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Effectiveness of Constructed Wetlands in Reducing Nitrogen and Phosphorus Export from Agricultural Tile Drainage

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ABSTRACT

Much of the nonpoint N and P entering surface waters of the Midwest is from agriculture. We determined if constructed wetlands could be used to reduce nonpoint N and P exports from agricultural tile drainage systems to surface waters. Three treatment wetlands (0.3 to 0.8 ha in surface area, 1200 to 5400 m^3 in volume) that intercepted subsurface tile drainage water were constructed in 1994 on Colo soils (fine-silty, mixed, superactive, mesic Cumulic Endoaquoll) between upland maize (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.] cropland and the adjacent Embarras River. Water (tile flow, precipitation, evapotranspiration, outlet flow, and seepage) and nutrient (N and P) budgets were determined from 1 Oct. 1994 through 30 Sept. 1997 for each wetland. Wetlands received 4639 kg total N during the 3-yr period (96% as NO_3–N) and removed 1697 kg N, or 37% of inputs. Wetlands decreased NO_3–N concentrations in inlet water (annual outlet volume weighted average concentrations of 4.6 to 14.5 mg N L^{-1}) by 28% compared with the outlets. When the wetlands were coupled with the 15.3-m buffer strip between the wetlands and the river, an additional 9% of the tile NO_3–N was apparently removed, increasing the N removal efficiency to 46%. Overall, total P removal was only 2% during the 3-yr period, with highly variable results in each wetland and year. Treatment wetlands can be an effective tool in reducing agricultural N loading to surface water and for attaining drinking water standards in the Midwest.

CONSTRUCTED wetlands can effectively treat a variety of point-source pollutants. Examples include nutrients, pesticides, and heavy metals, from sources such as municipal wastewater, liquid mine waste, urban stormwater drainage, and feedlot runoff (Hammer, 1989; Moshiri, 1993; Kadlec and Knight, 1996). Successful use of constructed wetlands to treat point-source pollution has led to broad interest in their potential to ameliorate the effects of nonpoint pollution from agricultural sources (USEPA, 1990a; Baker, 1992; Campbell, 1995).

Two of the most important agricultural nonpoint pollutants are N and P. Nitrogen enters ground water and surface water through seepage and lateral subsurface flow primarily as NO_3–N and is generally the limiting nutrient in estuaries. Phosphorus enters surface waters in both dissolved and particulate form and is generally the limiting nutrient in lakes. A large portion of the N and P now entering U.S. freshwater and marine ecosystems as nonpoint diffuse nutrient loading can be attributed to agriculture and associated land use practices in the Midwest (USEPA, 1989, 1990b). These practices have contributed to reduced ground water quality, reduced surface water quality, and are also believed to be a contributing cause of hypoxia in the Gulf of Mexico and anoxia in estuarine and ocean ecosystems (Turner and Rabalais, 1991; Vitousek et al., 1997; Jickells, 1998; Burkart and James, 1999).

Thirty-seven percent of the Cornbelt and Great Lakes cropland is artificially drained by surface channels, subterranean tiles, or a combination of the two (Fausey et al., 1995). Artificial drainage lowers water tables in many poorly drained soils of the Midwest so they can be farmed, but it also shunts NO_3–N laden drainage waters more rapidly and directly to surface waters, further exacerbating the NO_3–N problem (Fausey et al., 1995; Drury et al., 1996; Lowrance et al., 1997). Phosphorus export to surface waters has been conventionally associated with surface erosion and surface runoff; however, recent studies show that artificial drainage systems can also be a major source of nonpoint P (Sims et al., 1998).

In the upper Embarras River watershed of central Illinois, where tiles drain 70 to 85% of the cropland, total N losses averaged 39 kg ha^{-1} yr^{-1} from 1990 to 1996 (David et al., 1997a). Tiles contributed an estimated 75 to 91% of the total N load to the Embarras River in 1995 and 68 to 82% of the total N load in 1996. Dissolved P export in the Embarras River from 1993 to 1996 averaged 0.9 kg ha^{-1} yr^{-1}. During 1995 to 1996, an estimated 46 to 59% of the dissolved P export to the upper Embarras River resulted from tile drainage (Xue et al., 1998). These studies demonstrate that in heavily tile-drained watersheds most of the N and P added to surface waters is transported through the tiles.

To remedy the nonpoint-pollution problem, methodologies must be devised to reduce export to aquatic ecosystems. Several forms of output control have been discussed in the literature. Riparian buffer strip systems have the potential to reduce diffuse nutrient inputs into surface waters and are effective in many environments

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Abbreviations: DOC, dissolved organic carbon; MCL, USEPA maximum contaminant level; WA, Wetland A; WB, Wetland B; WC, Wetland C; WD, Wetland D.

(e.g., Groffman et al., 1991; Haycock and Pinay, 1993; Nelson et al., 1995). For buffer strips to function, however, drainage waters must pass through them as diffuse flow before entering streams. In many agricultural areas of the Midwest, tiles drain directly into streams and rivers causing buffer strips to be ineffective (Vought et al., 1994; Lowrance, 1996). In such instances where nonpoint agricultural pollution enters surface waters, wetland treatment has been suggested as a potential solution (Baker, 1992).

There is little known about the use of constructed wetlands for treating nonpoint pollutants derived from agricultural runoff and drainage in the agricultural Midwest. Several studies have shown that constructed wetlands can remove N and P loads from river waters draining mixed agricultural–municipal watersheds (Hey et al., 1994; Jansson et al., 1994; Mitsch et al., 1998). The field scale approaches discussed above treated agricultural drainage after it entered the stream. Crumpton et al. (1993) found that constructed wetland mesocosms could reduce NO_3–N concentrations ranging from 3 to 15 mg L^{-1} by >80% during residence times of approximately 1 wk. These findings imply that created wetlands could remove nonpoint pollution at its source, the cultivated field. Osborne and Kovacic (1993) suggested the creation of lateral constructed wetlands formed by berming an area adjacent to a stream and forming a small detention basin or treatment wetland. The wetland would treat pollutants in drainage water before they entered the stream, with the water slowly released to the stream through regulated flow or seepage.

The purpose of the present study was to determine if constructed agricultural treatment wetlands could be effective in removing nonpoint N and P pollution from agricultural tile drainage waters before entering surface waters of the upper Embarras River. In east-central Illinois, N losses to surface waters were of primary concern, therefore, the major focus of the study was on N removal; however, P and C retention were also studied. Nitrogen, P, and C mass balances were determined for water years 1995–1997 in three constructed treatment wetlands.

METHODS AND MATERIALS

Description of the Study Area

The study site was located adjacent to the Embarras River in Champaign County, Illinois (39°54′55″ N, 88°11′30″ W), approximately 20 km south of Urbana, Illinois. For a detailed description of the Embarras River watershed see David et al. (1997a). The land area is 91% row crop agriculture in both maize and soybean. In this region, tile drains are installed in approximately 75% of the cropland area to allow drainage and to reduce high water tables, which enhances agricultural production.

The constructed wetlands, Wetland A (WA), Wetland B (WB), Wetland C (WC), and Wetland D (WD), received tile drainage from areas of 15, 5, 17, and 25 ha, respectively. The WA, WB, and WC drainages were in a maize–soybean rotation with maize planted in 1995 and 1997. This drainage area incorporated Farm Fields 1 and 2 (Fig. 1) that were on the same crop rotation during our study and employed similar cultivation and management practices. See Gentry et al. (1998) for details on production practices. The two fields were planted with similar genetic material and received the same fertilizer rate of 135 kg N ha^{-1} (for maize). However, Field 2 received a solution of $(NH_4)_2SO_4$ (6–0–0–6, NPKS) in February of 1995 and January 1997 rather than anhydrous ammonia (Gentry et al., 2000). Wetland D received drainage from Field 3, which was planted yearly with a combination of both maize and soybeans. Anhydrous ammonia was applied at a rate of 202 kg N ha^{-1} for maize production to the southern one-third of Field 3 in 1996, while the northern two-thirds of Field 3 received applications of $(NH_4)_2SO_4$ (6–0–0–6) at a rate of 202 kg ha^{-1} in 1995 and 1997.

