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The role of seepage in constructed wetlands receiving agricultural tile drainage

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Abstract

Constructed wetlands positioned in the landscape between row crop agriculture and surface waters can be used to intercept tile drainage and serve as agricultural waste water detention basins. A potential exit pathway in constructed wetlands for detained water and possibly NO_3^--N is via seepage through and under an earthen berm. The objective of this study was to determine if seepage was an important pathway for NO_3^--N transport from two constructed wetlands receiving tile drainage from adjacent agricultural land (wetland A, surface area of 0.6 ha; wetland D, 0.78 ha). A mean apparent hydraulic conductivity (K) was calculated (10.8 cm h^{-1}, range 8.2–14.3 cm h^{-1}) using empirical water budgets. Using Darcy's law, which included the apparent hydraulic conductivity, effective seepage area and daily hydraulic gradient measurements, daily seepage volumes for both wetlands were calculated for the 1997 water year. Total seepage volumes for wetlands A and D were 26 and 22 million liters, respectively, for the 1997 water year, which represented 47 and 27% of the total inlet flow. The amount of NO_3^--N exiting wetlands A and D in seepage water was estimated to be 61 and 25 kg N, respectively, and represented 10 and 4% of the total inlet NO_3^--N load. Seepage connected the wetland with the riparian buffer strip and transported NO_3^--N to populations of denitrifiers deeper in the sediment profile and outside the wetland perimeter, thereby enhancing overall NO_3^--N removal efficiencies.

Keywords: Hydraulic conductivity; Nitrate; Tile drains; Water quality

1. Introduction

Constructed wetlands have been shown to be effective in the treatment of an assortment of polluted water sources (Hammer and Bastian,

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1993), and have also been incorporated into farm management practices for treatment of animal waste, such as effluent from swine operations (Hammer et al., 1993). Two advantages of constructed wetlands are: (1) relatively low construction costs and (2) low operating costs (Hammer et al., 1993). Recent research has attempted to evaluate constructed wetlands and their potential for treating water discharged from agricultural drainage tiles (Crumpton et al., 1993; Gersberg et al., 1983; Higgins et al., 1993). Although artificial drainage of natural wetlands has provided a suitable environment for productive farmland, it has uncoupled the filtering effect of riparian zones, allowing agricultural drainage waters to be shunted directly into surface waters.

Wetlands have been constructed within the Embarras River floodplain, in east-central Illinois, in areas where they once existed before artificial drainage (David et al., 1997b). These constructed wetlands intercept and temporarily detain tile drainage water from adjacent agricultural fields until wetland storage capacity is exceeded and water overtops the outlet weir crest. Until storage capacity is exceeded, agricultural drainage water is slowly released to the river through a small orifice in the outlet weir structure (Konyha et al., 1995), or by seeping through a 15.3-m wide riparian buffer strip. Constructed wetlands provide a suitable environment for the removal of NO_3^--N from drainage water through the processes of denitrification and plant uptake (Xue et al., 1999). Although studies have demonstrated that constructed wetlands can remove NO_3^--N from agricultural runoff, only a limited number of investigations have been conducted to determine their efficiency and practical use in the intensively tiled drained agricultural Midwest.

Brodie (1993) identified factors in site selection that can play an important role in the overall effectiveness of a constructed wetland. Considerations for site selection should account for topography, ground water hydrology and the existence of an underlying aquiclude. To reduce costs, wetlands can be built utilizing resources from the site and do not require the installation of expensive impermeable liners to prevent seepage. Lateral seepage of wetland water through the riparian buffer increases overall hydraulic loading capacity of this tandem system (wetland–riparian buffer) and may transport some NO_3^--N to populations of denitrifiers and plants outside the wetland perimeter. The objective of this study was to determine the daily seepage rate from two constructed wetlands that receive agricultural tile drainage and determine the mass of NO_3^--N contained in seepage water. This information was necessary to determine the overall NO_3^--N removal efficiencies of each wetland.

2. Materials and methods

2.1. Site description

Four wetlands (A, B, C and D) were constructed in 1994 in Champaign County, IL; only wetlands A and D were evaluated during the 1997 water year (1 October 1996 through 30 September 1997) in this study. The wetlands were created in pairs and positioned in the floodplain between the upland row crop and the Embarras River to intercept field drainage tiles, prior to discharge into the river. Earthen berms were used to create a detention structure and form the boundaries of each wetland. Prior to berm construction, a trench 0.9 m deep by 1.2 m wide was excavated along the center of the berm designed to retard seepage below the berm. The berms were set 15.3 m from the river to form a riparian buffer strip through which wetland seepage water would slowly migrate to the river. Soil media for the berms was obtained by using excavation equipment (bulldozer and scraper) to cut into the soil profile within each wetland. These excavated cuts were approximately 7.5 m wide, 7.5 m apart and 0.5–1.5 m deep. The soil from the excavation was mounded and a 'sheeps foot roller' was used to compact the berm to prevent erosion and decrease permeability. Berm lengths of wetlands A and D were 185 and 131 m, respectively. A partition berm separated wetland A from B and wetland C from D. The partition berm was positioned perpendicularly to the river and was not factored into the overall berm length. The surface areas and maximum volumes for wetlands A and D were 0.60 and 0.78 ha, and 5.4 and 5.2 million liters, respectively. Wetland D was located 600 m downstream from wetland A.

Field drainage tiles were intercepted and directed into each wetland using an inlet weir structure. Outlet weir structures were installed in each wetland at the opposite end away from the inlet. The inlet structures contained a weir plate with a slot and v-notch for low to moderate flow measurements and a rectangular crest for high flow (Gentry et al., 1998). The outlet structures were equipped with a flash board riser system which included an outlet orifice (3.81 cm^2) below a crest (Konyha et al., 1995). The distance between the bottom of the orifice and the crest (33 cm) served as detention storage volume of the wetland. These structures were equipped with automated samplers (ISCO, Inc., Lincoln, NE), pressure transducers and data loggers to constantly record flow and to provide samples for nutrient concentrations (Gentry et al., 1998). Inlet samples were collected on a flow proportional basis and outlet samples were collected once every 12 h.

