested that measures of TKN and TN export may provide a more complete understanding of N export. Total Kjeldahl nitrogen is a measure of organic N and NH_4-N in runoff, and, for a measure of TN, export of TKN and NO_3-N were summed. While NH_4-N and NO_3-N are most available to plants and other aquatic organisms, organic forms of N are in a less available form but are critical to developing a mass balance of N in pastured systems. For cumulative TN export at 30 min after runoff initiation, cover level ($P < 0.0001$) and rain event ($P < 0.05$) were significant, but slope was not (Table 24). In addition, interactions were observed between cover and slope ($P < 0.05$), cover and rain event ($P < 0.0001$), and rain event and slope ($P < 0.05$). A similar pattern of relationships was depicted for mean cumulative TN and TKN export (Table 23) at all time intervals. Soil inorganic N, amount of N removed at forage harvest, and rainfall rate

were significantly related to TKN and TN export, and were included in the model as covariates.

Due to the interactions, mean TN export was examined by cover at each combination of slope and rain event (Fig. 12). At 30 min after initiation of runoff, mean cumulative TN export was greater from bare ground plots than at low, medium, and high cover levels for all rain events and slopes, except during the June event. In June, TN export from bare ground plots did not differ from that at high cover for either slope. Mean TKN export responded similarly at the June event, in that no differences in mean TKN export at each cover level were indicated at either 30 min after runoff initiation or for the entire 1-h rain event. This is consistent with low runoff volume and sediment export from bare ground plots during the June rain (Butler et al., 2004a). Several studies have reported substantially reduced TKN when rainfall occurred 1 to 2 weeks after manure application (McLeod and Hegg, 1984; Sauer et al., 1999), but that may have been partly due to decreased levels of NH_4-N in runoff from subsequent rains after application. When NO_3-N is included for a measure of TN, the relationship may also differ as in this study, NO_3-N did not decrease as much as NH_4-N in the month following initial manure and urine application. When bare ground plots were removed from the analysis, neither cover, rain event, or slope were significantly related to TN export.

Concentration of TKN in runoff was shown to be significantly related to cover ($P < 0.0001$) at all runoff-sampling times (Table 25). Significance of rain event and slope, as well as interactions were not consistent across sampling times. Again, the greatest difference in concentration was observed at the initial runoff sampling, where mean concentration of TKN from bare ground plots was 81.9 mg N L^{-1}, and greater than the

concentration at all other cover levels. Initial runoff concentration averaged 10.7 mg N L^{-1}at low cover, 4.15 mg N L^{-1} at medium cover, and 5.26 mg N L^{-1} at full cover.

Soil Water Extracts

Difficulties were encountered in obtaining soil water extracts, which were collected only at Sept and Oct rain events. Insufficient soil moisture prohibited obtaining a measurable volume of sample from certain lysimeters. There was no difference in NO_3-N or NH_4-N concentration means by either cover or rain event, likely due to missing values that reduced degrees of freedom and statistical power. However, some large concentrations of NO_3-N were recorded in extracts from the Oct event (Table 26). This is consistent with the high export of mass NO_3-N reported in surface runoff during the Oct rain event. Results reported here are also consistent with work by Decau et al. (2003) and Stout et al. (1998) who reported greater leaching of NO_3-N from applied feces and urine later in the fall.

CONCLUSION

Mean cumulative export of NO_3-N at each time following initiation of runoff was higher from bare ground plots, than at low, medium, and high levels of cover. Only at 30 min were any differences seen in mean export between cover levels of low, medium, and high, perhaps due to limited range of canopy cover values. The highest export of NO_3-N was seen in Oct, which was somewhat unexpected considering no differences were observed between May, June, and Sept rain events. Beginning at 15 min after runoff initiation, all events had higher mean NO_3-N export than the baseline rain event in Apr.

This indicated that cover might not be as important for preventing NO_3-N losses from pasture when runoff persists for a short period of time.

During May and Sept, NH_4-N export was higher from bare ground plots than at all other levels of cover. However, low, medium, and high cover levels did not differ during any rain event. On bare ground and low cover plots, the Sept rain event had greater export of NH_4-N than all other rain events. During rain events when feces and urine were not applied, NH_4-N concentrations were negligible. This indicates that timing of grazing in relation to expected runoff events could be an important aspect of grazing management as NH_4-N export seems to be highly related to the time of rainfall. Riparian and other environmentally sensitive areas may be best utilized during drier seasons of the year. Ammonium-N comprised a large portion of TN, especially at those rain events that included application of feces and urine, and so is a very substantive potential impact from pastured systems.

This work confirmed that ground cover is an important determinant of NO_3-N, NH_4-N, TKN, and TN export from pastured riparian areas. Management for cover maintenance and elimination of lounging areas can minimize N export. Results from this study suggested that complete canopy cover is not necessary to reduce N export from riparian pasture, but that it may be possible to utilize forages in and near riparian areas as part of a rotational grazing system where canopy cover of 70% is maintained. While cover at lower levels may also be sufficient, such levels were not examined in this study. Future work examining levels of cover lower than those used in this study would provide useful information to those managing livestock in and near riparian areas. A range of

forage species could also be examined to determine nitrogen export from forages with differing growth habits and seasonal distribution of growth.

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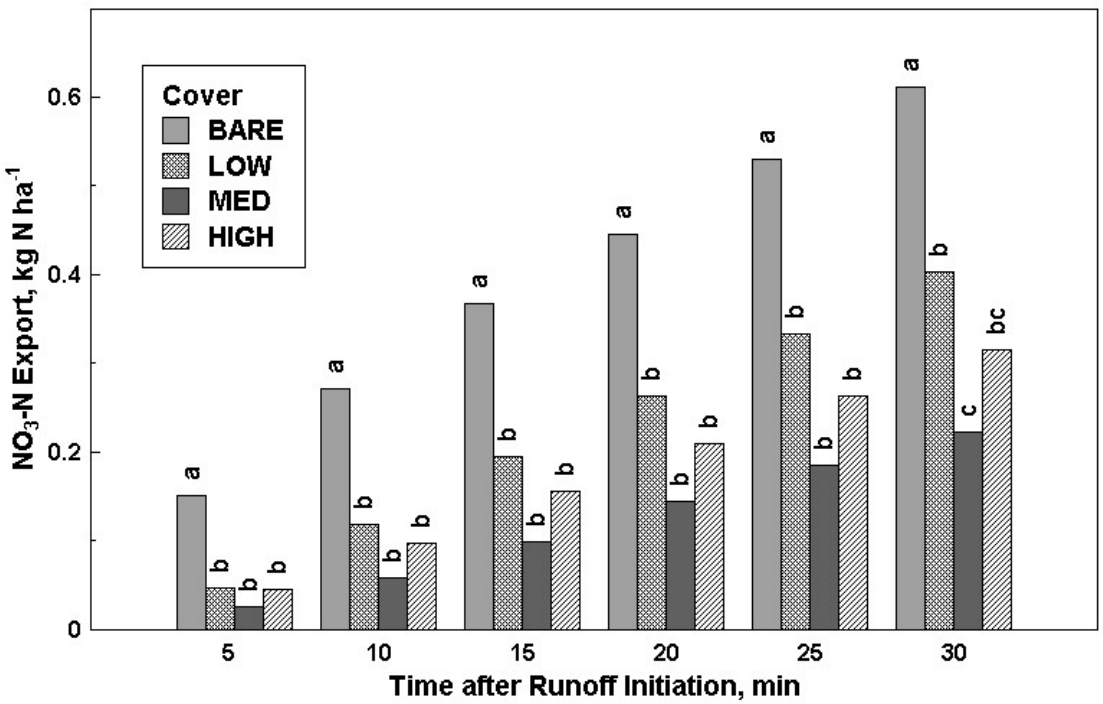

Fig. 8. Mean cumulative nitrate-nitrogen (NO_3-N) export as affected by cover (means within each time with the same letter are not significantly different, $P < 0.05$)

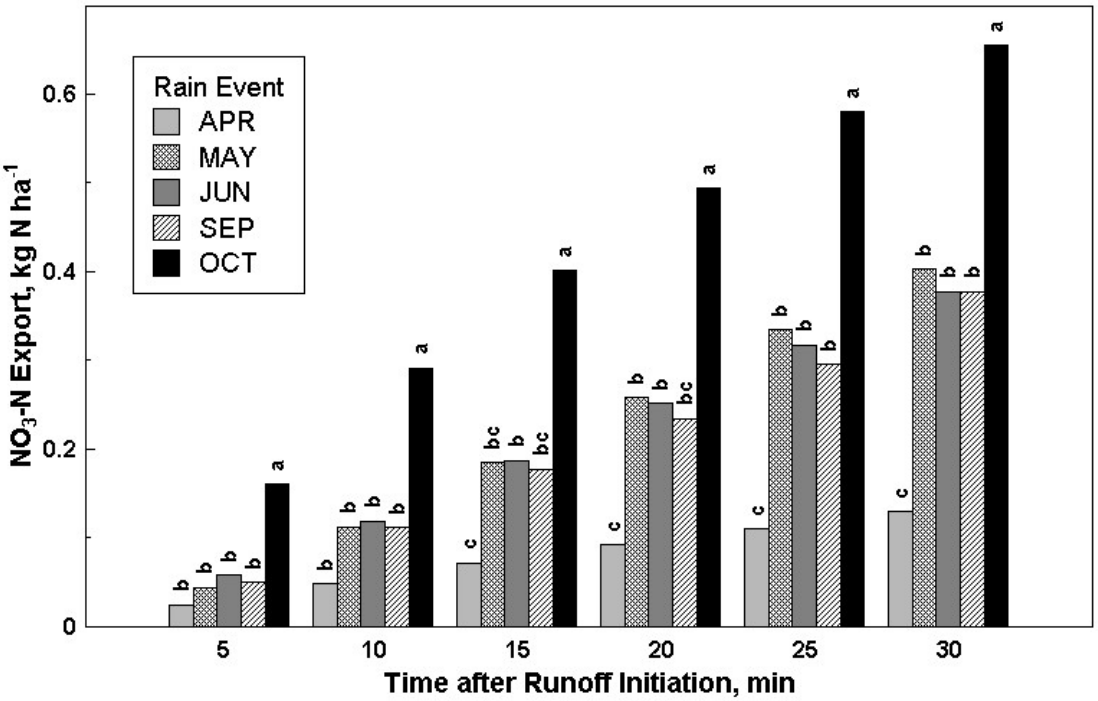

Fig. 9. Mean cumulative nitrate-nitrogen (NO_3-N) export as affected by rain event (means within the same time group with the same letter are not significantly different, $P < 0.05$)

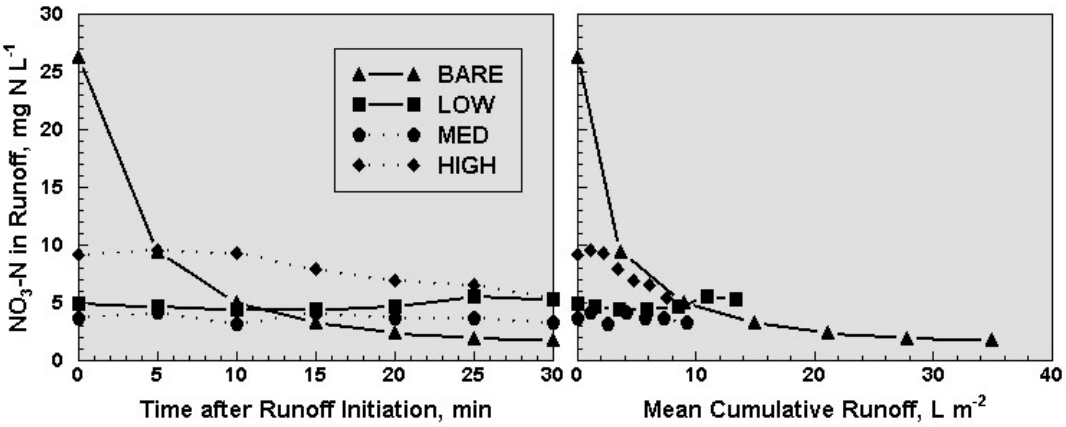

Fig. 10. Mean nitrate-nitrogen (NO_3-N) concentration in runoff at the Oct rain event as affected by cover (concentration at bare ground treatment significantly greater than concentration at all other treatments at time=0 and runoff=0)

