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Reconnecting Tile Drainage to Riparian Buffer Hydrology for Enhanced Nitrate Removal

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Riparian buffers are a proven practice for removing NO_3 from overland flow and shallow groundwater. However, in landscapes with artificial subsurface (tile) drainage, most of the subsurface flow leaving fields is passed through the buffers in drainage pipes, leaving little opportunity for NO_3 removal. We investigated the feasibility of re-routing a fraction of field tile drainage as subsurface flow through a riparian buffer for increasing NO_3 removal. We intercepted an existing field tile outlet draining a 10.1-ha area of a row-cropped field in central Iowa and re-routed a fraction of the discharge as subsurface flow along 335 m of an existing riparian buffer. Tile drainage from the field was infiltrated through a perforated pipe installed 75 cm below the surface by maintaining a constant head in the pipe at a control box installed in-line with the existing field outlet. During 2 yr, >18,000 m^3 (55%) of the total flow from the tile outlet was redirected as infiltration within the riparian buffer. The redirected water seeped through the 60-m-wide buffer, raising the water table approximately 35 cm. The redirected tile flow contained 228 kg of NO_3. On the basis of the strong decrease in NO_3 concentrations within the shallow groundwater across the buffer, we hypothesize that the NO_3 did not enter the stream but was removed within the buffer by plant uptake, microbial immobilization, or denitrification. Redirecting tile drainage as subsurface flow through a riparian buffer increased its NO_3 removal benefit and is a promising management practice to improve surface water quality within tile-drained landscapes.

Surface waters within the Upper Mississippi River basin contain some of the highest concentrations of nonpoint source NO_3 in the United States (Schilling et al., 2012; David et al., 2010). These NO_3 loads have potentially widespread impacts on ecosystem function and public health. Excessive NO_3 has been identified as a leading cause of hypoxia in the northern Gulf of Mexico (Rabalais et al., 2001; Dale et al., 2010). High NO_3 concentrations can also significantly affect local aquatic integrity, and the USEPA is encouraging states to establish numeric NO_3 criteria to protect aquatic life (USEPA, 2007). In addition, NO_3 concentrations in surface waters often exceed the USEPA maximum contaminant level for drinking water of 10 mg L^{-1} N and can threaten public water supplies that use surface water (Schilling and Wolter, 2009; Jha et al., 2010). Numerous studies at the field and watershed scale (David et al., 1997; Goolsby et al., 1999; Jaynes et al., 1999; Mitchell et al., 2000; Dale et al., 2010, David et al., 2010) have shown that much of the NO_3 in surface waters of the Midwest comes from corn (*Zea mays* L.)–soybean [*Glycine max* (L.) Merr.] production. These studies identify the primary pathway for NO_3 entering surface waters as discharge from artificial subsurface drains (popularly called and hereafter termed “tiles”) that are common across the Midwest cornbelt (Zucker and Brown, 1998). Thus, it is not surprising that the area within the Mississippi River watershed identified by Goolsby et al. (2001) as the primary source of NO_3 to the Gulf is the same area where corn production on artificially drained lands is prevalent.

Riparian buffers have been proven effective in reducing sediment and nutrients in runoff to surface waters when properly maintained (Lee et al., 2000; Schultz et al., 2004; Helmers et al., 2008). Riparian buffers can also remove NO_3 from shallow subsurface flow through a combination of plant uptake, immobilization in bacterial biomass, and denitrification (Mayer et al., 2007). One finding of this meta-analysis of published buffer studies was that subsurface removal of nitrate did not depend on buffer width or buffer vegetation type. The USDA has been very successful in inducing landowners to establish riparian conservation practices within the United States. As of October 2013, nearly 690,000 ha of filter strips and riparian forest buffers established with USDA technical assistance were currently

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under contract within the Conservation Reserve Program (FSA, 2013). However, in the 17.4 million ha of artificially drained land in the midwestern United States, much of the NO_3–laden water leaching from row crop fields is routed through the buffers in drainage pipe and discharged directly into surface waters. Thus, the potential for NO_3 removal within riparian buffers in tile-drained landscapes is greatly reduced because they are no longer hydraulically connected to the row crop area above the buffers and most of the NO_3 passes through them in pipes.

Whether NO_3 is lost via plant uptake, microbial immobilization, or denitrification, maximum NO_3 loss is associated with conditions when the water table saturates an organic rich layer within the buffer (Hill, 1996; Clément et al., 2002; Ranalli and Macalady, 2010). The organic rich material can be a relic of past depositional environments or continually renewed from senescence of roots and root exudates from deep-rooted buffer vegetation (Hill, 1996; Gold et al., 1998; Tufekcioglu et al., 1999; Schultz et al., 2004; Clague et al., 2013).