Inlet tiles originated in agricultural fields that were in a corn and soybean crop rotation. The tile for wetland A drained an estimated 15 ha and crossed two fields that were planted with corn in 1997. The tile for wetland D was estimated to drain a total of 25 ha; however, in 1997, only 5 ha were planted with corn and the remainder with soybean. All areas of the two tile systems that were planted with corn in 1997 received nitrogen (N) fertilization. On 27 January and 13 February 1997, N in the form of $(NH_4)_2SO_4$ fertilizer at a rate of 135 kg N ha^{-1} was applied to one of the two fields draining into wetland A (approximately 75% of the watershed). The remaining area of the watershed for wetland A received a 28% N solution of NH_4NO_3 and urea on 29 April at the same rate. The 5-ha portion of the watershed for wetland D received the $(NH_4)_2SO_4$ fertilizer on 27 January at a rate of 202 kg N ha^{-1} and the remaining area did not receive a N fertilizer. This fertilizer was rated as 6–0–0–6 (% N, P, K and S) and contained 7% dissolved organic carbon (DOC).

The site was also equipped with a weather station where precipitation was measured. An instream electronic staff gauge equipped with both a pressure transducer and data logger provided continuous river elevation. An estimate of evapotranspiration (ET) from the Illinois Climate Network in Champaign, IL, was used to calculate daily wetland evaporation loss.

2.2. Topographic and soil characteristics

This site is located within the Embarras River floodplain and prior to wetland construction was used for livestock pasture. David et al. (1997a) described the Embarras River watershed as very flat with 3–5 m of topographic relief from the upland to floodplain and a river slope of 0.1%. The soils are classified as a Colo series, fine-silty, mixed, mesic Cumulic Haplaquolls. These soils were high in organic matter with 8.9% in the top 10 cm declining to 4.9% at soil depths of 1 m (David et al., 1997b). Prior to wetland construction, floodplain soils were found to contain 19,700 kg N ha^{-1} and 236,000 kg C ha^{-1} in the upper 1 m of soil. This site contains common alluvial soils consisting of highly permeable sand layers deposited by sedimentation and river meandering. Layers of sand and gravel were found during well boring procedures at depths ranging from 0.5 to 2.5 m.

2.3. Monitoring well design and installation

Groundwater wells were installed to measure water table depths and to facilitate sampling of wetland seepage water. To estimate potential flow lines, each wetland area and riparian buffer strip was surveyed to produce topographic profiles. A total of ten wells were installed at wetland A, creating three transects approximately 50 m apart and perpendicular to the berm. Each transect included wells located within the wetland next to the berm (wetland wells), outside the wetland next to the berm (berm wells) and wells 3.0 m from the river (riparian wells). One transect contained two riparian wells located 3 m apart to provide duplication to evaluate variability in well construction, sampling procedures and chemical analysis. Wetland D had a total of five wells with four berm wells located approximately 30 m apart and one riparian well located midway along the berm.

The monitoring wells were designed and installed in June 1996 using parameters found in Nielson and Schalla (1991) and Kirkland et al. (1991). Wells were constructed from 5-cm diameter polyvinyl chloride pipes with 15-cm screens consisting of 0.1-cm slots. Bentonite was used to seal the annular space from the top of the gravel pack to the soil surface to prevent the vertical movement of water along the well casing.

In order to intercept estimated seepage flow lines, well depths increased from the wetland to the river. Average well depths at wetland A were 1.0, 1.6 and 2.3 m for wetland wells, berm wells, and riparian wells. For wetland D, average well depth for berm wells was 1.4 m and the riparian well was at a depth of 1.8 m.

2.4. Water sampling

Well, wetland, and river water samples were collected during the 1997 water year when the wetlands contained water (23 sampling days). Well sampling followed parameters in Kirkland et al. (1991). One day prior to sampling, well water levels were measured and the water purged, with sampling the following day. On well sampling days for both wetlands, inlet, outlet and wetland water grab samples were collected. Wetland water samples were collected at three points equidistant along the berm. Methods for all sample analysis followed guidelines in Clesceri et al. (1989). Samples for anions were filtered through a 1.2-μm glass-fiber filter and were frozen. Aliquots for DOC, NH_4^+-N and total N determination were filtered using a 1.2-μm glass-fiber filter, preserved with sulfuric acid (pH <2) and stored at 4°C.

Chemical analysis for water samples included NO_3^--N, chloride (Cl^-), sulfate (SO_4^{2-}), ammonium (NH_4^+), total N and DOC. Ion chromatography was used for analysis of anions. A Technicon Autoanalyzer was used to measure NH_4^+ and total N and a Dohrmann-Xertex DC-80 Analyzer was used to determine DOC. For total N, samples underwent persulfate digestion and were analyzed for NO_3^--N by cadmium reduction. A full QA/QC program was used including duplicates, internal QC samples, and external QC audit samples for all analyses. Duplicate variability and QC check samples were both $<5\%$.

2.5. Quantification of seepage water

The volume of seepage water exiting each wetland was determined by using a standard equation for water flow in a saturated soil (e.g. Freeze and Cherry, 1979; Jury et al., 1991):

$$K \times A \times i \tag{1}$$

where K is the apparent hydraulic conductivity, A is the total effective seepage area and i is the hydraulic gradient. Empirical water budgets were determined using inputs consisting of tile flow and direct precipitation and water outputs consisting of outlet flow and ET (David et al., 1997a). The wetland water budgets were found to be less accurate during periods with significant precipitation (>13 mm) due to the uncertainty of simultaneous inputs from direct rainfall and surface runoff. Therefore, five selected time periods without large rain events ranging from 5 to 12 days for wetland A were used to evaluate an apparent K. The relationship between wetland water level and volume was used to predict the change in wetland volume over time.

The seepage area for wetland A was determined using the length of the berm (185 m) and the depth of the soil profile to an impermeable layer (1.8 m) (Kurien et al., 1997). A length of 0.61 m was added to the porous vertical soil depth of wetland A to adjust for an increase in the effective permeable soil caused by standing water within the wetland giving a total effective seepage area (A) of 445 m^2. The hydraulic gradient (i) was measured using daily wetland and river water elevations divided by the measured distance between the two bodies of water (18.3 m). Using measured daily i and the determined A, the hydraulic conductivity value (K) was adjusted until the total predicted seepage volume for the time period equaled the missing value from the empirical water budget. The average of the five adjusted K values (apparent K) was then used in the equation for water flow to determine daily seepage volumes for both wetlands.

