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Nitrogen export from Coastal Plain field ditches

J.P. Schmidt, C.J. Dell, P.A. Vadas, and A.L. Allen

Abstract: Mitigating the adverse impact of nitrogen (N) fertilizer applications depends on an understanding of transport mechanisms and flow pathways. The objective of this study was to quantify N export from seven ditches on the Maryland Eastern Shore. Ditches were monitored between June 2005 and May 2006, including flow and sample analyses for storms and base flow. Mean total N and NO_3-N concentrations were 10.6 and 6.0 mg L^{-1} (10.6 and 6.0 ppm) for ditch 8, which were 2 times the total N and NO_3-N concentrations for any other ditch. Greater mean concentrations in ditch 8 translated to 43.5 kg ha^{-1} (38.8 lb ac^{-1}) total N loss and 24.9 kg ha^{-1} (22.2 lb ac^{-1}) NO_3-N loss, which were not consistent with losses observed for any of the other ditches. The elevated losses in ditch 8 coincided with the presence of a manure storage shed located in this drainage basin. The two ditches (7 and 8) nearest the manure storage shed had the greatest increase in organic N loss as a function of drainage outflow, increasing 0.062 kg ha^{-1} (1.56 lb ac^{-1}) per mm (in) drainage outflow compared to 0.017 kg ha^{-1} (0.45 lb ac^{-1}) per mm (in) outflow for the other five ditches. Ditches 2 and 3 had the greatest outflow of water (640 mm [25.2 in]), contributing to greater NO_3-N loads—a consequence of greater groundwater drainage. Implementing management strategies that mitigate N losses from agricultural fields should be considered in the context of ditch hydrology and drainage basin features.

Key words: ditches—drainage basin—hydrology—nitrogen—poultry manure

Artificial or improved drainage is used to increase agricultural production on many soils, representing as many as 40 million ha (100 million ac) throughout the United States (Pavelis 1987). Poor drainage is common on the Coastal Plain of the eastern United States because of wide, flat interfluves and very little local relief, so artificial drainage is an integral part of the agricultural landscape here. Recognizing that artificial drainage networks represent a conduit through which NO_3 from fertilized agricultural fields quickly reaches surface waters, researchers in North Carolina have sought ways to mitigate NO_3 losses using controlled drainage. Gilliam et al. (1979) observed a 50% reduction in NO_3 losses in drainage ditches of the Coastal Plain when flashboard risers were installed in the tile mains and used to elevate water table depth. They attributed the decreased NO_3 loss to increased denitrification (conversion of NO_3 to N_2O or N_2 under anaerobic conditions) as a result of water remaining longer in the soil profile. Burchell et al. (2005) demonstrated that more closely spaced, shallow (0.75 m [2.5 ft]) subsurface drains reduced NO_3 drainage losses more than widely spaced, deeper (1.5 m [5 ft]) subsurface drains during one year of a two-year study on the lower Coastal Plain of North Carolina, decreasing NO_3-N loss by 10 kg ha^{-1} (8.9 lb ac^{-1}) during the second year. Numerous studies have been conducted in North Carolina (Gambrell et al. 1975a, 1975b; Amatya 1998; Skaggs et al. 2005) evaluating the impact of subsurface and surface drainage spacing on the hydrology and, consequently, on NO_3 drainage

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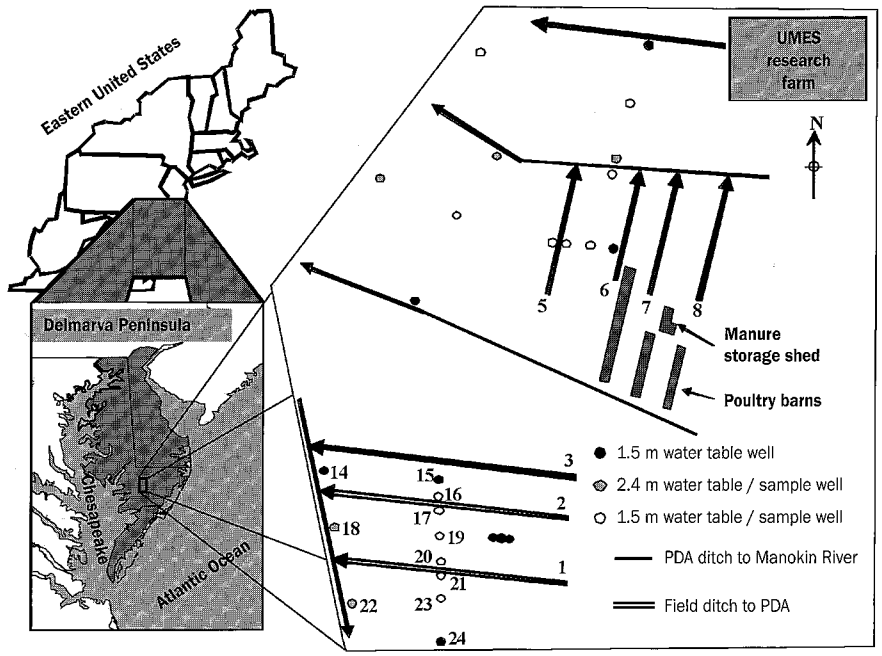
losses from these Coastal Plain soils; however, few drainage studies have been conducted on the Eastern Shore of Maryland, and perhaps fewer studies have focused on processes within ditches.

Nitrate that moves below the crop root zone will likely reach surface waters through lateral movement to streams and ditches of the Coastal Plain, unless the groundwater is intercepted by a riparian area—vegetated area along stream and ditch edges where the combination of soluble C, anoxic conditions, and NO_3 can lead to denitrification (Jacobs and Gilliam 1985; Greenan et al. 2006). While NO_3 removal from ground and surface waters is a desirable outcome, denitrification has a potentially negative consequence that should also be considered. Complete reduction of NO_3 to N_2 represents an environmentally benign gaseous emission to the atmosphere; however, incomplete reduction (Alexander 1977) to N_2O (a greenhouse gas) has implications for global climate change. The effectiveness of most riparian areas in mitigating excess groundwater NO_3 depends on a model of groundwater hydrology where shallow subsurface flow is horizontal and perpendicular (or nearly so) to the stream or ditch. This representation is reasonable in some landscapes and has been effective in describing the hydrology in North Carolina (Jacobs and Gilliam 1985; Skaggs et al. 2005); however, a recent study on the Coastal Plain of Maryland suggests that focused groundwater exfiltration has not been adequately considered in determining the effectiveness of riparian areas (Angier et al. 2005). Angier et al. were able to identify upwelling zones that displayed positive vertical hydraulic heads supplying about 4% of the stream outflow, but only comprising 0.006% of the riparian area. Management practices that maximize complete reduction of NO_3 to N_2 will depend on improving our understanding of the landscape processes affecting denitrification and identifying landscape locations that can be effectively targeted with improved management techniques to mitigate the export of excess NO_3 from the agricultural landscape.