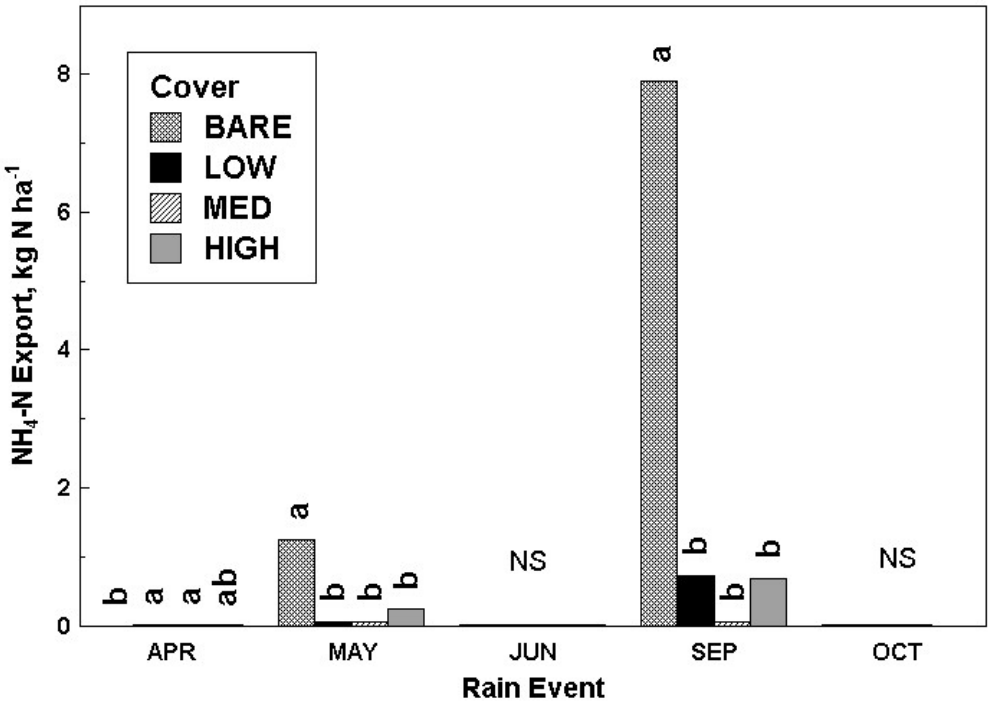

Fig. 11. Mean cumulative ammonium-N (NH_4-N) export 30 min after runoff initiation, as affected by cover at each rain event (means within the same time group with the same letter are not significantly different, $P < 0.05$)

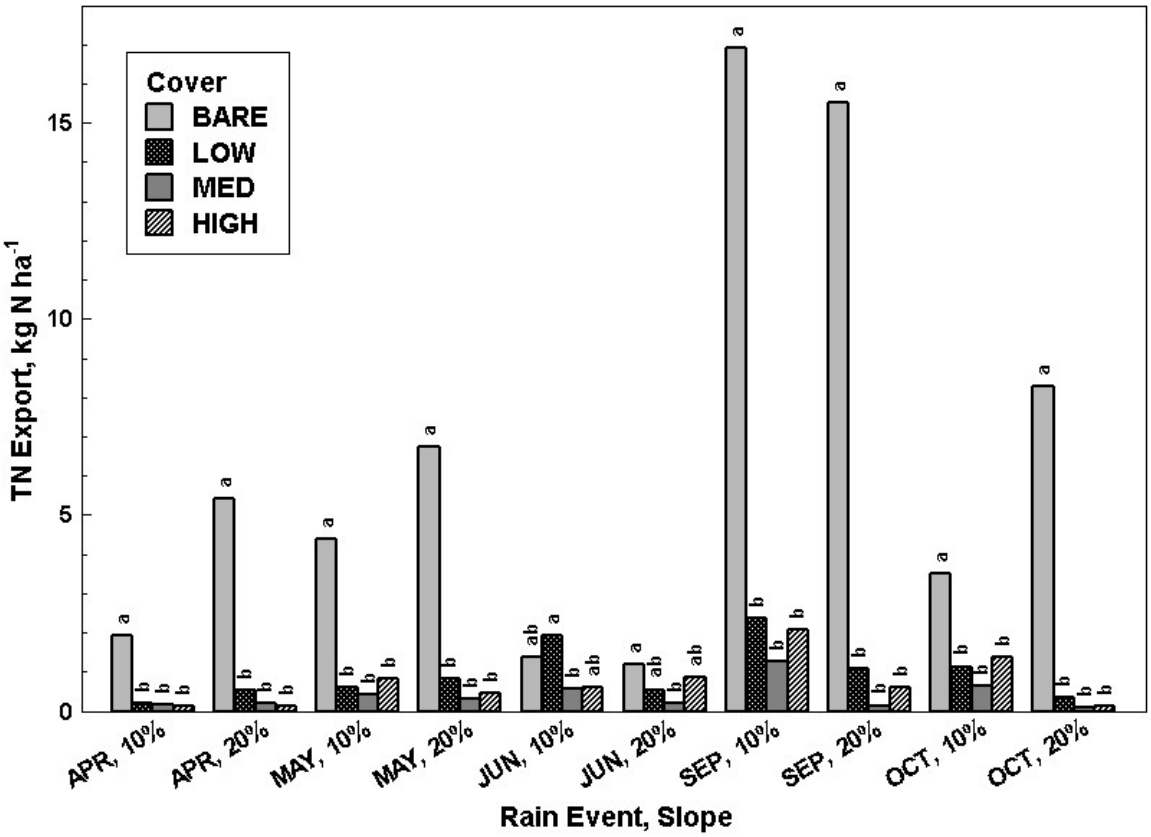

Fig. 12. Mean cumulative total nitrogen (TN) export 30 min after runoff initiation, as affected by cover at each combination of slope and rain event (means within the same grouping with the same letter are not significantly different, P < 0.05)

Table 13. Mean canopy cover as measured by line transect method at each rain event and ground cover level

Established Ground Cover	APR	MAY	JUN	SEP	OCT
	Month of Rain Event				
	Canopy Cover (%)				
0% (Bare)	0a†	0a	0a	0a	0a
45% (Low)	63b	65b	78b	75b	78b
70% (Medium)	76c	78c	83c	80c	80b
95% (High)	94d	95d	98d	92d	83b

†means within the same rain event with the same letter are not significantly different ($P < 0.05$)

Table 14. Mean forage nitrogen removed from plots at harvest, at each cover level, rain event, and slope

Cover			
BARE	LOW	MED	HIGH
Forage N removed (kg N ha^{-1})			
6.79	19.5	22.3	22.8

Month of Harvest					
APR	MAY	JUN	SEP	OCT	NOV
Forage N removed (kg N ha^{-1})					
5.80	4.24	7.28	66.8†	5.19	7.69‡

Slope	
10%	20%
Forage N removed (kg N ha^{-1})	
21.4	14.3

†includes forage harvested in July

‡not included in calculation of means by cover or slope

Table 15. Mean rainfall rate at each rain event and cover level

	Cover	APR	MAY	JUN	SEP	OCT
		Rain Event				
		Rainfall rate mm h^{-1}				
10% slope	BARE	86.0	129	96.3	104	99.9
	LOW	91.1	114	97.9	107	112
	MED	103	128	110	106	111
	HIGH	85.3	113	96.0	98.7	111
20% slope	BARE	103	86.3	95.1	97.2	108
	LOW	91.3	106	99.3	107	110
	MED	101	97.6	104	99.4	117
	HIGH	65.0	95.6	92.6	110	109

Table 16. Nitrogen (N) concentration in cattle feces and urine and application rate

Date	Manure Component	Total N	Inorg. N	Total N	Inorg. N	Total N	Inorg. N
		g N kg^{-1}†		g N $plot^{-1}$		kg N ha^{-1}	
21, 22, 27, 28 May	Feces	1.96	0.0260	4.72	0.0623	31.4	0.416
	Urine	2.92	N/A	2.92	N/A	19.5	N/A
9, 10, 11 Sept	Feces	1.63	0.0309	3.90	0.0742	26.0	0.494
	Urine	2.92	N/A	2.92	N/A	19.5	N/A

† wet basis

Table 17. ANOVA: Cumulative nitrate-nitrogen (NO_3-N) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	88	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	NS	NS	NS	NS	NS	NS	NS
Cover	3	***	**	**	*	*	*	*
Cover * slope	3	NS	NS	NS	NS	NS	NS	NS
Rain event	4	****	***	**	*	*	*	+
Cover * event	12	****	*	NS	NS	NS	NS	NS
Slope * event	4	NS	NS	NS	NS	NS	NS	NS
Cover * event * slope	12	+	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 18. ANOVA: Nitrate-nitrogen (NO_3-N) concentration in runoff

		Time after runoff initiation, min						
		0	5	10	15	20	25	30
Source	DF	P-values†						
Model	88	****	**	****	**	*	*	****
Hypothesis tests								
Slope	1	NS	NS	NS	NS	NS	NS	NS
Cover	3	***	NS	NS	NS	NS	*	NS
Cover * slope	3	+	NS	NS	NS	NS	NS	NS
Rain event	4	****	***	*	*	*	*	+
Cover * event	12	****	+	*	*	+	NS	NS
Slope * event	4	NS	NS	NS	NS	NS	NS	NS
Cover * event * slope	12	**	NS	*	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 19. Amount of nitrogen (N) in feces remaining on plots at Sept rain event

Cover	10% slope	20% slope
	kg N ha^{-1}	
BARE	22.3	20.9
LOW	14.9	21.3
MED	22.5	19.0
HIGH	25.1	19.7

Table 20. ANOVA: Cumulative ammonium-nitrogen (NH_4-N) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	88	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	NS	NS	+	+	+	+	+
Cover	3	*	*	*	*	*	*	+
Cover * slope	3	NS	NS	NS	NS	NS	NS	NS
Rain event	4	+	*	*	*	*	*	*
Cover * event	12	**	***	****	****	****	****	***
Slope * event	4	NS	+	*	*	*	*	*
Cover * event * slope	12	NS	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 21. Mean cumulative ammonium nitrogen (NH_4-N) export 30 min after runoff initiation as affected by slope, cover, and rain event

Month of	10% Slope				20% Slope			
	Cover							
Rain Event	BARE	LOW	MED	HIGH	BARE	LOW	MED	HIGH
	NH_4-N export, kg N ha^{-1}							
APR	0.00b†	0.01b	0.01	0.01	0.00b	0.02b	0.02	0.01
MAY	1.32b	0.03b	0.09	0.38	1.19b	0.08b	0.04	0.10
JUN	0.01b	0.01b	0.02	0.03	0.02b	0.01b	0.00	0.00
SEP	10.7a	1.00a	0.10	1.24	5.11a	0.45a	0.01	0.14
OCT	0.00b	0.00b	0.00	0.00	0.01b	0.00b	0.01	0.00

†Means in the same column followed by the same letter are not significantly different (P < 0.05)

Table 22. ANOVA: Ammonium-nitrogen (NH_4-N) concentration in runoff

		Time after runoff initiation, min						
		0	5	10	15	20	25	30
Source	DF	P-values†						
Model	88	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	NS	NS	NS	NS	NS	NS	NS
Cover	3	****	**	+	NS	NS	NS	NS
Cover * slope	3	NS	NS	NS	NS	NS	NS	NS
Rain event	4	+	**	*	*	+	*	+
Cover * event	12	****	****	****	***	**	NS	NS
Slope * event	4	NS	NS	NS	NS	NS	NS	NS
Cover * event * slope	12	NS	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 23. ANOVA: Cumulative total Kjeldahl nitrogen (TKN) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	90	****	****	****	****	****	****	****
Hypothesis tests								
Slope	1	NS	NS	NS	NS	NS	NS	NS
Cover	3	**	***	****	****	****	****	****
Cover * slope	3	NS	NS	+	*	*	*	**
Rain event	4	**	**	**	**	**	*	*
Cover * event	12	****	****	****	****	****	****	****
Slope * event	4	*	*	*	*	*	*	+
Cover * event * slope	12	NS	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 24. ANOVA: Cumulative total nitrogen (TN) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	**DF**	P-values†						
Model	**88**	****	****	****	****	****	****	****
Hypothesis tests								
Slope	**1**	NS	NS	NS	NS	NS	NS	NS
Cover	**3**	**	***	****	****	****	****	****
Cover * slope	**3**	NS	NS	+	+	*	*	**
Rain event	**4**	**	**	**	**	*	*	+
Cover * event	**12**	****	****	****	****	****	****	****
Slope * event	**4**	*	*	*	*	*	*	+
Cover * event * slope	**12**	NS	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, * = < 0.001, **** = < 0.0001**