2.6. Quantification of NO_3^--N losses through seepage

Nitrate flux from the wetland in seepage was determined by multiplying the average well NO_3^--N concentration, for a given set of wells, with daily seepage rates. Daily, monthly and seasonal N loads were calculated assuming that measured NO_3^--N concentration represented actual concentrations for variable periods of time before and after sampling. The average NO_3^--N concentration of the berm wells was used in determining total NO_3^--N load in seepage water for each wetland. The water chemistry of the duplicated riparian well at wetland A was always similar; therefore, the average of these two wells represented one riparian well value. Riparian wells were used to predict the NO_3^--N removal in the riparian buffer strip. Assuming that higher NO_3^--N concentrations may be found in areas of higher hydraulic conductivity, the highest well NO_3^--N concentration at each wetland was used to predict the worst case scenario for NO_3^--N exiting the wetland.

2.7. Chloride:nitrate ratio

Chloride (Cl^-) to NO_3^--N ratios were calculated for wetland inlets, wetland water and berm wells. The Cl^-:NO_3^--N ratio was calculated throughout the year for both inlets and wetland water; however, for wells, ratios were calculated only in the winter and early spring until NO_3^--N concentrations decreased below detection limits (0.1 mg NO_3^--N l^{-1}). At least one riparian well at wetland A had NO_3^--N concentrations below detection limits before 6 February and the riparian well at wetland D before 8 April, therefore ratios were not calculated for riparian wells.

3. Results

3.1. Water budgets

During the 1997 water year, the wetland area and associated agricultural watershed received 931 mm of precipitation. Tile inlet flows during the water year were 55 and 81.6 million liters for wetlands A and D, respectively (Table 1). Of the total inlet flow 77 and 96% occurred during the winter (January–March) and spring (April–June) months for wetlands A and D, respectively (Table 1). The highest daily inlet flows observed for wetland A was 1.9 million liters on 13 June 97 and for wetland D 3.4 million liters on 27 February 97 (Fig. 1). Although the highest tile flows did not coincide, these 2 days produced the largest flow volumes for both wetlands for the year. Outlet flow for both wetlands began on 21 January 1997 (Fig. 1) and totaled 28 and 76 million liters for wetlands A and D, respectively (Table 1). During high flow events, daily outlet flow volumes equaled inlet flow volumes

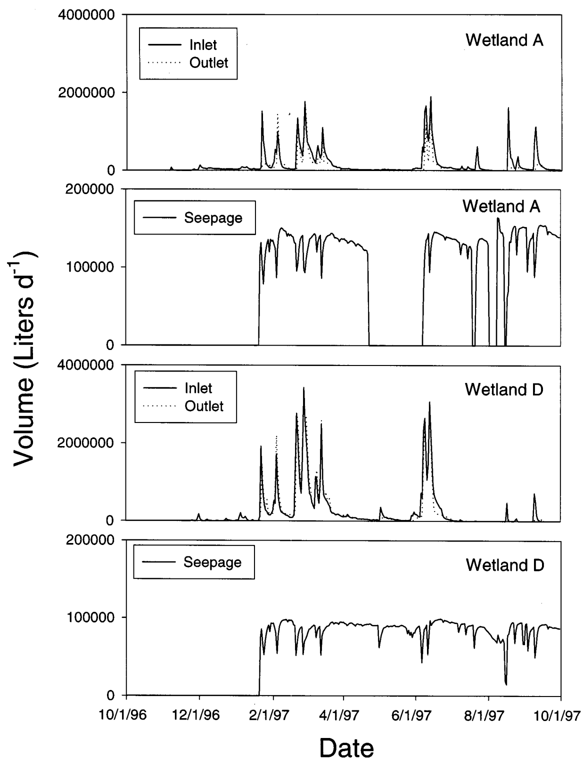

Fig. 1. Daily water flow for tile inlets, outlets and seepage for wetlands A and D during the 1997 water year. Water year begins 1 October of the previous year and ends 30 September of the identifed year.

Table 1
Seasonal inlet, outlet and seepage flow and forms of N along with total N for wetlands A and D during the 1997 water year[a]

Wetland and season	Inlet				Outlet					Seepage	
	Flow (million liters)	NO_3^--N (kg N)	NH_4^+-N (kg N)	Total N (kg N)	Flow (million liters)	NO_3^--N (kg N)	NH_4^+-N (kg N)	Organic N (kg N)	Total N (kg N)	Flow (million liters)	NO_3^--N (kg N)
Wetland A											
Fall	2	15	0	15	0	0	0	0	0	0	0
Winter	27	265	137	403	19	132	61	18	212	9	40
Spring	15	244	4	248	7	89	0	1	90	6	21
Summer	11	111	0	111	2	4	1	2	7	11	0.3
Total	55	635	141	777	28	225	62	21	309	26	61.3
Wetland D											
Fall	0.6	3	0	3	0	0	0	0	0	0	0
Winter	51	291	31	323	54	237	16	23	276	6	20
Spring	28	270	2	272	21	126	2	20	147	8	5
Summer	2	14	0	14	1	1	0	1	1	8	0.1
Total	81.6	578	33	612	76	364	18	44	424	22	25.1

[a] Seasons are Fall (October, November, December), Winter (January, February, March), Spring (April, May June) and Summer (July, August, September).

as a result of high inlet flow overwhelming the wetland storage capacities.

3.2. Apparent K determination

An apparent hydraulic conductivity was calculated using empirical water budgets from wetland A. Missing water volumes ranged from 0.98 to 1.5 million liters for the five time periods and produced an apparent mean K of 10.8 cm h^{-1} (S.D. 2.5). The range of apparent K values for the five periods was 8.2–14.3 cm h^{-1} with higher values in the summer than winter or spring. This mean apparent K was used in the standard flow equation to calculate daily seepage volumes for both wetlands.

3.3. Estimated daily seepage

Daily seepage volumes varied throughout the year and never exceeded 164,000 l day^{-1} for wetland A or 100,000 l day^{-1} for wetland D (Fig. 1). Total seepage for wetland A was 26 million liters and for wetland D was 22 million liters (Table 1). The lower daily seepage volumes for wetland D were due to a smaller effective seepage area of 276 m^2 caused by a shorter berm length and shallower wetland water depth. Daily seepage volumes for both wetlands declined after major rain events as river heights increased. Following these events, seepage rates quickly began to increase as river heights declined and wetland water elevations remained high due to high tile flow rates.