The experimental wetlands were constructed on Colo silty clay loam, a bottomland soil that formed adjacent to streams. The site has never been cultivated and was used as pastureland for cattle. Prior to drainage and grazing this soil supported wet prairie vegetation. The top 30 cm of wetland soil contained 10 480 kg ha^{-1} total N, 83 kg ha^{-1} total P, and 132 079 kg ha^{-1} total C (David et al., 1997b, Hoagland et al., 1999). Because of the subsurface tile and surface drainage channels of the region, the site is prone to flash flooding.

Wetland Construction

Experimental wetlands were constructed in 1994 by rerouting tile drainage lines to the ground surface above the floodplain and creating a 1.4-m earthen berm to form wetlands that intercept drainage waters (Fig. 2). In Wetlands A, C, and D, strips of soil were excavated to create a berm. Wetland bottoms consisted of a series of nonexcavated soils alternating with 3.0- to 3.7-m-wide excavated strips ranging from 0.4- to 0.9-m deep (Table 1). The WB site contained native mesic prairie species that originated on the site, therefore, the bottom of WB was not excavated but left undisturbed and soil for berm construction was from excavation of WA. Deep zones (1.2- to 1.5-m deep) covered approximately 25% of the area in WA, WC, and WD.

Wetland construction followed the conceptual design of

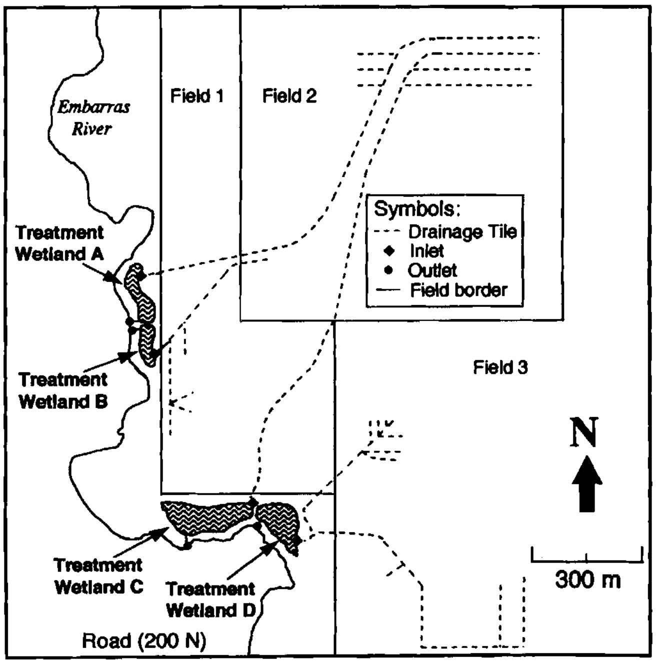

Fig. 1. Map of the study site depicting the location of tile drainage systems and constructed wetlands.

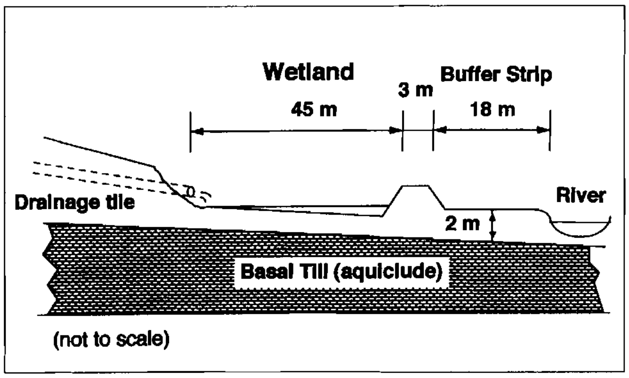

Fig. 2. Constructed treatment wetland design (modified from Osborne and Kovacic, 1993).

Osborne and Kovacic (1993) (Fig. 2). Wetland berms were constructed following USDA guidelines (Gray et al., 1992). A 0.9-m-deep by 1.2-m-wide trench was excavated the length of the planned berm to form a cutoff to retard seepage below the berm. Soil was applied to the berm in 15-cm layers and compacted with a sheep's foot roller until a height of 1.4 m was reached. A tracked bulldozer with a pushing blade (Caterpillar [Peoria, IL] Model D-8) was used to excavate and spread soil. The unexcavated strips within the wetlands and the surrounding river floodplain provided a seed bank to revegetate the site.

An emergency spillway, 2.4-m wide and 0.3-m deep, was constructed in each berm. Berm side slopes were 2:1, top width 2.7 m and base width 8.2 m, with a minimum free board (difference between height of emergency spillway and top of berm) of 0.3 m. The bases of the berms were established at a distance of 15.3 m from the stream bank to minimize constriction of river flow during flooding as required by the local Natural Resources Conservation Service guidelines. Wetland bottoms were not planted; however, berms were seeded with oat (*Avena sativa* L.) following construction in the fall to provide a quick soil stabilizing cover. The following spring, berms were seeded with perennial rye grass (*Lolium perenne* L.). Wetland surface areas, volumes, and drainage area to wetland surface area ratios are given in Table 1. Surface runoff events on the WA,WB, and WD sites were negligible because each wetland drained only one sub-basin. Wetland C, however, drained several sub-basins and frequent surface runoff events often inundated the WC inlet. Although WC functioned properly, accurate water and nutrient budget determinations were precluded; therefore, WC was eliminated from the study.

End of season standing crop biomass was taken from ten 0.5 m^2 random clip plots per site and dried at 80°C for 72 h. Average standing crop biomass was 721 and 529 g m^{-2} for WB and WD, respectively, in 1995; and 416, 865, and 503 g m^{-2} for WA, WB, and WD, respectively, in 1996 (1995 standing crop biomass was not determined in WA). The dominant plant species consisted of reed canary grass (*Phalaris arundinacea* L.), hop sedge (*Carex lupulina* Muhl. ex Willd.), barnyard grass [*Echinochloa crus-galli* (L.) P. Beauv.], prairie cordgrass (*Spartina pectinata* Link), lady's thumb (*Polygonum persicaria* L.), swamp smartweed (*Polygonum amphibium* L. var. emersum Michx.), yellow nutsedge (*Cyperus esculentus* L.), pigweed (*Amaranthus viridis* L.), and ironweed [*Vernonia gigantea* (Walter) Trel. ex Branner & Coville].

Instrumentation and Wetland Monitoring

Inlet and outlet structures were installed to monitor flow. Inlet structures were fabricated from culvert pipes with a rectangular opening cut to allow the addition of a combination slot, v-notch, and sharp crested weir. Inlet structures for WA, WC, and WD were constructed of 1.8-m-diameter culvert pipe, while that for WB (smallest area drained and lowest flow rate) was constructed of 1.5-m-diameter culvert pipe and attached to drainage tiles. Outlet structures were fabricated from 0.8-m-diameter culvert pipe cut in half longitudinally and fitted with channel iron and a bottom plate and outlet pipe to form a pipe drop structure fitted with stop logs (4 × 13 cm tongue and groove boards), an orifice, and a sharp crest weir.

Table 1. Descriptive physical characteristics of each constructed wetland.

Wetland	Surface area	Volume	Tile drainage area/ wetland surface area	Average depth
	ha	m^3		m
A	0.6	5400	25	0.9
B	0.3	1200	17	0.4
D	0.8	5200	32	0.7

Each inlet and outlet monitoring station had a Campbell (Logan, UT) CR10 data logger, a Keller PSI pressure transducer (KPSI, Hampton, VA), and an ISCO (Lincoln, NE) Model 2900 automatic water sampler. Wetland inlet and outlet water flow were measured using a pressure transducer and combination weir (described above). Data loggers continually recorded water levels within the inlet structure on a 15-min interval and automatic water samplers were used to collect flow proportional solution samples as frequent as every 15 min during runoff events. Outlet water samples were taken four times a day, with flow measured every 15 min. Analyses indicated that tile and outlet flow nutrient concentrations were stable between precipitation events and often increased following major precipitation events. Because concentration patterns were consistent, it was possible to determine concentrations for each 15-min inlet and outlet flow period by interpolating between known concentrations.

