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Managing natural processes in drainage ditches for nonpoint source nitrogen control

J.S. Strock, C.J. Dell, and J.P. Schmidt

Abstract: In watersheds dominated by agriculture, artificial drainage systems can efficiently and quickly transport excess water from agricultural soils. The application of more nitrogen (N) than a crop uses creates a surplus in the soil and increases the risk of N loss to the environment. We examine issues associated with agricultural N use, N transfer from artificially drained agricultural land to drainage ditches, N cycling within ditches, and options for management. Watercourses in agricultural watersheds often have high concentrations of N and are effectively N saturated. Numerous processes are involved in N cycling dynamics and transport pathways in aquatic ecosystems including N mineralization, nitrification, and denitrification. Flow control structures can lower N losses related to artificial drainage by increasing water retention time and allowing greater N removal. An ongoing study in Minnesota compares the impact of flow control structures on N losses from paired ditches with and without flow control. During the first year of observation, results were mixed, with lower N concentrations in nonstorm event samples from the ditch with the flow control structure, but no significant difference in annual total N load between the two ditches. Appropriate management of drainage ditches represents a potential opportunity to remove biologically available forms of N from drainage water through a combination of physical and biogeochemical processes.

Key words: ditch—nitrate—nitrogen—total nitrogen—water quality

Artificial drainage can increase crop yields, reduce risk of saturated soil at planting and harvest, and improve economic returns for crop producers in many regions of the United States (Pavelis 1987). Starting in the late 1700s, drainage in the United States has been improved by constructing open channel ditches (frequently by artificially deepening and straightening natural waterways) and by installing subsurface tiles that transfer excess water to ditches or natural waterways. In the regions with slowly permeable soils, like much of the Midwest, extensive networks of subsurface tile drains and vegetated ditches have been established. In other regions, such as the Delmarva Peninsula, which have highly permeable soils but also high regional water tables, improved drainage can be obtained using only open channel ditches.

Artificial drainage can considerably increase the amounts of sediment, nutrients, pathogens, and pesticides exported from agricultural fields to waterways (Gilliam et al. 1999; Randall and Goss 2001). One of the most significant water quality impairments within aquatic ecosystems is accelerated eutrophication caused by nutrient overenrichment, a problem often associated with agricultural production (USEPA 2002). Although many factors contribute to eutrophication, most attention has focused on the supply of carbon (C), nitrogen (N), and phosphorus (P). A comparison of N:P ratios among freshwater and estuary ecosystems indicated that freshwater ecosystems were

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P limited whereas estuaries were N limited (table 1) (Thomann and Mueller 1987). The total N and P discharge from the Mississippi River and its tributaries to the Gulf of Mexico is approximately 1.6×10^6 and 1.4×10^5 metric tons year^{-1} (1.8×10^6 and 1.5×10^5 tons year^{-1}) (Goolsby et al. 1999). The Upper Midwest, including the Upper and Middle Mississippi and Ohio River Basins, contribute about 70% of the total N and 60% of the total P discharged to the Gulf of Mexico. Indeed, the growth of the hypoxia zone in the Gulf of Mexico (Rabalais et al. 1994) has been linked to heavy precipitation in the Upper Midwest of the United States, along with loss of N from artificially drained soils used for row crop production (Turner and Rabalais 1994; Randall and Mulla 2001). Total N inputs to the Chesapeake Bay, which is also threatened with accelerated eutrophication and hypoxia, are approximately 9.1×10^4 metric tons year^{-1} (1×10^5 tons year^{-1}) (USGS 1995). An estimated 40% of N loadings to the Chesapeake Bay are attributed to agricultural sources (Magnien et al. 1995). Fortunately, there is considerable evidence that drainage management practices, such as controlled drainage and sub-irrigation, can reduce pollutant loads in landscapes favorable to their installation (Gilliam et al. 1979; Skaggs et al. 2005).

Given the importance of agricultural sources to nutrient loadings of impaired water bodies, one of the most critical challenges for agriculture in the United States is the development of agronomically, economically, and environmentally sensible N and P management strategies. This paper examines the issues associated with agricultural N use, N transfer from artificially drained agricultural land to vegetated ditches, and N cycling within vegetated ditches. The current state of research and options available for agricultural N management within vegetated ditches are presented.

Table 1
Limiting nutrients for various waters.

System	N:P ratio	Limiting nutrient
Rivers and streams	>>10	P
Lakes	>>10	P
Estuaries		
Freshwater region	>>10	P
Saline region	<<10	N

Source: Adapted from Thomann and Mueller (1987).

Table 2
Characteristics of undisturbed natural streams and channelized ditches.

	Natural stream	Channelized ditch
Channel characteristics		
Bankfull recurrence interval	Lower	Higher
Stream discharge	Lower	Higher
Stream velocity	Lower	Higher
Stream gradient	Lower	Higher
Width:depth ratio	Higher	Lower
Entrenchment ratio	Higher	Lower
Roughness of bed and bank	Higher	Lower
Geometry	Parabolic	Trapezoidal
Sinuosity	Sinuous	Linear
Floodplain	Present	Absent
Pools/riffles	Present	Absent
Habitat quality	Higher	Lower
Natural biodiversity	Higher	Lower

Source: Adapted from Brooks et al. (2003).