To better achieve the nutrient removal capabilities of riparian buffers established within tile-drained landscapes, the hydrology between the uplands drained by tiles and the buffer needs to be reconnected. To restore this connection, tile drainage could be diverted into the buffer as surface flow or subsurface flow. Diverting tile flow as surface flow in riparian buffers has been shown to be successful (Chescheir et al., 1991), but drawbacks could include increased soil erosion and development of concentrated flow with little NO_3 removal. An alternative is to intercept a field tile outlet where it crosses a riparian buffer and divert a fraction of the flow as shallow groundwater within the buffer. The infiltrated water would potentially raise the water table within the buffer into organic rich soil layers and provide an opportunity for NO_3 in the field tile drainage to be removed by denitrification and immobilization before entering the adjacent stream. In this study, we assess a practice to divert flow from a field tile outlet into the shallow subsurface of an existing riparian buffer. We measure the amount of flow that is diverted over a 2-yr period and quantify the loss of NO_3 as the water flows laterally through the buffer to the stream.

Materials and Methods

The research site was a 48-ha privately owned field located in Hamilton county, north-central Iowa (42°11′ N, 93°30′ W). The field is in the Bear Creek watershed, a third-order stream that drains 6810 ha, most of which are tile-drained and used for the production of corn and soybean. The field had more than 7 m of vertical relief, with the upland portion consisting of poorly drained Canisteo and Webster silty clay loams (fine-loamy, mixed, superactive, calcareous, mesic Typic Endoaquolls; fine-loamy, mixed, superactive, mesic Typic Endoaquolls), with a limited extent of somewhat poorly drained Nicollet loam (fine-loamy, mixed, superactive, mesic Aquic Hapludolls). Well-drained Clarion (fine-loamy, mixed, superactive, mesic Typic Hapludolls) and Storden (fine-loamy, mixed, superactive, mesic Typic Eutrudepts) loams comprised the side slopes of the field sloping down to a level area of Coland clay loam (fine-loamy, mixed, superactive, mesic Cumulic Endoaquolls) adjacent to Bear Creek.

A riparian buffer was established on both sides of Bear Creek in 1995. The buffer consisted of a 6-m-wide zone of silver maple trees (*Acer saccharinum* L.) planted along the stream followed by a 6-m-wide mixed shrub-grass planting consisting mainly of common ninebark (*Physicarpus opulifolius* L.), nannyberry viburnum (*Viburnum lentago* L.), and switchgrass (*Panicum virgatum* L.). The upper part of the buffer consisted of 8 m of switchgrass, giving a total width for the buffer of 20 m. Details of the riparian buffer design and plant species used are given in Schultz et al. (1995).

The poorly and somewhat poorly drained upland soils within the field were tile-drained for the production of corn and soybean grown in a 2-yr rotation with corn planted in even years and soybean planted in odd years. The tiles within the study field drain to the stream through three outlets (Fig. 1) that run through the buffer to Bear Creek. The specific outlet we selected for interception had the highest flow rate and longest duration of flow after rainfall events observed in the summer of 2010. Tile maps were not available for the field, but, on the basis of field topography, we estimated that the intercepted field tile outlet drains approximately 10.1 ha.

The tile outlet was intercepted just inside the buffer as it left the row crop portion of the field. The 15-cm-diameter tile was excavated and reconnected to an in-line water-level control box (AgriDrain Corp.). The control box consisted of three chambers separated by two sets of stoplogs that could be used to independently set the water level within the upstream and middle chamber of the box (Fig. 2). The control box extended from the depth of the tile (~1.2 m) to about 0.3 m above the soil surface. The field tile outlet was connected to the inlet chamber of the control box. The outlet end of the control box was reconnected to the existing pipe that emptied directly into Bear Creek. The middle chamber had outlets on both sides of the box that were connected to 10-cm-diameter slotted corrugated plastic drainage pipe. This new pipe was installed perpendicular to the field tile along the top of the buffer at a depth of 76 cm below the ground surface. The pipe served to introduce tile water as shallow groundwater within the buffer. The distribution pipe was installed as far as possible along the buffer before encountering another field tile outlet (Fig. 1). This distance was 30.5 m in the downstream (southeast) direction and 305 m in the upstream (northwest) direction. The pipe was laid at 0% slope and was wrapped in needle-punched polypropylene fabric (AgriDrain Corp.) to reduce root penetration and clogging of the pipe from the buffer vegetation.

The water level within the middle chamber of the control box determined the head within the distribution pipe. Water level within the control box was controlled with the two sets of stoplogs separating the three chambers. A combination of 12.7- and 17.8-cm-high stoplogs could be used to set the water height in the first two upstream chambers to any desired level. To facilitate accurate flow measurement a 45° v-notch weir was cut into the top stoplog in each set. Water levels within the two upstream chambers was measured with dedicated pressure transducers (AST4510, American Sensor Technologies) and recorded every hour with a CR10X datalogger (Campbell Scientific). Local relief at the site allowed us to raise the head in the control box and distribution pipe and thus raise the water table in the buffer to several decimeters below the ground without causing wet soil conditions in the adjoining field.