Although wetland seepage rates followed similar trends, there was a dry weather period when the pattern of seepage differed between the two wetlands. Daily seepage volumes for wetland A decreased from 143,000 l day^{-1} on 22 March to 116,000 l day^{-1} on 22 April. On 22 April, wetland water height declined below the pressure transducer height at the outlet of wetland A. At this time only two small areas of water remained and the wetland was considered empty which marked the cessation of seepage. Unlike wetland A, wetland D did not become empty and seepage rates remained nearly constant at 90,000 l day^{-1} (Fig. 1).

3.4. Wetland water nutrients

Inlet NO_3^--N concentrations ranged from < 0.1 to 52 mg NO_3^--N l^{-1} and from < 0.1 to 25 mg NO_3^--N l^{-1} for wetlands A and D, respectively. The highest NO_3^--N concentration found in both tiles occurred on 6 June and the low on 31 January. Typical NH_4^+, SO_4^{2-} and DOC concentrations in the tile inlets were similar, following seasonal patterns, and ranged from below detection (0.05) to 0.3 mg NH_4^+-N l^{-1}, 10–35 mg SO_4-S l^{-1} and 1–6 mg C l^{-1}. However, as a result of timing of the $(NH_4)_2SO_4$ fertilizer application and environmental conditions, high concentrations of nutrients entered wetlands A and D through inlet tile flow during the winter. During a snow melt event on 31 January, tile water transported NH_4^+-N, SO_4^{2-}-S and DOC concentrations as high as 278, 352 and 326 mg l^{-1} and 19, 95 and 43 mg l^{-1} for wetlands A and D, respectively.

Total N inputs from inlet flow for the 1997 water year were 777 kg N for wetland A and 612 kg N for wetland D (Table 1). Total N inputs were highly dependent on tile flow rates and ranged from 0 to 30 and 0 to 33 kg N day^{-1} for wetlands A and D, respectively. Of the total N entering the wetland through inlet flow, 84% for wetland A and 97% for wetland D occurred in the winter and spring months (Table 1). The major form of N entering both wetlands was NO_3^--N, equaling 82 and 95% of the total N for wetlands A and D, respectively. The remainder of N entered the wetlands as NH_4^+-N in January and February with the greatest single day input of 29 kg on 31 January 1997 for wetland A (Fig. 2). This was the only event during the water year that NH_4^+-N inputs were greater than NO_3^--N for either wetland. Organic N in tiles was found to be negligible.

During the 1997 water year, total N export (NO_3^--N + NH_4^+-N + organic N) through outlet flow was 309 kg for wetland A and 424 kg for wetland D (Table 1). Daily outlet export of N varied from 0 to 24 and from 0 to 21 kg N for wetland A and D, respectively (Fig. 2). Similar to seasonal inputs of N, nearly all of the outlet export occurred in the winter and spring months,

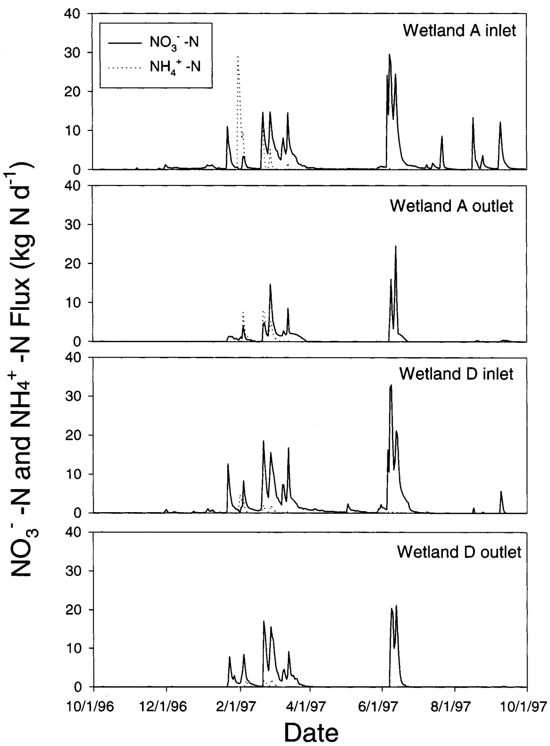

Fig. 2. Daily NO_3^--N values and NH_4^+-N flux in tile inlets and outlets for wetlands A and D during the 1997 water year.

with 98% for wetland A and 99% for wetland D (Table 1). The major form of N exiting the wetlands was NO_3^--N, with 73% for wetland A and 86% for wetland D. The remainder of N leaving the wetlands was divided into organic N and NH_4^+-N (Table 1). During high inlet flow when the wetland storage capacities were overwhelmed, daily total N export equaled daily N inputs as was seen on 13 June 1997 for both wetlands (Fig. 2). However, on 4 February outlet flow of both wetlands exceeded inlet flow due to surface runoff from a precipitation and snowmelt event.

3.5. Seepage water nutrients

Throughout the 1997 water year, there were negligible differences in nutrient concentrations between wetland water and water from wells within wetland A (wetland wells); therefore, water chemistry data from the wetland wells were not reported. The average NO_3^--N concentrations for the berm wells at wetland A ranged from <0.1 to 7.6 mg N l^{-1} (Fig. 3). Following the pulse of $(NH_4)_2SO_4$ fertilizer through wetland A, higher SO_4^{2-}-S and DOC concentrations were observed in berm wells with the largest influence on the well closest to the inlet. Well SO_4^{2-}-S concentrations immediately before and approximately 2 months after the nutrient pulse were typically 15–25 mg SO_4^{2-}-S l^{-1}; however, in late February concentrations increased to 62, 38 and 33 mg SO_4^{2-}-S l^{-1} for the three berm wells. DOC concentrations, which were generally below 3 mg C l^{-1}, increased for the three berm wells, and on 20 February 1997 the berm well closest to the inlet was 7.1 mg C l^{-1}. Conversely, NO_3^--N concentrations decreased for all wells at wetland A following 31 January and the berm well closest to the inlet was <0.1 mg N l^{-1} for three sampling days in February.