Although precautions were taken to prevent seepage from the wetlands by compacting the berms, seepage was a major component of hydraulic export from the wetlands. In a separate study, seepage volumes and seepage NO_3–N export loads were determined for WA and WD during the 1997 water year (see Larson et al., 2000 for details). Seepage export estimates derived from 1997 measurements were factored into wetland N budgets for the 1995 and 1996 water years. For 2 mo in 1997 (6 February to 8 April) the surface application of a $(NH_4)_2SO_4$ (6–0–0–6) fertilizer solution to Field 2 (see Gentry et al., 2000, for details) enhanced the NO_3–N removal efficiency of WA (but not WB or WD), creating conditions that did not occur in 1995 or 1996. Prior to the fertilizer pulse, NO_3–N concentrations in wells outside the berm of WA paralleled NO_3–N concentrations along the berm inside the wetland and were higher near the inlet source and lower toward the outlet (Larson et al., 2000). After this event, increases in concentrations of SO_4^{2-}–S and dissolved organic carbon (DOC) were observed in the berm wells closest to the inlet. The decrease in NO_3–N concentrations observed in berm wells closest to the inlet and the reversal of the well NO_3–N gradient along the berm from inlet to outlet was probably due to the influence of the $(NH_4)_2SO_4$ fertilizer. Using prefertilizer pulse NO_3–N concentrations from the berm wells, adjusted NO_3–N removal rates were estimated for WA. These more conservative rates were used to determine NO_3–N export under nonpulse conditions in WA during 1995 and 1996.

During the 1997 year, seasonal seepage export of NO_3–N retained in WA was 0, 30, 14, and 0.3% for fall, winter, spring, and summer, respectively, and total seepage export was 15% (retained NO_3–N = inlet load − outlet export) (Larson et al., 2000). The adjusted rates increased the winter seepage export to 48%, the spring to 17%, and the total to 22% (Larson,

Table 2. Tile and wetland hydrologic characteristics for the water years 1995 to 1997.

Wetland	Number of flow days			Days to 75% of hydrologic loading†			Retention time—year‡			Retention time—April through June		
	1995	1996	1997	1995	1996	1997	1995	1996	1997	1995	1996	1997
	d											
A	254	175	207	50	51	54	35	8	19	20	10	32
B	272	252	172	58	79	66	17	14	11	9	7	19
D	314	299	228	35	45	44	26	13	11	13	5	15

† Minimum number of days to reach 75% of annual tile inlet flow.
‡ Wetland volume/sum of inlet flow, precipitation, surface runoff, and seepage from adjoining wetlands (for D only) multiplied by number of days of tile flow.

1998). Seepage export loads in WB were assumed to be proportional to those in WA. Wetland B, however, was not affected by the fertilizer pulse in 1997, therefore the adjusted seepage export percentages from WA were used to calculate seepage export in WB for water years 1995 through 1997. Seasonal seepage NO_3–N export rates determined in WD during the 1997 water year were 0, 41, 3, and 0.8% for fall, winter, spring, and summer (Larson et al., 2000), respectively, and were used to determine WD seepage export for water years 1995 through 1997.

Inlet and outlet flows along with estimates of seepage (Larson et al., 2000), evaporation, and precipitation were used to determine water budgets for the wetlands. Precipitation was measured on site with a Campbell Scientific YE 525 tipping bucket rain gauge with rainfall recorded every 30 min by a Campbell Scientific CR10 data logger. Evaporation losses were determined using daily potential evapotranspiration estimates provided by the Illinois State Water Survey in Champaign, IL, and extrapolated over the surface of the wetlands to determine volume. Fifteen-minute inlet and outlet flows were multiplied by inlet and outlet N, P, and C concentrations to determine corresponding budgets.

Sample Analyses

Water samples were preserved for storage on the same day as collection following standard methods (American Public Health Association, 1995). Subsamples were filtered (Whatman GF/C glass fiber 1.2-μm pore size) and analyzed for NO_3^-–N by ion chromatography, NH_4^+–N by automated phenate, NO_2^-–N by colorimetric techniques, dissolved P by colorimetric techniques, and DOC by a Dohrmann–Xertes (Cincinnati, OH) DC-80 Analyzer (American Public Health Association, 1995). No differences were found between 1.2- and 0.45-μm filter sizes for dissolved P. Unfiltered subsamples were analyzed for total N by the persulfate digestion method and total P by digestion with ammonium persulfate and H_2SO_4 with detection of dissolved P. Organic N was calculated as total N minus the sum of nitrate, nitrite, and ammonium. Organic P was determined as the difference between total P and dissolved P. After the first year of analysis, inlet NO_2^-–N, organic N, and organic P levels were found to be negligible and were discontinued.

RESULTS AND DISCUSSION

Water Budget

In 1995, 1996, and 1997 precipitation was 790, 990, and 991 mm, respectively, while the corresponding yearly inflow was 23, 42, and 35% of the total 3-yr hydraulic load. Although 52% of all precipitation from water years 1995–1997 occurred in winter and spring, the two seasons accounted for 95% of the annual wetland hydraulic loading. Inflow occurred annually in all three wetlands during all four seasons over the entire 3-yr study period, while outflow occurred only in winter, spring and early summer. As crops matured, however, evapotranspiration in the watershed increased and tile flow decreased as the shallow water table was drawn down below the depth of drainage tiles, resulting in minimal summer flow for only three of nine wetland/water years and negligible flow in the fall (budgets determined for 3 yr on three sites for a total of nine wetland/water year combinations). Wetlands usually dried up in late summer and early fall when tile lines ceased flowing.

Retention, determined by dividing total annual hydraulic load by wetland volume, was estimated to range from 11 to 35 d (Table 2). Using this approach, long periods with relatively low flow volumes inflated retention time estimates. A second retention time estimate was determined using only winter and spring budgets because these seasons accounted for nearly all inlet flow. Estimated retention times using the latter method were shorter for 1995 and 1996 (5–20 d), yet longer for 1997 (15–32 d). Results can be explained by less evenly dispersed pulse rainfall events in 1995 and 1996, more evenly dispersed 1997 rainfall, and less rainfall during April and May 1997 (Fig. 3).

Although tile inflow ranged from 172 to 314 d annually (Table 2), during most days flow volumes were low. Overall, 75% of total hydraulic load was reached within 24, 29, and 15% of the total flow duration in WA, WB, and WD, respectively (Table 2). This demonstrates the flashy and rapid nature of subsurface runoff in the watershed.

Inflow was proportional to the land area drained. During the 3-yr study, approximately 0.5 million m^3 of combined flow entered the three wetlands, with the maximum flow entering WD (245 600 m^3, Table 3). This was nearly double the load entering WA (144 000 m^3) and nearly five times the load entering WB (47 700 m^3). Expressed on a daily area basis, hydraulic loads were 0.04, 0.03, and 0.04 m for WA, WB, and WD, respectively. Daily loading was similar to that reported by Hey et al. (1994) and Davis et al. (1981), but negligible when compared with that reported by Fleischer et al. (1994).

Flow events for WA are depicted in Fig. 3 and represent the typical inlet and outlet flow patterns observed in the three wetlands during the study. Under nonflooded conditions, outlet flow volumes were lower than inlet flow volumes and water was temporarily retained within the wetlands. However, following peak precipitation events, entering water was rapidly shunted through the wetlands so that outlet flow was equal to inlet flow as wetland capacity was exceeded.