Agricultural Nitrogen in the Environment

There is justifiable concern about the delivery of N from agricultural sources into the environment. Nitrogen is one of the most mobile compounds in the soil-water-atmosphere system. Nitrate (NO_3^-) is the dominant form of N in the soil-water solution and is highly susceptible to leaching from landscapes with artificial drainage. A 20-fold increase in N fertilizer use in the United States over the last 50 years has led to increased crop production, but reported inefficient use of fertilizer N on row crops such as corn, ranging from 14% to 65% (Meisinger et al. 1985), allows a substantial portion of applied N to be lost to water and air (Keeney and DeLuca 1993; Tilman 1999). Nitrogen delivered to the soil-water-atmosphere system has been linked to soil acidification, eutrophication and hypoxia of coastal estuaries and the Gulf of Mexico, elevated levels of NO_3^- in drinking water, stratospheric ozone depletion, and greenhouse gas effects (Kinzig and Socolow 1994).

A variety of studies indicate that artificial drainage can result in elevated levels of NO_3-N from row crop systems regardless of fertilizer management practices. A drainage study in Minnesota indicated that soil-derived N (attributed to mineralization of N from organic matter) from fallow plots, that received no fertilizer and only periodic cultivation, averaged about 20 mg L^{-1} during an eight-year period (Randall 2000). In an earlier study of drainage from continuous corn in Minnesota, Gast et al. (1978) applied N fertilizer at 20, 112, 224, and 448 kg ha^{-1} (18, 100, 200, and 400 lb ac^{-1}). They reported no difference in flow-weighted mean NO_3-N concentrations during the first year of the study. During the last two years of the study, NO_3-N concentrations averaged 19, 24, 40, and 74 mg L^{-1}, respectively, demonstrating the increasing risk of NO_3^- export to water supplies with increasing fertilizer additions.

Nitrogen Cycling in Fluvial Systems

Although very little information exists on N cycling in drainage ditches, numerous studies have examined N uptake, retention, release, and export from relatively undeveloped and undisturbed fluvial systems (Meyer et al. 1988; Hamilton et al. 2001; Peterson et al. 2001; Dodds et al. 2002). These undeveloped and undisturbed systems differ from fluvial systems found in moderate to highly agricultural watersheds. By comparison, natural streams tend to be more stable and have higher biologic diversity than artificially deepened, straightened ditch channels

Table 3
Examples of nitrogen loss from headwater streams and ditches.

State	Land use	Watercourse	Discharge ($L\ s^{-1}$)	NH_4^+ ($\mu g\ L^{-1}$)	NO_3^- ($\mu g\ L^{-1}$)	Reference
Kansas	Tall grass prairie	South Kings Creek	41.8	2.4	3.1	Peterson et al. (2001)
Minnesota	Agricultural	County Ditch 5	3,700	–	5,000 to 7,000	Magner et al. (2004)
Ohio	Northern deciduous forest	E. Fork Little Miami River	849	31.1	543.5	Peterson et al. (2001)
Tennessee	Southern deciduous forest	Walker Branch	9.8	2.7	15.6	Peterson et al. (2001)
Oregon	Northern coniferous forest	Mack Creek	55.8	3.6	54.3	Peterson et al. (2001)
New Mexico	Southern coniferous forest	Gallina Creek	4.2	5.2	7.5	Peterson et al. (2001)

(table 2). The N concentration of undeveloped headwater streams is considerably lower than found in ditches draining agricultural watersheds (table 3). Even so, considerable insight can be gained from these systems in understanding critical processes controlling N cycling in drainage ditches.

Nitrogen cycling in fluvial ecosystems requires consideration of N inputs and their fate. Because biological N fixation in flowing waters is generally limited (Vitousek et al. 2002), it can be assumed most N entering fluvial systems comes from terrestrial sources. The major controls on N supply to a watercourse (natural or man-made) include land use practices, landscape vegetation, atmospheric loading, soil processes, and hydrology including artificial drainage. The defining characteristic of fluvial systems is the unidirectional flow of water occurring downstream. The N status of a watercourse will largely depend on N storage, uptake, release, and exchange by abiotic and biotic processes, within and between sediments and the water column, and subsequent transport longitudinally downstream (Bernot and Dodds 2005). Because subsurface sediments are biologically active, they may be sinks or sources of N, depending on the magnitude of processes generating, releasing, or immobilizing N (Keeney 1973).