Starting on 1 Jan. 2011, the water level was set at 29 cm below the soil surface at the control box. The stoplogs were pulled to lower the water table for row crop planting on 11 May 2011 as a precaution against raising the water table in the field adjacent to the buffer and causing soil conditions that were too wet for crop planting. After

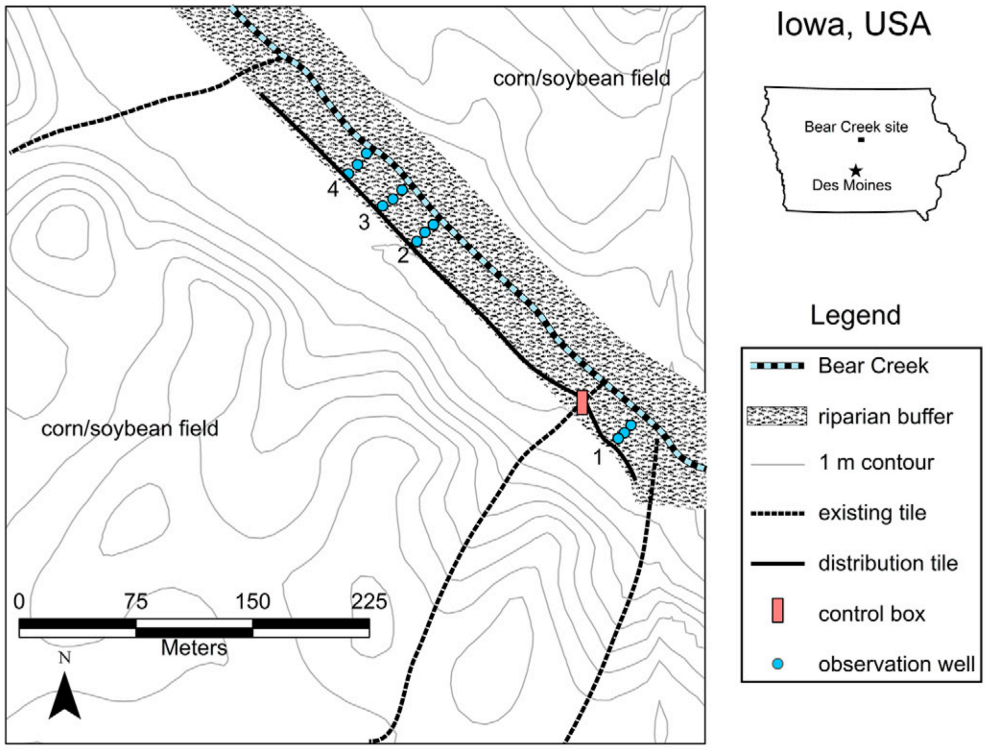

Fig. 1. Schematic of the experimental field and riparian buffer next to Bear Creek, the existing tile outlets, installed control box, distribution pipe, and wells in transects 1 through 4.

planting the stoplogs were reinserted on 26 May 2011 to establish a water level in the middle chamber 47 cm below the ground surface. In 2012, the stoplogs were again pulled for 6 d as a precaution against wet soil conditions along the edge of the buffer before field planting on 8 May and pulled again on 1 June and 13 June for short periods to allow for verification of the v-notch weir calibrations. The water level in the middle chamber was maintained at 47 cm below the ground surface at all other times. Stoplogs controlling water level in the inlet chamber receiving water from the field tile were set to keep the water level 12.7 cm higher in this chamber than in the middle chamber of the control box.

The weirs were calibrated in the laboratory and fitted to the curve:

$$q = 0.00511(d - 1.582)^{2.5}$$

where q is flow rate (L s^{-1}), and d is depth of water above the v-notch (cm). Flow from the field was measured as flow from the first chamber of the control box into the middle chamber over the first set of stoplogs. Water flow discharging directly to the stream was measured by flow over the second set of stoplogs separating the middle and third chamber of the control box. Flow that infiltrated into the buffer via the distribution pipe was determined by the difference between flows over the two sets of stoplogs. Flow rates for both v-notch weirs were verified in the field against timed catch-can volumes to determine the correct elevation of the bottom of the v-notches as measured with the pressure transducers. Accuracy of individual flow measurements were within 11% relative error but were larger for the computed buffer infiltration rate, which was determined as the difference between two flow measurements. During peak events this was often a small difference between two large values.

Soil cores taken from the southeast and northwest ends of the grassed portion of the buffer were used to measure soil texture (using the sedimentation method of Gee and Bauder [1986]) and soil organic carbon content on 30-cm sections down to 240 cm depth. Total organic C (after removal of carbonates with 1 mol L^{-1} H_2SO_4) was measured using a dry combustion method in a Carlo-Erba NA1500 NCS elemental analyzer (Haake Buchler Instruments). Additionally, soil coring was continued down to 6.2 m depth to probe for hydraulically restricting layers below the buffer.