The highest daily peak flow in WA occurred in 1995, with the majority of flow in May; however, annual flow and rainfall were lowest that year (Fig. 3). Overall flow

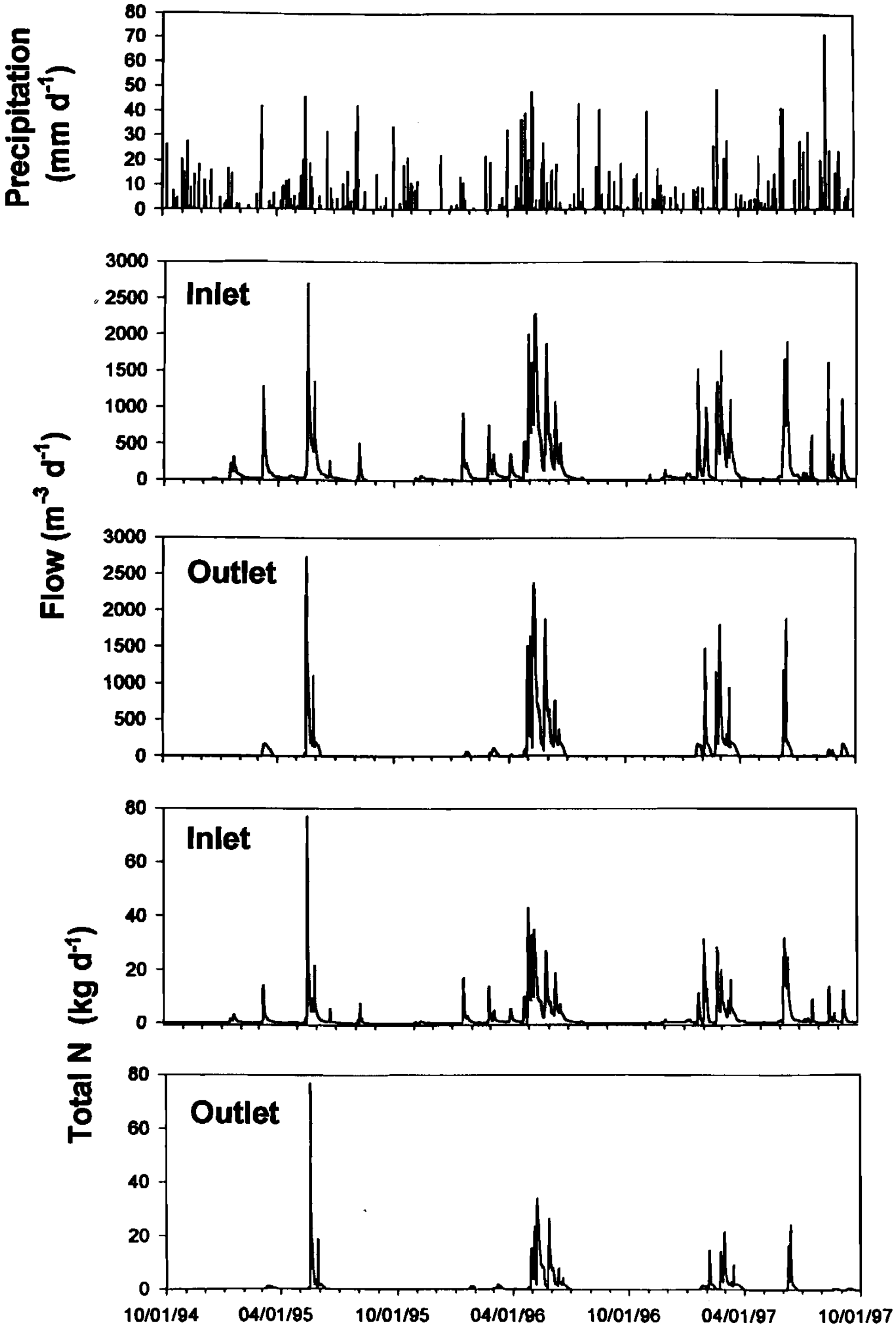

Fig. 3. Daily precipitation at the wetland site, daily inlet and outlet flows for Wetland A, and daily inlet total N loading and outlet export from 1 Oct. 1994 to 30 Sept. 1997.

was greatest in 1996, although WA exhibited the second highest peak flow with most of the annual flow entering the wetlands in May and June. In 1997 the rainfall was nearly identical to 1996, but WA inflow in 1997 ranked second overall while the wetland exhibited the lowest overall annual peak flow. The discrepancy between rainfall and flow events was probably a result of major rainfall events dispersed in February, March, and June, 1997.

During the 3-yr period, water exited the wetlands as outlet flow (64%), seepage (28%), and evaporation (8%) (Table 3). Evaporation was linked to wetland retention time, the surface area to volume ratio, and temperature. Outlet flow was a direct function of inlet flow (or area drained) and size of wetland. For example, in WD, with the largest watershed to wetland area ratio, 78% of the total hydraulic load exited the wetland as outlet flow, while in WB, with the smallest watershed

Table 3. Water balances for Wetland A, B, and D for water years 1995 to 1997.

Wetland & water year	Tile inlet	Precipitation	Outlet	ET†	Seepage	Balance‡
	m³					
A						
1995	30500	3500	14100	5000	19700	−4700
1996	59200	4200	37800	4400	20300	800
1997	54300	4900	27700	5500	23600	2500
B						
1995	13000	1600	7200	2100	9600	−4300
1996	19300	2400	10000	2800	8600	200
1997	15400	2800	7200	3100	7800	200
D						
1995	58400	3700	39900	5500	15300	1300
1996	10600	4800	99000	5200	16000	−9000
1997	81200	6100	76000	7000	20700	−16300

† Estimated evapotranspiration.
‡ Balance refers to total inflow (tile inlet flow + precipitation) minus total outflow (outlet flow + ET + seepage). Negative values result from surface runoff events not accounted for in the balance.

to wetland area ratio, 42% exited as outlet flow (Tables 1 and 3).

Overall seepage volume was greatest in WA (63 600 m³) and lowest in WB (26 000 m³, Table 3). Proportionally, however, seepage accounted for 40% of the total flow leaving WA, 44% of that leaving WB, and only 18% of that leaving WD. Although flow entering WD was greatest, the hydraulic head pressure of this shallower wetland was lower than that in WA. Greater outlet flow, reduced retention, and lower hydraulic head pressure in WD reduced the proportion of flow available for seepage, while increased retention in WA and WB increased their seepage.

Estimated evapotranspiration losses were slightly larger than direct precipitation inputs to the wetlands (Table 3). Surface runoff into the wetlands was negligible in this relatively flat watershed except under a few intense rain events, which caused the negative water balances for WA and WB in 1995 (Table 3). Water budget determinations in WD were confounded by seepage of water through a partition berm dividing WD from WC (Fig. 1). We found the effective width of the dividing berm was decreased by muskrat (*Ondatra zibethicus*) burrowing. In addition, water in WC was slightly higher, causing positive headpressure and movement of water through the berm and into WD.

Chemical Budgets and Concentrations

Nitrogen

Nitrate Nitrogen (NO_3–N). During the 3-yr study, subsurface tile drainage flow carried a total of 4445 kg NO_3–N into the wetlands, with 2749 kg exported through the outlet and by seepage flow (Table 4). Nitrate N removal by the wetlands was therefore 1696 kg, or 38% of the total 3-yr load. This was equal to a removal of 13 kg ha^{-1} yr^{-1} based on watershed area drained, and 333 kg ha^{-1} yr^{-1} based on wetland area removal.

Annual inflow NO_3–N entering each wetland was consistently greater than that exiting the wetland (Table 4). Annual total inflow NO_3–N for all wetlands was 985, 2098, and 1362 kg, while outflow NO_3–N was 646, 1350, and 753 kg in 1995, 1996, and 1997, respectively (Table 4), with quantities paralleling hydraulic flows. The largest inflow and outflow NO_3–N loads (observed in WD-96) were an order of magnitude greater than the lowest inflow and outflow loads (observed in WB-95, Table 4). Annually, inflow NO_3–N was always greatest in WD and lowest in WB, as would be expected from their corresponding tile drainage areas (Table 1).

Because NO_3–N is highly mobile in soils, it was the only nutrient to leave the wetlands in seepage. Therefore, NO_3–N export was a combination of both seepage and outlet flow. For the 3-yr period, outlet flow accounted for 85% of the combined wetland outflow NO_3–N, while seepage flow accounted for the remaining 15%.