Table 25. ANOVA: Total Kjeldahl nitrogen (TKN) concentration in runoff

		Time after runoff initiation, min						
		0	5	10	15	20	25	30
Source	**DF**	P-values†						
Model	**88**	****	****	****	****	****	****	****
Hypothesis tests								
Slope	**1**	NS	NS	NS	NS	NS	NS	NS
Cover	**3**	**	****	****	****	****	****	***
Cover * slope	**3**	NS	NS	**	+	*	**	+
Rain event	**4**	***	NS	*	NS	NS	NS	NS
Cover * event	**12**	****	****	****	***	****	****	**
Slope * event	**4**	NS	NS	NS	NS	NS	NS	NS
Cover * event * slope	**12**	NS	NS	NS	NS	NS	**	+

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, * = < 0.001, **** = < 0.0001**

Table 26. Concentration of ammonium-nitrogen (NH_4-N) and nitrate-nitrogen (NO_3-N) in soil water extracts 80 cm beneath application area of feces and urine

Rain		NH_4-N (mg L^{-1})			NO_3-N (mg L^{-1})		
Event	Cover	Pre†	Post	+24h	Pre	Post	+24h
SEP	BARE	0.54	1.45	4.10	3.31	3.55	3.19
	LOW	3.01	3.39	1.87	3.15	0.58	1.11
	MED	2.67	0.95	1.57	4.24	3.92	4.83
	HIGH	0.87	0.53	0.80	0.66	0.43	0.09
OCT	BARE	0.69	0.22	0.25	3.25	17.1	13.3
	LOW	2.04	3.03	1.06	1.61	5.68	7.72
	MED	0.08	0.29	0.46	10.2	8.05	10.7
	HIGH	0.50	4.64	1.60	0.12	22.0	17.3

†Pre=before rainfall simulation, Post=after rainfall simulation, +24h=day after rainfall simulation

CHAPTER 4

RUNOFF, SEDIMENT, NITROGEN, AND PHOSPHORUS EXPORT FROM MANURED RIPARIAN WETLAND VEGETATION AS AFFECTED BY SIMULATED RAIN AND GROUND COVER

D.M. Butler, N.N. Ranells, D.H. Franklin, M.H. Poore, and J.T. Green, Jr.

ABSTRACT

The impact of livestock pastures on sediment and nutrient export to surface waters in North Carolina is not well documented. The objective of this work was to determine the effect of ground cover on sediment and nutrient export from pastured riparian areas. Plots 0.75 m by 2.0 m were established on stands of existing wetland vegetation that consisted of pickerelweed (*Pontederia cordata* L.), showy goldenrod (*Solidago erecta* (Pursh) MacM.), Japanese honeysuckle (*Lonicera japonica* Thunb.), arrowleaf tearthumb (*Polygonum sagittatum* L.), and leathery rush (*Juncus coriaceus* Mackenzie), with stands modified to establish either bare ground or full cover treatments. The bare ground treatment was also compacted to simulate a cattle heavy use area. Rainfall simulators were used to evaluate total suspended sediment (TSS), dissolved reactive phosphorus (DRP), total Kjeldahl phosphorus (TKP), nitrate-nitrogen (NO_3-N), ammonium-N (NH_4-N), total Kjeldahl N (TKN), and total N (TN) export from plots with applied cattle feces and urine representing a stocking rate of ~1.4 cows ha^{-1} yr^{-1}. Rainfall simulation events were conducted in April 2003 to evaluate baseline runoff conditions. Feces and urine were applied in May and Sept 2003 and immediately followed with rainfall simulations. Additional rainfall simulations without additional application of feces and urine occurred in June and Oct 2003. Mean runoff volume was generally greater from bare ground than from full cover plots, and from the wetland plots compared to upland mixed tall fescue

(*Festuca arundinacea* Schreb.) / dallisgrass (*Paspalum dilatatum* Poir.) plots on a similar 10% slope. Plots at full cover were remarkably effective at minimizing mass export of TSS during all simulated rain events. There was a greater mean export of DRP from plots with full cover compared to bare ground plots, though mean export at both levels of cover was negligible at < 0.20 kg P ha^{-1}. Unlike DRP, mean TKP export was much greater from bare ground plots than at full cover, which is not surprising given the large amount of TSS exported from bare ground plots. Greater mean NO_3-N export was observed from the bare ground treatment than from full cover. There was also greater mean NO_3-N export one month after application of feces and urine compared to immediately after application of feces and urine. Conversely, rain events that included feces and urine application had a far greater export of NH_4-N, TN, and TKN than did rain events without application. Results indicate that livestock heavy use areas in riparian zones may export substantial sediment, N, and P, but when cover is maintained sediment and nutrient losses from cattle feces and urine may be minimal.

INTRODUCTION

Nitrogen and phosphorus from agricultural nonpoint pollution can contribute to eutrophication of surface waters (Carpenter et al., 1998). Eutrophication can lead to algal blooms, reduced dissolved oxygen levels, fish kills, reduction in biodiversity, and other potential negative impacts which may reduce the suitability of surface waters for ecosystem functioning and use by humans. Increased sediment loads have been widely reported to reduce populations of benthic organisms and fish, as well as overall primary productivity of aquatic ecosystems (Cooper, 1993).

Grazing cattle (*Bos* spp.) can be a source of nutrients and sediment to surface waters in the eastern USA (White et al., 1980). However, the impact of grazing livestock on nutrient and sediment export to North Carolina surface waters is not well quantified. In 1997, it was estimated that over 710 thousand hectares or ~16% of North Carolina agricultural land area was used for grazing (USDA, 1997), which makes the impact of grazing to water quality critically important. Specifically, the importance of riparian area management to quality of receiving surface waters underscores the need for thorough scientific investigation of environmental impacts when these riparian areas are grazed.

Areas in and near the riparian zone are often grazed because they are typically unsuitable for row crop production due to topography and seasonal flooding, but can have relatively high forage productivity because of favorable moisture conditions during drier periods of the year. However, poor grazing management can lead to variation in stand density and forage ground cover (Alderfer and Robinson, 1947), which in turn can negatively influence infiltration, runoff, erosion, and sediment deposition (McGinty et al., 1979; Self-Davis et al., 2003) and limit the ecosystem services provided by the riparian area. However, while there are data relating to runoff volume, sediment, and nutrient export from grasslands in the Southeast, there are few data that report the level of cover necessary in order to protect water quality when livestock graze in and near riparian areas.

In the North Carolina Piedmont, wet soils in close proximity to surface waters are often dominated by wetland plants such as rushes (*Juncus* spp.), and sedges (*Cyperaceae* spp.) and do not support production of typical forage species. Because of the slow growing nature of many of these wetland plants, the high level of soil moisture, and the

susceptibility of the soil surface to treading damage, these areas may be easily degraded by grazing cattle and thus be sources of high export of sediment and nutrients.

Several studies have reported the effectiveness of vegetated buffer strips at controlling non-point sources of pollution from agriculture (Bingham et al., 1980; Chaubey et al., 1995; Daniels and Gilliam, 1996). Sediment loss from grazed riparian areas presents a slightly different research problem because significant amounts of sediment can be exported from within the grazed buffer, which can be adjacent to surface waters. Generally, cattle will congregate in moist areas for available forage and water, cooler temperatures, and shade (Belsky et al., 1999). These factors combine to increase the likelihood of heavy use areas in grazed riparian zones. Several studies in diverse environments have suggested threshold levels of 70% to 75% pasture ground cover, below which significantly greater runoff and sediment loss can occur (Costin, 1980; Lang, 1979; Mwendera and Saleem, 1997).

In Montana, Hook (2003) examined riparian buffer plots to evaluate the influence of several factors to sediment retention. Mean retention of applied sediment increased with buffer width: 83% at 1-m, 94% at 2-m, and 99% at 6-m buffer widths. Increasing slope from 2 to 20% reduced sediment retention from 96 to 91%. At a 1-m buffer width ~20% and at 2-m ~10% more sediment was retained in the dense vegetation of the wetland and transitional plots than in the relatively sparse vegetation of upland plots. Wetland plots contained ~6 to 7% bare soil area, whereas upland plots had ~55 to 60% bare soil area. Generally, measurements of biomass, basal cover, and bare soil area explained sediment retention fairly well.

Several studies have examined the impact of cattle manure to nutrient export from grasslands (Edwards et al., 2000; Kleinman and Sharpley, 2003; Kuykendall et al., 1999; McLeod and Hegg, 1984; Sauer et al., 1999). However, few studies have examined the impact of wetland riparian vegetation to nutrient export from cattle feces and urine or compared the export from wetland vegetation to that observed from typical forage species in a more upland setting.

This paper is the third of three companion papers from a study that evaluated the impact of ground cover in manured riparian pasture plots to the export of sediment and nutrients. This paper examines the impact of two levels of wetland cover of riparian pasture plots on runoff volume, total suspended sediment (TSS), dissolved reactive phosphorus (DRP), total Kjeldahl phosphorus (TKP), nitrate-nitrogen (NO_3-N), ammonium-N (NH_4-N), total Kjeldahl N (TKN), and total N (TN) export in runoff following application of cattle feces and urine. The first two papers examined the impact of bare ground, and low, medium, and high levels of ground cover to runoff, DRP, TKP, NO_3-N, NH_4-N, TKN, and TN export on two sites of mixed tall fescue (*Festuca arundinacea* Schreb.) / dallisgrass (*Paspalum dilatatum* Poir.) vegetation on 10% and 20% slope (Butler et al., 2004a; Butler et al., 2004b).

MATERIALS AND METHODS

In February 2003, research plots were established on existing wetland vegetation at North Carolina State University's Lake Wheeler Road Field Laboratory, Raleigh, NC (35°43'N; 78°41'W; elevation =100 m). Vegetation consisted of pickerelweed (*Pontederia cordata* L.), showy goldenrod (*Solidago erecta* (Pursh) MacM.), Japanese

honeysuckle (*Lonicera japonica* Thunb.), arrowleaf tearthumb (*Polygonum sagittatum* L.), and leathery rush (*Juncus coriaceus* Mackenzie). A randomized complete block experiment was established on a slope of ~10%. Soils were mapped as Wehadkee sandy loam (fine-loamy, mixed, active, nonacid, thermic Fluvaquentic Endoaquepts). Ground cover treatments included full cover (~95%) and bare ground (0% cover), with four replications of each treatment, for a total of eight plots. The bare ground treatment was established to simulate a compacted cattle heavy use area. Soil cores were examined prior to plot establishment to ensure consistency of soils within replications.

Experimental plots were 0.75 m by 2.0 m and delineated with flashing 23 cm wide, placed into the soil to a depth of 18 cm to isolate the surface hydrology of the plots. A runoff collection gutter was placed at the downslope edge of each plot.

Simulated lounging treatments on bare ground plots were established by using black plastic to cover the whole plot and solarize all vegetation. A steel compaction device with an impact surface area of ~100 cm^2 was used to simulate cattle hoof compaction over the entire plot, using methods described by Clary (1995). Plots were compacted before the first simulated rain event in the spring and again before the first fall rain event. There was no attempt to simulate compaction from cattle hoof action in vegetated plots.

Plots were harvested to a 41-cm stubble height prior to each rainfall simulation beginning in May 2003, as well as in July 2003. The 41-cm height was an average of height measurements taken at the baseline runoff event in April 2003. Vegetation samples were weighed, dried, and analyzed by the North Carolina Dept. of Agriculture and Consumer Services (NCDA&CS) Agronomic Division for total N (TN) by

combustion as described in AOAC method 990.03 and total P (TP) by the photometric method described in AOAC method 965.17 (Cunniff, 1995). Percent canopy cover was determined using the line transect method (Laflen et al., 1981) with 40 points after the vegetation was harvested and just prior to the rainfall simulations.