Monitoring wells were installed within the buffer along four transects (Fig. 1). Each transect consisted of three wells equally spaced between the distribution pipe used to convey

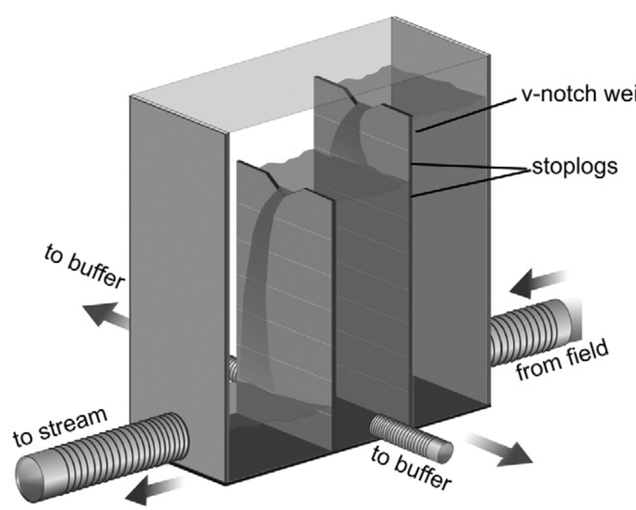

Fig. 2. Schematic of the control box used to redirect tile flow showing the three chambers separated by two sets of stoplogs, each topped with a v-notch weir (not drawn to scale).

water along the top of the buffer and Bear Creek. Wells were installed 2.3 m deep and were fully screened. Each well was equipped with a pressure transducer and datalogger (Global Water Instrumentation Sales) to measure and record water table depth every 6 h. Water samples were collected from each well on a roughly weekly schedule when the field tile was flowing and returned to the laboratory for the determination of NO_3 concentration. At the same time, water samples from Bear Creek and tile water flowing within the control box were collected for measurement of NO_3 concentrations. Water samples were returned to the laboratory and stored at 4°C until analysis. Water samples were analyzed for NO_3 using a Lachat 8000 (Zellweger Analytics, Lachat Instrument Division) wherein NO_3 was quantitatively reduced to NO_2, and the NO_2 concentration was determined colorimetrically (Keeney and Nelson, 1982). The method quantitation limit was 0.3 mg N L^{-1} as NO_3. Annual mass load of NO_3 was calculated by multiplying the NO_3 concentration times the volume of water that infiltrated the buffer between water sampling dates and summing over all samples in a calendar year. Chloride concentrations were also measured in the same water samples for seven sampling dates from spring through summer of 2012 using a 4500i ion chromatograph (Dionex). The quantitation limit for Cl was 0.1 mg L^{-1}.

Rainfall was measured at the site with a tipping bucket located at the control box. Missing data and precipitation data during winter was filled in from data collected at an Iowa State University agricultural research farm located 14.5 km away (Herzmann, 2012).

Results

Soils and Weather

The southwest portion of the buffer had soil textures consistent with Coland soils, whereas the southeast portion had markedly higher sand contents in the top 150 cm and did not conform to a Coland soil (Table 1). Both locations had sand contents >50% below the 150-cm depth. Soil organic carbon content was higher on the southwest side, exceeding 2% C down to 150 cm depth. Organic carbon content in the southwest portion of the buffer was lower but still exceeded 2% C from 45 to 75 cm and again from 105 to 120 cm depth. Thus, the soil appeared to have sufficient carbon content to support denitrification if saturated (Burford and Bremner, 1975). Coring continued to 5.2 m, and, although no samples were collected, soils were observed as becoming more sandy and gravelly starting at 3.35 m to about 4.57 m, where the soil became massive and reduced in color with a loam texture consistent with basal till of the Dows Formation (Eidem et al., 1999). This till typically has hydraulic conductivities two orders of magnitude lower than the more weathered upper zones and likely serves as a restricting layer causing groundwater to flow laterally from the field to the stream (Seo et al., 1996; Wineland, 2002).

Annual precipitation in 2011 was 813 mm and was 577 mm in 2012. Both years were less than the 40 yr average (1961–2010) of 876 mm as measured at the Ames, Iowa airport. The year 2012 was classified as severe to exceptional drought from July through the end of the growing season in mid-Iowa and across much of the midwestern United States.

Field Tile Nitrate and Flow

Similar to other drainage studies in mid-Iowa, tile drainage started when the frost went out of the ground in late February or early March in 2011 and 2012 (Fig. 3). Drainage flow peaks corresponded to rainfall events until about the first of July or August when evapotranspiration from developing row crops exceeded rainfall, resulting in the water able dropping below the depth of the field tiles. There was little to no tile flow in late summer or fall in both years despite several large precipitation events. This tile drainage pattern is consistent with historic observations from mid-Iowa, although fall drainage has become more common over the last 10 yr in response to increased fall precipitation after harvest (Helmers et al., 2005; Jaynes et al.,

Table 1. Soil texture and organic carbon content with depth in the southeast and southwest sides of the buffer zone.