Biologically available forms of N (ammonium [NH_4^+], nitrate [NO_3^-], and organic N) are subject to an extensive combination of physical and biogeochemical processes and transformations in aquatic systems (Novotny 2003) (figure 1). Once in a fluvial system, N undergoes numerous chemical and biological transformations as it partitions between dissolved and particulate phases, sediment and water column, and biotic and abiotic environments (figure 2). Ammonium-nitrogen may be adsorbed to negatively charged surfaces of fluvial sediments or organic matter where most is held in an exchangeable form. Nitrogen can also be retained in the system as organic N through the biological assimilation of both NH_4^+ and NO_3^-, with NH_4^+ being the preferred form for assimilation by bacteria, biofilms, and aquatic plants. Organic N dissolved or suspended in the water column or associated with organic material deposited in stream channel sediment will, over time, be converted back to NH_4^+ through mineralization. Under aerobic conditions, mineralized NH_4^+ is rapidly nitrified to NO_3^-.

The processes of nitrification and denitrification are influenced by the oxidation-reduction (redox) status of the system. Nitrification is important because it involves the oxidation of immobile NH_4^+ to highly mobile NO_3^-, which is easily transported downstream. Nitrification is also a key process linked to productivity in aquatic systems and, in systems where N concentrations are low, can control denitrification rates (Bernot and Dodds 2005). Nitrification occurs primarily on the oxidized surface of bottom sediment and minimally in the water column. Oxidation of NH_4^+ to NO_3^- serves as an energy source for microorganisms. Nitrification can also have a regulatory effect on pH; as nitrification rates increase, pH decreases because nitrification produces hydrogen ions. As pH decreases, nitrification rates decrease. Denitrification involves the reduction of NO_3-N to dinitrogen gas (N_2) or nitrogen oxides (i.e., nitrous oxide, N_2O). In the presence of NO_3^-, the rate and extent of denitrification are controlled by the supply of oxygen and available energy (i.e.,

Figure 1
Processes involved in nitrogen cycling dynamics and transport pathways in a vegetated open ditch.

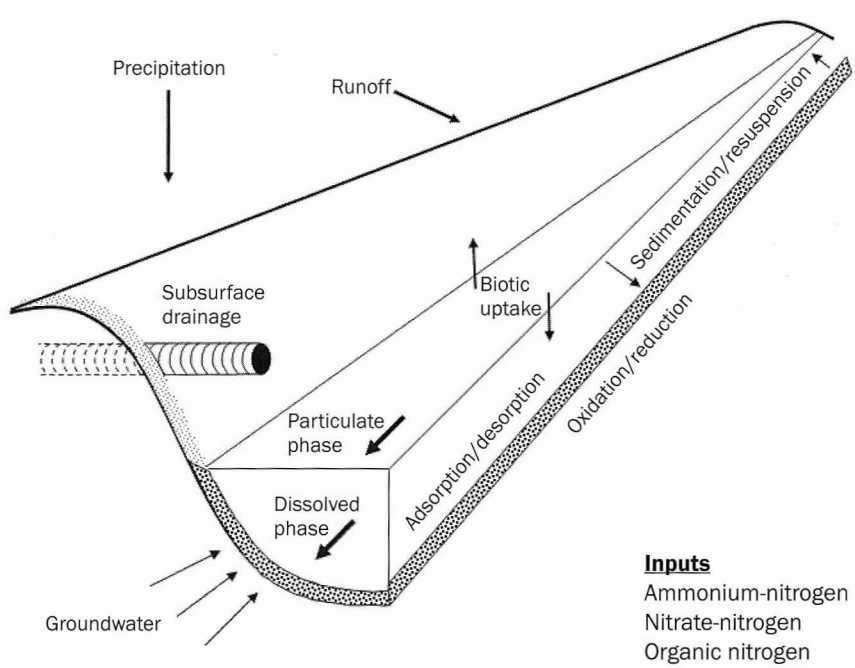
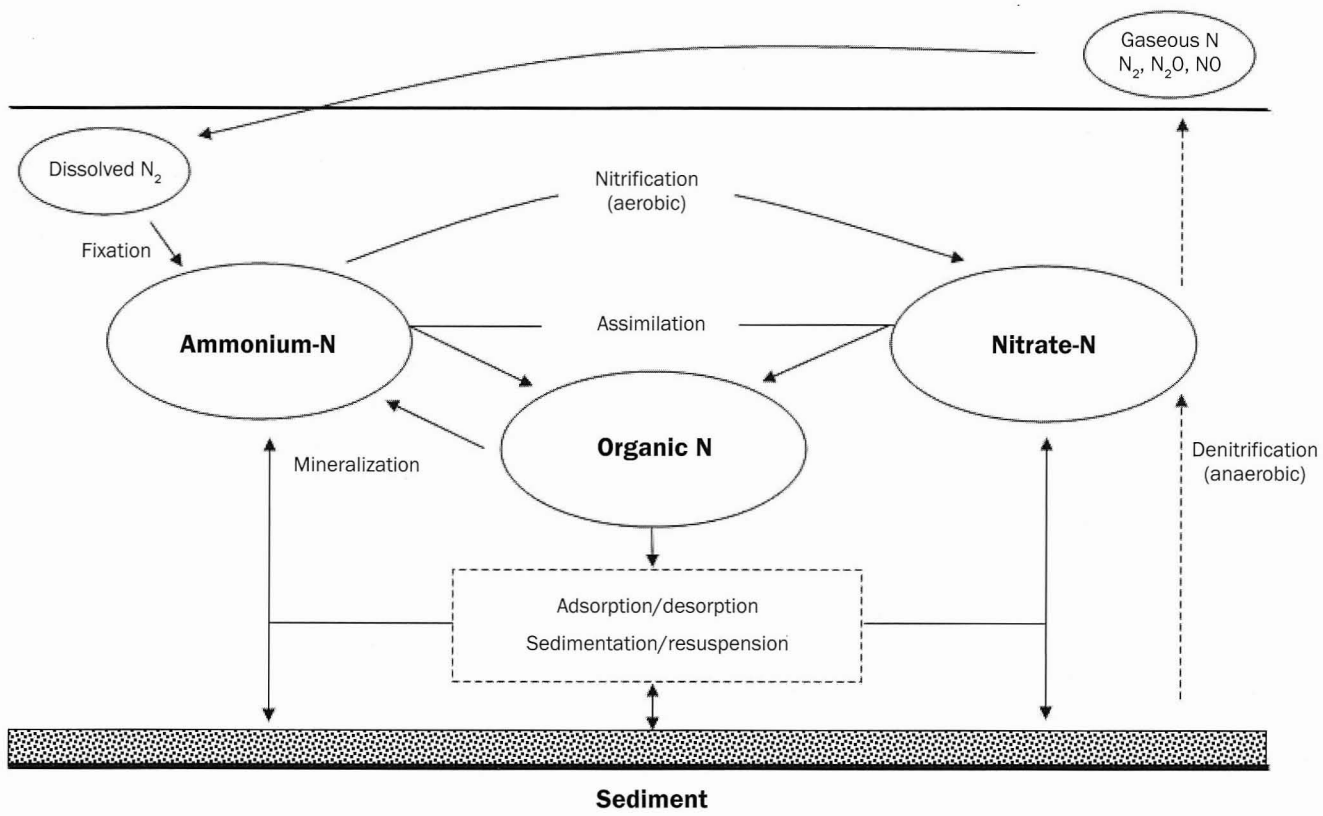