In our study, seepage passed through the wetland berms and then through a 15.3-m soil buffer zone before entering the river. Seepage flow accounted for 20, 33, and 7% of the overall NO_3–N flux from WA,WB, and WD, respectively, or 398 kg N for the 3 yr (Table 4). The small proportion of seepage from WD resulted from excess inlet flow, which rapidly shunted large proportions of water carrying NO_3–N through the wetlands and over the outlet weir during pulse flow events. This indicated that WD was undersized for its inlet flow.

Annual average NO_3–N concentrations (all concentrations herein are reported as flow-weighted averages) for individual wetland inflows ranged from 7.5 to 14.5 mg L^{-1} and outflows ranged from 4.6 to 14.5 mg L^{-1} (Fig. 4). Average outflow NO_3–N concentrations were consistently lower than those of inflow in eight of nine wetland/water year combinations. For those eight combinations, outlet NO_3–N concentrations were 11 (WD-95) to 37% (WD-97) lower than inlet concentrations (Fig. 4). In two of four wetland/water years where inflow NO_3–N concentration exceeded the USEPA maximum contaminant level (MCL) for drinking water (10 mg L^{-1}), NO_3–N concentrations were reduced below the MCL (WA-97 and WB-96) following wetland treatment.

In 1995, the average outlet NO_3–N concentration of WA (14.5 mg L^{-1}) exceeded that of the inlet (12.3 mg L^{-1}, Fig. 4), suggesting that wetlands were concentrating NO_3–N or that autogenic NO_3–N was increasing the concentration. However, the high flow-weighted outlet concentration for WA was the result of a single high concentration pulse flow event on 17 May 1995 (Fig. 3)

Table 4. Constituent inlet load, outlet export, seepage export, and percent removal from three wetlands for water years 1995, 1996, and 1997.

Wetland	1995					1996					1997				
	Inlet load	Outlet export	Seepage export	Total export	% removal	Inlet	Outlet	Seepage export	Total export	% removal	Inlet	Outlet	Seepage export	Total export	% removal
	kg yr^{-1}														
A															
NO_3^--N	374	203	50	253	33	861	463	107	571	34	635	225	61	286	55
NH_4^+-N	3.4	3.2	0	3.2	6	0.3	1.7	0	1.7	(567)	141.5	62.5	0	62.5	56
Organic N	0	3	0	3	ND†	0	2	0	2	ND	0	21	0	21	ND
Total N	378	209	50	259	32	861	467	107	574	33	777	309	61	370	52
PO_4-P	5.4	4.3	0	4.5	17	9.9	8	0	8	20	15	8	0	8	47
Organic P	0	0.3	0	0.3	ND	0	0.7	0	0.7	ND	0	1.2	0	1.2	ND
Total P	5.4	4.5	0	4.5	17	9.9	8.9	0	8.9	10	14.7	9.6	0	9.6	35
TOC	52	68	0	68	(32)‡	130	116	0	116	10	264	147	0	147	44
B															
NO_3^--N	105	49	17	66	37	238	85	43	128	46	149	47	30	77	48
NH_4^+-N	0.7	0.8	0	0.8	(14)	0.4	1.0	0	1.0	(1.5)	4	1	0	1	75
Organic N	0	1	0	1	ND	0	2.8	0	2.8	ND	0	2	0	2	ND
Total N	106	51	17	68	36	238	88	43	131	45	153	50	30	80	48
PO_4-P	1.5	0.2	0	0.2	90	2.4	2.3	0	2.3	42	2.1	1.3	0	1.3	38
Organic P	0	0.2	0	0.2	ND	0	0.5	0	0.5	ND	0	0.1	0	0.1	ND
Total P	1.5	0.3	0	0.3	80	2.4	2.7	0	2.7	(13)	2.1	1.4	0	1.4	38
TOC	27	16	0	16	41	35	38	0	38	(8)	37	29	0	29	23
D															
NO_3^--N	506	303	24	327	35	999	610	41	651	35	578	363	25	390	33
NH_4^+-N	6.9	3.4	0	3.4	51	2.5	3.7	0	3.7	(48)	33.0	17.5	0	17.5	38
Organic N	0	6	0	6	ND	0	20	0	20	ND	0	43	0	43	ND
Total N	513	313	24	337	34	1002	634	41	675	33	611	424	25	449	27
PO_4-P	5	3	0	3	60	11	8	0	8	27	11	14	0	14	(27)
Organic P	0	0.8	0	0.8	ND	0	5.6	0	5.6	ND	0	3.3	0	3.3	ND
Total P	4.8	4.1	0	4.1	14.6	11.4	13.1	0	13.1	(14.8)	11.4	17.5	0	17.5	(54)
TOC	146	155	0	155	(06)	431	445	0	445	(3)	319	398	0	398	(25)

† Export greater than loading but retention cannot be determined because of division by zero.
‡ Parentheses indicate negative value (export from system greater that input).

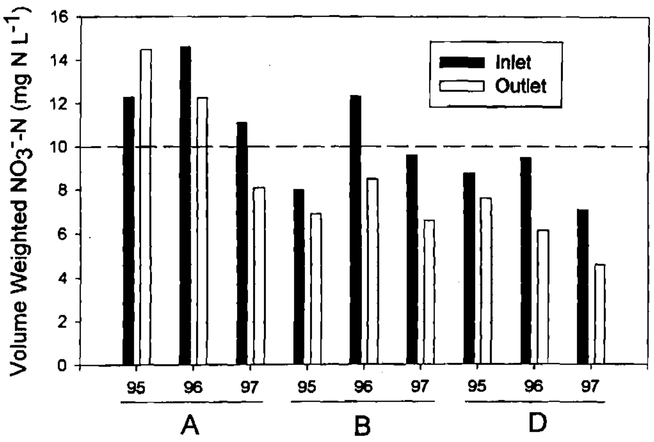

Fig. 4. Wetland/water year flow-weighted NO_3–N concentrations of inlet and outlet flow from Wetlands A, B, and D. Dashed line equals USEPA maximum contaminant level for drinking water nitrate N.

that dominated the outlet average. During the event the nonweighted instantaneous outlet concentration in WA-95 equaled but never actually exceeded the inlet concentration. Overall concentrations for inlets and outlets during the 3-yr period ranged from 8.5 to 13.0 mg L^{-1} and 5.9 to 11.2 mg L^{-1}, respectively (Table 5).

Our results correspond with those of constructed wetlands treating river water, where NO_3–N concentration reduction ranged from 25 to 99% (Hey et al., 1994; Mitsch et al., 1998; Hunt et al., 1999). However, inlet NO_3–N concentrations in these studies were generally lower than ours, ranging from 1.2 to 6.6 mg L^{-1} (Hey et al., 1994; Mitsch et al., 1998; Hunt et al., 1999).

In the Hey et al. (1994) study, wetlands were operated only during the growing season when temperatures were increased and more optimal for denitrification. Additionally, although NO_3–N export due to seepage did occur in the wetlands (Hey et al., 1994), it was not included in the NO_3–N budget, resulting in higher removal estimates. The wetland total N removal efficiency (0.3 g m^2 d^{-1}) reported by Hunt et al. (1999) might also be a result of milder temperatures.

Previous studies in our wetlands identified denitrification as the major process removing NO_3–N, with a maximum denitrification rate of 0.28 g m^{-2} d^{-1} (Xue et al., 1999). Xue et al. (1999) found that approximately 90% of the N lost was denitrified while approximately 10% was retained in wetland plants. Using the overall wetland average NO_3–N loading rate of 0.50 g m^{-2} d^{-1} and an overall removal rate of 38%, the average daily loss was 0.19 g m^{-2} d^{-1}. This is within the potential denitrification range reported by Xue et al. (1999), and also within the range (0.001–0.48 g m^{-2} d^{-1}) reported for river water treatment wetlands (Fleischer et al., 1994).