Depth	Southeast side					Southwest side				
	Clay	Silt	Sand	Soil texture†	C	Clay	Silt	Sand	Soil texture	C
cm	%				%	%				%
0–15	15.0	25.4	59.6	SL	1.77	39.5	57.4	3.1	SiC	3.40
15–30	21.6	21.3	57.1	SCL	1.33	39.3	58.0	2.7	SiC	3.35
30–45	26.0	45.3	28.6	L	2.59	31.1	66.1	2.8	SiC	2.71
45–60	32.1	38.4	29.5	CL	2.57	39.2	57.7	3.1	SiC	2.42
60–75	26.3	42.2	31.5	L	2.16	41.2	55.0	3.9	SiC	2.99
75–90	24.8	42.8	32.5	L	1.96	41.4	56.2	2.4	SiC	3.72
90–105	18.6	33.4	48.0	L	1.45	38.8	57.9	3.3	SiC	3.55
105–120	22.8	45.2	32.0	L	2.02	41.4	55.6	3.0	SiC	2.75
120–135	14.0	24.7	61.3	SL	1.93	41.9	55.6	2.5	SiC	2.27
135–150	13.7	21.0	65.3	SL	0.97	23.4	41.5	35.2	L	2.24
150–165	18.9	22.5	58.6	SL	0.56	6.9	17.2	75.9	SL	1.13
165–180	12.9	8.6	78.5	SL	0.38	16.0	32.0	52.0	L	2.57
180–195	8.3	4.2	87.6	LS	0.21	11.5	28.1	60.4	SL	1.19
195–210	9.7	7.3	83.1	LS	0.27	–‡				1.66
210–225	11.1	8.8	80.1	SL	0.62	10.4	17.3	72.3	SL	1.03
225–240	9.8	5.9	84.3	LS	0.52	9.1	14.7	76.2	SL	1.07

† CL, clay loam; L, loam; LS, loamy sand; SCL, sandy clay loam; SiC, silty clay; SL, sandy loam.

‡ Sample lost.

2008). There were several breaks in the measured inflow to the control box in 2011 and 2012 when no tile drainage was diverted into the buffer. This was due to pulling the stoplogs for short intervals during planting of row crops (May) or when validating the weir calibration. In 2011, nearly 20,000 m^3 (or the equivalent depth of 199 mm) were drained from the 10.1-ha contributing area. Although considered a severe drought year, sufficient precipitation fell in early 2012 to produce over 12,900 m^3 in flow or an equivalent depth of 128 mm of drainage from the field. These drainage rates were similar to measured rates for other drained row crop fields within central Iowa (Jaynes and Colvin, 2006).

Tile outflow increased during precipitation events, with much of the tile flow during larger events being discharged directly to the stream when the infiltration capacity of the buffer was exceeded (Fig. 3). At lower field tile flow rates, there was no outflow from the control box to the stream even though flow into the control box continued. This represented times when the buffer was infiltrating all of the water coming from the field.

Water flow redirected into the buffer from the field tile outlet is represented by the difference between the flow into and out of the control box (Fig. 4). The difference was highly variable especially during large flow events when infiltration represented a small difference between two large flow measurements. The difference was 0 when the control box stoplogs were removed in May of 2011 and 2012 and during calibration in 2012. During steady flow conditions, the infiltration into the 335-m-long buffer averaged 0.35 $m^2\ d^{-1}$ in 2011 and 2012. Assuming a 1-m-high lateral transmission zone for flow within the buffer, the 0.35 $m\ d^{-1}$ infiltration rate compares well with the hydraulic conductivity measured by Ryan (1993) for this soil ($\bar{x}$ = 0.25 $m\ d^{-1}$; s = 0.11 $m\ d^{-1}$). For 2011, 11,280 m^3 of tile flow was diverted into the buffer, representing 56% of the total tile flow for that year. In 2012, only 7145 m^3 was diverted into the buffer, reflecting the much drier weather conditions in that year. However, the diverted flow represented 55% of the total annual flow, which was nearly the same percentage as in 2011.

The NO_3 mass infiltrated into the buffer was computed by multiplying NO_3 concentration in the field tile water by the buffer infiltration rate and summing over time. Nitrate concentration in the tile water varied little during the year and averaged 11.0 $mg\ L^{-1}$ in 2011 (range, 3.7 $mg\ L^{-1}$) and 14.8 $mg\ L^{-1}$ (range, 3.1 $mg\ L^{-1}$) in 2012. These concentrations are typical for nitrate in tile drainage in fields in mid-Iowa (Jaynes al., 1999; Jaynes et al., 2008). The computed mass of NO_3 diverted into the buffer totaled 122 kg N in 2011 and 106 kg N in 2012. This gave a total of 228 kg N as NO_3 being diverted during the 2 yr. Virtually all this N was diverted to the buffer during the April through July time period, which corresponds to the long-term tile flow patterns observed in central Iowa (Helmers et al., 2005). Because the NO_3 concentrations in the tile water were greater in 2012, the mass of NO_3 diverted each year was nearly the same despite less water being diverted into the buffer in 2012. Higher concentrations in 2012 may be from fertilizer N applied to corn grown that year, whereas in 2011 soybean was grown and no N fertilizer was applied. However, greater NO_3 concentrations in tile flow during corn years than during soybean years have not been a consistent pattern in other studies from mid-Iowa (Jaynes and Colvin, 2006; Jaynes et al., 2008).

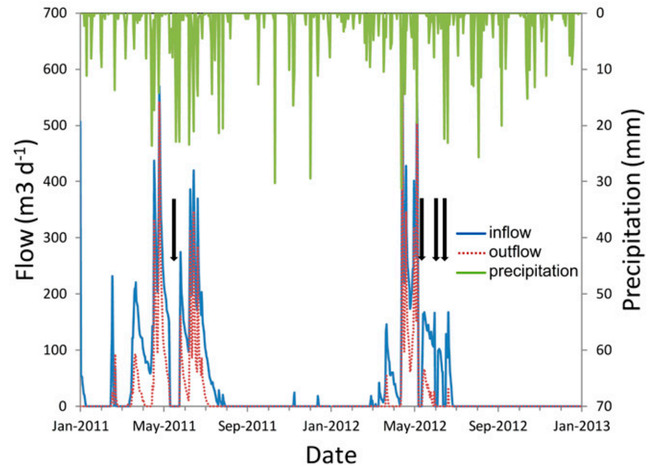

Fig. 3. Flow into the control box from the field tile, flow out of the control box into Bear Creek, and daily precipitation for 2011 to 2012. Vertical arrows indicate times when the stoplogs were removed from the control box and flow was not measured or water was diverted into the buffer.

