Wetland Management Reduces Sediment and Nutrient Loading to the Upper Mississippi River

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Wetland Management Reduces Sediment and Nutrient Loading to the Upper Mississippi River

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Restored riparian wetlands in the Upper Mississippi River basin have potential to remove sediment and nutrients from tributaries before they flow into the Mississippi River. For 3 yr we calculated retention efficiencies of a marsh complex, which consisted of a restored marsh and an adjacent natural marsh that were connected to Halfway Creek, a small tributary of the Mississippi. We measured sediment, N, and P removal through a mass balance budget approach, N removal through denitrification, and N and P removal through mechanical soil excavation. The marsh complex had average retention rates of approximately 30 Mg sediment ha^{-1} yr^{-1}, 26 kg total N ha^{-1} yr^{-1}, and 20 kg total P ha^{-1} yr^{-1}. Water flowed into the restored marsh only during high-discharge events. Although the majority of retention occurred in the natural marsh, portions of the natural marsh were hydrologically disconnected at low discharge due to historical over-bank sedimentation. The natural marsh removed >60% of sediment, >10% of P, and >5% of N loads (except the first year, when it was a N source). The marsh complex was a source of NH_4^+ and soluble reactive P. The average denitrification rate for the marsh complex was 2.88 mg N m^{-2} h^{-1}. Soil excavation removed 3600 Mg of sediment, 5.6 Mg of N, and 2.7 Mg of P from the restored marsh. The marsh complex was effective in removing sediment and nutrients from storm flows; however, retention could be increased if more water was diverted into both restored and natural marshes before entering the river.

In the Mississippi River basin, nonpoint runoff primarily from agricultural activities and secondarily from urban areas (Goolsby et al., 1999, 2000; Alexander et al., 2008) is having detrimental effects on the water quality and ecology of local streams and rivers (Griffith et al., 2009) and coastal Gulf of Mexico ecosystems (Turner and Rabalais, 1991; Committee on Environment and Natural Resources, 2000; Rabalais et al., 2002; USEPA, 2008; Broussard and Turner, 2009; Battaglin et al., 2010; Sprague et al., 2011). Increased sediment loading leads to the filling of off-channel areas (Bhowmik and Adams, 1989), and greater loads of suspended solids limit light transmission into the water column, hindering the growth of aquatic macrophytes (Moore et al., 2010), phytoplankton (Whalen and Benson, 2007), and invertebrates (Griffith et al., 2009). Excess N and P inputs contribute to localized eutrophication, and the combination of increasing nutrients and sediment degrades the floodplain (Nietch et al., 2005; Griffith et al., 2009; Houser and Richardson, 2010). Further downstream, nutrients, particularly NO_3^-, are associated with a large annual hypoxic zone in the Gulf of Mexico (Turner et al., 2008). The economic impact of such environmental degradation is large; shellfish and ground fish fisheries are dwindling and nonexistent in the area of hypoxic water (Rabalais et al., 2002). With the increased production of corn (*Zea mays* L.)-based ethanol, this problem is likely to grow, and it has been estimated that the amount of dissolved inorganic N entering the Gulf will increase by 1 to 34% (Donner and Kucharik, 2008). Because of these local and regional environmental problems, many state and national management agencies are developing more comprehensive mitigation and remediation strategies.

One common management strategy is the construction or restoration of riparian wetlands along small tributaries to trap and retain sediments and nutrients (Mitsch and Day, 2006). Along with the burial of sediment and particulate nutrients, wetlands can immobilize nutrients through temporary storage in plant biomass, the binding of P to cations in the

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Abbreviations: CTH, County Trunk Highway; SRP, soluble reactive phosphorus; SSC, suspended sediment concentration; TN, total nitrogen; TP, total phosphorus.

soil, and the permanent removal of N through denitrification (Johnston, 1991; Mitsch and Gosselink, 2007). The removal of nutrients from the tributaries is ideal because once N and P reach the Mississippi River, little in-stream nutrient removal occurs and the majority of the nutrients are transported to the Gulf of Mexico (Goolsby et al., 1999, 2000; Richardson et al., 2004; Alexander et al., 2008).

The U.S. Fish and Wildlife Service manages the Upper Mississippi River floodplain and much of the historical riparian zone for the benefit of wildlife and fisheries resources. Such management serves to protect much of the floodplain from urban and agricultural development and may act as an intercepting buffer, collecting and removing sediment and nutrient loads transported from the uplands. One common management activity is wetland restoration on agricultural fields. The managed moist soil and seasonal wetlands provide feeding grounds for migratory birds and increase waterfowl production (U.S. Fish and Wildlife Service, 2011), and within 10 yr of restoration, waterfowl usage in restored wetlands can be similar to natural wetlands (Stevens et al., 2003). As added benefits, the restoration and construction of wildlife habitat increases sediment and nutrient retention in the floodplain and improves water quality (Kadlec and Knight, 1996; Johnston et al., 1997; Craft and Schubauer-Berigan, 2006; Ardon et al., 2010).