Before each simulated rainfall, three soil cores of 0- to 5-cm depth (1.75-cm inside diameter) were obtained from each plot. The cores from individual plots were combined to form a composite sample, which was divided into two subsamples. One sample was air-dried and ground, while the second was placed in a soil tin and dried at 105ºC to determine gravimetric soil moisture content. Mehlich-3 soil test P was determined on the dried and ground sample using methods described by Mehlich (1984). A portion of the dried and ground sample was also extracted by shaking 10-g soil samples in 25 mL of 1M potassium chloride (KCl) solution. The resulting filtrate was analyzed by the salicylate-hypochlorite method for NH_4-N (Crooke and Simpson, 1971) and by the Griess-Ilosvay method (Keeney and Nelson, 1982) for NO_3-N and nitrite-N (NO_2-N). Concentrations of NH_4-N, NO_3-N, and NO_2-N were then added to obtain a measure of soil inorganic N.

Three rainfall simulators (Tlaloc 3000, Joern's Inc., West Lafayette, IN) were used to simulate rainfall at an intensity of 70 mm h^{-1} for a 1-h duration. This is just above a 10-yr, 1-h rain event of ~65 mm h^{-1} for Raleigh, NC, whereas a 25-yr, 1-h rain event for Raleigh is ~85 mm h^{-1} (National Weather Service: Eastern Region Headquarters (NOAA-ERH), 2004). Each simulator rained on two plots simultaneously. In April 2003, an initial rainfall simulation was conducted to determine baseline runoff conditions from each plot prior to application of feces and urine. Deionized water was used as source

water for the rainfall simulator. Great care was taken to ensure equal volume and distribution of rainfall from each simulator each time used. Simulators were calibrated by measuring volume and distribution of rain and adjusting the pressure valve at each simulator accordingly before moving simulators onto plots. Rain gauges were placed in plots to verify rainfall rate for each rain event.

Time was recorded as runoff began to drip from the gutter and the first 125-mL of runoff was collected. When steady flow began, time was again recorded and a timer started so that runoff could be sampled beginning at 5, 10, 15, 20, 25, and 30 min after initiation of steady runoff flow. At each sampling time interval, 500-mL of runoff was obtained. Runoff between samplings was collected into large bins and weighed every 5 min after the initiation of steady runoff state until the end of the 1-h simulated rain event. This runoff was sampled at 30 min after runoff initiation and the bin emptied, then sampled again at the end of the rain event. The same methods of rainfall application and runoff collection were used for all rainfall simulations.

On May 20, 21, 27, or 28, 2003, feces and urine were applied to plots immediately before rainfall simulations at a rate that approximated 10% of the average daily output for mature cattle. This equated to a 2.4-kg deposit of feces (85% moisture) applied on a 550-cm^2 area and a 1-L urine deposit applied over an equal area. The center of the fecal deposit was placed 30.5 cm from the top of the plot and centered between the plot sides. The 1-L urine deposit was placed directly down slope of the fecal deposit.

The feces and urine applied to plots were collected from four beef steers fed switchgrass (*Panicum virgatum* L.) and gamagrass (*Tripsacum dactyloides* (L.) L.) hay as part of an unrelated study. Feces were mixed and formed into 2.4-kg portions (0.33-kg

dry matter) before being frozen until thawed for plot application. Urine was collected in buckets placed under the steers while in metabolism crates. All collected urine was mixed, adjusted to a pH level between 5 and 6 with hydrochloric acid to prevent precipitation of solids or nitrogen losses to ammonia (NH_3), and then frozen until thawed for plot application. Feed grade urea was used to adjust to 1% N content directly before application to plots.

Fecal samples were analyzed by the NCDA&CS Agronomic Division for TN by the combustion method as described in AOAC method 990.03 (Cunniff, 1995). Urine-N was determined by Kjeldahl digestion by methods described in US-EPA (United States Environmental Protection Agency) method 351.2 (US-EPA, 1993). Fecal- and urine-P were determined by inductively coupled plasma-atomic emission spectrometry (ICP-AES) as described in US-EPA (United States Environmental Protection Agency) method 200.7 (US-EPA, 1992). A sample of feces that remained on each plot until the September rain event was similarly analyzed for TN and TP. Water extractable P (WEP) was determined by methods described by Kleinman et al. (2002b). Fresh manure samples (1-g dry weight equivalent) were shaken with 200 mL of deionized water for 1 h, centrifuged, filtered through 0.45-µm cellulose nitrate membranes, and analyzed for dissolved reactive P (DRP) by the molybdate blue method (Murphy and Riley, 1962). The filtrate was also analyzed for NH_4-N, NO_3-N, and NO_2-N by the same methods as determination of soil inorganic N. The resulting values were summed to obtain a measure of soluble N, which would be most available for immediate transport in runoff during rain events.

Following rainfall simulations in May 2003, plots were covered with translucent plastic attached to a hoop frame ~1 m above the plot surface to prevent precipitation from reaching the plots during natural rain events. Plots remained covered until the final forage harvest of the experiment in November 2003.

Rainfall simulations were conducted June 9 to 11, 2003 to determine runoff constituents from experimental plots without additional application of feces and urine. Plots were again treated with feces and urine before rainfall simulations on September 9 to 11, 2003, which were followed by rainfall simulations October 6 to 8, 2003 without additional plot treatment with feces and urine. Feces and urine application in Sept was on the same location on the plot, but fecal deposit remaining from the spring was manually removed from the plot, weighed, and a 30-g sample removed for TN determination. The remaining fecal deposit was manually broken up into ~25 pieces of similar size and distributed in the 2500-cm^2 area surrounding the original fecal deposit immediately prior to the new application of feces and urine in Sept. Total feces and urine application to plots represented a stocking rate of ~1.4 cows ha^{-1} yr^{-1}.

Runoff sample vials were placed in ice in the field until transport to the lab. Vacuum filtration of 100 mL of runoff sample through 0.45-µm cellulose nitrate membranes was used to determine concentration of TSS. Filters were dried at 105ºC and weighed before and after filtration. The filtered sample was analyzed for DRP by the molybdate blue method (Murphy and Riley, 1962), NH_4-N by the salicylate-hypochlorite method (Crooke and Simpson, 1971), and NO_3-N by the Griess-Ilosvay method (Keeney and Nelson, 1982). Total Kjeldahl P was determined similarly to DRP following Kjeldahl digestion of an unfiltered sample according to US-EPA method 365.4 (US-EPA,

1979). Total Kjeldahl N was also determined by colorimetric methods after digestion according to EPA method 351.2 (US-EPA, 1993).

Samples collected at 5-min intervals represented point estimates of concentrations and were plotted versus cumulative runoff volume. The points were joined with straight lines and the area under the straight lines was integrated using the PROC EXPAND procedure to determine cumulative mass of sediment and nutrients lost at each collection time (SAS Institute Inc., 1994). For an estimate of total export for the 1-h rain event, cumulative mass export was added to export estimates obtained from the post 30-min composite sample.

The effect of cover, slope, and month of rain event to runoff volume, TSS, DRP, TKP, NO_3-N, NH_4-N, TKN, and TN mass export was determined using the PROC GLM procedure (SAS Institute Inc., 1994). Means were separated using Fisher's least significant difference. Unless otherwise noted, all differences were considered to be significant at $P < 0.05$.

RESULTS AND DISCUSSION

Cover

Line transect cover measurements of the forage canopy were recorded at each rain simulation and generally were closely matched to established level of cover. The full cover plots had canopy cover ranging from 79% in Apr to a high of 97% in June (Table 27).

The amount of N and P removed at forage harvests can be to some extent related to cover level as more dense cover is likely to produce more forage dry matter.

Generally, there was not great variation in total forage N and P uptake in this study, although a far greater amount of N and P are represented as removed at the Sept event, as this included total N and P uptake for the entire summer, after the June rain event (Table 27). Total N uptake averaged 34 kg N ha^{-1} and total P uptake averaged 4.7 kg P ha^{-1} for this summer period.

Rainfall Rate and Distribution

At simulated rain events, a rainfall rate of 70 mm h^{-1} was the output target for each rainfall simulator. However, rainfall rate determined from rain gauges at the plots recorded a higher mean rainfall of 98 mm h^{-1}. There was some variation at each rain event (Table 28), though the numbers should be examined with caution, as rain gauges were of a different design and somewhat less reliable at the April and May events compared to the June, Sept, and Oct events. Mean rainfall rate and rainfall rate adjacent to the feces and urine deposit were included as covariates in initial statistical models to account for any variation, and were significantly related to export of some runoff constituents, which will be noted.

Feces and Urine

The rate of N application in feces and urine was ~50 kg N ha^{-1}, at both the May and Sept rain events (Table 29). Of this amount, ~20 kg N ha^{-1} was applied in urine, which because in a liquid phase is very susceptible to runoff or leaching. The amount of N extracted by water from the feces, which would also be more likely to be transported in runoff was 0.416 kg N ha^{-1} in May and 0.494 kg N ha^{-1} in Sept. Relatively low agronomic rates of P, < 10 kg P ha^{-1}, were applied to plots at each application in feces and urine. Though considerable effort was made to apply an equal amount of P in May

and Sept, slightly less P was present in samples of Sept applied feces and urine than in May samples. This may be due to inherent variability within the feces and urine, considering feces and urine for both May and Sept applications were collected, mixed, and packaged for storage at the same time. The amount of WEP in manures has been reported to be highly correlated to runoff P (Kleinman et al., 2002a). The amount of WEP in feces was 2.12 kg P ha^{-1} in May and 2.43 kg P ha^{-1} in Sept.

Soil Nitrogen and Phosphorus

Mean soil inorganic N was 20.1 mg N kg^{-1} (15.1 kg N ha^{-1}, 0- to 5-cm depth) on full cover plots, while on bare ground plots mean soil inorganic N was 26.9 mg N kg^{-1} (20.2 kg N ha^{-1}). This was expected, as the greater biomass of forage in the plots of greater cover utilized more N for growth, leaving less readily available N in the soil. Mehlich-3 soil P was very similar on plots of both covers, averaging 18.9 mg P kg^{-1} (14.2 kg P ha^{-1}). Soil N and P were included as covariates in analysis of runoff N constituents if found to be significant for particular runoff constituents.

Runoff Volume

Analysis of cumulative runoff volume 30 min after runoff initiation indicated little significance of rain event ($P < 0.1$), therefore means of cumulative runoff volume were examined by cover ($P < 0.01$), averaged across each level of rain event (Table 30). Soil moisture was significantly related to runoff volume and was included as a covariate. Runoff volume was similar between bare ground and full cover plots until 15 min after runoff initiation, when bare ground plots had a greater volume of runoff (Fig. 13). Total runoff volume for the 1-h rain event was greater from bare ground than from full cover plots (64.6 L m^{-2} vs. 46.6 L m^{-2}).

When runoff volume from the wetland plots was compared with volume from upland plots on a similar ~10% slope (Butler et al., 2004b), site was not significant for any cumulative volume of runoff (data not shown). However, when soil moisture was excluded as a covariate, site was shown to be significant ($P < 0.05$). There were interactions between rain event and site, rain event and cover, and a three-way interaction between site, cover, and rain event. However, averaged over levels of rain event and cover, mean runoff volume for the wetland site was 55.4 L m^{-2} for the 1-h rain event, which was greater than the upland site with mean runoff volume of 33.0 L m^{-2} (Butler et al., 2004b).

Total Suspended Sediment

A strong interaction between rain event and cover ($P < 0.0001$) was evident in analysis of TSS export at 30 min after runoff initiation, and both main effects were significant (Table 31). Soil moisture content was significantly related to TSS export and was included in the model as a covariate. Due to the interaction, differences in mean TSS export by rain event were examined at both levels of cover. Minimal mass export of TSS, < 30 kg ha^{-1}, was produced from full cover plots, and this relationship was consistent over all rain events. However, greater export of TSS from bare ground plots was not consistent for each rain event (Fig. 14). Export of TSS was typically highest during the fall months, with the greatest export in Sept, which differed from Oct beginning at 20 min after runoff initiation. Differences in TSS export were not noted between April, May, and June rain events.

Mean TSS export for the entire 1-h rain event was similarly related among rain events, as no differences were seen from plots at full cover (data not shown). On bare

ground plots, Sept and Oct resulted in the greatest TSS export, and mean export in Apr, May, and June did not differ.