Ammonium Nitrogen (NH_4^+–N). Annual NH_4–N tile inputs for individual wetlands ranged from 0.3 to 142 kg, while that of export ranged from 0.8 to 63 kg (Table 4). When the liquid fertilizer $(NH_4)_2SO_4$ (6–0–0–6) was applied to Field 2 (Fig. 1) in 1995, NH_4–N loads were only 3.4 kg in WA and 6.9 kg in WD. Following the same fertilizer application to snow covered soil of Field 2 (February 1997), rapid thawing resulted in high NH_4–N loads to WA (142 kg) and WD (33 kg). Gentry et al. (2000) determined that the large pulse of NH_4–N in 1999 entered Field 2 tile drains through preferential flow. In 1996, when no NH_4 fertilizer was applied, NH_4–N loads in the three wetlands were relatively low, ranging from 0.3 to 2.5 kg, with concentrations ranging from 0.01 mg L^{-1} to 0.02 mg L^{-1}.

Combined wetland NH_4–N tile inputs were 11, 3, and 179 kg in 1995, 1996, and 1997, respectively, while export was 7, 6, and 81 kg, respectively. During the 3-yr period, 193 kg NH_4–N entered the wetlands and 95 kg was exported, resulting in an overall removal efficiency of 51% (Table 4). This high removal efficiency was the result of atypically high NH_4–N loads in 1997, when NH_4–N removal efficiency was 55%. Under lower loads in 1995 and 1996, the overall NH_4–N removal efficiency was only 3%. Overall inflow and outflow average concentrations of NH_4^+–N for the 3-yr period were 0.4 and 0.3 mg L^{-1}, respectively (Table 5).

Ammonium is usually nitrified in field soils, forming NO_3–N. Only then does N derived from NH_4^+–N subsequently leach through the soil and enter tile drains. Wetlands constructed for the treatment of tile drainage would not typically receive the large loads of NH_4–N seen in February 1997; however, results indicated that these high pulses of NH_4 could be reduced with wetland treatment. The wetland reduction of NH_4–N was attributed to adsorption to sediment, plant uptake, and the nitrification of NH_4–N to NO_3–N (under aerobic conditions).

In a prairie pothole marsh draining a maize–soybean agricultural watershed in Iowa, Davis et al. (1981) reported NH_4–N loading similar to that found in our study in 1995 and 1996, but marsh removal efficiency was much greater (78%). This difference may have been due to lower hydraulic loading in the marsh and sediment-associated NH_4–N in surface runoff, which was generally absent in the our wetlands. Conversely, the Herring Marsh wetland in North Carolina exported 47% more NH_4–N than entered in streamflow (Hunt et al., 1999).

Organic Nitrogen. During the 3-yr study, 101 kg or-

Table 5. Flow-weighted average concentrations for inlet and outlet constituents, 1995 to 1997 water years.

	NO_3^-–N		NH_4^+–N		Organic N		Ortho-P		Organic P		DOC†	
Wetland	Inlet	Outlet	Inlet	Outlet	Inlet	Outlet	Inlet	Outlet	Inlet	Outlet	Inlet	Outlet
	mg L^{-1}											
A	13.0	11.2	1.0	0.8	–‡	0.3	0.21	0.26	–	0.03	3.1	4.2
B	10.3	7.4	0.1	0.1	–	0.2	0.12	0.14	–	0.03	2.1	3.4
D	8.5	5.9	0.2	0.1	–	0.3	0.11	0.12	–	0.04	3.6	4.6

† Dissolved organic carbon.
‡ Concentration below detection limits.

ganic N exited the wetlands; 65% was exported in 1997. The higher organic N export may be associated with increased microbial and plant growth as well as increased decomposition of organic matter enhanced by the ammonium sulfate fertilizer inorganic N pulse. Organic N in outlet water was probably derived from preexisting plant and microbial pools as well as from the conversion of inorganic N to the organic N pool. Only negligible organic N entered the wetlands from the tiles. Individual wetland average outflow concentrations for the 3-yr period ranged from 0.2 to 0.3 mg L^{-1} (Table 5), with an overall 3-yr average N of 0.3 mg L^{-1}.

Total Nitrogen. To determine the true effectiveness of wetlands in removing NO_3–N from surface waters, it is essential to examine all forms of N that could potentially yield NO_3–N in the water column. Annual total N tile inputs for individual wetlands ranged from 106 to 1002 kg N, while export ranged from 68 to 674 kg (Table 4). Total N export was lower than inputs for all wetland/water years (Table 4). The observed fluxes closely paralleled hydraulic loading during the 3-yr period with 21, 45, and 33% of total N load, and 23, 47, and 30% of total N export, in 1995, 1996, and 1997, respectively.

During the 3-yr period, WA removed 766, WB removed 221, and WD removed 715 kg N, with removal efficiencies of 40, 44, and 31%, respectively. Wetland D received the most total N overall, and its removal efficiency was the lowest of the three wetlands. Wetland B, however, both received and removed the least total N, but exhibited the highest removal efficiency.

The combined wetlands removed 1697 kg N, or 37% of the overall 4639 kg total N load for the 3-yr period. Total N removal was significantly correlated with loading, indicating consistent removal efficiencies at loading rates up through 1.3 g m^2 d^{-1} (Fig. 5). This linear trend suggested a constant total N removal rate of 37% for the wetlands with watershed to wetland area ratios ranging from 17 to 32.

For the 3-yr period, winter and spring total N load was 4400 kg N, or 95% of the total N entering the wetlands. Winter and spring removal was 1544 kg N, or 87% of the overall total N removed. During these two seasons wetland functions were of greatest benefit in the overall reduction of N to surface waters, even though total N removal efficiencies were low (Fig. 6). Low removal efficiencies were a result of the large load entering the wetlands and the low winter and spring temperatures that reduced the effectiveness of the denitrification process. Although wetland removal efficiencies were high in fall (100%) and summer (95%), the total N removal was 230 kg N, or 13% of the overall total N removed. Without fall and summer removal, overall wetland total N removal efficiency would have dropped by only 5%. When evaluating wetland performance, it is important to understand the differences between removal efficiency and total removal. Insignificant loads may enter and be removed by treatment wetlands (e.g., fall and summer) resulting in high removal efficiencies but contributing little to overall nutrient removal.

Total N removal in our wetlands was lower than that reported in the Des Plaines wetlands (Hey et al., 1994)

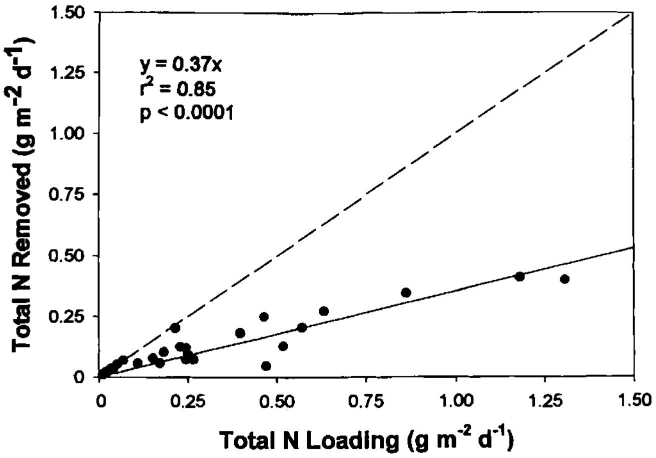

Fig. 5. Total N removed by each wetland by season compared with total N loading. Regression line shown (solid line) along with 1:1 line (dashed line).

and the Eagle Lake marsh (Davis et al., 1981). However, as discussed earlier, the Des Plaines wetlands received similar total N loading rates to ours in 1990 but much lower loads in 1991. In addition, the Des Plaines wetlands were operated only during the growing season from April to September with warmer temperatures (greater denitrification rates), hydraulic loading to the wetlands was controlled, and seepage export of N was not included in the N budget. Eagle Lake marsh exhibited similar inlet concentrations but much lower N and hydraulic loading than our wetlands (Davis et al., 1981). In addition, the Eagle Lake wetland accounted for 14% of the watershed area. These factors would result in a large retention time and thus a greater potential to remove N. Hunt et al. (1999) found that Herring Marsh wetland removed 37% of its total N load during a 3-yr period, nearly identical to our wetland overall removal efficiency. While hydraulic loading in the Herring Marsh wetland averaged approximately three times that of our study, total N loading on a wetland area basis was only 20% greater than that of our wetlands.

