

In Situ Bioreactors and Deep Drain-Pipe Installation to Reduce Nitrate Losses in Artificially Drained Fields

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Nitrate in water removed from fields by subsurface drain ('tile') systems is often at concentrations exceeding the 10 mg N L⁻¹ maximum contaminant level (MCL) set by the USEPA for drinking water and has been implicated in contributing to the hypoxia problem within the northern Gulf of Mexico. Because previous research shows that N fertilizer management alone is not sufficient for reducing NO₃ concentrations in subsurface drainage below the MCL, additional approaches are needed. In this field study, we compared the NO₃ losses in tile drainage from a conventional drainage system (CN) consisting of a free-flowing pipe installed 1.2 m below the soil surface to losses in tile drainage from two alternative drainage designs. The alternative treatments were a deep tile (DT), where the tile drain was installed 0.6 m deeper than the conventional tile depth, but with the outlet maintained at 1.2 m, and a denitrification wall (DW), where trenches excavated parallel to the tile and filled with woodchips serve as additional carbon sources to increase denitrification. Four replicate 30.5- by 42.7-m field plots were installed for each treatment in 1999 and a corn-soybean rotation initiated in 2000. Over 5 yr (2001–2005) the tile flow from the DW treatment had annual average NO₃ concentrations significantly lower than the CN treatment (8.8 vs. 22.1 mg N L⁻¹). This represented an annual reduction in NO₃ mass loss of 29 kg N ha⁻¹ or a 55% reduction in nitrate mass lost in tile drainage for the DW treatment. The DT treatment did not consistently lower NO₃ concentrations, nor reduce the annual NO₃ mass loss in drainage. The DT treatment did exhibit lower NO₃ concentrations in tile drainage than the CN treatment during late summer when tile flow rates were minimal. There was no difference in crop yields for any of the treatments. Thus, denitrification walls are able to substantially reduce NO₃ concentrations in tile drainage for at least 5 yr.

NONPOINT source contamination is a major surface water quality concern in the Midwest cornbelt (Humenik et al., 1987). The 1992 national water quality inventory (USEPA, 1992) noted that in the rivers studied, 72% of the water quality problems were attributed to agriculture. Plant nutrients have been identified as contaminants of surface water throughout the Midwest (Baker, 1988; Thurman et al., 1992; USEPA, 1992; Goolsby and Battaglin, 1993). Increased NO₃ loading in the Mississippi River has been linked to the spread and increased severity of hypoxia within the northern Gulf of Mexico (Rabalais et al., 1996).

Nitrogen, in the form of NO₃, contributes to surface water contamination in the Midwest primarily from the discharge of drainage water and shallow groundwater. Numerous studies have shown significant edge-of-field losses of NO₃ (Benoit, 1973; Logan et al., 1980; Baker and Johnson, 1981; Bergström, 1987; Kanwar et al., 1988; Drury et al., 1996; David et al., 1997; Goolsby et al., 1999; Jaynes et al., 2001) and that the primary pathway for these losses is discharge from subsurface drains (tiles) that are common across the Midwest cornbelt (Zucker and Brown, 1998). Our studies of a 5130-ha watershed in central Iowa (Jaynes et al., 1999) showed flow-weighted NO₃ concentrations in field and district drains were often greater than the 10 mg N L⁻¹ maximum contaminant level (MCL) for drinking water set by the USEPA. Yearly NO₃ losses from this predominantly agricultural watershed ranged from 4 to 66 kg N ha⁻¹.

Attempts to reduce NO₃ concentrations in field drains have focused on nitrogen rate, placement, and timing issues. However, Baker et al. (1975) and Gast et al. (1978) showed that even under low N-fertility management, NO₃ concentrations in drainage water often exceeded 10 mg N L⁻¹. These observations were confirmed by Jaynes et al. (2001) who found that lowering N fertilizer rates by 67% still did not consistently result in NO₃ concentrations lower than 10 mg N L⁻¹ in drainage water, while significantly reducing corn yields. Insensitivity of NO₃ concentrations in drainage water to fertilizer rate was also cited by Keeney and DeLuca (1993) who found ~10% increase in NO₃ levels in the Des Moines River between 1945 and the 1980s despite a 10-fold increase in N fertilizer use during this period. They concluded that intensive agriculture as a whole rather than just N fertilizer use was the major source of NO₃ in water. Thus, while fine-tuning N fertilizer application for corn production has yielded reductions in NO₃ concentrations in drainage waters, it seems that this approach

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Abbreviations: CN, conventional; DT, deep tile; DW, denitrification wall; EC_a, apparent electrical conductivity; MCL, maximum contaminant level; UAN, urea ammonium nitrate; USEPA, United States Environmental Protection Agency.

alone will not suffice in lowering drainage water concentrations below the 10 mg N L⁻¹ MCL (Dinnes et al., 2002).