The objective of this study was to quantify nitrogen (N) export from shallow surface ditches on Maryland's Eastern Shore, focusing on losses incurred from seven ditches during one year, and to consider implications for improving agricultural management practices that minimize N export from this landscape.

Figure 1
Field site map illustrating ditches and wells at the University of Maryland Eastern Shore research farm.



Note: This illustration is not to scale; ditch dimension are provided in table 1.

Materials and Methods

Site Description. This research was conducted on the research farm located at the University of Maryland Eastern Shore (UMES), Princess Anne, Maryland (38°12'22" N and 75°40'35" W). Geographic and climatic characteristics of the farm are provided by Kleinman et al. (2007). Prior to 1997, the UMES research farm was a commercial broiler (poultry) operation for more than 20 years. Three broiler barns and a manure storage shed are currently located on the farm (figure 1). Broilers and sometimes goats are still raised in these barns. The manure storage shed is regularly used as the name implies. The barns and manure storage shed have a packed dirt floor, and sometimes manure is temporarily stacked and/or spilled outside the manure storage shed. The poultry barns are within the drainage basins of ditches 6, 7, and 8. The manures storage shed is midway between ditches 7 and 8, and it probably contributes water to both ditches. Roof runoff for all of these buildings is uncontrolled.

Generally, the crop rotation for the UMES research farm includes corn (*Zea mays* L.) and soybean (*Glycine max* (L.) Merr.). In 2005, corn was planted in the area west of ditch 6 and in roughly half of the area south of ditch 3 (including around ditches 1 and 2); soybean was planted adjacent to ditches 7 and 8, north of ditch 3, and in the other half of the area south of ditch 3. In 2006, corn was planted in all the areas adjacent to ditches 5, 6, 7, and 8, north of ditch 3, and in half of the area south of ditch 3. Soybeans were planted in 2006 in the other half of the area south of ditch 3. Soils are enriched with P (Kleinman et al. 2007) due to a long history of receiving poultry manure at rates often exceeding annual crop removal, which is an indication that N mineralization potential could also be high in these fields.

Soils on the farm belong to the poorly-drained Othello series (fine-silty, mixed, active, mesic Typic Endoaquult) derived from silty eolian sediments underlain by coarser marine sediments (Matthews and Hall 1966). An extensive ditch system (figure 1) is present, with most fields bounded by at least one taxed ditch (>2 m [>7 ft]) maintained by the local Public Drainage Association (PDA). Most fields include additional shallower (0.3 to 1 m [1.0 to 3.3 ft]) ditches that are managed by the farm operator. Between 2001 and 2006, fertilizer was applied before planting corn using the poultry litter gen-

Table 1
Dimensions for seven drainage ditches and soil NO_3 and NH_4 concentration (0 to 5 cm) in the fields around these ditches.

Ditch	Ditch dimensions l × w × d (m)	Inorganic soil N* (mg kg^{-1}) NO_3	Inorganic soil N* (mg kg^{-1}) NH_4	Inorganic soil N† (mg kg^{-1}) NO_3	Inorganic soil N† (mg kg^{-1}) NH_4
1	280 × 1.0 × 0.5	12.2	2.0	–	–
2	300 × 1.0 × 1.0	10.5	2.4	–	–
3	320 × 1.0 × 1.0	9.4	3.5	–	–
5	160 × 0.3 × 0.3	14.4	2.8	–	–
6	200 × 0.5 × 0.5	20.2	3.6	–	–
7	160 × 0.3 × 0.3	20.8	2.0	2.6	220.4
8	180 × 0.6 × 0.6	17.0	6.1		

* Soil samples were collected in October 2006.

† Soil samples around the manure storage shed were collected in November 2006.

erated from the farm (50 to 150 kg N ha^{-1} [45 to 135 lb N ac^{-1}] and 40 to 120 kg P ha^{-1} [35 to 105 lb P ac^{-1}]) and supplementing with sidedress of urea ammonium nitrate (UAN) to achieve about 170 kg N ha^{-1} (150 lb N ac^{-1}).

A solar powered meteorological station at the UMES research farm was used to record wind speed and direction, temperature, precipitation, relative humidity and solar radiation on five-minute intervals.

Water Samples. Seven ditches on the UMES Farm were monitored for flow between June 2005 and May 2006, with water samples collected for analyses. These include ditches 1 through 8, excluding 4 (figure 1). Ditch dimensions are provided in table 1. Since May 2005, the outlet of each ditch has been equipped with a monitoring station consisting of an H-flume (size 0.5 m to 0.8 m [1.6 to 2.6 ft]) and a solar-powered automatic sampler (Sigma 900max, Hach Corporation, Loveland, Colorado). The automatic sampler was controlled by a pressure transducer and programmed to collect flow proportional samples (samples 1 to 4 every 95 L [25 gal], samples 5 to 8 every 190 L [50 gal], samples 9 to 96 every 380 L [100 gal]) that were later combined to form a single, composite sample for each runoff event. Samplers were removed in January due to freezing concerns and then reinstalled near the end of March. However, flow stage was continuously recorded throughout the year using a float-pulley shaft encoder. Storm event samples obtained by the automatic samplers were retrieved within 48 to 72 hours of each event. A water sample was split—half was filtered (0.45 μm) and the other half remained unfiltered. Samples were then stored at 4°C (39°F) prior to analysis. In addition, grab samples representing base flow conditions were obtained every two to four weeks, including during the winter months when the automatic samplers were absent. Grab samples were processed for analyses following the same procedure as already described. A monthly load (kg or lb) was determined based on total flow for the month multiplied by mean concentration of one to three storms (and base flow) within each month. An annual flow-weighted mean concentration was determined by summing monthly loads and then dividing by total flow.