Total N removal reported by the Wetlands and Lakes as Nitrogen Traps (WLNT) studies in Sweden, however, were considerably lower than the Embarras River study (Fleischer et al., 1994). Five wetlands in the WLNT study received shunted river water draining agricultural land, whereas two wetlands were sites for municipal wastewater polishing. These wetlands were created by damming river sections to create wetlands or by shunting river water into wetlands or ponds adjacent to a river. Rivers drained either agricultural lands or urban areas. Total N loading was extremely high, ranging from 620 to 29 650 mg m^{-2} d^{-1}, or 1.2 to 56 times the largest loads found in the Embarras River study. Wetlands exhibited watershed to wetland area ratios ranging from 333 to 4700 and they received hydraulic loads 2.5 to 79 times the largest average daily hydraulic load observed in our study. This led to low removal rates of −1 to 10%.

Phosphorus

Dissolved Phosphorus. Individual wetland annual loading rates ranged from 1.5 to 15.0 kg dissolved P,

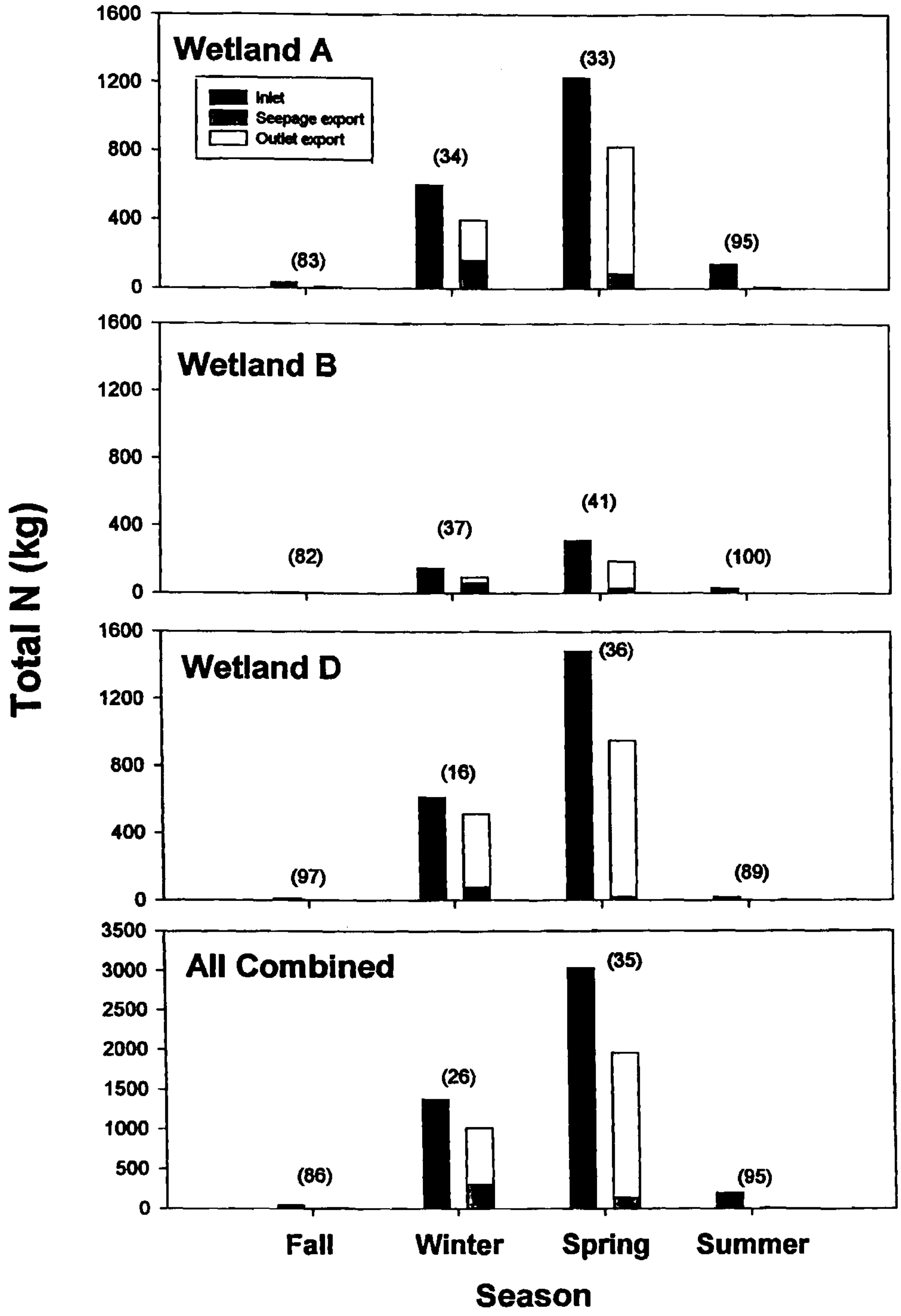

Fig. 6. Cumulative seasonal inlet load, seepage export, outlet export, and percent removal of total N (in parentheses) for each wetland during the 3-yr study and for all wetlands combined.

while export ranged from 0.2 to 14.0 kg (Table 4). Overall dissolved P removal was 32, 36, and 7% in WA, WB, and WD, respectively (Table 4). During the 3-yr period, 63.3 kg dissolved P entered the wetlands and 49.1 kg were exported for a total removal efficiency of 22%. In only one wetland/water year (WD-97) did export exceed the annual load (Table 4). Three-year average tile inlet concentrations for individual wetlands ranged from 0.11 to 0.21 mg L^{-1}, while outlets ranged from 0.12 to 0.26 mg L^{-1} (Table 5).

Davis et al. (1981) reported a similar removal efficiency of 20% in a prairie pothole wetland receiving a load of 3.52 kg ha^{-1} yr^{-1} dissolved P. Our wetlands, however, received two to five times this amount of dissolved P with similar removal efficiencies. In contrast to our study, the Des Plaines wetlands were more effective, with dissolved P removal of 52 to 99% (Hey et al., 1994).

Organic Phosphorus. Overall 3-yr organic P export for individual wetlands ranged from 0.1 to 5.6 kg P (Table 4). The overall cumulative organic P export from the wetlands was 12.7 kg P. Organic P entering the wetlands was negligible, although individual wetland outflow concentrations averaged from 0.03 to 0.04 mg P L^{-1} during the 3 yr. (Table 5).

Total Phosphorus. Annual total P loading for individual wetlands ranged from 1.5 to 14.7 kg, and exports ranged from 0.3 to 17.5 kg (Table 4). For six of nine wetland/water years total P inputs were greater than outputs, with removal efficiencies ranging from −64 to 80%. Overall wetland removal during the study was 1.5 kg, or 2% of the overall 63.6 kg total P load. This is in contrast to the overall dissolved P removal of 22%.

Removal of total P varied within the wetlands depending on flow and retention time. Little surface runoff and sediment-associated P entered the wetlands except during times of pulse flooding, and was not estimated. In only one wetland/water year (WD-97) did P export exceed import. We speculate this resulted from pulse flow conditions where surface runoff carrying sediment and associated P was not accounted for by inlet weir measurements. Three-year combined overall seasonal total P import loads were 0.2, 18.9, 41.8, and 2.6 kg; removal efficiencies were 100, 5, −4, and 78% for fall, winter, spring, and summer, respectively. Loads and removal efficiencies followed a pattern similar to that of total N, with the lowest removal rates in winter and spring, when 96% of the total P load entered the wetlands. In winter and spring organic P export typically offset dissolved P removal. Overall, these results indicated that wetlands were neither a significant source nor a sink for P. The total biomass P content for WB in 1997 was 57 kg ha^{-1} (Hoagland et al., 1999). Most of the inlet dissolved P was proabably of little use to wetland plants because pulse flows shunted much of the P out of the wetlands in the winter and early spring when the P cycle was inoperable (Kadlec and Knight, 1996) and before annual herbaceous plant growth began. Following plant death and the return of temperatures conducive to decomposition in the spring, dissolved P was slowly released (turnover of 1× to 2× per year; Mitsch and Gosselink, 1993) and assimilated by new plant growth. During this time, decomposition of aboveground litter and standing dead biomass probably contributed to an increase in outlet organic P.