Because N fertilizer management alone cannot reduce NO₃ contamination sufficiently, additional methods of NO₃ removal from subsurface drainage water are needed. Numerous methods for removing NO₃ from water have been identified including ion exchange, biological denitrification and assimilation, chemical denitrification, reverse osmosis, electro dialysis, and catalytic denitrification (Kapoor and Viraraghavan, 1997). Of these, only biological denitrification seems practical for permanently removing NO₃ from nonpoint source waters.

Naturally occurring NO₃ concentration gradients in shallow groundwater below agricultural fields can be substantial. Patni et al. (1998) measured decreasing NO₃ concentrations with depth in the water table below corn fields in Ontario, Canada. Similarly, observations of shallow groundwater below fields in central Iowa showed a dramatic decrease in NO₃ concentrations below the water table surface (Hatfield et al., 1995; Cambardella et al., 1999). Placing drainage pipe at the depth of lower NO₃ concentrations may result in lower NO₃ losses in tile drainage.

Presumably, the observed decrease in NO₃ concentrations with depth below the water table is due to denitrification removing NO₃ from the groundwater. For example, Robbins and Carter (1980) noted that denitrification in a silt loam soil in south central Idaho reduced NO₃ concentrations in water leaving the root zone by about 10 mg N L⁻¹ before entering subsurface drains. While denitrification can remove significant amounts of NO₃ before water drains through subsurface drainage systems, only a few studies have looked at the relationship between drainage system design and NO₃ concentration in the drainage water. Calvert and Phung (1971) compared NO₃ losses from a tile that was either open ended or had an elbow on the outflow end that allowed outflow only after the tile was filled with water—effectively keeping the tile below the water table surface. They found slightly lower NO₃ concentrations in drainage from the submerged tiles vs. the free-flowing tiles in nearly all months.

Gilliam et al. (1979) investigated controlled drainage systems for reducing NO₃ losses in drainage. They found no indication of increased denitrification in well-drained soils when the tiles were kept submerged below the water table by installing flashboard riser-type water level control structures. Conversely, in a poorly drained soil, NO₃ concentrations decreased even though the oxidation–reduction potentials in the soil showed little change. They attributed the decrease in NO₃ concentrations not to increased denitrification above the tile, but to changes in the flow path of water entering the drains. Submerging the tiles resulted in increased movement of water into and through deeper soil horizons, where denitrification was known to occur and NO₃ concentrations were lower, before entering the tiles. Keeping the tiles submerged below the water table may enhance denitrification (Kliewer and Gilliam, 1995), although this was not observed by Gilliam et al. (1979) in their field study in North Carolina.

While denitrification in the subsoil can be substantial under natural conditions, it has often been found to be limited by the availability of C for denitrifying bacteria (Yeomans et al., 1992). Sotomayor and Rice (1996) found higher NO₃ concentrations,

denitrifier populations, and denitrification potential in a soil under cultivation vs. grassland, but that denitrification was severely limited by lack of available C in the subsoil of the cultivated site. This is not surprising given the usual decrease in organic C with depth in most soils and the findings by Siemens et al. (2003) that dissolved organic matter that leaches to shallow water tables has limited bioavailability and does not support denitrification. Thus, much work has been done on designing ways to increase natural denitrification rates (McCleaf and Schroeder, 1995; Reising and Schroeder, 1996; Shanableh et al., 1997). Most approaches have used a supplemental carbon source such as glucose (Obenhuber and Lowrance, 1991), sucrose (Sison et al., 1995), ethanol (Weier et al., 1994), acetic acid (Constantin and Fick, 1997), methane (Thalasso et al., 1997), or vegetable oil (Hunter, 2001) to stimulate denitrification and would require a high level of management for in-field or edge-of-field treatment of subsurface drainage water.

Solid carbon sources have also been tested and would appear to be more amenable to application in the field (Williford et al., 1969). These sources have included peat (Kao and Borden, 1997), pine bark, and almond and walnut shells (Diaz et al., 2000). Volokita et al. (1996) used shredded newspaper as the only carbon source in laboratory columns to obtain nitrogen removal rates from 0.056 to 0.875 mg g⁻¹ newspaper d⁻¹. Blowes et al. (1994) used a fixed bed bioreactor filled with a sand, tree bark, woodchips, and leaf compost mixture to treat drainage water at the outlet of a tile. Over a year, a 200-L bioreactor was able to remove nearly all NO₃ from a 10 to 60 L d⁻¹ discharge of field drainage water containing 3 to 6 mg N L⁻¹ of NO₃.