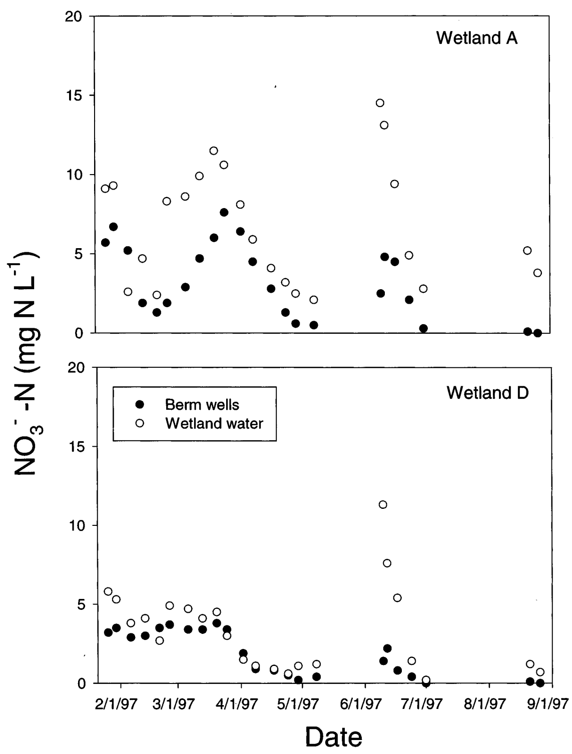

Fig. 3. Berm well and wetland water NO_3^--N concentrations for wetlands A and D during the 1997 water year. Also shown are data for the berm well with highest NO_3^--N concentration.

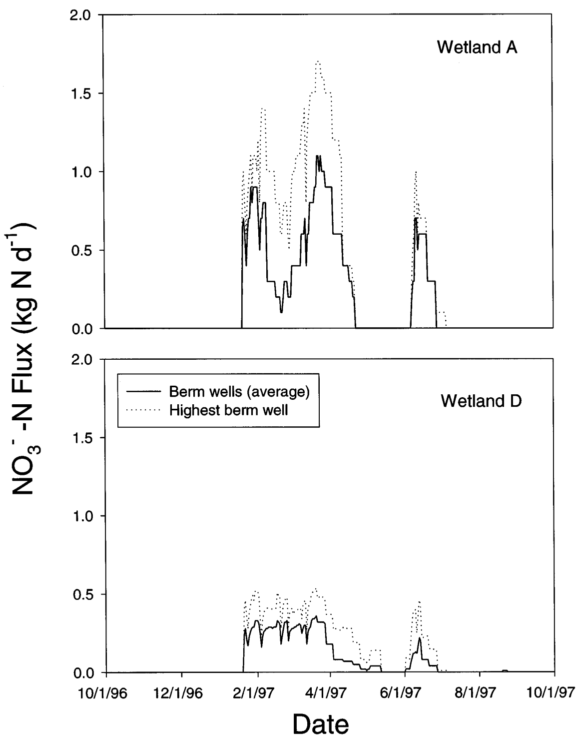

Fig. 4. Daily NO_3^--N flux in seepage water using average concentrations for berm wells compared with the berm well containing the highest NO_3^--N concentration for wetlands A and D during the 1997 water year.

The average daily well NO_3^--N concentrations at wetland A were usually lower than the wetland water and followed a similar pattern throughout the year (Fig. 3). During the study period, the concentrations of the berm wells ranged from <0.1 to 8.5 mg N l^{-1} and the riparian wells ranged from <0.1 to 8.0 mg N l^{-1}. The highest NO_3^--N concentration for either berm or riparian well occurred on 30 January. The berm well closest to the inlet typically had the highest NO_3^--N concentration of the three wells and was used to predict the worst case scenario for N exiting the wetland through seepage (Fig. 4). The daily average NO_3^--N concentration of the three riparian wells never exceeded the daily average wetland water concentration, and these concentrations were used to predict the potential effect of the riparian strip on additional NO_3^--N removal.

Although some $(NH_4)_2SO_4$ fertilizer did enter wetland D, the lower inlet and wetland nutrient concentrations did not appear to affect well samples because well SO_4^{2-}-S concentrations remained constant between 15 and 25 mg l^{-1} throughout the water year. Berm wells at wetland D had average NO_3^--N concentrations that ranged from 0.3 to 3.8 mg l^{-1}, whereas the concentration of wetland water on well sampling days ranged from 0.2 to 11.3 mg l^{-1} (Fig. 3). As in wetland A, both the wetland water and well concentrations followed the same pattern and the highest NO_3^--N concentrations were found in the berm well closest to the inlet. Therefore, this well was used to predict the worst case scenario for N exiting wetland D through seepage (Fig. 4). It generally had NO_3^--N concentrations higher than the average wetland water, except for an event in August. The riparian well at wetland D had NO_3^--N concentrations that ranged from 0.2 to 4.6 mg N l^{-1}. This well was used to predict the effect that the riparian strip had on NO_3^--N removal.

Using the average NO_3^--N concentrations of the berm wells at wetland A, it was estimated that 61.3 kg of NO_3^--N exited the wetland through seepage during the 1997 water year (Table 1). Using the average concentrations of the riparian wells, only 33 kg NO_3^--N were exported in seepage through both the berm and riparian buffer strip. Of the 410 kg of NO_3^--N that were retained (inlet NO_3^--N minus outlet NO_3^--N) in wetland A during the 1997 water year, 15% exited the wetland through seepage. Using the well closest to the inlet for a worst case scenario gave a total export of 109 kg N, equaling 27% of the retained N. Using the riparian wells lowered the percentage of retained N exported through seepage to 8%. Regardless of the estimate used, the highest percentage of retained N that was exported through seepage occurred during the winter and spring months (Table 1).

For wetland D, the average NO_3^--N concentration of the berm wells predicted a total export of N through seepage of 25.1 kg which represented 12% of the 214 kg retained in wetland D (Table 1). Using the highest well concentrations from the berm well closest to the inlet, total export through seepage was 45 kg N, or 22% of the retained N. The riparian well NO_3^--N concentrations predicted N export through seepage after the riparian strip of 20 kg. Similar to wetland A, the majority of the retained N exported through seepage was during the winter and spring months (Table 1).