Water Table Observations

Diverting field tile flow into the buffer affected the water table within the buffer during both years as illustrated in Fig. 5 for the second well transect. Well 2–1 was closest to the distribution pipe and furthest from the stream, well 2–2 was midway between pipe and stream, and well 2–3 was closest to Bear Creek. The water table depth fluctuated similarly in all transects and generally rose in spring with precipitation events and fell in late summer as evapotranspiration exceeded rainfall, especially in the drought year of 2012. The effects of diverting tile water into the buffer was most easily observed in May of both years when the water table dropped 30 to 40 cm in 2011 and about 30 cm in 2012 when stoplogs were removed during rowcrop planting. Water table response corresponded to distance from the distribution pipe because it was closer to the ground surface at the distribution pipe (well 2.1) and decreased in height approaching the stream (well 2.3). Thus, the tile water diversion did raise the water table within the buffer and saturated a portion of the soil that would not otherwise have been saturated without

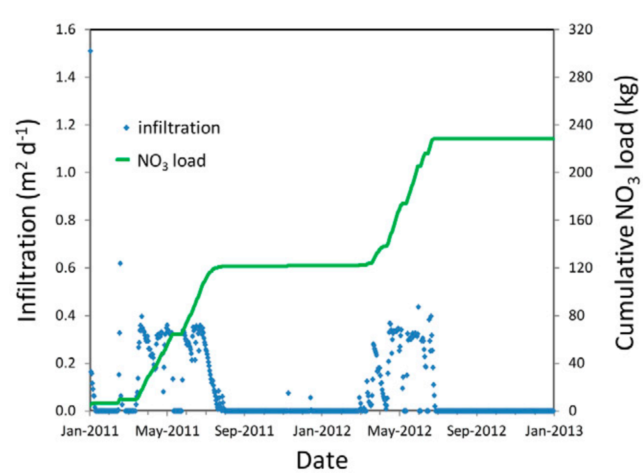

Fig. 4. Daily flow rate of field tile outlet water diverted into the buffer calculated as the difference between the measured flow over the first set of stoplogs and the flow over the second set of stoplogs in the control box (Fig. 2) and the cumulative load of NO_3 contained in the diverted water.

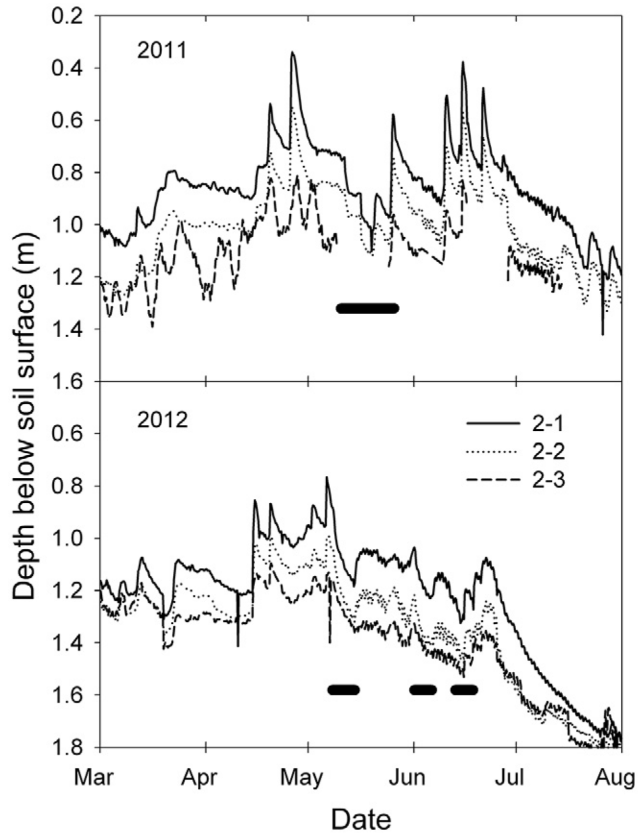

Fig. 5. Water table elevations along transect 2 within the riparian buffer for 2011 and 2012. Well 2–1 was closest to the distribution pipe, well 2–2 was in the middle of the buffer, and well 2–3 was closest to the stream. Heavy horizontal lines indicate times when drainage from the field outlet was not being diverted into the buffer.

the diversion. While the field tile was flowing, the water table within the buffer varied within a range of 70 to 120 cm deep, which had soil C contents of at least 1.45%. Tufekcioglu et al. (1999) measured considerable root density at these depths in a similar riparian buffer near this site that could facilitate NO_3 removal from the shallow groundwater within the buffer. Root biomass and root exudates could also serve as an important C source for denitrifiers within the buffer (Hill, 1996).