Shallow groundwater wells were installed at various points (figure 1) within the site in 2003 and are described in detail by Vadas et al. (2007). For this study, groundwater height measured almost weekly was considered in the context of interpreting drainage outflow from the various ditches.

Soil Samples. Soil samples were collected from the field around each ditch in October 2006 and from the area near the manure storage shed in November 2006 to a depth of 5 cm (2 in). Field soil samples were obtained at roughly 30 m (100 ft) intervals along transects parallel to the ditches, approximately halfway between the ditch and the midpoint between ditches. Samples around the manure storage shed were collected at 5 m (17 ft) intervals in the area between the shed and ditches 7 and 8. Subsamples were combined to form a single composite sample.

Analyses. Total N was measured on unfiltered water samples by digesting the sample using the alkaline persulfate digestion (Patton and Kryskalla 2003), converting all N forms to NO_3. Total N, as NO_3, and NO_3-N for filtered samples were determined using a Lachat autoanalyzer (Quick Chem FIA+ 8000 Series, Lachat Instruments, Loveland, Colorado) following the method described by Wendt (2000). Organic N was determined as the difference between total N and NO_3-N.

Soil samples were air dried and sieved (2 mm or 0.08 in) prior to analysis. Inorganic N in soil samples was determined by flow injection analysis of 2 *M* KCl extracts (QuikChem Methods FIA+ 8000 Series, Lachat Instruments, Loveland, Colorado). Linear and quadratic regression analyses were completed using PROC REG (SAS Institute 1999).

Results and Discussion

Drainage Outflow. Drainage outflow varied from 135 to 643 mm (5.3 to 25.3 in) for these seven ditches on the UMES research farm (table 2). This represents between 15% and 70% of the total amount of precipitation received within each of the drainage areas and reflects a measure of variability in drainage basin features and depth of ditches. Drainage outflows for five of the ditches—1, 5, 6, 7, and 8—were similar, while outflows from ditches 2 and 3 were greater than from the other ditches, particularly between October and January (figure 2).

The greater outflows from ditches 2 and 3 can be attributed to greater subsurface flow in these ditches because they are deeper ditches (1 m [3.3 ft], table 1) and outflows exceeded precipitation between October and January (figure 2)—a consequence of continued outflow despite small precipitation amounts. Although these two ditches are adjacent to ditch 1, ditch 1 is not as deep (0.5 m [1.6 ft]) as ditches 2 and 3. A deeper (>2 m [>6.6 ft]) PDA ditch south of well 24 (figure 1, this PDA ditch is not shown on the map) controls the direction of groundwater flow near ditch 1 (Vadas et al. 2007), decreasing outflow in ditch 1 as compared to ditches 2 and 3. Between 2003 and 2006, mean groundwater height at wells 16 and 17 (figure 1, ditch 2) was 7 cm (2.8 in) higher than mean groundwater height at wells 20 and 21 (ditch 1), which was 120 cm (4 ft) higher than mean groundwater height at well 24 (based on weekly measurements from wells identified in figure 1). The prolonged flows in ditches 2 and 3 that were observed between October and January after the larger October precipitation events (figure 2) illustrate the impact of groundwater hydrology on the deeper ditches at this site.

Despite the presence of barns in the drainage basins of ditches 6, 7, and 8, which

Table 2
Drainage outflow and nitrogen characterization for seven drainage ditches between June 2005 and May 2006.

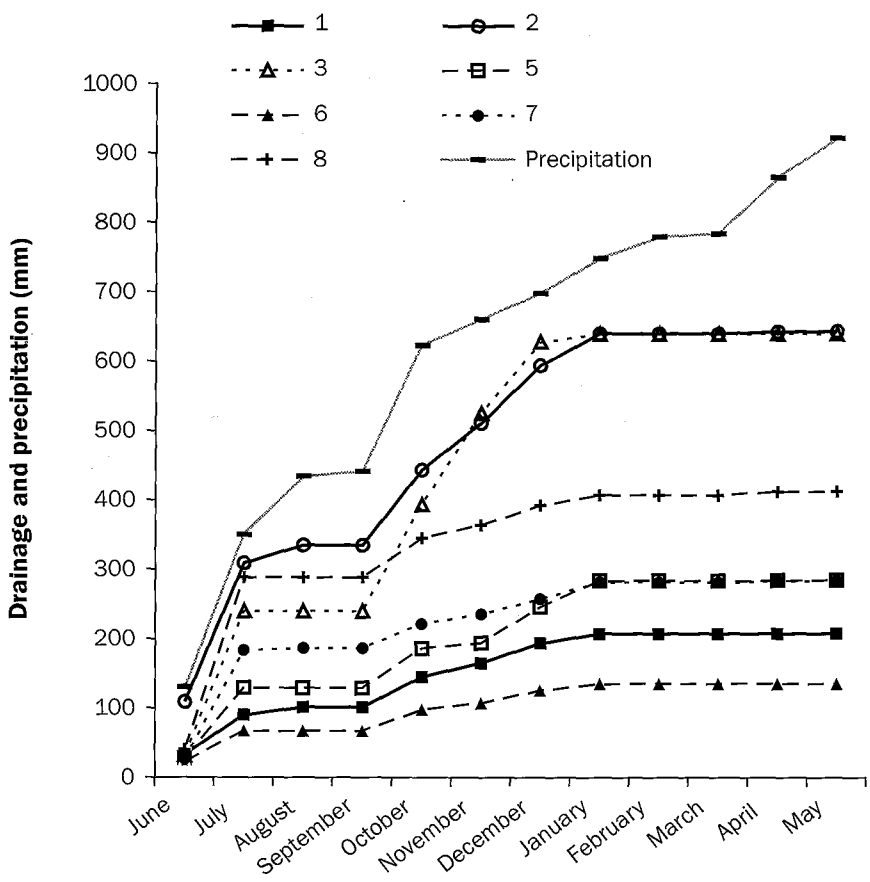
Ditch	Precipitation (mm)	Catchment (ha)	Outflow* (mm)	Total load (kg)		Loss (kg ha^{-1})		Proporation of N loss as NO_3-N	Mean concentration‡ (mg L^{-1})	
				NO_3-N	Total N†	NO_3-N	Total N†		NO_3-N	Total N†
1	922	2.7	207	9.5	23.7	3.5	8.6	0.40	1.7	4.2
2	922	1.8	643	19.4	39.9	11.0	22.6	0.49	1.7	3.5
3	922	2.0	639	32.7	58.0	16.3	28.9	0.56	2.6	4.5
5	922	1.1	284	7.5	14.6	6.8	13.3	0.51	2.4	4.7
6	922	1.0	135	1.8	5.3	1.7	5.1	0.33	1.3	3.8
7	922	1.2	284	5.7	18.6	4.8	15.5	0.31	1.7	5.5
8	922	0.8	412	20.4	35.6	24.9	43.5	0.57	6.0	10.6
Mean		1.5	372	13.8	27.9	9.8	19.7	0.45	2.5	5.2

* These outflows are slightly smaller than reported by Kleinman et al. (2007) because one or two very small flow events were omitted for which N analyses were missing.