Similar to runoff volume, TSS export was compared between these wetland plots and upland plots on the same slope (Butler et al., 2004b). Again, site was not a significant explanatory variable in the model until soil moisture was eliminated as a covariate, which indicated that soil moisture differences between site was significantly related to TSS export. Greater export of TSS in the fall may be related to soil conditions during the Sept compaction, such as moisture content, that may have created a more friable soil surface that was more susceptible to soil loss as suspended sediment in runoff, similar to a cultivated crop field.

Export of TSS in this study was generally lower than that in a Utah study of sediment yield from a 30-min simulated rainfall at a rate of 76 mm h^{-1}. A sediment yield of 3,405 kg ha^{-1} was reported from bare ground and 60% trampling disturbance and a 665 kg ha^{-1} yield was reported from plots at 80% cover and no trampling disturbance (Dadkhah and Gifford, 1980).

Dissolved Reactive Phosphorus

Mean DRP export was not significantly related to cover level or rain event at any time of cumulative DRP export (Table 32). However, models did have larger F-values as time after runoff initiation increased and the model for total DRP export for the 1-h rain event was significant ($P < 0.05$). For all 1-h rain events, there was a higher mean export of DRP from plots with full cover (0.19 kg ha^{-1}) compared to bare ground plots (0.071 kg ha^{-1}). This may due to DRP sorbing to suspended sediment in runoff samples from the

bare ground plots (Daniel et al., 1994). DRP was also not significantly related to site when both the wetland and upland sites were included in the analysis (data not shown).

Total Kjeldahl Phosphorus

Unlike DRP export, cover ($P < 0.1$) was significantly related to TKP export at each 5-min interval after runoff initiation, whereas rain event was not (Table 33). Mehlich-3 soil P was significantly related to TKP export and was included as a covariate. Mean soil P was 18.9 mg P kg^{-1}, which was relatively low compared to the upland plots examined as part of this study (Butler et al., 2004b). Unlike DRP, mean TKP export was much greater at bare ground than at full cover (Fig. 15). Because TKP included sediment-bound P, this may be expected given the large amount of TSS exported from bare ground plots.

TKP export was slightly greater from the wetland site compared to the upland site on a similar slope (Butler et al., 2004b). However, when analyzed, site was not significant (data not shown).

Nitrate-Nitrogen

Statistical models of mean cumulative NO_3-N export were marginally significant ($P < 0.1$) at 15, 20, 25, and 30 min after runoff initiation (Table 35). While earlier in the runoff period neither cover nor rain event were significantly related to mean cumulative NO_3-N export, at 30 min after runoff initiation both parameters were significant ($P < 0.05$), and there was a slight interaction between the two ($P < 0.1$). A greater mean NO_3-N export was observed from the bare ground treatment (0.505 kg N ha^{-1}) than from full cover (0.228 kg N ha^{-1}). There was also variation among rain event months, with the greatest NO_3-N export observed in June and the least observed in Sept (Fig. 16). There

was a general pattern of increased NO_3-N export one month after application of feces and urine. This may be related to nitrification of manure NH_4-N and mineralization of organic N in the weeks after feces and urine application, making more NO_3-N available for transport in runoff. An examination of the other parameters of N export clarify the N balance from both treatments evolving in runoff. Soil inorganic N did not significantly improve any models of nitrate export for the wetland plots. When NO_3-N export from the wetland and the upland site on similar slope were examined (Butler et al., 2004a), site was not a significant explanatory variable in the model.

Ammonium-Nitrogen

While cover was not significantly related to export of NH_4-N, rain event was highly significant ($P < 0.0001$) (Table 36). Therefore, means of NH_4-N export were examined by rain event, averaged over both levels of cover. The lack of significance of cover on wetland plots is consistent with DRP export and the low significance observed from cover as related to NO_3-N export. Both soil moisture and rainfall rate adjacent to the manure deposit were included as covariates in the model.

Trlica et al. (2000), reported higher NH_4-N concentrations of runoff following grazing. Similarly, in this study, May and Sept events that included feces and urine application depicted greater export of NH_4-N than did rain events without feces and urine application (Fig. 17). At no time following runoff initiation did cumulative export differ between the May and Sept rain events, or between the Apr, June, and Oct rain events. This pattern of lower export of NH_4-N during rain events one month after manure application was consistent with findings on upland plots in an associated study (Butler et al., 2004a) as well as with the findings of Franklin et al. (2004) who reported NH_4-N

losses one month after poultry litter application to be 1.4% of losses seen immediately after litter application. Pierson et al. (2001) also reported a rapid decrease in NH_4-N concentrations following application of poultry litter.

Soil moisture seems to account for a great deal of the difference between the wetland site and the upland site on similar slope (Butler et al., 2004a). Only when soil moisture was excluded from the model, was site significantly related to NH_4-N export ($P < 0.01$) at 30 min after runoff initiation. Rain event ($P < 0.0001$) and the interactions between rain event and site ($P < 0.0001$) and rain event and cover ($P < 0.01$) were also significant. At events that did not include application of feces and urine, there were no differences between the two sites. However, at those events that did include application of feces and urine there was a greater mean export of NH_4-N from the wetland plots than from the upland plots. In May, mean export at 30 min after runoff initiation was 15.79 kg N ha^{-1} from wetland plots compared to 0.85 kg N ha^{-1} from upland plots. In Sept, mean export was 13.97 kg N ha^{-1} on wetland plots and just 5.97 kg N ha^{-1} on upland plots.

Total Kjeldahl Nitrogen and Total Nitrogen

The different responses of NH_4-N and NO_3-N with cover, site, and rain event suggested that TKN and TN export may provide a more complete understanding of N export. Total Kjeldahl nitrogen is a measure of organic N and NH_4-N in runoff, and, for a measure of TN, export of TKN and NO_3-N were summed. This is an especially important component on the wetland plots in this study, as cover has shown minimal effect on export of inorganic forms of N.

Export of both TKN and TN indicated a high degree of significance on rain event ($P < 0.0001$), as well as significance of cover ($P < 0.05$) at 30 min after runoff initiation

(Tables 37 & 38). The interaction between cover and rain event was not highly significant. Means were examined at each level of cover and rain event. As with other N constituents in runoff, soil moisture and rainfall rate recorded adjacent to the manure deposit were included as model covariates.

Similar to export of NH_4-N, there was greater mean export of TN at events that included feces and urine application (Fig. 18). The relationship of mean TKN export to rain event is nearly identical (data not reported), given the relatively low mean export of NO_3-N (< 1 kg N ha^{-1}) as compared to NH_4-N. Of the rain events that did not include feces and urine application, June and Oct depicted a trend of increased mean cumulative TN export over the baseline event in April, but the relationship was not significant at any 5-min interval following runoff initiation. At 5 and 10 min after runoff initiation, export at May and Sept events did not differ from each other. Beginning at 15 min, mean TN export at the Sept event was greater than at the May event. This result may be partially explained by the accumulation of nutrients from the first application of feces and urine to the second application. An average of total N remaining on plots in above-ground fecal material before additional application of feces and urine in Sept was 1.39 kg N ha^{-1} at bare ground and 8.71 kg N ha^{-1} at full cover (Table 34). Though full cover plots had a greater amount of N remaining in above-ground feces, the amount of N in the soil N pool was likely greater at the Sept event, considering that close contact with moist soil may have facilitated more rapid decomposition of fecal deposits and mineralization of organic N on bare ground plots. The relatively low export in June and Oct would seem to refute this argument, but while NO_3-N export was relatively high during these months as compared to other months, NH_4-N export was minimal in June and Oct. This causes a

relatively low value of TN export for June and Oct compared to months with high NH_4-N export.

Mean cumulative export of TN was greater on plots at bare bround than on plots with full cover, a relationship that was established shortly after runoff initiation (Fig. 19). This result was expected considering the large amount of TSS recorded in the runoff samples from 0% cover plots. Forms of organic-N attached to sediment account for the proportion of TN that is not represented as NO_3-N or NH_4-N, and organic-N was more affected by cover than either of the inorganic forms. As with P, controlling N losses from riparian wetland areas may be in large part a function of controlling TSS export. Similar to NH_4-N, soil moisture accounted for a large amount of the variability between the wetland and upland sites on similar slopes.

CONCLUSION

There was a great difference in the response of fully vegetated plots and bare, "lounging" plots to simulated rain and application of feces and urine. Mean total runoff volume was similar at both bare ground and full cover treatments until 15 min after runoff initiation, when bare ground had a greater volume of runoff for the remainder of the 1-h rain event. Mean total runoff volume for the 1-h rain event from the wetland site was greater than from an upland site on a similar slope, which was significantly correlated to soil moisture content. Full cover plots were remarkably effective at minimizing mass export of TSS at all rain events (< 30 kg ha^{-1} at 30 min after runoff initiation). However, for the 1-h rain event, there was a greater mean export of DRP from full cover plots compared to bare ground plots, though loads were generally

minimal. Unlike DRP, mean TKP export was much greater at bare ground than at full cover, which is not surprising given the large amount of TSS exported from bare ground plots.

Greater mean NO_3-N export was observed from the bare ground treatment than from full cover. There was also variation among rain event months, with the greatest NO_3-N export observed in June and the least observed in Sept. There was a general pattern of increased NO_3-N export one month after application of feces and urine. This may be related to nitrification of manure NH_4-N after feces and urine application, making more NO_3-N available for transport in runoff. May and Sept events, which included feces and urine application, depicted a greater export of NH_4-N, TN, and TKN than did rain events without feces and urine application.

Results of this study indicate that cattle heavy use or "lounging" areas in the riparian zone can be a significant source of sediment and sediment-bound nutrients. However, when cover was maintained, feces and urine application generally did not greatly impact sediment or nutrient export. The sole exception was NH_4-N export, which was elevated during rain events when feces and urine were applied regardless of ground cover, and minimal at other times. Management to limit manure deposited in the riparian zone in times when runoff is likely may prevent large export of NH_4-N to surface waters.

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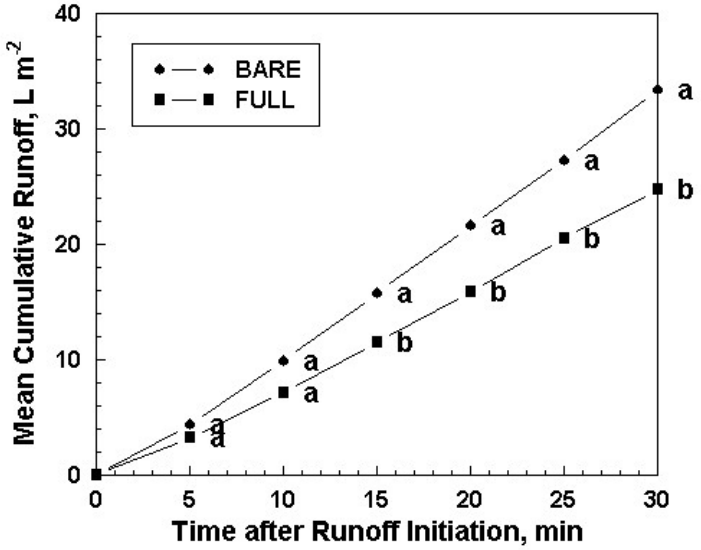

Fig. 13. Mean cumulative runoff volume as affected by cover (means at the same time with the same letter are not significantly different, P < 0.05)

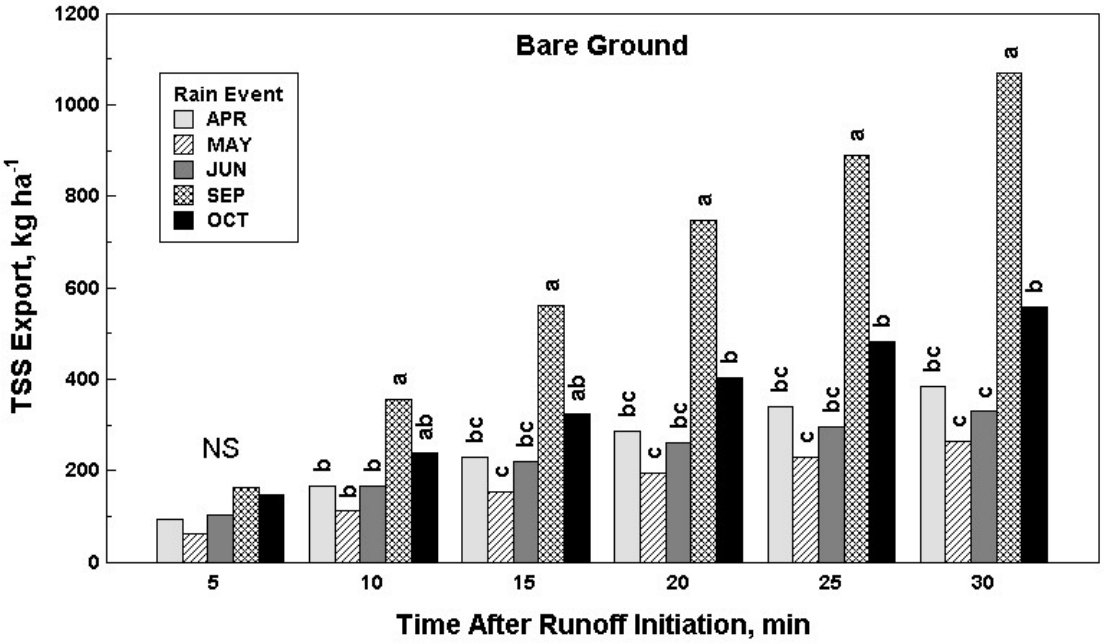

Fig. 14. Mean cumulative total suspended sediment (TSS) export at bare ground as affected by rain event (means at the same time with the same letter are not significantly different, P < 0.05)