Robertson and Cherry (1995) demonstrated the NO₃ removal potential of a constructed in situ bioreactor. They filled a 0.6-m-wide trench that extended 0.75 m below a shallow water table with sand containing 20% v/v coarse sawdust and measured the concentration of NO₃ in groundwater before and after flowing through the mixture. Very high NO₃ concentrations (57–62 mg N L⁻¹) were reduced to much lower concentrations (2–25 mg N L⁻¹) in groundwater passing laterally through the bioreactor. They attributed the removal of NO₃ to heterotrophic denitrification with the sawdust serving as the labile carbon source and estimated that this “denitrification wall” would have an effective lifetime of from 20 to 200 yr. Similarly, Schipper and Vojvodic-Vukovic (1998) found that a denitrification wall constructed of sawdust mixed with soil reduced NO₃ concentrations from 5 to 16 mg N L⁻¹ to less than 2 mg N L⁻¹ in shallow groundwater passing through it. They too attributed the removal process to denitrification and measured denitrification rates of ~100 mg N m⁻³ d⁻¹ after 5 yr of operation (Schipper and Vojvodic-Vukovic, 2001).

Use of readily available, low value solid carbon sources may have applicability to in situ field treatment systems. Solid organic carbon sources could be added to fields to increase denitrification. Ideally they would be incorporated below the root zone throughout the field to capture and remove excess NO₃ leaching below the root zone. However, a more practical approach would be to follow the concept of Robertson and Cherry (1995) and add the carbon sources as backfill or mixed with soil as backfill adjacent to tile drains. This would in essence create denitrification walls on both sides of the tile that would continuously remove NO₃ from

water before it entered the tile. In a laboratory study, Greenan et al. (2006) studied four readily available organic materials—woodchips, woodchips amended with soybean oil, cornstalks, and cardboard—as carbon substrates for supporting denitrification. They found that all carbon sources tested stimulated NO₃ removal from solution with removal over a 180-d period greater for cornstalks and least for woodchips. However, the removal rates for the woodchip material were more consistent throughout their experiment and they concluded that NO₃ removal would continue longer with woodchips than with the other materials.

Thus, there is evidence that NO₃ in subsurface drainage water can be removed by denitrification under anaerobic conditions by incorporation of additional carbon. Under favorable conditions, keeping tiles submerged may also lead to lower NO₃ concentrations in tile drainage. While the literature seems promising regarding these approaches, research is needed to determine the potential removal rates in agricultural fields over the short and long term. This article reports results from a 5-yr experiment where these approaches were tested in a cropped field.

Materials and Methods

A research site was selected on an Iowa State University research farm in central Iowa (42.04° N, 93.71° W) in 1999. Soils at the site are mapped primarily as Canisteo silty clay loam (fine-loamy, mixed, superactive, calcareous, mesic Typic Endoaquolls) and Nicollet loam (fine-loamy, mixed, superactive, mesic Aquic Hapludolls). These permeable soils are poorly to somewhat poorly drained because they are underlain by an unoxidized, low permeability till at about the 3-m depth. Field plots were laid out in a randomized complete block design with four replications. Ideally, the plots would have been assigned to blocks based on tile flow volume. However, the deep tile treatment had to be installed at the same time as the tile drains, precluding this approach. As an alternative, an electromagnetic induction survey of the field was conducted to measure the average apparent soil electrical conductivity (EC_a) of each plot (Jaynes, 1996). Blocks were assigned by grouping the plots by increasing EC_a with the assumption that EC_a was an effective surrogate measure for soil drainage class (Jaynes, 1996). Three treatments were established—a conventional or check treatment (CN), a deep tile placement treatment (DT), and a denitrification wall (DW). Three other treatments were also established for a total of 24 plots. Results for the other treatments are not presented here but are discussed in a companion paper (Kaspar et al., 2007).

Each plot was 30.5 m wide by 42.7 m long with a drainage pipe installed lengthwise bisecting the plot. The tile for the CN and DW treatments consisted of a perforated, 7.62-cm diameter corrugated pipe installed 1.2 m below the surface. The DT treatment had the same perforated pipe installed at 1.83 m below the surface or 0.6 m deeper than the control. The outlet for the deep tile was maintained at the 1.2-m depth so that drainage occurred only when the pipe was at least 0.6 m below the water table. Any preexisting drain pipe in the field was cut and blocked during installation of the new pipe. In addition, a 25.4-cm diameter drain pipe was installed around the perimeter of the site

Table 1. Planting and harvest dates for corn and soybean.

Year	Crop	Planting date	Harvest date
2000	corn	25 Apr.	12 Oct.
2001	soybean	10 May	27 Sept.
2002	corn	25 Apr.	30 Sept.
2003	soybean	12 May	30 Sept.
2004	corn	28 Apr.	4 Oct.
2005	soybean	6 May	30 Sept.

to reduce subsurface flow into the plots. A trench was excavated between rows of plots and a 12-mil-thick plastic sheet installed to a depth of 1.83 m to act as a flow barrier before backfilling the trenches. For the DW treatment, two trenches, 0.6 m wide by 1.83 m deep were excavated 3.05 m on either side of the tile and backfilled with woodchips obtained from an oak-pallet recycling center. The woodchips ranged from several mm to several cm in size. Surface soil was randomly mixed with the woodchips during backfilling to act as a microbial inoculant. Woodchips were backfilled to within 30 cm of the surface and then surface soil added to level the trench with the soil surface.