† Persulfate digestion (Patton and Kryskalla 2003).

‡ Flow-weighted mean concentration based on a sum of monthly load divided by total flow. A monthly load is based on total flow multiplied by mean concentration of 1 to 3 storms within each month.

Figure 2
Precipitation and cumulative drainage for seven drainage ditches between June 2005 and May 2006.



could contribute to additional surface runoff, outflows from these ditches were similar, or only slightly greater (ditch 8), to outflows from ditches 1 and 5, all of which are shallow ditches (<0.6 m [<2 ft], table 1). These shallow ditches function predominantly to move surface water away from the field and perhaps to move lateral-flowing shallow groundwater that occurs during and immediately after a runoff-generating rainfall. Water in ditches 1, 5, 6, 7, and 8 was observed only during and immediately after a runoff-generating rainfall.

Nitrogen Load. Mean N concentration, total load, and loss (table 2) represent single observations for each of these seven ditches; consequently, an interpretation based on a statistical comparison of means among these ditches, e.g., a means separation procedure, was not possible. However, an evaluation of several relationships helps identify outliers from which we might attribute unique characteristics to individual ditches or their drainage basins.

Total N load from these seven ditches during the study year increased from 5.3 kg N (11.7 lb) in ditch 6 to 58 kg N (128 lb) in ditch 3 (table 2). Similarly, NO_3-N and organic N loads were least in ditch 6 and greatest in ditch 3, with NO_3-N load ranging from 1.8 to 32.7 kg N (4.0 to 72.1 lb) and organic N load ranging from 3.5 to 25.3 kg N (7.9 to 55.6 lb). Total N, NO_3-N, and organic N load all increased linearly with increasing drainage outflow (figures 3A, 3B, and 3C), indicating that the driver for N load from all of these drainage basins

was primarily dependent upon the amount of water outflow per unit area (outflow corrected for the drainage basin size).

Nitrogen Loss. When total N load is converted to total N loss, i.e., based on a per unit area measure for each drainage basin, the relationship between total N loss and drainage outflow indicated that total N loss from ditch 8 was inconsistent with losses observed for the other ditches. Total N losses for ditches 1, 2, 3, 5, 6, and 7 were linearly related to drainage outflow (figure 4A). With each additional 100 mm (4 in) of outflow for all of these six ditches, there was a 3.8 kg N ha^{-1} (3.4 lb ac^{-1}) increase in total N loss (r^2 = 0.91; P > F = 0.003). The total N loss from ditch 8 was 25.8 kg N ha^{-1} (23.0 lb ac^{-1}) greater than would be suggested by the relationship derived for the other six ditches (figure 4A), suggesting that ditch 8 has some unique characteristic that is contributing to total N loss that is not present in any of the other ditches. Ammonium-N concentration in the surface soil around the manure shed was 220 mg L^{-1} (220 ppm), which was more than 60 times the level of NH_4-N observed for soil samples collected around any of the other ditches (table 1). This high level of NH_4-N in the soil around the manure storage shed implicates this area as a point source in contributing to total N loss from ditch 8.

Total N losses from most of the ditches at this Eastern Shore site were within the range of results noted in previous studies (described below) on the Coastal Plain of the Mid-Atlantic and southeastern United States. In North Carolina, where most of the published drainage research on the Coastal Plain originates, Gambrell et al. (1975b) estimated that the two-year mean N loss from surface runoff was 22 to 29 kg N ha^{-1} yr^{-1} (19.6 to 25.9 lb ac^{-1} yr^{-1}) for two different study sites in continuous corn. Surface outflow at these two sites was estimated at 250 to 300 mm, while annual N applications were about 180 kg N ha^{-1} (160 lb N ac^{-1}). The N losses attributed to surface runoff in this North Carolina study were slightly more than N losses for ditches 1, 5, 6, and 7. The extra N applied in a continuous corn rotation could account for the additional losses observed in the North Carolina study. Staver and Brinsfield (1995) estimated annual total N loss in surface water runoff at <5 kg N ha^{-1} (<4.5 lb ac^{-1}) for a no-till continuous corn rotation study on the Delmarva Peninsula. The lower N losses

Figure 3
Total N load, NO_3-N load, and organic N load as a function of drainage outflow for the seven ditches.