Nitrate Observations in Wells

The concentration of NO_3 diverted into the buffer was monitored along the four groundwater well transects within the buffer. Each transect spanned the distance from the distribution pipe along the top of the buffer to the stream (Table 2). Measurements of NO_3 concentrations in the diverted tile water ranged from 16.2 mg L^{-1} in 2012 to 9.3 mg L^{-1} in 2011. Measureable NO_3 was observed in the wells closest to the distribution pipe in all transects but only early in 2011 for transect 2. Nitrate concentrations in these downgradient wells were always lower than the concentration being diverted from the field tile (except for one observation on 16 June 2011 at Well 1–1 [Table 2]). Nitrate concentration within groundwater at the second well in each transect was almost always below detection, except for Transect 1. However, even for Transect 1, NO_3 concentrations in the second well were always below concentrations in the well nearest the distribution tile. The wells closest to the stream had no NO_3 concentrations above the detection limit. Thus, there was a strong decreasing trend in NO_3 concentration across the buffer at all times and locations.

Chloride concentrations in wells, field tile drainage, and Bear Creek ranged from 15.6 to 54.7 mg L^{-1} and averaged 25.6 mg L^{-1} (SE, 0.61 mg L^{-1}), which is very similar to concentrations found in a survey of 10 Iowa rivers from 2004 to 2008 (Garrett, 2012) and in nearby groundwater monitoring wells (Wineland, 2002). The primary source of Cl in this landscape is KCl, which is widely used as a potassium fertilizer in corn production. There were no significant differences ($p = 0.05$) in Cl concentrations across the transect from distribution pipe to stream. This lack of a trend in Cl concentrations across the buffer indicates that dilution within the buffer was not the cause of the decreasing NO_3 concentration trend across the buffer (Lowrance, 1992).

Discussion

Over 2 yr, we were able to redirect over 18,000 m^3 of flow from a field tile as subsurface flow along 335 m of an existing riparian buffer. This flow represented about 55% of the total flow coming from a tile outlet draining 10.1 ha of a field in corn–soybean. The redirected water seeped through the 20-m-wide buffer, raising the water table approximately 30 to 40 cm. The redirected tile flow contained 228 kg of NO_3, and, on the basis of the strong decrease in NO_3 concentrations within the shallow groundwater across the buffer, we conclude that all of this NO_3 was removed within the buffer and did not enter the stream. Thus, over 2 yr, the saturated buffer removed 228 kg N of NO_3 that otherwise would have entered Bear Creek as tile discharge. Materials and labor for installing the control box and additional tile were US$3500. With an estimated life expectancy of 20 yr and an interest rate of 4%, the total cost of implementing the practice in an existing riparian buffer is US$4960, or US$248 yr^{-1}. Given the observed NO_3 removal rate, the cost of removal for a kg of N is US$2.17 kg^{-1}. This compares favorably to other NO_3 remediation practices, such as wetlands (US$3.26 kg^{-1}) or rye cover crops (US$11.06 kg^{-1}) (Schipper et al., 2010).

The control box used here for diverting water into the buffer may not be feasible in nearly level fields because raising the water table within the buffer would also raise the water table within the field and negate the purpose of draining the field. However, in many areas of Iowa and other midwestern states, drainage systems are installed in poorly drained upland soils, and there is frequently a few meters of elevation relief between these drained soils and the stream outlet. This elevation difference presents the opportunity to raise the water table within a streamside buffer without affecting the drainage within the field.

In our design, tile flow that did not infiltrate the buffer was discharged directly into the stream. We feel that this is an important feature for this practice because water in the field was not being backed up in the field. Thus, field drainage was not affected; this is a critical consideration for farmers who rely on subsurface drainage to maintain a well-aerated root zone for their crop and are leery of any changes that may restrict field drainage.

We observed a rapid decrease in NO_3 concentrations within the groundwater as it entered the buffer. This pattern has been observed in many other riparian buffers (Cooper, 1990; Simmons et al., 1992; Ranalli and Macalady, 2010) and indicates that these systems have a large capacity for NO_3 removal. Using only the wells with NO_3 concentrations above the detection limit, we computed a NO_3 removal rate of (mean ± SE) 1.01 ± 0.07 mg N L^{-1} m^{-1} within the buffer. This is about 2.5 greater than the average rate found by Mayer et al. (2007) for 53 studies of subsurface flow through riparian buffers. The higher rate found here may be a result of saturating a 30- to 40-cm section of the buffer soil high in soil organic matter and roots that would normally be above the water table.

Denitrification permanently removes NO_3 from the system and is the most desirable mechanism for NO_3 removal. Although we did not measure the fate of the lost NO_3, most studies of riparian buffers have found denitrification to be the main removal mechanism when the water table is present in a high-organic-matter soil layer (Simmons et al., 1992; Ranalli and Macalady, 2010). Although denitrification represents a permanent sink for NO_3 (i.e., the atmosphere), incomplete denitrification could lead to the formation and release of N_2O, a powerful greenhouse gas that would not be desirable from a global warming perspective. However, any N_2O produced during denitrification within the buffer would be balanced somewhat by reductions in N_2O production in the riverine system if the NO_3 had entered the stream with the field tile discharge (Mosier et al., 1998).