The cropping system in the field was a corn–soybean rotation (Table 1) with a high, but not unusual N fertilizer application rate for the area. Liquid urea-ammonium nitrate (UAN) was sidedressed in the corn growing years with a spoke-wheel fertilizer injector (Baker et al., 1989) at 224 kg N ha⁻¹ on 14 Apr. 2000 and 30 May 2002 and 217 kg N ha⁻¹ on 21 May 2004. Phosphorus and potassium fertilizers were surface applied based on soil tests. Weed control was implemented using herbicides and practices typical for no-till systems. Plots were deep-ripped in fall 1999 after tile installation and disked in the spring 2000 and 2002 to level the surface because of subsidence resulting from the tile installation. Plots were not tilled in 2001 before planting soybean or in years after 2002. Seeding rates were 79,000 seeds ha⁻¹ for corn and 445,000 seeds ha⁻¹ for soybean. In 2001, 2003, and 2005, soybean grain yields were determined using a modified combine with a weigh tank and moisture meter mounted inside the combine grain storage tank (Colvin, 1990) by harvesting the entire plot area and dividing total grain weight by harvested plot area. In 2002, corn yield was determined by harvesting the entire area of each plot and weighing the grain in a weigh wagon with load cells and taking samples to measure grain moisture. In 2004 because a wind storm had knocked down corn in some areas of the plots, undamaged corn in four strips 2.29 m wide and 42.67 m long from each plot were harvested with the modified combine with weigh tank. The remaining area was bulk harvested. All corn and soybean shoot residues were left on the soil surface after harvest. Yields were calculated based on harvested plot area and were adjusted to 0.155 and 0.130 g g⁻¹ grain moisture for corn and soybean, respectively.

Drainage from each plot was conducted by solid plastic pipe to one of three large pits. Within each pit, drainage from eight plots was collected into dedicated sumps that a pump emptied whenever the water level exceeded a preset level. Flow from each pump went through a combination digital and mechanical totalizing flow meter with flow volume versus time recorded hourly with a data logger. The mechanical water meters were read periodically and used to correct the digital flow rates to the

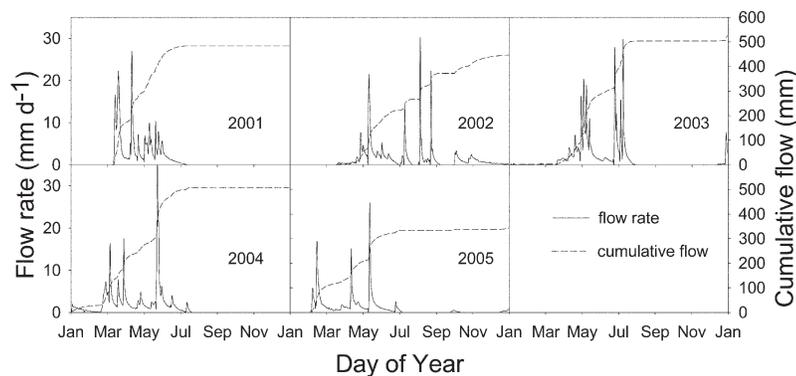


Fig. 1. Daily flow rate and cumulative tile flow for 2001–2005 for the control treatment (CN) plot with the most drainage.

correct total flow. Missing flow data caused by system failures were interpolated based on similar flow events. Flow values at higher flow rates were more uncertain as the district drainage pipe servicing the site would often be over capacity during high flow preventing adequate pumping of plot sumps.

Flow proportional water samples were composited over approximately weekly intervals via a capillary tube connected to each sump pump outlet. Water samples were returned to the laboratory, refrigerated, and analyzed for NO_3^- using a colorimetric method after first reducing NO_3^- to NO_2^- (USEPA Method 353.2). Quantitation levels for NO_3^- in all samples were 0.3 mg N L^{-1} .

Monthly precipitation totals and average monthly air temperatures were calculated from daily values collected at the Iowa State University research farm located 5.4 km southwest of the study area (Herzmann, 2006). Nitrate loads were calculated by multiplying the NO_3^- concentration for the composite sample by the total volume of drainage during the compositing period. Annual flow-weighted NO_3^- concentrations were computed by dividing the cumulative annual load by the annual drainage volume. The experimental design was a randomized complete block design with four blocks or repetitions. Data for individual years were analyzed separately and then combined for the combined years analysis. Data for individual years were analyzed for treatment and block effects using PROC ANOVA (SAS, 1999). Data for all 5 yr were combined and analyzed for year, treatment, block, and year-by-treatment effects using the PROC MIXED procedure (SAS, 1999) with years as a repeated measure following the guidelines by Littell et al. (2000). A protected least significant difference (LSD) test at the 0.05 probability level was used to compare treatment or year means when the ANOVA indicated significant effects at the 0.05 probability level (SAS, 1999). Regressions of NO_3^- mass flux vs. tile flow were computed with log-transformed data using PROC GLM model (SAS, 1999).