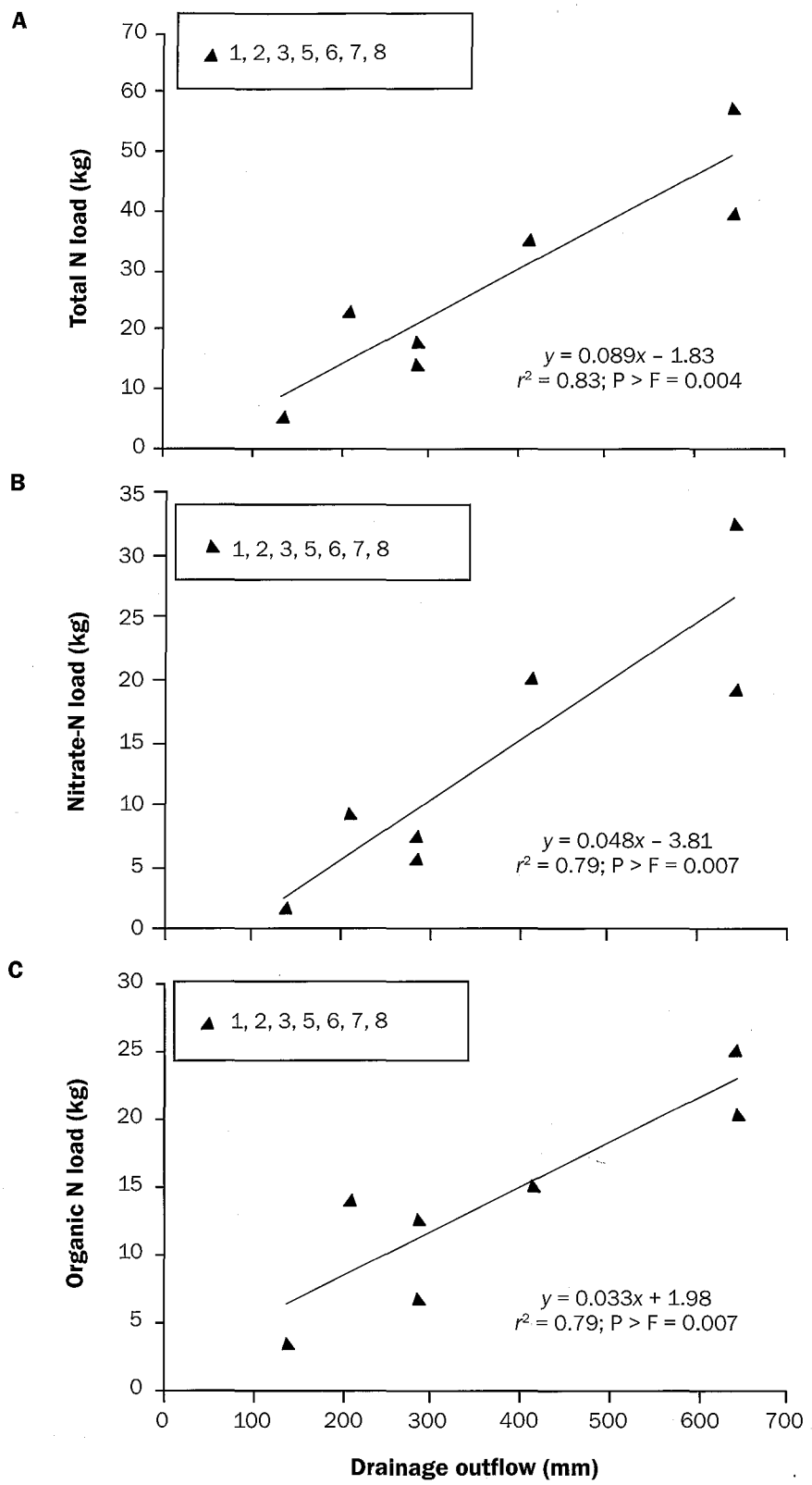
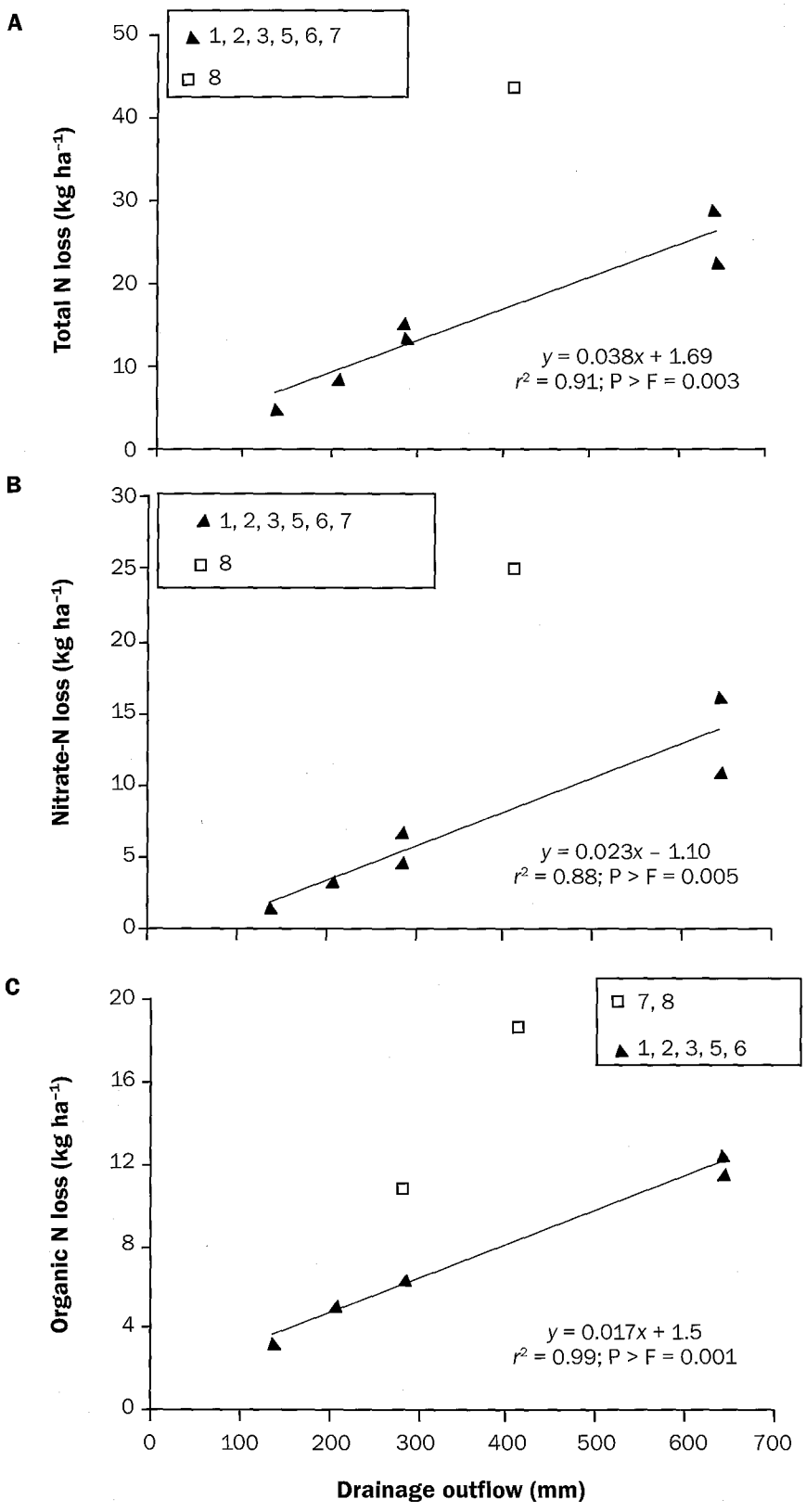


Figure 4
Total N loss, NO3-N loss, and organic N loss as a function of drainage outflow for the seven ditches.



observed in this latter study reflect the use of N management practices that reduce N losses, such as a split N application and the use of grass waterways. Nitrogen losses in ditches 1, 5, 6, and 7 (table 2) represent losses for surface drainage that were comparable to losses observed in other studies on this agricultural Coastal Plain landscape.