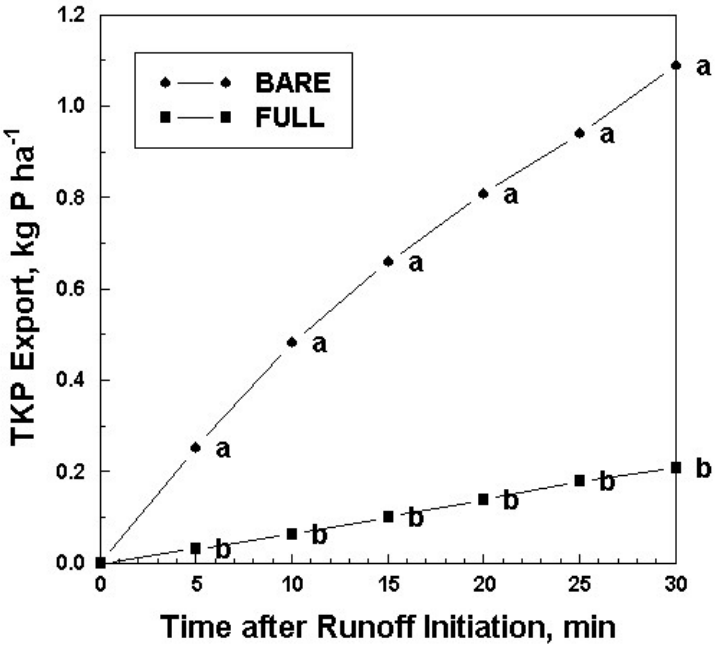

Fig. 15. Mean cumulative total Kjeldahl phosphorus (TKP) export as affected by cover (means at the same time with the same letter are not significantly different, P < 0.05)

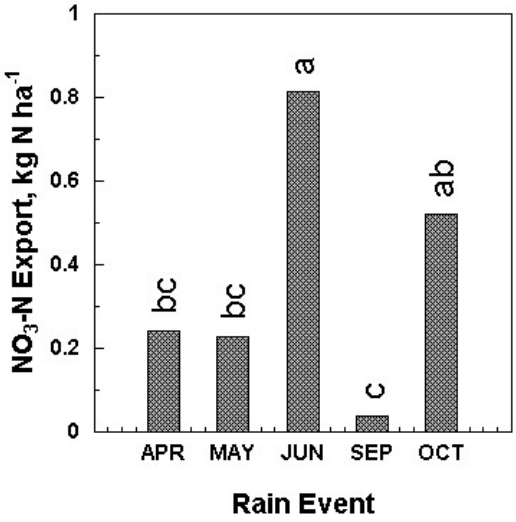

Fig. 16. Mean cumulative nitrate-nitrogen (NO_3-N) export 30 min after runoff initiation, as affected by rain event (means at the same time with the same letter are not significantly different, P < 0.05)

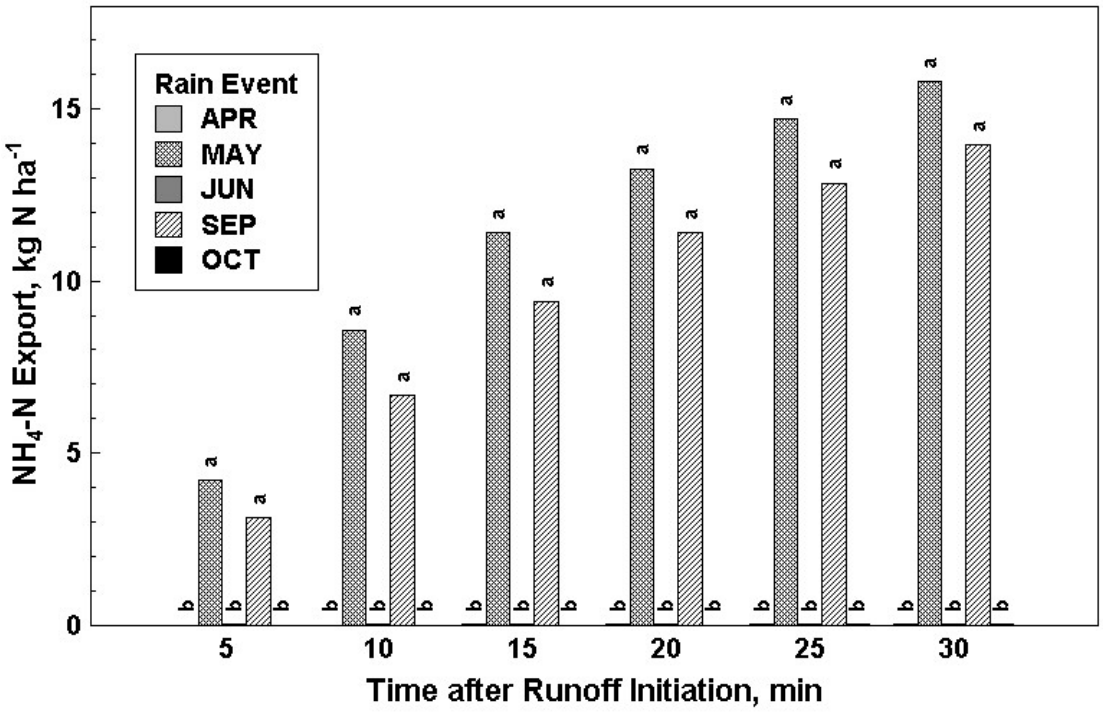

Fig. 17. Mean cumulative ammonium-nitrogen (NH_4-N) export as affected by rain event (means at the same time with the same letter are not significantly different, $P < 0.05$)

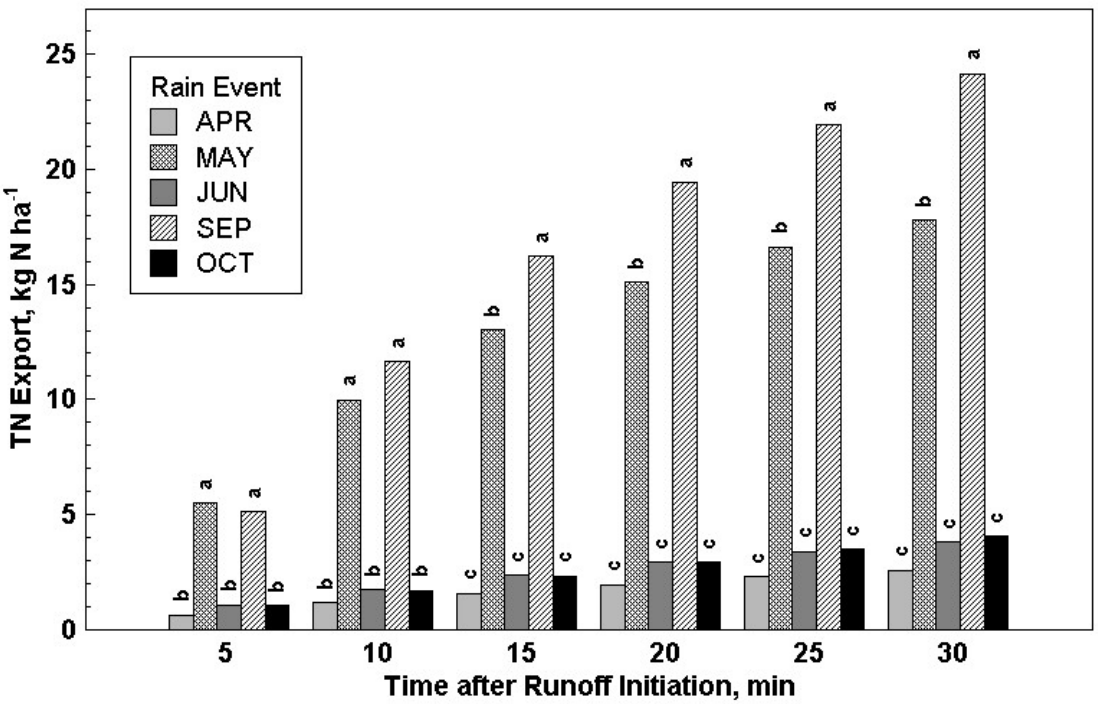

Fig. 18. Mean cumulative total nitrogen (TN) export as affected by rain event (means at the same time with the same letter are not significantly different, P < 0.05)

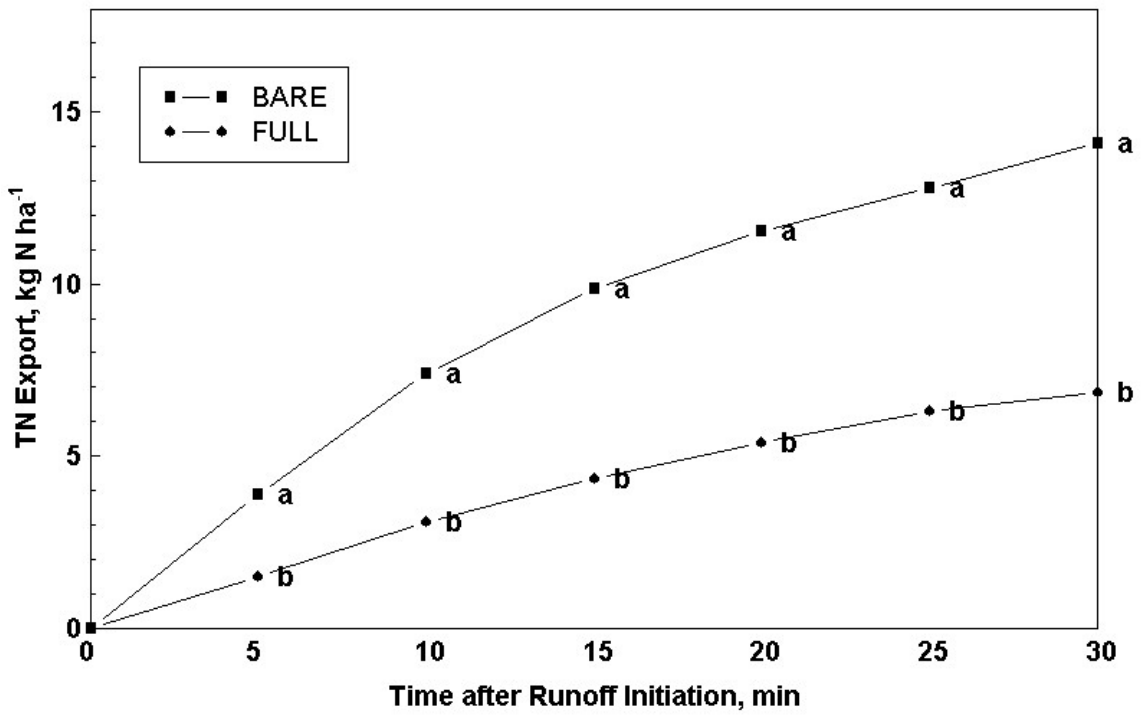

Fig. 19. Mean cumulative total nitrogen (TN) export as affected by cover (means at the same time with the same letter are not significantly different, P < 0.05)

Table 27. Line transect measure of canopy cover and nitrogen (N) and phosphorus (P) uptake by wetland plots at full cover