Inlet flow weighted mean NO_3^--N concentrations for the 1997 water year were 11.2 and 7.1 mg N l^{-1} for wetlands A and D, respectively. Outlet flow weighted mean NO_3^--N concentrations were 7.5 and 4.5 for wetlands A and D, respectively. When comparing the average NO_3^--N concentration for only the 23 sampling days, for both wetlands: inlet > wetland water > outlet > berm wells > riparian wells. For wetland A, average NO_3^--N concentrations for wetland water, berm wells and riparian wells for the 23 sampling days were 6.7, 3.1 and 1.6 mg N l^{-1}, respectively, and for wetland D these averages were 3.5, 2.0 and 1.3 mg N l^{-1}. The average river NO_3^--N concentration on the 23 sampling days was 8.5 mg N l^{-1}.

3.6. Wetland nitrate removal efficiencies

Nitrate removal percentages for wetland A during the 1997 water year ranged from 47 to 60% using the various estimates of well NO_3^--N concentrations (Table 2). Berm well concentrations predicted seasonal wetland NO_3^--N removal percentages of 100, 35, 55 and 96% for the fall, winter, spring and summer months, respectively, with a total removal of 55% (Table 2). Seasonal and annual removal percentages were increased when using the riparian well NO_3^--N concentrations, with the largest increase observed in the winter.

Wetland D had lower overall NO_3^--N removal percentages than wetland A for the 1997 water year. For wetland D, the total NO_3^--N removal efficiency was 33%, which included seasonal values of 100, 12, 51 and 92% for the fall, winter, spring and summer months, respectively (Table 2). Using concentrations in the well closest to the inlet, annual NO_3^--N removal percentages decreased to 28% for wetland D. Based on the riparian well data, seasonal NO_3^--N removal percentages were increased in the winter and spring months.

3.7. Chloride:nitrate ratio

Inlet water Cl^-:NO_3^--N ratio was consistent within a given wetland; however, the average ratio was lower for wetland A than for wetland D (3.3 vs 7.4). The difference was due to higher Cl^- concentrations in wetland A inlet and lower NO_3^--N concentrations in the inlet of D. This difference was also reflected in the Cl^-:NO_3^--N ratios of the wetland water; however, the Cl^-:NO_3^--N ratio was usually higher for wetland water compared with inlet water. Cl^-:NO_3^--N ratios for both wetlands were always higher in well samples than in wetland water samples except for two time periods in late March for wetland D. For the ten sampling dates when all berm wells at wetland A had detectable NO_3^--N levels, the average Cl^-:NO_3^--N ratio was 10.1 compared with a ratio of 6.0 for wetland water during that same time. For wetland D, there were 11 sampling dates where a ratio could be calculated and the Cl^-:NO_3^--N ratio was 17.1 compared with 13.9 for the wetland water. An increase in the Cl^-:NO_3^--N ratio in either wetland water or well water indicates that NO_3^--N was biologically removed and was not a result of dilution.

Table 2
Percent removal of NO_3^--N during the 1997 water year for two constructed wetlands[a]

Wetland and season	NO_3^--N removal (%)		
	Berm wells	Highest	Riparian wells
Wetland A			
Fall	100	100	100
Winter	35	21	43
Spring	55	51	59
Summer	96	95	95
Total	55	47	60
Wetland D			
Fall	100	100	100
Winter	12	8	12
Spring	51	48	53
Summer	92	92	93
Total	33	28	34

[a] Seasons are Fall (October, November, December), Winter (January, February, March), Spring (April, May, June) and Summer (July, August, September). Removal percentages are given for inlet and outlet alone and with various seepage NO_3^--N values.

4. Discussion

The higher hydraulic gradient, larger effective seepage area and longer berm length of wetland A produced greater daily seepage rates than wetland D. Daily seepage volumes, however, varied little during the 1997 water year for both wetlands, compared to the flashy characteristics of daily inlet and outlet flow. The loss of water from wetland A as a result of seepage represented 47% of the inlet flow during the 1997 water year, whereas in wetland D the volume of seepage water was equal to only 27% of the inlet flow. Rushton (1996) determined that 31% of the annual water lost from a wetland exited through seepage, which lowered net water export through outlet flow. The loss of water from wetlands A and D through seepage lowered outlet flow and increased hydraulic loading capacity within the wetland–riparian system.

Wetland water retention time, although not quantitatively determined, was strongly influenced by inlet flow and wetland storage capacity. Wetlands A and D have similar water storage volumes, but 1997 yearly inlet flow for wetland D was 33% higher than wetland A. Wetlands need to have sufficient water storage capacity to provide proper hydrological conditions to facilitate microbial processes (Kadlec and Knight, 1996). Nichols (1983) found that increased hydraulic loading of wetlands decreased N removal efficiencies. The flashy flow characteristics of the inlet tiles for wetlands A and D often exceeded storage capacities and small increases in seepage rates were unable to offset these large flow events.

The smaller capacity of wetland D in relation to the high inlet flow is a design parameter that decreased water retention time. Ratios for agricultural drainage area to wetland surface area for wetlands A and D were 25:1 and 32:1, respectively. In addition, comparing wetland surface area to wetland volume showed that the average depth of wetland A was 35% greater than wetland D (90 vs 67 cm) at full volume. These two factors influenced residence time and limited the ability of wetland D to retain a large percentage of its total inlet flow.

The small difference between yearly inlet (81.5 million liters) and outlet (76 million liters) values for wetland D suggest that it was undersized for its total inlet flow volume (Table 1). However, there was also evidence that the water budget of wetland D was influenced by an adjacent wetland. Using the apparent K from wetland A with the hydraulic gradient at wetland D revealed that an extra 16 million liters of water exited wetland D through both seepage and outlet flow. Water was apparently seeping through the partition berm that divides wetland D from wetland C. The slightly higher water height of wetland C was creating a positive pressure gradient toward wetland D and we discovered the effective width of the partition berm had been decreased by burrowing muskrats. The additional water from wetland C increased outlet flow for wetland D and further reduced its ability to retain inlet tile water. This problem did not allow a water budget approach to be used to estimate an apparent K for wetland D.