Total N loss from ditch 8, 43.5 kg N ha^{-1} yr^{-1} (38.8 lb N ac^{-1} yr^{-1}), was considerably greater than the amount attributed to surface runoff by Staver and Brinsfield (1995) or Gambrell et al. (1975b). Whether outflow from ditch 8 (table 2) can be attributed only to surface runoff or surface water and shallow, laterally flowing groundwater, this high level of N loss seems likely a consequence of the manure storage shed within the drainage basin. Nitrogen losses in ditch 8 were 3 to 8 times the losses attributed to surface runoff from other ditches at this study site (excluding ditches 2 and 3 because of subsurface contribution to outflow) and identify a source problem that should be mitigated with roof and soil surface runoff diversions and/or improving manure storage management by avoiding spillage around the shed.

Nitrate-N losses from ditches 1, 5, 6, and 7 were 6.8 kg N ha^{-1} yr^{-1} (6.1 lb ac^{-1} yr^{-1}) or less (table 2), which were comparable to the surface runoff losses in NO_3-N observed for the lower North Carolina Coastal Plain watershed (Jacobs and Gilliam 1985). Nitrate-N losses in ditches 2 and 3 were 11.0 and 16.3 kg N ha^{-1} yr^{-1} (9.8 and 14.5 lb ac^{-1} yr^{-1}), which were similar to the losses observed for subsurface drainage in the middle North Carolina Coastal Plain watershed (Jacobs and Gilliam 1985). Greater drainage outflow (figure 2) and greater NO_3-N losses for ditches 2 and 3 suggest that subsurface flow was impacting N losses in these two ditches. However, greater NO_3 loss was observed even for ditch 8 (24.9 kg N ha^{-1} yr^{-1} or 22.2 lb ac^{-1} yr^{-1}), which was much greater than losses from any other ditch (figure 4B). Although NO_3-N levels in the soil around the manure storage shed were not very high when sampled in November 2006 (table 1), the very high soil NH_4-N here provides an N source for microbial nitrification—a continual source of NO_3-N throughout the year. Ditch 8 is a shallow ditch (0.6 m [2 ft]), so the high NO_3 loss for ditch 8 could not be attributed to additional groundwater outflow that results from extended periods of base flow (as observed

with ditches 2 and 3). Outflow in this ditch occurs during runoff-generating rainfall events and is a consequence of surface runoff and/or shallow, laterally flowing groundwater during and immediately following such events. High NO_3 loss from ditch 8 seems to implicate the latter of these two possibilities.

Similar to results for total N loss, NO_3-N loss appeared to behave somewhat similarly among six of the ditches but different from the NO_3-N results for ditch 8, implicating the unique characteristic of the ditch 8 drainage basin (figure 4B). With each additional 100 mm (4 in) increase in drainage outflow, NO_3-N loss in ditches 1, 2, 3, 5, 6, and 7 increased 2.3 kg N ha^{-1} (1.9 lb ac^{-1}) (r^2 = 0.88; P > F = 0.005). Nitrate-N loss from ditch 8 was 16.4 kg ha^{-1} (14.6 lb ac^{-1}) greater than would be expected if ditch 8 had been behaving similarly to the other six ditches (figure 4B). Because NO_3-N loss is the product of mean NO_3-N concentration and outflow (corrected for area), greater mean NO_3-N concentration in ditch 8 (6.0 mg L^{-1} [6.0 ppm], table 2) must be responsible for the unusually greater NO_3-N loss from this ditch. Elevated NO_3-N concentration in ditch 8 was most likely a consequence of the manure storage shed located in the ditch 8 drainage basin.

Nitrate-N losses are generally considered a consequence of subsurface flow, as a result of NO_3 leaching through the soil profile and lateral movement to ditches and streams via tile drainage or lateral flow through the soil matrix. However, in this Coastal Plain watershed contributions from both subsurface and surface runoff appear to be implicated in NO_3-N losses. Jacobs and Gilliam (1985) estimated that mean NO_3-N losses in two tile-drained Coastal Plain watersheds were 23.5 and <0.01 kg N ha^{-1} yr^{-1} (21.0 and <0.01 lb ac^{-1} yr^{-1}) from subsurface drainage and 1.5 and 7.9 kg N ha^{-1} yr^{-1} (1.3 and 7.1 lb ac^{-1} yr^{-1}) from surface drainage. The greater NO_3-N loss occurred by subsurface drainage in the middle Coastal Plain watershed, which included well- to moderately well-drained soils. Conversely, greater NO_3-N loss was observed for surface runoff at the lower Coastal Plain watershed, which was dominated by surface runoff from somewhat poorly drained soils. The current study site, on Maryland's Eastern Shore, is probably more comparable to the lower Coastal Plain watershed described by Jacobs and Gilliam (1985), and the observed NO_3-N losses reflect a shallow drainage system that is dominated by surface runoff, but not entirely (i.e., ditches 2 and 3).

Organic N losses among these seven ditches did not consistently follow the same pattern as observed for NO_3-N or total N losses. Organic N losses from ditches 1, 2, 3, 5, and 6 appeared to depend on similar drainage basin characteristics, based on the linear relationship between organic N loss and drainage outflow for these five ditches (figure 4C). With an additional 100 mm (4 in) outflow for these five ditches, organic N loss increased 1.7 kg N ha^{-1} (1.4 lb ac^{-1}) (r^2 = 0.99, P > F = 0.001). By contrast, the relationship between organic N loss and outflow for ditches 7 and 8 indicated that organic N losses for these ditches increased 6.2 kg ha^{-1} (4.9 lb ac^{-1}) for each additional 100 mm (4 in) increase in drainage outflow. This represents more than 3 times increase in organic N loss per unit outflow for ditches 7 and 8 compared to ditches 1, 2, 3, 5, and 6, implicating the organic N point source of the manure storage shed located midway between ditches 7 and 8 (figure 1).