Event	Cover	N uptake	P uptake
		kg ha^{-1}	
APR	78.7	N/A	N/A
MAY	93.1	1.99	0.283
JUN	96.9	1.80	0.145
SEP	91.9	34.0	4.69
OCT	87.5	0.953	0.0775
NOV	N/A	N/A	N/A

Table 28. Mean rainfall rate at each simulated rain event and cover level

	Rain Event				
Cover	APR	MAY	JUN	SEP	OCT
	Rainfall rate, mm h^{-1}				
BARE	87.0	94.8	109	88.4	101
FULL	110	95.7	106	90.4	96.6

Table 29. Phosphorus (P) and nitrogen (N) concentration in cattle feces and urine and application rate

Date	Manure Component	Total P	WEP†	Total P	WEP†	Total P	WEP†
		g P kg^{-1}‡		g P plot^{-1}		kg P ha^{-1}	
21, 22, 27, 28 May	Feces	0.478	0.133	1.15	0.318	7.65	2.12
	Urine	0.165	N/A	0.165	N/A	1.10	N/A
9, 10, 11 Sept	Feces	0.349	0.152	0.835	0.364	5.57	2.43
	Urine	0.118	N/A	0.118	N/A	0.784	N/A

Date	Manure Component	Total N	Inorg. N	Total N	Inorg. N	Total N	Inorg. N
		g N kg^{-1}‡		g N plot^{-1}		kg N ha^{-1}	
21, 22, 27, 28 May	Feces	1.96	0.0260	4.72	0.0623	31.4	0.416
	Urine	2.92	N/A	2.92	N/A	19.5	N/A
9, 10, 11 Sept	Feces	1.63	0.0309	3.90	0.0742	26.0	0.494
	Urine	2.92	N/A	2.92	N/A	19.5	N/A

† water extractable phosphorus
‡ wet basis

Table 30. ANOVA: Cumulative runoff volume

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	**DF**	P-values†						
Model	**28**	NS	NS	*	**	**	**	***
Hypothesis tests								
Cover	**1**	NS	+	*	*	*	**	**
Rain event	**4**	NS	+	+	+	+	+	*
Cover * event	**4**	NS	NS	NS	+	+	+	*

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 31. ANOVA: Cumulative total suspended sediment (TSS) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	**DF**	P-values†						
Model	**28**	**	****	****	****	****	****	****
Hypothesis tests								
Cover	**1**	+	*	*	*	*	*	*
Rain event	**4**	NS	NS	*	*	**	**	**
Cover * event	**4**	NS	**	****	****	****	****	****

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 32. ANOVA: Cumulative dissolved reactive phosphorus (DRP) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	**DF**	P-values†						
Model	**28**	NS	NS	NS	NS	NS	+	*
Hypothesis tests								
Cover	**1**	NS	NS	NS	NS	NS	NS	NS
Rain event	**4**	NS	NS	NS	NS	NS	NS	NS
Cover * event	**4**	NS	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 33. ANOVA: Cumulative total Kjeldahl phosphorus (TKP) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	28	**	**	**	**	**	***	***
Hypothesis tests								
Cover	1	+	+	*	*	*	*	*
Rain event	4	NS	NS	NS	NS	NS	+	*
Cover * event	4	NS	NS	NS	NS	NS	NS	+

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 34. Amount of nitrogen (N) and phosphorus (P) in feces remaining on plots at Sept rain event

Cover	Total N ($kg\ N\ ha^{-1}$)	Total P ($kg\ P\ ha^{-1}$)
BARE	1.39	0.135
FULL	8.71	1.61

Table 35. ANOVA: Cumulative nitrate-nitrogen (NO_3-N) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	27	NS	NS	+	+	+	+	NS
Hypothesis tests								
Cover	1	NS	NS	NS	NS	+	*	NS
Rain event	4	NS	NS	NS	+	*	*	*
Cover * event	4	*	+	+	*	*	+	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 36. ANOVA: Cumulative ammonium-nitrogen (NH_4-N) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	27	NS	*	**	**	***	***	***
Hypothesis tests								
Cover	1	NS	NS	NS	NS	NS	NS	NS
Rain event	4	***	****	****	****	****	****	****
Cover * event	4	NS	NS	NS	NS	NS	NS	NS

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 37. ANOVA: Cumulative total Kjeldahl nitrogen (TKN) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	27	NS	*	**	***	**	***	****
Hypothesis tests								
Cover	1	+	*	*	*	*	*	*
Rain event	4	***	***	****	****	****	****	****
Cover * event	4	NS	NS	+	*	NS	*	*

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

Table 38. ANOVA: Cumulative total nitrogen (TN) export

		Time after runoff initiation, min						
		5	10	15	20	25	30	Total
Source	DF	P-values†						
Model	27	NS	*	**	***	**	***	****
Hypothesis tests								
Cover	1	+	*	*	**	*	*	*
Rain event	4	***	***	****	****	****	****	****
Cover * event	4	NS	NS	+	+	NS	+	*

†NS = not significant, + = < 0.1, * = < 0.05, ** = < 0.01, *** = < 0.001, **** = < 0.0001

APPENDIX

Appendix 5.1. **Mean soil moisture before application of simulated rain**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	Soil Moisture, %				
10% slope	BARE	24.9	23.8	23.1	21.5	21.9
	LOW	23.1	23.0	22.7	20.0	19.2
	MED	24.4	24.3	22.6	22.1	22.0
	HIGH	22.0	21.9	20.7	18.0	16.6
20% slope	BARE	24.1	23.4	19.9	18.0	17.6
	LOW	22.9	22.2	19.9	14.7	12.6
	MED	23.5	21.3	24.4	11.2	9.35
	HIGH	24.5	21.9	23.0	12.0	11.3
Wetland site						
10% slope	BARE	38.8	37.3	37.7	32.9	38.7
	FULL	37.9	39.1	39.1	35.3	40.5

Appendix 5.2. **Mean Mehlich-3 soil phosphorus prior to simulated rain events**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	Mehlich-3 soil P, mg kg^{-1}				
10% slope	BARE	40.5	41.7	50.5	44.1	47.7
	LOW	40.9	46.5	48.6	39.4	40.1
	MED	42.4	46.9	51.2	43.1	37.9
	HIGH	44.3	41.9	50.1	41.9	42.6
20% slope	BARE	35.5	38.3	38.7	40.5	39.4
	LOW	34.9	39.6	37.0	34.7	35.6
	MED	37.1	37.4	37.6	38.8	35.8
	HIGH	31.7	28.9	32.7	29.3	29.1
Wetland site						
10% slope	BARE	18.4	14.8	9.85	12.2	12.0
	FULL	20.5	16.0	14.7	11.1	12.4

Appendix 5.3. **Mean soil inorganic nitrogen prior to simulated rain events**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	Soil inorganic N, mg kg $^{-1}$				
10% slope	BARE	11.1	11.3	50.9	41.3	34.0
	LOW	13.6	7.01	11.5	14.1	10.8
	MED	10.5	8.04	10.8	14.0	13.9
	HIGH	8.83	4.74	8.05	12.1	13.2
20% slope	BARE	17.4	14.3	57.7	41.2	45.2
	LOW	12.4	7.30	12.1	9.68	9.02
	MED	16.5	8.34	18.37	9.97	9.76
	HIGH	17.9	6.51	26.5	12.0	11.5
Wetland site						
10% slope	BARE	23.8	19.3	23.3	15.5	19.1
	FULL	16.0	12.5	21.0	13.5	12.6

Appendix 5.4. **Mean total dry matter removed at forage harvest**

		Rain Event					
		APR	MAY	JUN	SEP	OCT	NOV
Upland sites	Cover	Forage dry matter harvested, kg ha^{-1}					
10% slope	BARE	0.00	0.00	0.00	1,620	0.00	0.00
	LOW	743	112	318	5490	140	530
	MED	1020	515	512	7060	220	357
	HIGH	1270	222	550	5230	254	645
20% slope	BARE	0.00	0.00	0.00	2,470	0.00	0.00
	LOW	466	273	122	4,170	177	199
	MED	594	242	145	4020	257	328
	HIGH	673	415	228	3760	463	447
Wetland site							
10% slope	BARE	0.00	0.00	0.00	638	0.00	0.00
	FULL	0.00	170	98.3	3040	58.5	0.00

Appendix 5.5. **Mean total phosphorus removed at forage harvest**

			Rain Event				
		APR	MAY	JUN	SEP	OCT	NOV
Upland sites	Cover		Forage TP removed, kg ha^{-1}				
10% slope	BARE	N/A	N/A	N/A	2.61	N/A	N/A
	LOW	1.08	0.270	0.853	11.6	0.300	1.01
	MED	1.45	1.16	1.45	14.4	0.443	0.663
	HIGH	1.15	0.593	1.65	11.7	0.548	1.21
20% slope	BARE	N/A	N/A	N/A	4.27	N/A	N/A
	LOW	0.430	0.713	0.273	7.52	0.353	0.335
	MED	0.685	0.633	0.358	6.89	0.438	0.553
	HIGH	0.785	0.955	0.530	6.63	0.820	0.650
Wetland site							
10% slope	BARE	N/A	N/A	N/A	1.42	N/A	N/A
	FULL	N/A	0.283	0.145	4.69	0.0775	N/A

Appendix 5.6. **Mean total nitrogen removed at forage harvest**

			Rain Event				
		APR	MAY	JUN	SEP	OCT	NOV
Upland sites	Cover		Forage TN removed, kg ha^{-1}				
10% slope	BARE	N/A	N/A	N/A	27.7	N/A	N/A
	LOW	8.28	2.31	10.4	93.9	4.28	14.5
	MED	10.9	10.2	16.2	96.4	6.67	8.97
	HIGH	8.26	5.02	16.8	102	8.25	15.1
20% slope	BARE	N/A	N/A	N/A	40.2	N/A	N/A
	LOW	4.29	5.02	3.54	58.9	4.04	4.61
	MED	7.15	5.03	4.61	60.3	5.68	8.13
	HIGH	7.54	6.38	6.69	54.4	12.6	10.3
Wetland site							
10% slope	BARE	N/A	N/A	N/A	1.42	N/A	N/A
	FULL	N/A	1.99	1.80	8.55	0.778	N/A

Appendix 5.7. **Mean time to first runoff drip from plots**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	Time to runoff drip, s				
10% slope	BARE	166	234	277	173	200
	LOW	274	641	511	430	449
	MED	253	332	586	252	455
	HIGH	264	824	588	214	290
20% slope	BARE	110	173	201	175	115
	LOW	117	210	778	244	255
	MED	854	352	137	623	175
	HIGH	281	312	380	269	258
Wetland site						
10% slope	BARE	192	188	141	187	126
	FULL	160	202	239	281	222

Appendix 5.8. **Mean time to steady flow of runoff from plots**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	Time to steady flow, s				
10% slope	BARE	264	251	549	268	223
	LOW	988	1160	1070	1110	1630
	MED	793	910	1000	1270	1540
	HIGH	978	935	1170	1630	1750
20% slope	BARE	140	194	563	197	142
	LOW	746	1100	1560	1630	1650
	MED	725	1050	748	1400	1460
	HIGH	1410	1130	1630	1330	1720
Wetland site						
10% slope	BARE	250	200	174	367	209
	FULL	219	339	278	292	503

Appendix 5.9. **Mean cumulative runoff volume 30 minutes after initiation of runoff**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	Cumulative runoff, L m^{-2}				
10% slope	BARE	23.8	23.5	15.1	26.5	24.9
	LOW	13.6	12.5	13.7	14.0	10.8
	MED	13.1	12.1	14.3	9.27	9.00
	HIGH	12.9	13.0	12.7	12.1	8.00
20% slope	BARE	26.1	23.7	12.9	21.3	21.7
	LOW	12.3	9.20	12.5	7.27	7.00
	MED	8.33	4.93	2.55	1.82	3.29
	HIGH	8.67	8.93	7.40	5.31	2.07
Wetland site						
10% slope	BARE	32.9	33.1	35.6	33.1	31.7
	FULL	29.9	26.5	25.0	20.3	21.6