Although tile installation was completed in August 1999 and installation of pumps, sampling equipment, and dataloggers was completed in April 2000, very little drainage flow was collected in 2000 because of dry soil conditions and below normal precipitation between Sept. 1999 and May 2000. Because of little drainage in 2000 and because 2000 was the first cropping year after establishing the crop and nitrogen treatments, 2000 was considered a transition year and data from that year are not presented.

Results

Hydrology

Annual rainfall for 2001 and 2002 was 30 and 6 mm below the 50-yr average of 837 mm at this site. Annual rainfall exceeded the long time average by 54 mm in 2003, 28 mm in 2004, and 56 mm in 2005. Rainfall exceeded evapotranspiration in the spring of each year causing substantial tile flow in all plots. Rainfall during the March through June period when drainage tiles typically are flowing in Iowa totaled 364, 315, 414, 456, and 352 mm for 2001 through 2005, respectively. Compared to the 50-yr average rainfall of 380 mm for these months, rainfall in 2002 was well below average and rainfall in 2004 was well above average.

Tile flow typically started in the spring of each year as the frozen soil thawed. Tile flow rate and cumulative annual flow for the CN treatment plot with the greatest flow is shown as an example in Fig. 1. Tiles started flowing in March 2001. Tile flow continued until mid-July when decreasing precipitation and increasing transpiration from the growing soybean crop combined to lower the groundwater table below the tiles. Tile flow in 2002 started in mid-April and continued through August due to much greater than normal rainfall in that month. The tiles continued to flow during the fall of 2002 unlike the other 4 yr. Tile flow ceased shortly into January of 2003 when the soil froze, resumed in March after the ground thawed, and ceased again in August. Only in 2004 was there measurable flow throughout the winter months as a result of greater than normal precipitation in November and December of 2003. In 2005, tile flow started in February and ended in July with small discharges in October and December.

Annual drainage from the individual plots was extremely variable ranging from 97 to 482 mm in 2001, 157 to 612 mm in 2002, 143 to 591 mm in 2003, 105 to 597 mm in 2004, and 64 to 344 mm in 2005. Substantial variability in tile discharge is common for these size plots (Lawlor et al., 2005) despite the use of subsurface flow barriers and perimeter drains. Half of the annual tile flow occurred by mid-April in 2001, 2004, and 2005. Half of the annual tile flow occurred by mid-May in 2003 and mid-June in 2002. Thus, in most years, half of the tile flow had occurred before the main crop had emerged. When averaged across all treatments, substantially more water drained from the plots in 2003 than the other 4 yr (Table 2). When averaged by treatment, there were no significant differences in cumulative annual flow between any of the treatments ($P = 0.05$) for any year or for all years averaged (Table 2).

Flow-Weighted Nitrate Concentration

Nitrate concentrations in tile discharge from the plots varied from <0.3 to 35 mg N L^{-1} over the 5-yr period. While there was considerable variability between plots, the DW plots had consistently lower NO_3^- concentrations in drainage. Data from all plots in 2004 are shown in Fig. 2 as examples of measured NO_3^- concentrations. In 2004, average nitrate concentrations in the DT treat-

Table 2. Average annual cumulative drainage by treatment.

Year	Treatment			Avg.
	Control (CN)	Denitrification wall (DW)	Deep tile (DT)	
	mm			
2001	258†	228	246	244BC
2002	227	226	258	237BC
2003	346	332	381	353A
2004	248	304	269	274B
2005	175	209	196	193C
Avg.	251	260	270	260

† Numbers within a row followed by the same lowercase letter and numbers within a column followed by the same uppercase letter are not significantly different at the 0.05 probability level. Rows or columns without letters indicate that main effects or interaction effects were not significant in the analysis of variance.

ment were similar to the CN treatment until about 10 June when tile flow rates decreased and NO₃ concentrations for the DT treatment started to decrease in relation to the CN concentrations. This trend of NO₃ concentrations decreasing for the DT treatment relative to the CN treatment when tile flow rates slowed at the end of the drainage season was repeated in the other years as well (data not shown) and may be indicative of increased denitrification at depth as the soil warmed, or residence times increased due to lower fluxes, or of a greater proportion of the water entering the tile drains coming from deeper in the profile where anaerobic conditions are more prevalent and denitrification is greater.