Figure 2
Mechanisms for nitrogen cycling dynamics, retention, uptake, and removal in aquatic systems.



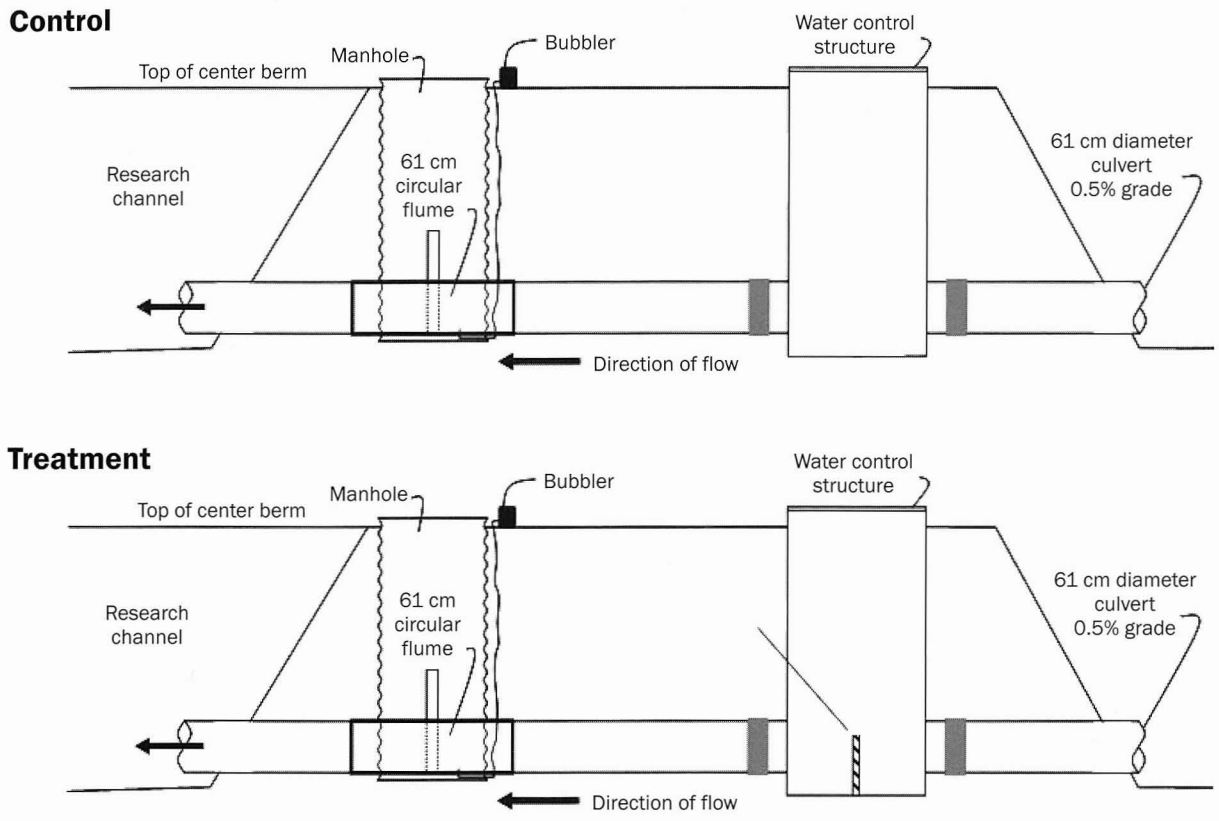
organic carbon). The rate of denitrification is influenced by pH, being much slower in acid than in neutral or alkaline systems (Keeney 1973). Denitrification reaches maximum rates of 100 to 500 mg N m^{-2} d^{-1} (2.05×10^{-5} to 1.02×10^{-4} lb N ft^{-2} d^{-1}) regardless of NO_3^- concentrations (Christensen et al. 1990; Kemp and Dodds 2002), therefore denitrification can remove only a limited portion of the NO_3^- in N-saturated fluvial systems. The rate of transport of NO_3^- into sediment or the rate of its formation within the sediment determines the rate of denitrification (Kamp-Nielsen and Andersen 1977). Nitrate may also be removed from sediment by biotic dissimilatory NO_3^- reduction that results in the reduction of NO_3^- to NH_4^+ (Tiedje 1988). Ammonium is then available to be assimilated and incorporated into amino acids or used for other metabolic purposes. The ammonification pathway results in microbial excretion of NH_4^+ into the environment where it is available as a substrate for nitrification. Compared to denitrification, dissimilatory nitrate reduction is a less significant process for nitrate reduction.