The range in apparent K values calculated during the five time periods showed a trend related to season. Three of the five time periods were in the late winter and early spring and the other two were in mid-summer. Results from the empirical water balance showed that seepage rates were higher in the summer than in the winter or early spring. Similar seasonal variations have been reported in the literature. For example, Asare et al. (1993) measured lower field saturated hydraulic conductivity and matrix flux potential in the spring, as compared with summer, on a silt loam soil. They attributed the higher values in the summer to drier conditions that create more macropores. We speculate that hydraulic conductivity may increase in the summer as flow paths are created by channels from roots and burrowing insects as well as a lower viscosity of water.

Although seepage of water out of the wetlands and into the riparian buffer strip increased overall hydraulic loading capacity of each wetland–riparian complex, it provided another route for NO_3^--N export. Seepage occurred whenever there was a sufficient hydraulic gradient present; however, NO_3^--N export occurred predominantly during the winter and spring months. It is during this time period when the greatest amount of inlet tile flow and N loading to both wetlands occurred (David et al., 1997a; Gentry et al., 1998).

Colder water and sediment temperatures decrease microbial activity and plant growth and lower wetland NO_3^--N removal efficiency. David et al. (1997b) demonstrated that both water and soil temperature could influence NO_3^--N removal within constructed wetlands. Pfenning and McMahon (1996) found that potential denitrification rates in a riverbed sediment were 75% lower at 4°C than at 22°C. Lower denitrification rates within wetlands A and D, during the winter and spring months, caused increased amounts of NO_3^--N to exit the wetland through seepage. Lowrance (1992) also found higher outputs of NO_3^--N in subsurface flow during the colder months. Decreases in NO_3^--N exiting with seepage water were observed in both wetlands as ambient and soil temperatures increased, which would coincide with higher denitrification rates and plant growth within the wetlands. During the summer months less than 1% of the NO_3^--N retained in the wetlands exited the system through seepage. The amount of NO_3^--N exiting wetlands A and D in seepage water during the 1997 water year was estimated to be 61 and 25 kg, respectively, which represented 10 and 4% of the total inlet NO_3^--N load.

Annual NO_3^--N removal efficiency was higher for wetland A than wetland D (55 vs 33%) during the 1997 water year. Wetland D probably experienced reduced water retention times and may have received additional NO_3^--N as a result of seepage water from wetland C. Although this likely decreased the removal efficiency of wetland D, we speculate that the higher removal efficiency for wetland A was due to its ability to retain a greater percentage of inlet flow, thereby enhancing the role of seepage in NO_3^--N removal. Seepage coupled the wetlands with the riparian buffer strip and carried NO_3^--N to populations of denitrifiers and plants outside the perimeter of the wetland. Without seepage more inlet water would exit the wetland, thus decreasing retention time and overall NO_3^--N removal efficiency.

The influx of $(NH_4)_2SO_4$ fertilizer in early February into the wetlands was not a common occurrence and had a larger impact on wetland A than on wetland D. Prior to the event, NO_3^--N concentrations in the wells paralleled NO_3^--N concentrations in the wetland water along the berm and were higher near the inlet source and lower toward the outlet. After this event, increases in concentrations of SO_4^{2-}-S and DOC were observed in the berm wells while NO_3^--N concentrations decreased, especially in the wells closest to the inlet. The elevated SO_4^{2-}-S and DOC concentrations indicated the duration of influence that the pulse of fertilizer had on the wetland–riparian system. Results indicated that the pulse of nutrients associated with the $(NH_4)_2SO_4$ fertilizer enhanced the NO_3^--N removal efficiency of wetland A.

The decrease in NO_3^--N concentration observed in the berm wells closest to the inlet and the reversal of the well NO_3^--N gradient along the berm from inlet to outlet was likely due to higher denitrification rates caused by the influx of DOC from the $(NH_4)_2SO_4$ fertilizer. This finding suggested that this available C may have increased microbial activity during colder temperatures. Burford and Bremner (1975) found higher denitrification rates in saturated soils were highly related to availability of water-soluble organic carbon. Well NO_3^--N concentrations began to increase gradually after the pulse of fertilizer, as the available carbon source was depleted.

The movement of water out of the wetlands and into the riparian buffer strip may be providing an environment for additional NO_3^--N removal. Once NO_3^--N exits the wetlands through subsurface flow, it enters the riparian system. Therefore, the berm wells add a small portion of the riparian component to the wetland efficiencies, whereas the riparian wells include the entire wetland–riparian system. The difference in NO_3^--N concentrations between the berm and riparian wells gave an indication that further NO_3^--N reduction occurred after water passed through the riparian buffer strip.

Many studies have shown that riparian systems have the ability to remove NO_3^--N from groundwater. Ambus and Christensen (1993) found the limiting factor of denitrification in a riparian system to be the vertical diffusion of NO_3^--N into the soil. The movement of NO_3^--N out of wetlands A and D, as the result of the hydraulic gradient, overcame the limitations of diffusion. Schnabel et al. (1996) found that denitrification occurred in a grassed riparian system and was higher than rates measured in a wooded site. Osborne and Kovacic (1993) and Maag et al. (1997) also concluded that riparian vegetation removed NO_3^--N from ground water. However, Hill (1996) reviewed the role of stream riparian zones, and noted that although many studies found reductions in groundwater NO_3^--N, the hydrologic flow paths were generally not known.

Water sampling of the berm wells was adequate to quantify seepage NO_3^--N and determine wetland NO_3^--N removal efficiencies. The increase in Cl^-:NO_3^--N ratios of well water when compared with wetland water indicates a biological removal rather than dilution of groundwater. After the pulse of the $(NH_4)_2SO_4$ fertilizer, the high SO_4^{2-}-S concentration in the berm wells at wetland A was considered a tracer of wetland water movement; however, the riparian wells did not show an increase in SO_4^{2-}-S concentration. Because SO_4^{2-}-S concentration did not increase in the riparian wells and they were located another 10 m farther from the wetland berm, the flow path of water to these wells was less certain. However, based on NO_3^--N concentrations in the river compared with concentrations in the riparian wells, none of the wells along the river at either wetland appeared to be influenced by river water. Results are promising that the riparian buffer strip provided suitable conditions for additional NO_3^--N removal, especially during the colder months. More intensive sampling of seepage water, however, is needed to verify the fate of water and NO_3^--N through the riparian buffer strip.