Nitrogen Concentration. Mean N (total, NO_3, and organic) concentrations in the runoff corroborated the N load and loss results, suggesting that ditches 7 and 8 were characteristically similar and impacted by a point source (manure storage shed), while ditches 1, 2, 3, 5, and 6 were behaving similarly and not impacted by a point source. Mean total N and NO_3-N concentrations for ditches 1, 2, 3, 5, 6, and 7 (figures 5A and 5B) were 4.4 and 1.9 mg L^{-1} (4.4 and 1.9 ppm), respectively. While drainage outflow for ditch 8 was comparable to the other shallow ditches, 412 mm (16.5 in), total N concentration was 10.6 mg L^{-1} (10.6 ppm) and mean NO_3-N concentration was 6.0 mg L^{-1} (6.0 ppm). These concentrations were greater than observed for the six other ditches (figures 5A and 5B), suggesting that total N and NO_3-N concentrations in the surface runoff of ditch 8 was adversely affected by the manure storage shed. Organic N concentrations for ditches 7 and 8 increased linearly with increasing drainage outflow (figure 5C); however, a negative linear trend in organic N concentration was observed for ditches 1, 2, 3, 5, and 6. These disparate trends (figure 5C) in organic N concentrations as outflow increased distinguish the impact of point sources (particularly manure) and non-point N sources on this Coastal Plain landscape. Very few studies have partitioned N losses between NO_3-N and organic N losses, although Gambrell et al. (1975b) did separate surface runoff losses (22 and 29 kg N ha^{-1} yr^{-1} [19.6 and 25.9 lb ac^{-1} yr^{-1}]) for two North Carolina Coastal Plain watersheds into either sediment or organic N. The N loads, losses, and concentrations observed in ditches 1, 2, 3, 5, and 6 appear to be representative of and the consequence of field management practices that are currently typical on the Coastal Plain, whereas N loads, losses, and concentrations observed for ditch 8 (and ditch 7 for organic N) implicate an N point source problem.

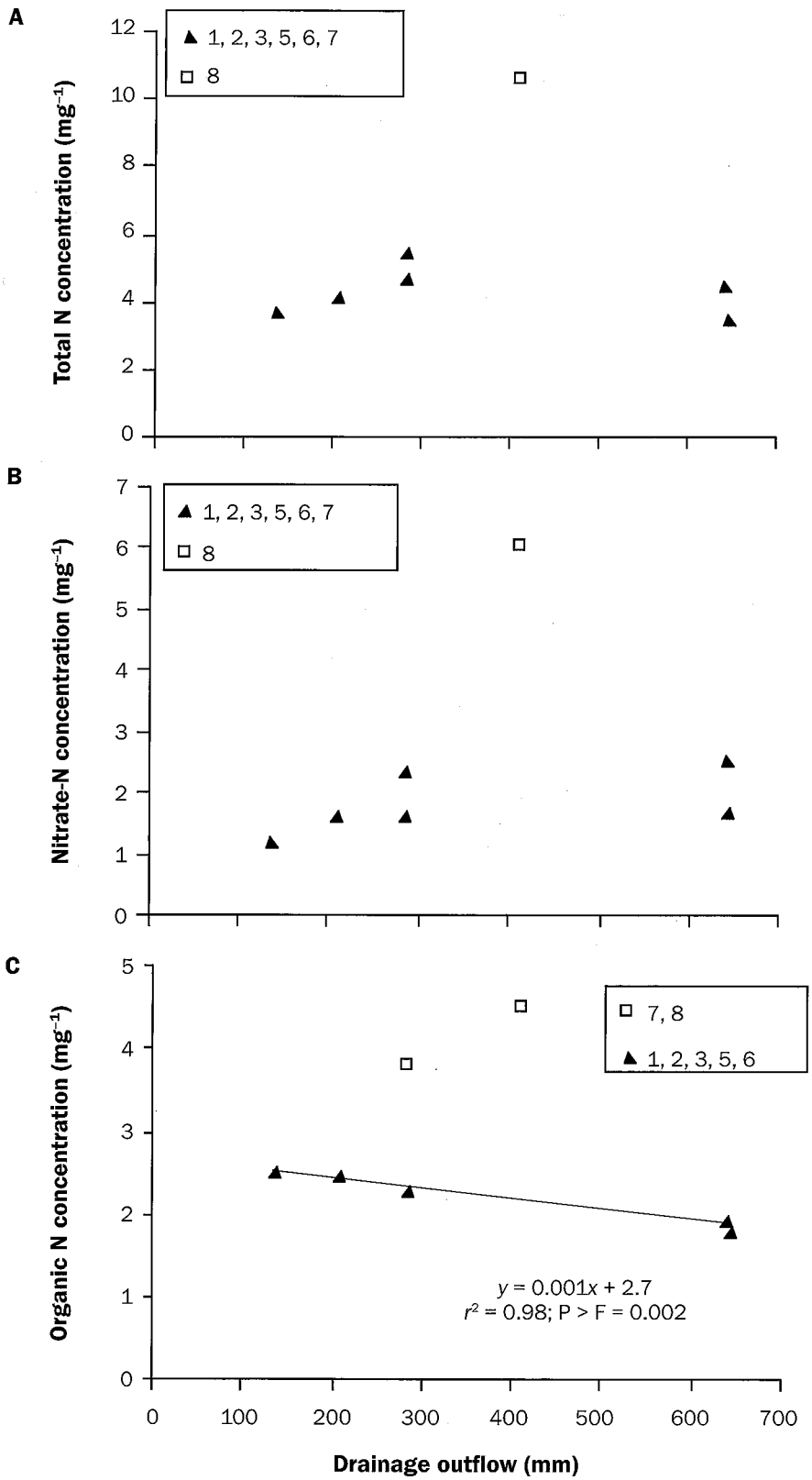
Management Implications. Improving N management, particularly for lands that potentially contribute pollutants to vulnerable water bodies such as the Chesapeake Bay, will depend on a better understanding of the processes and pathways responsible for the movement of N from the landscape.