Alexander et al. (2000) concluded that the rate of N removal in streams was inversely related to channel size. Their modeling results indicated that small streams and rivers influence N export to large rivers because small watercourses have more contact and exchange of N between water and sediment. Therefore, N removal through denitrification and in-stream storage of particulate N is more effective for small watercourses. Alexander et al. (2000) defined small streams as streams with mean stream flow rates less than 28.3 m^3 s^{-1} (989 ft^3 s^{-1}) and an in-stream N loss rate coefficient of 0.455. The rate coefficient represents the rate of in-stream N loss per unit of water travel time. Expressed as a percentage, the rate coefficient 0.455 represents 45.5% removal of N per day of water travel time.

In agricultural landscapes, low order streams are often channelized and straightened (ditched) to promote field drainage. For example, there are more than 43,500 km (27,000 mi) of vegetated open-ditches in Minnesota. During winter and summer baseflow conditions, ditches function like a linear wetland, biologically and physically, with relatively long hydraulic residence times. However, during wet seasons, ditches function more like fluvial, transport-dominated systems. In particular, a ditch must transport and distribute sediment similarly to a natural stream to effectively remove water from agricultural landscapes. Maintenance activities disrupt natural channel forming processes and any biological communities associated with the open-ditch system. Very little is known about the nutrient cycling dynamics of ditches. Vegetated open ditches may develop natural nutrient (N and P) and sediment removal capacity. The nutrient removal capacity of ditches may be further enhanced under alternate management strategies.

Nitrate removal from natural watercourses as well as from artificially straightened and channelized drainage systems occurs through biotic uptake of nitrogen into plant and/or microbial cellular tissue, denitrification, and/or dissimilatory nitrate reduction. Artificial subsurface drainage systems that outlet into natural and constructed channels export excess water and nutrients into these systems. Natural watercourses have biologically and hydrologically active flood-

Figure 3
Diagram of structures in control and treatment channels used in ditch management.



plains whereas ditches are devoid of active floodplains as a result of maintenance. These floodplains contribute to sediment and nutrient removal. Ditches are periodically maintained to remove woody vegetation, sediment deposits, and to repair unstable bank slopes (Powell et al. 2007). When sediment is excavated from ditch channels, vegetation is simultaneously removed. These disturbances result in loss of substrate, carbon, and microbial communities and will have considerable detrimental impact on nutrient cycling process. In particular, nitrogen removal by denitrification will likely be diminished for some period after maintenance activities. Pre-maintenance conditions will be more stable than post-maintenance conditions until the channel is restabilized and revegetated.

Management of Nitrogen Export from Ditch-Drained Systems

The use of agronomic practices that maximize crop utilization of applied N is a key component for the minimization of N losses for drained agricultural systems. Splitting fertilizer applications, pre-side dress soil NO_3^- testing, using nitrification inhibitors, adjusting fertilizer application rates to account for the contribution of previous legume crops and the mineralization of organic N, and a range of other practices can help to optimize crop N utilization. Since the N use efficiency of row crops seldom exceeds 65% (Meisinger et al. 1985), managing fertilizer inputs is only a partial solution to limit N losses from drained systems.