Average annual flow-weighted concentrations for the CN treatment were >19 mg L⁻¹ every year (Table 3). Across all treatments, the highest annual average flow-weighted NO₃ concentration occurred in 2001 and 2003 for the CN and DT treatments—years when soybean was grown and no fertilizer N was applied. That NO₃ concentrations in tile drainage can be as high or higher under soybean than corn have been noted by others (Kanwar et al., 1997; Jaynes et al., 2001; Randall et al., 2003; Kladvik et al., 2004) and demonstrates that NO₃ concentrations in tile drainage are more a reflection of the cropping system (annual row cropping) than of N fertilizer use. The annual averaged NO₃ concentrations for the DT treatment were at least 14% lower than the control in 2002 and 2005, but the differences were only significant in 2002. Although the NO₃ concentrations in the tile flow from the DT treatments decreased with respect to the CN treatment near the end of each drainage season, the decrease did not result in significant differences in flow-weighted NO₃ concentrations between the treatments for 4 of the 5 yr and when averaged for all 5 yr. Thus, if the deeper tile placement increased NO₃ loss by denitrification, the increased denitrification rate was insufficient to cause a significant reduction in the annual average concentration.

Average annual flow-weighted NO₃ concentrations were significantly lower for the DW treatment than the CN treatment every year and when averaged for all 5 yr (Table 3). Annual concentrations for the DW treatment averaged at least 10 mg N L⁻¹ less than the CN treatment in every year. Nitrate concentrations were reduced by an average of 60% for the DW treatment compared to the CN treatment during the 5-yr period.

Nitrate concentrations in the tile drainage of the CN treatment plots were independent of tile flow rate. This is illustrated in Fig. 3a

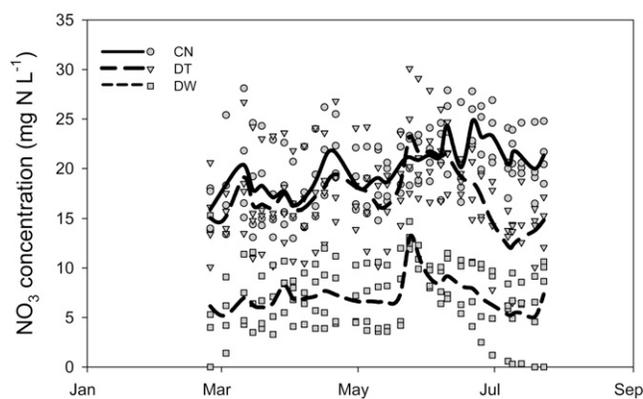


Fig. 2. Nitrate concentration in tile drainage from four plots each of the control (CN), denitrification wall (DW), and deep tile (DT) treatments and the average concentration (lines) for each treatment in 2004.

where the average NO₃ mass flux is plotted vs. the average tile flow on a log-log plot for each sampling period. A line fitted to this data indicates no change in NO₃ concentration with change in flow rate if the slope is equal to 1, with the intercept for this case equaling the log of the average NO₃ concentration in the tile drainage. Fitting a log-linear line to the control treatment data resulted in a slope of 1.007 with a 95% confidence interval that included 1.0, thus the NO₃ concentration did not change with flow rate. For the DT treatment (Fig. 3b), a slope of 1.075 was computed with a 95% confidence limit that did not include 1.0, thus NO₃ concentrations increased with increasing tile flow rate for this treatment. This is in agreement with the observed decrease in NO₃ concentration with respect to the control treatment at the end of each drainage season when the tile discharges were lower (Fig. 2). Additionally, the fitted slope for the DW treatment was 1.147, which was also significantly different than 1.0 (Fig. 3c). Thus, NO₃ concentrations increased as tile flow increased for this treatment as well. Higher NO₃ concentrations at higher drainage flows may reflect decreased residence time for removal of NO₃ from water passing through the woodchip-filled trenches as tile discharge increased or possibly due to water channeling through the trenches (Ghodrati and Jury, 1990) or bypassing around the trenches (Schipper et al., 2004), decreasing woodchip–NO₃ contact.

Table 3. Average annual flow-weighted NO₃ concentration of drainage water by treatment.

Year	Treatment			Avg.
	Control (CN)	Denitrification wall (DW)	Deep tile (DT)	
	mg N L ⁻¹			
2001	25.3a†	10.0b	24.7a	20.0A
2002	19.1a	6.3c	16.4b	13.9C
2003	24.7a	10.3b	23.9a	19.7A
2004	19.8a	9.2b	20.0a	16.4B
2005	21.6a	8.4b	17.4a	15.7BC
Avg.	22.1a	8.8b	20.5a	17.1

† Numbers within a row followed by the same lowercase letter and numbers within a column followed by the same uppercase letter are not significantly different at the 0.05 probability level. Rows or columns without letters indicate that main effects or interaction effects were not significant in the analysis of variance.