Groffman et al. (1996) found that microbial immobilization was not an important removal mechanism of NO_3 in a forested buffer. In contrast, a number of studies have shown that plant uptake can be important, especially during times when the water table is seasonally low (Groffman et al., 1992; Simmons et al., 1992). Uptake and sequestration by riparian plants or immobilization by soil bacteria would temporally remove the NO_3 from water and store it within the buffer as organic N. However, a fraction of this stored N would eventually be remineralized through decomposition and could enter the stream as organic or inorganic N, and thus uptake and sequestration would only delay the movement of N to the stream. If sequestration is an important pathway for NO_3 removal, then biomass harvesting, perhaps as a bioenergy crop, could be pursed to maintain the long-term viability of this N removal pathway.

Success in removing NO_3 from subsurface field drains by rerouting some of the flow through riparian buffers depends on the soil properties within the buffer, particularly soil permeability, organic C content, and proximity of the water table to organic-rich material. This initial study has shown great potential for reestablishing the hydrologic connection between riparian buffers and drained croplands and removing NO_3 before it can enter surface waters. If the success observed here can be replicated in other landscapes, this practice could be used to prevent substantial amounts of NO_3 from entering surface streams throughout the midwestern United States.

Table 2. Nitrate concentrations redirected into the redistribution tile from the field and observed in wells along four transects between the field-edge of the buffer and Bear Creek.

Date	Input from field tile	Transect–well number											
		1–1	1–2	1–3	2–1	2–2	2–3	3–1	3–2	3–3	4–1	4–2	4–3
		Distance from distribution pipe, m											
		5.7	12.7	18.9	5.7	12.9	21.4	6.6	14.1	22.9	6.0	14.1	22.2
		NO_3 concentration, mg N L^{-1}											
28 Feb. 2011	9.8	7.9	<0.3†	<0.3	0.8	<0.3	<0.3	4.1	<0.3	<0.3	1.8	5.1	<0.3
17 Mar. 2011	9.3	8.9	0.5	<0.3	0.4	<0.3	<0.3	6	<0.3	<0.3	1.3	<0.3	<0.3
20 Apr. 2011	10.1	8.1	<0.3	<0.3	<0.3	<0.3	<0.3	4.8	<0.3	<0.3	3.7	0.8	<0.3
3 May 2011	11	8	1.6	<0.3	<0.3	<0.3	<0.3	2.5	<0.3	<0.3	2.5	<0.3	<0.3
19 May 2011	11.6	8.2	1.4	<0.3	<0.3	<0.3	<0.3	2.3	0.7	<0.3	1.9	<0.3	<0.3
3 June 2011	10.9	7.7	4.8	<0.3	<0.3	<0.3	<0.3	2.3	<0.3	<0.3	2.9	<0.3	<0.3
16 June 2011	11.8	13.1	3.6	<0.3	<0.3	<0.3	<0.3	1.5	<0.3	<0.3	4.4	<0.3	<0.3
28 June 2011	11.1	7.2	2.4	<0.3	<0.3	<0.3	<0.3	1.5	<0.3	<0.3	3.1	<0.3	<0.3
14 July 2011	13	8.2	3.8	<0.3	<0.3	<0.3	<0.3	3.1	<0.3	<0.3	5.1	<0.3	<0.3
26 July 2011	11.9	7.7	5.5	<0.3	<0.3	<0.3	<0.3	4.6	<0.3	<0.3	2.5	<0.3	<0.3
27 Mar. 2012	14.1	3.8	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
2 Apr. 2012	13.2	6.9	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
10 Apr. 2012	13.4	4.6	0.8	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
16 Apr. 2012	15.1	6.1	0.9	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
23 Apr. 2012	14.9	8.4	0.5	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
30 Apr. 2012	13.9	8.6	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	0.3	<0.3	<0.3
7 May 2012	15.9	9.7	0.5	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	0.6	<0.3	<0.3
14 May 2012	14.7	8.4	2.4	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	0.6	<0.3	<0.3
21 May 2012	16.3	9.8	3.1	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	0.5	<0.3	<0.3
29 May 2012	14.6	9.6	3.6	<0.3	<0.3	<0.3	<0.3	0.6	<0.3	<0.3	0.6	<0.3	<0.3
4 June 2012	15.8	10.6	4.7	<0.3	<0.3	<0.3	<0.3	2.7	<0.3	<0.3	2.4	<0.3	<0.3
11 June 2012	14.3	8.8	6	<0.3	<0.3	<0.3	<0.3	2.7	<0.3	<0.3	3	<0.3	<0.3
18 June 2012	16.2	14	5.6	<0.3	<0.3	<0.3	<0.3	4.7	<0.3	<0.3	1.2	<0.3	<0.3
25 June 2012	NS‡	12.9	7.3	<0.3	<0.3	<0.3	<0.3	3.8	<0.3	<0.3	1	<0.3	<0.3

† Detection limit was 0.03 mg N L^{-1}

‡ No sample.

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