An undeveloped, potentially worthwhile strategy for improving water quality is the use of vegetated open ditches for treatment of agricultural runoff. Nutrient and sediment load reduction in vegetated open ditches may be achieved through a variety of treatment methods and/or ditch modifications. Current research in Ohio suggests the potential exists for using vegetated ditches as best management practices (BMPs) for mitigating potential agricultural contaminants (Powell et al. 2007). Using an engineering approach, it has been hypothesized that a compound open-ditch channel would create a linear zone of plants and soil within open-ditch geometry for enhanced denitrification potential.

Controlled drainage has been shown to reduce N losses via drainage (Gilliam et al. 1979; Skaggs et al. 2005). The installation of flow control structures within ditches and/or at tile outlets and the insertion of flash boards restrict flow and alter the water table height in the ditch and adjacent soil. Increasing retention time of water within the ditch or the soil profile promotes sedimentation and permits greater removal of NO_3^- through denitrification. Gilliam et al. (1979) demonstrated that a 50% reduction in NO_3^- export from ditches on the Coastal Plain of North Carolina could be achieved by controlling flow.

Reduction curtains and infiltration filters have been proposed to decrease N loads entering ditches (Jaynes et al. 2004; Greenan et al. 2006). With these approaches, trenches are filled with wood chips or other organic materials to create a "bioreactor" where enhanced denitrification can occur. If properly positioned to intercept ground or surface water, the denitrification within curtain or filters can remove substantial amounts of NO_3^- before the water flows into the ditch. Riparian and other vegetated buffer strips

Table 4
Summary of monitoring periods and number of samples collected during 2006.

Monitoring period	Dates	Storm events	Precipitation (mm)	Control (no. of samples)	Treatment (no. of samples)
Calibration	March 23 to June 30, 2006	5	307	60	74
Treatment	July 1 to August 11, 2006	2	96	11	13

Table 5
Annual runoff volume and nitrogen loads during 2006.

Monitoring period	Runoff (cm)		NH_4-N (kg ha^{-1})		NO_3-N (kg ha^{-1})		Total N (kg ha^{-1})	
	Control	Treatment	Control	Treatment	Control	Treatment	Control	Treatment
Calibration	10.4	12.6	0.1	0.2	22.6	23.8	25.0	24.8
Treatment	0.6	0.8	0.01	0.01	1.1	1.7	1.2	1.8

are also known to be active zones of denitrification which can be effective in reducing N loads entering ditches and other surface waters (Groffman et al. 1992; Hill 1996; Vidon and Hill 2004). While the necessary width may limit the situations where riparian buffers will be practical, they offer an additional alternative to decrease N losses.

Vegetated Open-Ditch Research: Minnesota Case Study

Strock et al. (2000) describe the development of a vegetated open-ditch research facility in the glacial till plain within the Northern Corn Belt Plain region of Minnesota. This site represents physiographic features and land use typical of southwest and south-central Minnesota. Average annual precipitation at the site is 670 mm. Average annual temperature is 7°C (45°F), with monthly extremes ranging from 21°C (70°F) in July to -9°C (16°F) in January. Two side-by-side, 200 m (656 ft) long, vegetated open ditches receive surface runoff and subsurface drain flow from 113 ha (280 ac). These experimental channels discharge into the Cottonwood River. This experimental site was established to identify the effectiveness of open-ditch management strategies to increase water storage and decrease sediment and nutrient discharge from an agricultural landscape. Water flow in the channel is seasonal, with higher flows from April through June when spring snowmelt combines with spring rainfall and seasonally high subsurface drainage flow. The contributing watershed comprises 74% cropland (row crops), 20% pasture, and 6% farmstead.

The experimental design consists of conventional (typical) and improved management (targeted drainage ditch management strategy) treatments and two periods of study—a calibration and treatment period. The calibration period started in 2004 and the initial treatment period began in 2006. The initial focus of treatment was drainage water retention and N reduction. Currently, a flow control structure is being used to retain water in the treatment ditch while there is no flow control in the conventionally managed ditch (figure 3).

Identical instrumentation for water sampling and flow measurement was installed on the upstream and downstream ends of the open-ditch channels. Campbell Scientific Inc., DB1 liquid level sensors were used to continuously measure water level and record it at 5 min intervals using dataloggers. Each site was equipped with an ISCO Inc. 3700 auto-sampler that was triggered by a datalogger during storm events to collect water samples. Sampling was initiated by a 2.54 cm (1 in) rise in stage within a 30 min period. Samples (1 L, 0.26 gal) were collected using a flow proportional sampling method. Grab samples were collected once per week. Monitoring was conducted during ice-free periods between April and November. A Texas Electronics Inc. 0.25 mm (0.01 in) tipping bucket rain gauge was used to record rainfall at the site.