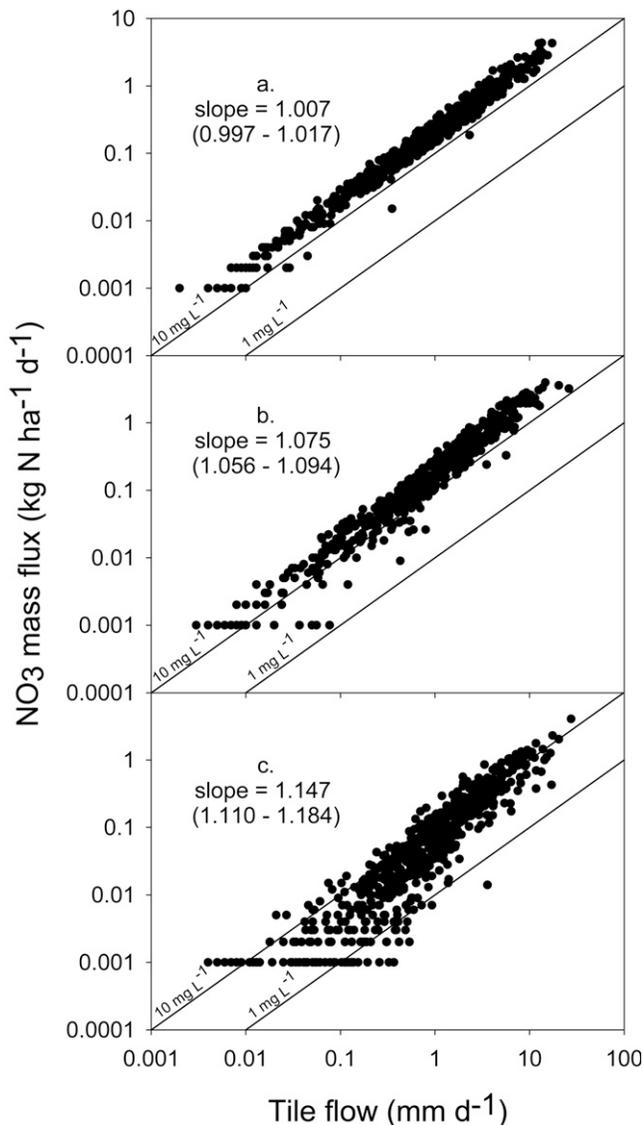


Fig. 3. Relationships between tile flow and NO_3 mass flux for the (a) control (CN), (b) deep tile (DT), and (c) denitrification wall (DW) treatments from 2001–2005. Isolines (1 and 10 mg L^{-1}) provide orientation to interpret NO_3 concentration. Values in figure are the best fit slope and 95% confidence levels of the log-transformed data.

Nitrate Mass Loss

Total NO_3 loss in tile drainage water from the individual plots was extremely variable mostly because of the large variation in drainage volumes. Total NO_3 mass loss in tile drainage per plot ranged from 9 to 113 kg N ha^{-1} in 2001, 7 to 101 kg N ha^{-1} in 2002, 20 to 120 kg N ha^{-1} in 2003, 17 to 106 kg N ha^{-1} in 2004 and from 5 to 60 kg N ha^{-1} in 2005. There was no significant difference in the average NO_3 mass loss between the DT and CN treatments in any year or averaged over all years (Table 4). Conversely, the average NO_3 mass loss from the DW treatment was significantly lower than the CN treatment in 2003 (Table 4). While the mass loss of NO_3 for the DW treatment was numerically lower than the control in the other years as well, the differences were not significant at $P = 0.05$ because of the large variability in tile discharge among plots and the limited number

Table 4. Average annual NO_3 load in drainage water by treatment.

Year	Treatment			Avg.
	Control (CN)	Denitrification wall (DW)	Deep tile (DT)	
	kg N ha^{-1}			
2001	63	22	60	48B†
2002	40	15	43	33CD
2003	81a	35b	87a	68A
2004	47	28	50	42BC
2005	34	19	31	28D
Avg.	53a‡	24b	54a	44

† Numbers within a row followed by the same lowercase letter and numbers within a column followed by the same uppercase letter are not significantly different at the 0.05 probability level. Rows or columns without letters indicate that main effects or interaction effects were not significant in the analysis of variance.

‡ Numbers within this row followed by the same lowercase letter are not significantly different at the 0.10 probability level.

of replications (4). When averaged over all years, the mass loss of NO_3 from the DW treatment was significantly lower at $P = 0.10$ by 29 $\text{kg N ha}^{-1} \text{ yr}^{-1}$ than from the CN treatment, which represents a 55% reduction in mass loss. This reduction in NO_3 mass loss is considerably greater than the 19 $\text{kg N ha}^{-1} \text{ yr}^{-1}$ average reduction found in tile discharge by Jaynes et al. (2001) from reducing N fertilizer application rates by two thirds.