Appendix 5.10. **Mean total runoff volume from the 1-h rain event**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	Total runoff, L m^{-2}				
10% slope	BARE	48.8	49.9	31.5	54.9	51.4
	LOW	26.3	22.9	25.7	19.3	15.8
	MED	25.1	23.5	24.9	13.2	13.1
	HIGH	22.2	23.3	18.2	17.4	11.9
20% slope	BARE	50.6	52.2	31.4	45.1	48.6
	LOW	25.7	16.7	23.2	11.9	11.3
	MED	13.3	8.67	5.79	1.89	6.05
	HIGH	18.7	15.1	13.7	8.53	2.63
Wetland site						
10% slope	BARE	63.7	66.1	70.0	59.7	61.3
	FULL	58.5	51.3	46.9	37.9	38.3

Appendix 5.11. **Mean cumulative total suspended sediment (TSS) export at 30 minutes after runoff initiation**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover		TSS, kg ha^{-1}			
10% slope	BARE	266	189	55.8	344	219
	LOW	12.5	14.3	16.8	6.16	2.47
	MED	7.62	6.73	7.88	5.55	1.60
	HIGH	7.75	2.84	4.82	6.46	1.51
20% slope	BARE	588	515	134	856	718
	LOW	57.3	32.7	49.2	7.30	3.30
	MED	20.2	9.59	2.15	0.700	3.00
	HIGH	13.5	15.6	11.5	2.30	0.700
Wetland site						
10% slope	BARE	384	265	329	1070	558
	FULL	29.5	26.6	15.7	21.3	29.7

Appendix 5.12. **Mean total suspended sediment (TSS) export for the 1-h rain event**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover		TSS, kg ha^{-1}			
10% slope	BARE	451	393	119	617	406
	LOW	43.7	23.0	28.2	8.70	2.57
	MED	11.3	11.2	12.5	8.20	2.70
	HIGH	10.1	5.67	6.17	7.70	2.05
20% slope	BARE	1060	1000	323	1480	1670
	LOW	225	49.4	76.5	10.6	5.30
	MED	49.6	15.4	5.25	0.700	5.30
	HIGH	36.2	26.2	18.3	3.30	0.700
Wetland site						
10% slope	BARE	594	439	499	1820	959
	FULL	44.0	38.1	22.5	33.1	36.9

Appendix 5.13. **Mean cumulative dissolved reactive phosphorus (DRP) at 30 min after runoff initiation**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	DRP export, kg P ha^{-1}				
10% slope	BARE	0.0858	0.124	0.0752	0.254	0.139
	LOW	0.0254	0.0837	0.150	0.257	0.0649
	MED	0.0282	0.0791	0.140	0.184	0.0649
	HIGH	0.0256	0.0747	0.166	0.332	0.0584
20% slope	BARE	0.0532	0.0797	0.0533	0.106	0.110
	LOW	0.0128	0.0522	0.0431	0.0440	0.0123
	MED	0.00854	0.0300	0.0114	0.0154	0.00738
	HIGH	0.00542	0.0520	0.0342	0.0550	0.00499
Wetland site						
10% slope	BARE	0.0191	0.0428	0.0635	0.0336	0.0387
	FULL	0.0103	0.0989	0.101	0.190	0.0913

Appendix 5.14. **Mean total dissolved reactive phosphorus (DRP) export for the 1-h rain event**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	DRP export, kg P ha^{-1}				
10% slope	BARE	0.132	0.198	0.159	0.562	0.258
	LOW	0.0395	0.139	0.248	0.398	0.101
	MED	0.0436	0.133	0.213	0.269	0.0868
	HIGH	0.0368	0.128	0.222	0.455	0.0877
20% slope	BARE	0.101	0.162	0.125	0.276	0.241
	LOW	0.0318	0.0876	0.0604	0.0640	0.0162
	MED	0.0190	0.0493	0.0212	0.0154	0.0162
	HIGH	0.00999	0.0797	0.0573	0.0737	0.00533
Wetland site						
10% slope	BARE	0.0294	0.0961	0.128	0.0346	0.0689
	FULL	0.0143	0.239	0.166	0.376	0.154

Appendix 5.15. **Mean cumulative total Kjeldahl phosphorus (TKP) at 30 min after runoff initiation**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	TKP export, kg P ha^{-1}				
10% slope	BARE	0.451	0.576	0.187	0.792	0.558
	LOW	0.0532	0.122	0.312	0.263	0.0683
	MED	0.0455	0.0955	0.172	0.191	0.0658
	HIGH	0.0397	0.0915	0.126	0.322	0.0578
20% slope	BARE	0.952	0.903	0.132	1.24	1.03
	LOW	0.105	0.121	0.120	0.0555	0.0146
	MED	0.0406	0.0546	0.0324	0.0156	0.0105
	HIGH	0.0282	0.0937	0.168	0.0615	0.00525
Wetland site						
10% slope	BARE	1.06	0.903	0.911	1.48	1.09
	FULL	0.0829	0.272	0.163	0.371	0.148

Appendix 5.16. **Mean total Kjeldahl phosphorus (TKP) export for the 1-h rain event**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	TKP export, kg P ha^{-1}				
10% slope	BARE	0.896	1.06	0.420	1.55	1.01
	LOW	0.0940	0.201	0.403	0.403	0.102
	MED	0.0747	0.165	0.270	0.281	0.0892
	HIGH	0.0547	0.156	0.179	0.451	0.0861
20% slope	BARE	1.71	1.59	0.290	2.28	2.09
	LOW	0.356	0.192	0.242	0.0824	0.0200
	MED	0.101	0.0984	0.0780	0.0156	0.0230
	HIGH	0.0651	0.142	0.236	0.0809	0.00561
Wetland site						
10% slope	BARE	1.67	1.57	1.48	2.73	1.95
	FULL	0.130	0.496	0.258	0.603	0.221

Appendix 5.17. **Mean cumulative ammonium-nitrogen (NH_4-N) export at 30 min after runoff initiation**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover		NH_4-N export, kg N ha^{-1}			
10% slope	BARE	0.00	1.32	0.0115	10.7	0.00257
	LOW	0.0137	0.0343	0.00784	0.999	0.00161
	MED	0.0119	0.0868	0.0241	0.101	0.000753
	HIGH	0.0100	0.377	0.0296	1.24	0.00155
20% slope	BARE	0.00	1.19	0.0203	5.11	0.00835
	LOW	0.0174	0.0807	0.00583	0.446	0.00103
	MED	0.0167	0.0386	0.00126	0.0126	0.00517
	HIGH	0.00807	0.100	0.00116	0.143	0.000465
Wetland site						
10% slope	BARE	0.00551	15.9	0.0175	15.8	0.00302
	FULL	0.000128	15.7	0.0372	12.1	0.00340

Appendix 5.18. **Mean total ammonium-nitrogen (NH_4-N) export for the 1-h rain event**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover		NH_4-N export, kg N ha^{-1}			
10% slope	BARE	0.00	1.33	0.0159	11.3	0.00284
	LOW	0.0140	0.0715	0.0850	1.30	0.00174
	MED	0.0137	0.0897	0.0265	0.130	0.000995
	HIGH	0.0100	0.379	0.0602	1.24	0.00166
20% slope	BARE	0.00	1.19	0.0396	5.53	0.0103
	LOW	0.0324	0.107	0.00629	0.573	0.00103
	MED	0.0331	0.0437	0.00288	0.0126	0.00567
	HIGH	0.0148	0.103	0.00324	0.184	0.000465
Wetland site						
10% slope	BARE	0.00931	17.1	0.0245	17.5	0.00466
	FULL	0.000128	17.7	0.0520	16.3	0.00401

Appendix 5.19. **Mean cumulative nitrate-nitrogen (NO_3-N) export at 30 min after runoff initiation**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover		NO_3-N export, kg N ha^{-1}			
10% slope	BARE	0.379	0.936	0.632	0.117	1.06
	LOW	0.0626	0.352	0.566	0.747	0.935
	MED	0.0562	0.257	0.331	0.761	0.483
	HIGH	0.0279	0.401	0.386	0.461	1.21
20% slope	BARE	0.340	0.638	0.735	0.108	1.17
	LOW	0.0892	0.329	0.251	0.467	0.239
	MED	0.0439	0.103	0.0398	0.0904	0.0632
	HIGH	0.0314	0.208	0.0775	0.269	0.0869
Wetland site						
10% slope	BARE	0.382	0.138	1.28	0.0117	0.712
	FULL	0.0958	0.313	0.343	0.0612	0.326

Appendix 5.20. **Mean total nitrate-nitrogen (NO_3-N) export for the 1-h rain event**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover		NO_3-N export, kg N ha^{-1}			
10% slope	BARE	0.572	1.22	0.906	1.70	1.38
	LOW	0.114	0.594	0.844	1.14	1.22
	MED	0.0834	0.476	0.535	1.19	0.640
	HIGH	0.0461	0.831	0.483	0.939	1.52
20% slope	BARE	0.551	1.02	1.62	0.732	1.64
	LOW	0.222	0.622	0.362	0.774	0.354
	MED	0.0765	0.166	0.0699	0.0904	0.0995
	HIGH	0.0618	0.421	0.112	0.454	0.0918
Wetland site						
10% slope	BARE	0.591	1.83	1.84	0.260	1.05
	FULL	0.162	2.35	0.454	0.0612	0.469

Appendix 5.21. **Mean cumulative total Kjeldahl nitrogen (TKN) export at 30 min after runoff initiation**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	TKN export, kg N ha^{-1}				
10% slope	BARE	1.54	3.46	0.759	16.8	2.46
	LOW	0.129	0.247	1.37	1.61	0.205
	MED	0.101	0.175	0.237	0.516	0.163
	HIGH	0.0873	0.428	0.236	1.62	0.173
20% slope	BARE	5.09	6.10	0.462	15.4	7.13
	LOW	0.443	0.484	0.298	0.631	0.100
	MED	0.162	0.207	0.148	0.0553	0.0442
	HIGH	0.0971	0.259	0.774	0.334	0.0366
Wetland site						
10% slope	BARE	4.12	20.2	5.34	31.9	6.40
	FULL	0.519	14.9	0.591	16.4	0.624

Appendix 5.22. **Mean total Kjeldahl nitrogen (TKN) export for the 1-h rain event**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover	TKN export, kg N ha^{-1}				
10% slope	BARE	3.28	5.68	1.71	20.4	4.29
	LOW	0.228	0.435	1.60	1.95	0.273
	MED	0.159	0.311	0.384	0.769	0.206
	HIGH	0.108	0.583	0.313	1.87	0.216
20% slope	BARE	9.21	10.6	0.986	21.6	14.0
	LOW	1.49	0.752	0.700	0.856	0.149
	MED	0.354	0.347	0.281	0.0553	0.0648
	HIGH	0.229	0.415	0.945	0.456	0.0387
Wetland site						
10% slope	BARE	7.43	25.2	8.04	42.9	11.6
	FULL	0.756	16.9	0.822	21.8	0.857

Appendix 5.23. **Mean cumulative total nitrogen (TN) export at 30 min after runoff initiation**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover		TN export, kg N ha^{-1}			
10% slope	BARE	1.92	4.40	1.39	16.9	3.52
	LOW	0.191	0.599	1.94	2.36	1.14
	MED	0.157	0.432	0.568	1.28	0.646
	HIGH	0.115	0.829	0.622	2.08	1.38
20% slope	BARE	5.43	6.74	1.20	15.5	8.29
	LOW	0.532	0.813	0.549	1.10	0.340
	MED	0.205	0.310	0.187	0.146	0.107
	HIGH	0.128	0.467	0.851	0.603	0.123
Wetland site						
10% slope	BARE	4.50	20.4	6.62	31.9	7.11
	FULL	0.615	15.2	0.934	16.4	0.949

Appendix 5.24. **Mean total nitrogen (TN) export for the 1-h rain event**

		Rain Event				
		APR	MAY	JUN	SEP	OCT
Upland sites	Cover		TN export, kg N ha^{-1}			
10% slope	BARE	3.85	6.90	2.62	22.1	5.68
	LOW	0.341	1.03	2.45	3.09	1.50
	MED	0.242	0.787	0.919	1.96	0.846
	HIGH	0.154	1.41	0.796	2.81	1.73
20% slope	BARE	9.76	11.6	2.61	22.3	15.7
	LOW	1.71	1.37	1.06	1.63	0.503
	MED	0.430	0.513	0.351	0.146	0.164
	HIGH	0.291	0.836	1.06	0.910	0.131
Wetland site						
10% slope	BARE	8.02	27.0	9.88	43.2	12.7
	FULL	0.918	19.3	1.28	21.8	1.33