Water samples were gathered within 24 h of collection and immediately frozen until prepared for analysis. Water samples were filtered upon thawing and analyzed for NO_3-N and NH_4-N. Nitrate-N was analyzed using colorimetric analysis by the cadmium reduction method using a Lachat Quickchem 8,000 Flow Injection Analyzer (Hach Company, Loveland, Colorado) at 520 nm (Wendt 2000). Data are reported for NO_3-N + nitrite-N as NO_3-N, as the concentration of nitrite-N was assumed negligible. Ammonium-N was measured with the Berthelot reaction method (Diamond 2001) at 630 nm using a Lachat. For total N, an unfiltered 15 mL (0.5 oz) aliquot of a water sample was digested with potassium persulfate and sodium hydroxide in a 40 mL (1.35 oz) vial (Clesceri et al. 1998). Samples were digested for 50 minutes in an autoclave at 121°C (250°F) at 117 kPa (17 psi). The digested samples were analyzed colorimetrically for total N as NO_3^- at 520 nm using a Lachat. Total nutrient flux through the drainage system was calculated by multiplying nutrient concentration for each sample by total calculated flow for the same

Table 6
Annual flow-weighted mean nitrogen concentration in 2006.

Monitoring period	NH_4-N (mg L^{-1})		NO_3-N (mg L^{-1})		Total N (mg L^{-1})	
	Control	Treatment	Control	Treatment	Control	Treatment
Calibration	0.10	0.14	21.8	18.9	24.2	19.7
Treatment	0.16	0.14	19.3	20.3	20.7	21.8

time period. Flow-weighted mean nutrient concentration was calculated by dividing the total nutrient flux for the period of interest by total flow volume.

Monitoring conducted during 2006 included three calibration months and 1.5 treatment months. During this time period, there were a total of 7 storm events in which water samples were collected for water quality analysis (table 4). A combined 134 and 24 samples were collected and analyzed during the 2006 calibration and treatments periods, respectively. Additional samples will be collected and analyzed during storm and non-storm periods during subsequent treatment years.

Total precipitation during the monitoring period was 6% below the 30-year normal observed at the Southwest Research and Outreach Center weather station (<1 km from the site). Flow through both channels ceased in mid-August. August through December was abnormally dry during 2006. The majority of storm events observed during 2006 occurred from April to June (figure 4). During this time, the risk of runoff is high because soils generally have low residue levels due to tillage for seed bed preparation, evaporation and transpiration are low, and the crop canopy is not well established. Peak flow occurred on June 5 at a rate of 0.25 $m^3 s^{-1}$ (8.8 $ft^3 s^{-1}$).

During 2006, large differences in annual N loads between the calibration and treatment periods (table 5) were attributed mainly to the difference in discharge volume, with smaller differences associated with declining nutrient concentration (figure 5). Flow volume during the calibration period was 16 to 17 times greater than during the treatment period. During the calibration period, NH_4-N and NO_3-N loads were slightly greater in magnitude from the treatment channel than the control channel. This was attributed to the June 15 storm event. While drainage flows from both channels responded quickly to rainfall, recession of drainage from peak flow occurred more slowly for the treatment channel (figure 4). Increased drainage volume resulted in higher NH_4-N and NO_3-N loads from the treatment compared to the control channel (table 5). There was no difference in annual total N load between the two channels during the calibration period. Similar results for annual N loads were observed during the treatment period between the two channels. Although

Figure 4
Magnitude and distribution of flows for control and treatment channels during 2006.