The deeper (1 m [3.3 ft]) ditches, such as ditches 2 and 3, had drained proportionally more water than the other ditches (figure 2), which can be attributed to additional subsurface drainage in these deeper ditches. Effective management that should contribute to a decrease in N export from these types of ditches depends on techniques such as riparian buffers, controlled drainage, or subsurface biological curtains (as described below), all of which contribute effectively to denitrification of NO_3 in the groundwater when appropriately installed and implemented.

Previous research that has demonstrated the effectiveness of riparian buffers was conducted by Jordon et al. (1993), who observed a decrease in groundwater NO_3-N from 8 to less than 0.4 mg L^{-1} (8 to less than 0.4 ppm) within a horizontal distance of 30 m (98 ft) in a wooded riparian area. Jacobs and Gilliam (1985) also demonstrated the effectiveness of riparian areas in reducing NO_3-N concentrations in the groundwater; however, in both cases the width of the riparian area was more than 30 m, which is impractical in areas where the ditches are more closely spaced than the necessary width of such a riparian area.

The use of controlled drainage to effectively reduce N losses from Coastal Plain landscapes, either through decreases in NO_3-N concentrations or decreases in outflow, has been effectively demonstrated by researchers in North Carolina (Gambrell et al. 1975b; Gilliam et al. 1979; Amatya et

Figure 5
Total N concentration, NO3-N concentration, and organic N concentration as a function of drainage outflow for the seven ditches.



al. 1998; Burchell et al. 2005; Skaggs et al. 2005). Controlled drainage, while an effective tool in reducing N losses from this landscape, depends on very flat topography. The UMES study site (particularly ditches 2 and 3) is a candidate for controlled drainage management because the landscape is flat. Outflow that occurs between October and January, coinciding with the fallow season and representing 55% of the total outflow for ditches 2 and 3 during this study, could be easily managed with an elevated water table.

Biological curtains—e.g., wood chips in a trench that intercept groundwater lateral flow—are a more recent management technique that has been shown to effectively reduce NO_3-N concentration in tile outflow (Jaynes et al. 2006). These curtains should also be effective when placed adjacent to open ditches, provided that groundwater flow is intercepted laterally. Both the riparian buffer and biological curtain concepts depend on a situation in which the groundwater travels through the area of increased denitrification potential, and perhaps for the riparian buffer, size requirements for effective denitrification may exceed space availability near the ditch.

If groundwater flow pathways are parallel to the drainage tile or ditch, the riparian buffer or biological curtain may not intercept the groundwater and, consequently, have little or no impact on the NO_3 concentration. The importance of understanding the groundwater hydrology was demonstrated by Angier et al. (2005), who identified concentrated flow pathways, described as upwelling zones within a stream, that served as a bypass to a riparian buffer, thus negating the ameliorating impact of the riparian area. Riparian buffers, controlled drainage, and biological curtains all depend on an understanding of the landscape hydrology to implement an effective strategy in reducing N losses as NO_3-N in the groundwater.

Ditch 1, which upon visual observation might be considered similar to ditches 2 and 3, is a less likely candidate for effective N management using the strategies just described (controlled drainage, riparian buffer, or biological curtain). Because ditch 1 is shallower than ditches 2 and 3, a greater proportion of the water outflow occurred as surface runoff, and subsurface structures designed to ameliorate N losses will be ineffective in mitigating surface N export.

Management to minimize N losses from ditches 1, 5, 6, 7, and 8 depends on a slightly

different set of circumstances compared to those that were present for ditches 2 and 3. Ditches 1, 5, 6, 7, and 8 are shallow, so surface flow is the predominate contributor to total outflow. Additionally, the manure storage shed (figure 1) represents a point source that was implicated in contributing to the additional N losses observed for some of these shallow ditches, especially ditch 8. The most appropriate management response to the N losses from these ditches is to divert roof runoff directly to a ditch and then divert all other surface runoff from this area through a vegetative treatment system. Koelsch et al. (2006) recently completed a review of vegetative treatment systems, indicating that a vegetative treatment system can reduce pollutant load of surface runoff by as much as 99%. Because ditches 5, 6, 7, and 8 are shallow ditches and collect very little, if any, subsurface flow, controlled drainage, riparian areas, and biological curtains are unlikely to provide much remediation for N export from these ditches.

Summary and Conclusions

At the Maryland Eastern Shore site, which relies on a network of ditches to improve drainage for crop production, there were ditches that could be placed in two distinct management categories.

The shallow (<0.6 m [<2 ft]) ditches (1, 5, 6, 7, and 8) were primarily conduits for surface water, with N losses for ditches 1, 5, 6, and 7 (5.1 to 15.5 kg N ha^{-1} [4.6 to 13.8 lb N ac^{-1}]) comparable to losses observed for previous research on this type of agricultural landscape (Gambrell et al. 1975b; Staver and Brinsfield 1995). Total N loss for ditch 8 was 43.5 kg N ha^{-1} (38.8 lb N ac^{-1}), much greater than might be expected from a shallow ditch that is a conduit for only surface runoff. The primary management concern for these shallow ditches should be to minimize N losses from any point sources, so diverting roof and surface runoff from the manure storage shed area should be an effective strategy. Management practices designed to impact groundwater flow, such as controlled drainage or biological curtains, would be ineffective for these shallow ditches. Mitigating NO_3 in the outflow from ditches 1, 5, 6, 7, and 8, if not addressed with a vegetative treatment system, might be best managed with control structures in the downstream PDA ditch (>2 m [>6.6 ft] depth).

Ditches 2 and 3 were similar ditches with greater outflow (on a per area basis) than observed for the other ditches (figure 2), and N export was a direct consequence of the additional outflow observed in these ditches (figures 3 and 4). This difference should be considered in guiding management decisions for these ditches. Ditches 2 and 3 are candidates for more effective control of N losses through the use of controlled drainage, riparian areas, or biological curtains, all of which are intended to reduce the NO_3-N concentration of groundwater prior to entering the ditches or by reducing total water outflow. Selecting appropriate management strategies for mitigating N losses from agricultural lands will depend on understanding management techniques in the context of the landscape and hydrology.

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