There were no obvious trends in the NO_3 concentrations or mass losses over the 5 yr for the DW treatment. This is comparable to other studies of denitrification walls (Schipper and Vojvodic-Vukovic, 2001; van Driel et al., 2006), where efficacy did not change over time. Eventually, the decomposition of the woodchips should lead to a decrease in the treatment's ability to remove NO_3 . We will continue to monitor this treatment to quantify any decrease in efficacy to estimate the effective lifetime for this system.

Nitrogen Removal Rate

We assume the reduction of NO_3 in the DW treatment compared to the CN treatment was due to increased denitrification sustained by the addition of the woodchips (Schipper and Vojvodic-Vukovic, 2000; Greenan et al., 2006). By assuming that the water flowing to the tile drains in the DW plots had the same NO_3 concentration as the CN plots before entering the woodchip-filled trenches, we can estimate the approximate rate of NO_3 removal by the woodchip trenches. By assuming uniform water movement through the $0.61 \times 1.52 \times 45.7$ m trenches, remembering that there was a trench on each side of the tile and that groundwater from the central 6.1-m-wide strip between the wood-filled trenches did not flow through the trenches, a N removal rate can be computed. Based on these assumptions, we computed the flow-weighted N removal rate of the woodchip-filled trenches to be 622 $\text{mg N m}^{-3} \text{ d}^{-1}$ averaged over the four DW plots and 5 yr of the study. This value is much greater than the removal rates observed by Jacinthe et al. (1998) and Groffman et al. (1996) in shallow groundwater in natural riparian areas (6–9 $\text{mg N m}^{-3} \text{ d}^{-1}$), which is not surprising given the addition of a carbon source in our plots. However, the average NO_3 removal rate for the DW treatment also exceeds the average found by Schipper and Vojvodic-Vukovic (2000) of 252 $\text{mg N m}^{-3} \text{ d}^{-1}$ for a sawdust-filled denitrifica-

tion wall near a stream bank in New Zealand treating groundwater with NO₃ concentrations generally under 10 mg N L⁻¹. Conversely, Robertson and Cherry (1995) observed a NO₃ removal rate of 3200 to 6000 mg N m⁻³ d⁻¹ (as calculated by Schipper and Vojvodic-Vukovic, 1998) for a denitrification wall treating groundwater with NO₃ concentrations exceeding 55 mg L⁻¹. As the NO₃ concentration in the groundwater flowing through the denitrification walls in this study was between 20 and 25 mg N L⁻¹ and because the NO₃ removal rate in similar denitrification walls are typically NO₃ limited and thus a function of NO₃ concentration (Schipper and Vojvodic-Vukovic, 1998), the N removal rate found here is consistent with these other studies.

Crop Yields

Corn and soybean yields in the plots were slightly greater than the county average each year (National Agricultural Statistics Service, 2006). There was no significant difference in yields among the treatments in any year, nor when years were averaged by crop (Table 5). No yields were recorded for the DT treatment in 2005 as the soybean plants were mowed in August after tile drainage ceased to prepare the plots for new treatments starting late summer. Overall, neither the DW nor DT treatment had any effect on crop yields, nor do these treatments take farmland out of production.

Conclusions

Modification of tile drains may be a potential method for reducing NO₃ concentrations and loads in tile drainage from corn/soybean fields. Results of a 5-yr field study illustrated that construction of denitrification walls composed of woodchips on both sides of a tile was effective in reducing flow-weighted NO₃ concentrations by ≥55%. Moreover, yearly grain yields from the DW treatment were equivalent to conventionally drained plot yields and no land needs to be taken out of production to install the denitrification walls. The other alternative treatment tested, deeper tile placement (DT), was not effective in reducing NO₃ concentrations. Continued monitoring of the DW treatment for additional years is necessary to clearly quantify the effectiveness of the treatment given the natural variability in weather and crop growth. Extended monitoring is also required to quantify the effectiveness over time of the DW. Eventually, the buried carbon in the DW treatment will be exhausted by microbial activity and will no longer be effective in increasing denitrification. The effectiveness of this system over the long term will determine the cost per unit of NO₃ removed and thus its potential as a management tool for removing NO₃.

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Table 5. Average grain yields each year by treatment.

Year	Treatment		
	Control (CN)	Denitrification wall (DW)	Deep tile (DT)
Mg ha ⁻¹			
2001– soybean	3.0†	3.0	2.9
2002– corn	12.4	11.8	11.7
2003– soybean	2.7	2.6	2.5
2004– corn	13.3	13.5	13.3
2005– soybean	4.5	4.1	ND‡
Avg.– soybean	3.4 (2.8)§	3.2 (2.8)	– (2.7)
Avg.– corn	12.0	11.9	11.8

† No significant differences in any year or when averaged for all years as indicated by *F*-test > 0.05.

‡ No data, crop was mowed in August before harvest to prepare plots for new experiment.

§ Average of 2001 and 2003 in parentheses.

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