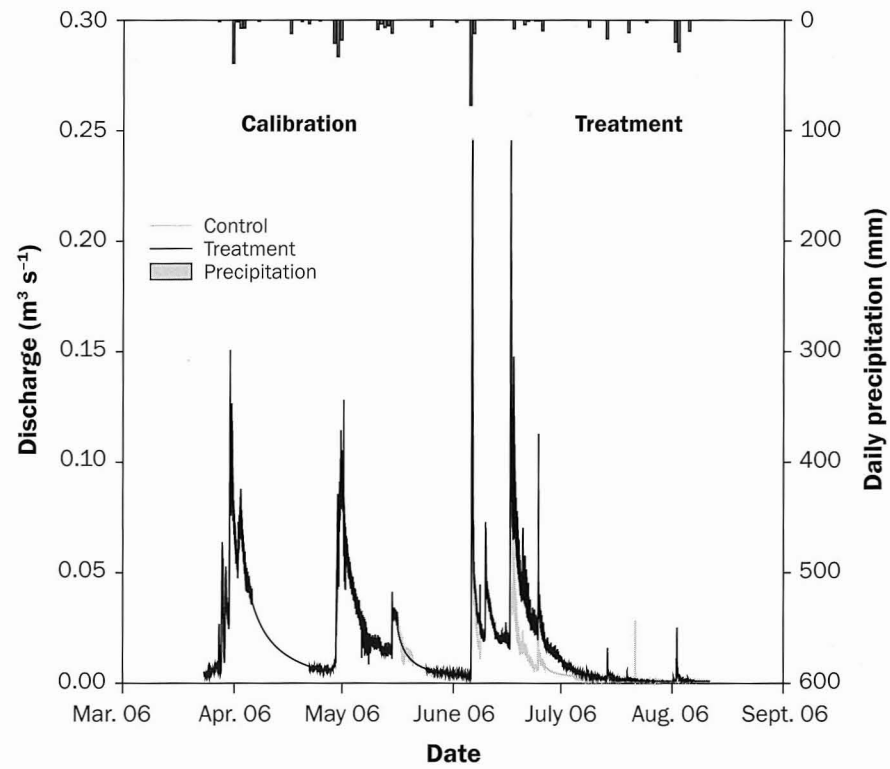
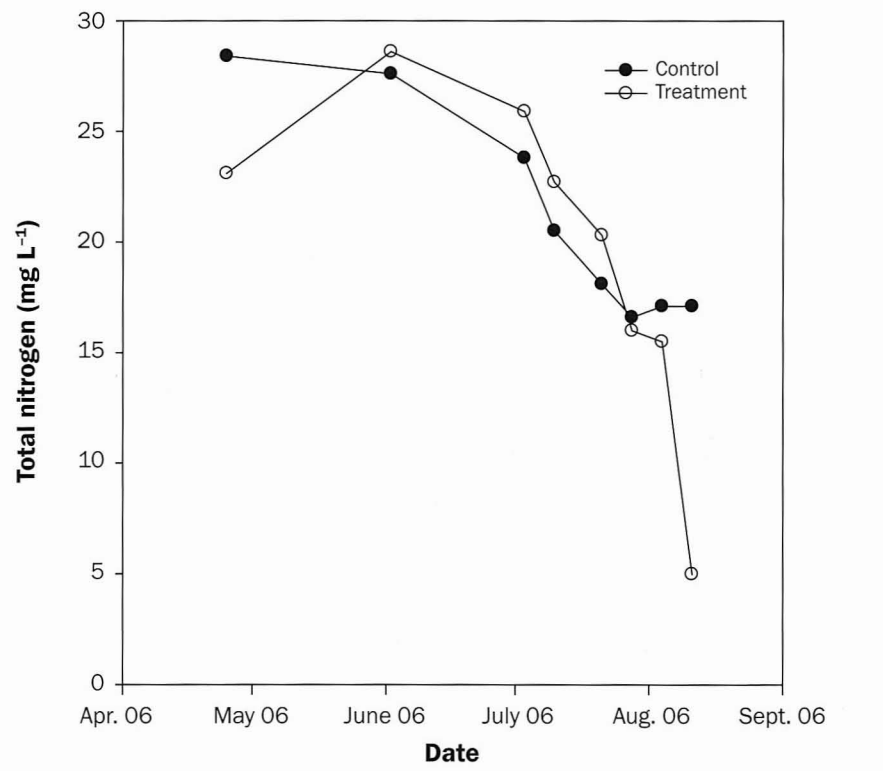


Figure 5
Distribution of total nitrogen concentrations in control and treatment channels from nonstorm events during 2006.



flow volumes were similar, slightly more water flowed through the treatment channel than the control channel, resulting in higher NO_3-N and total N loads from the treatment compared to the control channel (table 5).

Annual flow weighted mean N concentrations were variable between the two channels and the two monitoring periods (table 6). Analysis of nonstorm event samples indicated a continuous decline in total N concentration during the monitoring period (figure 5), which began in early July. Total N concentration in the treatment channel for August decreased 71% compared to the control channel. Future results are expected to show improved performance of the treatment channel over the control channel in reducing nitrogen load and concentration through a combination of flow and nitrogen concentration reduction.

Summary and Conclusions

The challenge of N management is to develop strategies that satisfy the food, feed, fiber, and energy demands of the world's growing population while also protecting human and ecosystem health. Environmental concerns such as eutrophication and greenhouse gas emissions demand the development of integrated solutions that incorporate atmospheric, terrestrial, and aquatic elements of landscapes. There is an immediate and continuing need for agricultural practices and management systems to reduce nutrient and sediment losses from agricultural lands. Most states have already developed a long list of current water resource use impairments and additional water quality criteria (e.g., total maximum daily loads [TMDLs]), which, when developed and implemented, will add to this list of impairments. A combination of agronomic, ecological, and engineering approaches to mitigate nonpoint source pollution from agricultural sources are being developed and implemented across the United States.

Managed vegetated ditches and flow control structures, in conjunction with improved N fertilizer management, offer opportunities for reducing N export from artificially drained agroecosystems. Careful placement of riparian zones and other vegetated buffers can effectively reduce nutrient loading into ditches, and the installation of reduction curtains and infiltration filters are potential alternatives to promote chemical or biological nutrient reduction when vegetative buffers alone are not sufficient to control nutrient loading to waterways. Innovative ditch designs, such as the two-stage design of Powell et al (2007), have the potential to reduce the need for maintenance and subsequent disturbance and allow the nutrient storage and removal capacities of natural waterways to develop in man-made ditches. How well will vegetated open-ditch management work? It is too early to tell how much impact vegetated ditch management will have on reducing N, P, and sediment loads, but the distribution of drainage during the year will make management of the system a critical issue.

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