5. Conclusion

The additional loss of water from the constructed wetlands and into the riparian zone through seepage increased the capacity of the wetland to remove N and reduced NO_3^--N exported through outlet flow. Seepage connected the wetland with the riparian buffer strip, which helped to increase NO_3^--N removal efficiencies and may have provided a suitable environment for additional treatment.

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References

Ambus, P., Christensen, S., 1993. Denitrification variability and control in a riparian fen irrigated with agricultural drainage water. Soil Biol. Biochem. 25, 915–925.

Asare, S.N., Rudra, R.P., Dickinson, W.T., Wall, G.J., 1993. Seasonal variability of hydraulic conductivity. Trans. ASAE 36, 451–457.

Brodie, G.A., 1993. Selection and evaluation of sites for constructed wastewater treatment wetlands. In: Hammer, D. (Ed.), Constructed Wetlands for Wastewater Treatment. Lewis Publishers, Chelsea, MI, pp. 307–317.

Burford, J.R, Bremner, J.M., 1975. Relationships between the denitrification capacities of soils and the total, water-soluble and readily decomposable soil organic matter. Soil Biol. Biochem. 7, 389–394.

Clesceri, L.S., Greenberg, A.E., Rhodes Trussel, R. (Eds.), 1989. Standard Methods for the Examination of Water and Wastewater, 17th ed. American Public Health Association, American Public Health Association, American Water Works Association, and Water Pollution Control Federation, Washington, DC.

Crumpton, W.G., Isenhart, T.M., Fisher, S.W., 1993. Controlling agricultural runoff by use of constructed wetlands. In: Moshiri, A. (Ed.), Constructed Wetlands for Water Quality Improvement. Lewis Publishers, Ann Arbor, MI, pp. 283–291.

David, M.B., Gentry, L.E., Kovacic, D.A., Smith, K.M., 1997a. Nitrogen balance in and export from an agricultural watershed. J. Environ. Qual. 26, 1038–1048.

David, M.B., Gentry, L.E., Kovacic, D.A., Smith, K.M., 1997b. Carbon, plant and temperature control of nitrate removal from wetland mesocosms. Trans. Ill. Acad. Sci. 90, 103–112.

Freeze, R.A., Cherry, J.A., 1979. Groundwater. Prentice-Hall, Engelwood Cliffs, NJ.

Gentry, L.E., David, M.B., Smith, K.M., Kovacic, D.A., 1998. Nitrogen cycling and tile drainage nitrate loss in a corn/soybean watershed. Agric. Ecosyst. Environ. 68, 85–97.

Gersberg, R.M., Elkins, B.V., Goldman, C.R., 1983. Nitrogen removal in artificial wetlands. Water Res. 17, 1009–1014.

Hammer, D.A., Bastian, R.K., 1993. Wetland ecosystems: natural water purifiers? In: Hammer, D. (Ed.), Constructed Wetlands for Wastewater Treatment. Lewis Publishers, Chelsea, MI, pp. 5–19.

Hammer, D.A., Pullin, B.P., McCaskey, T.A., Eason, J., Payne, V.W.E., 1993. Treating livestock wastewaters with constructed wetlands. In: Moshiri, A. (Ed.), Constructed Wetlands for Water Quality Improvement. Lewis Publishers, Ann Arbor, MI, pp. 343–347.

Higgins, M.J., Rock, C.A., Wengrezynek, B., 1993. Controlling agricultural runoff by use of constructed wetlands. In: Moshiri, A. (Ed.), Constructed Wetlands for Water Quality Improvement. Lewis Publishers, Ann Arbor, MI, pp. 359–367.

Hill, A.R., 1996. Nitrate removal in stream riparian zones. J. Environ. Qual. 25, 743–755.

Jury, W.A., Gardner, W.R., Gardner, W.H., 1991. Soil Physics, 5th ed. Wiley, New York.

Kadlec, R.H., Knight, R.L., 1996. Treatment Wetlands. Lewis Publishers, New York.

Kirkland, S.D., Jones, R.L., Norris, F.A., 1991. Well installation and sampling procedures for monitoring groundwater beneath agricultural fields. In: Nash, R., Leslie, R. (Eds.), Groundwater Residue Sampling Design. ACS, pp. 214–221.

Konyha, K.D., Shaw, D.T., Weiler, K.W., 1995. Hydrologic design of a wetland: advantages of continuous modeling. Ecol. Eng. 4, 99–116.

Kurien, V.M., Cooke, R.A., Hirschi, M.C., Mitchell, J.K., 1997. Estimating drain spacing of incomplete systems. Trans. ASAE 40, 377–382.

Lowrance, R., 1992. Nitrogen outputs from a field-size agricultural watershed. J. Environ. Qual. 21, 602–607.

Maag, M., Malinovsky, M., Nielson, S.T., 1997. Kinetics and temperature dependence of potential denitrification in riparian soils. J. Environ. Qual. 26, 215–223.

Nichols, D.S., 1983. Capacity of natural wetlands to remove nutrients from wastewater. J. Water Pollut. Control Fed. 55, 495–503.

Nielson, D.M., Schalla, R., 1991. Design and installation of ground water monitoring wells. In: Nielsen, D. (Ed.), Practical Handbook of Ground Water Monitoring. Lewis Publishers, Chelsea, MI, pp. 239–331.

Osborne, L.L., Kovacic, D.A., 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biol. 29, 243–258.

Pfenning, K.S., McMahon, P.B., 1996. Effect of nitrate, organic carbon, and temperature on potential denitrification rates in nitrate-rich riverbed sediments. J. Hydrol. 187, 283–295.

Rushton, B., 1996. Hydrologic budget for a freshwater marsh in Florida. Water Resour. Bull. 32, 13–21.

Schnabel, R.R., Cornish, L.F., Stout, W.L., Shaffer, J.A., 1996. Denitrification in a grassed and a wooded, valley and ridge, riparian ecotone. J. Environ. Qual. 25, 1230–1235.

Xue, Y., Kovacic, D.A., David, M.B., Gentry, L.E., Mulvaney, R.L., Lindau, C.W., 1999. In situ measurements of denitrification in constructed wetlands. J. Environ. Qual. 28, 263–269.