In contrast to our study, Davis et al. (1981) found that Eagle Lake marsh removed 11% of the overall 4.1 kg total P ha^{-1} entering as agricultural runoff. The higher removal reflects the relatively lower total P loading in the marsh. Total P removal in the Des Plaines wetlands ranged from 52 to 99% (Hey et al., 1994). Although loads were similar to those in this study, higher removal efficiencies were probably related to the sediment and suspended solid loads entering the Des Plaines wetlands, where much of the total P probably settled out and was removed by sedimentation. In a 0.6-ha wetland designed to remove total P from potato (*Solanum tuberosum* L.) field surface runoff, 88% of a 48 kg total P load was removed during a 2-yr period (Higgins et al., 1993). This system was highly effective because the P was associated with surface runoff and sediment and because it was designed with a grassed buffer for sediment reduction.

Carbon

Dissolved Organic Carbon. In our study, C entering wetlands as DOC was considered an essential component for wetland denitrification. In five of nine wetland/water years there was a net export of DOC from the wetlands. Yearly wetland DOC loading ranged from 27 to 431 kg and retention ranged from −32 to 44% (Table 4). For the 3-yr period, however, DOC loading (1441 kg) and export (1412 kg) were similar, with the wetlands retaining only 2% of the DOC. Overall retention was positive in fall, winter, and summer (12.2, 19.7, and 17.1 kg, respectively) and negative (−28.4 kg) in the spring. Inflow average DOC concentrations for individual wetlands during the 3-yr period ranged from 2.6 to 3.6 mg L^{-1}, while those of the outflow ranged from 3.4 to 4.6 mg L^{-1} (Table 5).

Total organic C from tile flow entered the wetlands primarily as DOC, which was probably resistant to breakdown and not important to denitrification. However, photosynthetic production by plants and algae also fixed C. In WB the aboveground 1997 fixed C was 3150 kg ha^{-1}, while belowground fixed C was 10 350 kg ha^{-1} (Hoagland et al., 1999, determined assuming C = 45% of total organic matter). This autogenically derived C was probably of most importance in denitrification. Wetland soils were also a likely source of organic C. Following initial wetland construction, soils contained 3.6, 2.9, and 2.7% organic C, respectively, at depths of 0 to 10, 10 to 30, and 30 to 50 cm (David et al., 1997b). Carbon from $(NH_4)_2SO_4$ (6–0–0–6) fertilizer also increased denitrification rates and N removal (Larson et al., 2000). When compared with the large 3-yr DOC load, overall retention of DOC was insignificant; wetlands were neither a major source nor sink for DOC.

Davis et al. (1981) reported much higher concentrations of DOC (18.0 mg L^{-1}) in wetland inflow from a maize–soybean watershed in Iowa; however, inflow included both surface and subsurface drainage. Our results contrast with those of Davis et al. (1981), who found a prairie pothole wetland to be a major source of C, receiving 302 kg ha^{-1} of soluble organic C but exporting more than 516 kg ha^{-1}. The major source of C export was attributed to high biomass production in the pothole wetland as well as pulse flows following a period of drought.

Constructed Treatment Wetlands and Water Quality

A critical difference between pollutant loading from typical wastewater treatment and that from croplands is that cropland drainage flow and pollutant concentrations are event driven (Baker, 1992). This is a characteristic of nonpoint pollution, which is often intermittent and related to the intensity of intermittent runoff events, making it difficult to quantify and even more difficult to control (Vigon, 1985). Pulse events lead to runoff, floodpeaks, and mobilization of pollutants from agricultural lands. This was the case with our constructed wetlands, where pulse flows occurred in the spring and winter seasons, resulting in the greatest tile discharges and nutrient fluxes.

Evidence of Additional Nitrogen Removal

Larson et al. (2000) found that NO_3–N moved with seepage water through the berm. Nitrate concentrations in seepage water passing through the berm, however, were initially reduced within the treatment wetlands. Chloride to NO_3 ratios indicated that NO_3–N reduction was not a result of ground water dilution or stream flow dilution and that denitrification in the subsurface soil was the probable mechanism decreasing NO_3–N concentration to zero adjacent to the stream (Larson et al., 2000). This work indicated that although water carrying NO_3–N did seep through the wetland berms, more NO_3–N was removed through denitrification before it entered the stream. The underlying tile drainage that once rendered buffer strips ineffective in removing N was modified by the creation of wetlands forming an adjacent *riparian buffer* that was effective in removing an additional 9% of NO_3–N exiting the wetland as seepage water (Larson et al, 2000). When the wetland and buffer strip were considered as a *coupled system*, the overall total N removal was increased to 46% of inputs.

Contribution to Agricultural Watersheds in Illinois

The greatest contribution of wetlands to water quality in Illinois may be in municipalities where drinking water reservoirs typically are in violation of drinking water standards during spring and summer (e.g., Bloomington, Decatur, and Danville). In many such areas, water discharged from treatment wetlands may be well below the MCL for drinking water. In two of four wetland/water years, when inlet NO_3–N levels were above 10 mg L^{-1}, outlet NO_3–N levels were reduced to below 10 mg L^{-1}. Absolute wetland loads were reduced, however, in all nine wetland/water–years.

Assuming that tile drainage accounts for nearly all the N entering the upper Embarras River watershed above Camargo, IL (David et al., 1997a), treatment wetlands with a wetland to watershed ratio of approximately 20:1 constructed on 50% of the tile systems would have a capacity to remove 18 to 23% of the inflow N. These wetlands would have lowered the annual nitrate concentration from 7.8 to between 6.0 and 6.5 mg NO_3–N L^{-1} in 1995 and from 10.1 to between 7.8 and 8.3 mg NO_3–N L^{-1} in 1996. This is a significant reduction considering these systems are inexpensive, low-maintenance, gravity-fed systems that are long lived due to low sediment loading. Tile drainage wetlands are a practical, low cost method to improve surface water quality. They also allow farmers to maintain their current production practices while reducing nonpoint pollution export to surface waters.

Constructed wetlands contribute to the environment in several ways besides water quality improvement. They can reduce flood peaks and increase habitat for wildlife and rare flora (Feierabend, 1989; Knight, 1992; Baker, 1992). In states such as Illinois, where 99.5% of the natural wetlands and wetland prairies have disappeared because of drainage and agricultural conversion (Mitsch and Gosselink, 1993), there is a critical need for the reestablishment of wetland habitat. Constructed wetlands also increase areas for aesthetics and both nonconsumptive and consumptive (hunting and fishing) recreation offering alternative agricultural business opportunities. As an example, the Embarras River wetland site was also operated as a private hunting preserve.

CONCLUSIONS

Results of this study indicated that constructed wetlands can effectively reduce overall NO_3–N loading from tile drainage systems to surface waters. In an east-central Illinois watershed, three constructed wetlands were found to reduce overall NO_3–N and total N loading to surface waters by 38 and 37%, respectively, during water years 1995–1997. The flow weighted NO_3–N concentration in tile drainage water was reduced by 28% overall. In two of four wetland/water years where average inlet concentrations exceeded the MCL for drinking water (10 mg L^{-1}), wetland treatment reduced the average outlet concentration below the MCL. The coupled effect of wetland NO_3–N removal and supplemental berm seepage NO_3–N removal increased the overall wetland N removal efficiency to 46%. In contrast, the wetlands were neither a major source nor a sink for either P or C.

Wetlands with low watershed to wetland area ratios have greater storage capacities and longer retention times and are most effective in removing NO_3–N. In this study, WB, with the lowest watershed to wetland area ratio (17) was most effective in NO_3–N removal. Although treatment wetlands with larger ratios will contribute to NO_3–N removal, it is suggested that optimum constructed treatment wetland designs should aim for watershed to wetland area ratios between 15 and 20. Constructed wetlands are passive low maintenance systems that may be useful tools in reducing NO_3–N loading to surface waters in artificially drained agricultural watersheds of the Midwest.

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