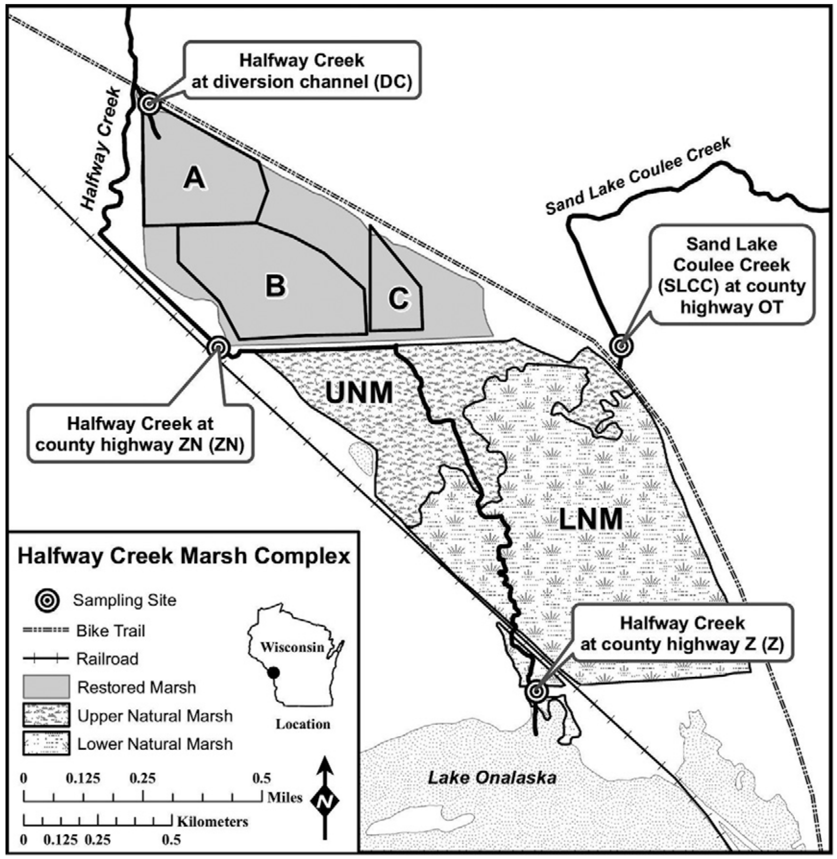

Fig. 1. Map of Halfway Creek Marsh Complex, Onalaska, WI, indicating Cells A, B, and C in the restored marsh and the upper natural marsh (UNM) and lower natural marsh (LNM). Sampling sites are for stream discharge and water quality.

One wetland along the Upper Mississippi River floodplain that is managed for waterfowl and as a sediment retention basin is the Halfway Creek Marsh Complex in Onalaska, WI. The upper restored section of the marsh complex is a former agricultural field that receives water from Halfway Creek only during high-discharge events. Although the lower natural section receives flow directly from Halfway Creek, portions of it are hydrologically disconnected during low-discharge events due to historical overbank sedimentation of Halfway Creek (Fitzpatrick et al., 2007, 2009). The design of the marsh complex created a unique opportunity to assess the effectiveness of using restored wetlands as sediment and nutrient traps during high stream discharge events and to compare sediment and nutrient retention capabilities of a restored marsh with an adjacent natural marsh. Our objectives were to determine (i) the effectiveness of a restored marsh to retain sediment and nutrients under a range of hydrologic conditions, (ii) the effectiveness of a restored marsh to retain sediment and nutrients compared with an adjacent natural marsh, and (iii) the effectiveness of removal of N and P by way of soil or sediment excavation compared with natural biogeochemical processes.

Materials and Methods

Site Description

Halfway Creek and Sand Lake Coulee Creek are adjacent watersheds in southwestern Wisconsin, near the city of Onalaska (Fig. 1; Table 1). Both of these watersheds drain into Lake Onalaska (Navigation Pool 7 of the Mississippi River). Land use in the two watersheds is primarily agricultural and forest, but commercial and residential development is occurring rapidly. Sediment transport models estimated movement of approximately 46,000 Mg of sediment through the Halfway Creek watershed in 1995, with an additional 4100 Mg of sediment moving through the Sand Lake Coulee Creek watershed (Vierbicher Associates, 1995). The entire complex is comprised of two primary sections: the restored marsh (Cells A, B, and C) and the natural marsh (upper and lower sections; Fig. 1; Table 1). The current site of the restored marsh was drained by ditching and dike construction in the early 1900s for planting and harvesting of row crops and hay. In 1999, a diversion channel and a connected three-celled marsh (Cells A, B, and C) was built on a portion of the Halfway Creek Marsh Complex managed by the U.S. Fish and Wildlife Service. The marsh complex was restored primarily to help capture and

Table 1. Areas of Halfway Creek and Sand Lake Coulee Creek drainage basins and sections within the Halfway Creek Marsh Complex.

Location	Area
	ha
Halfway Creek basin	8830
Sand Lake Coulee Creek basin	2590
Cell A of the restored marsh	9.6
Cell B of the restored marsh	17.9
Cell C of the restored marsh	4.7
Upper natural marsh	33.4
Lower natural marsh	78.8

retain sediments during increased flow periods from Halfway Creek and to replace lost waterfowl habitat. The natural marsh remained in a relatively natural state; however, it was subjected to some channel modification to increase water conveyance and has been affected by historical sedimentation of Halfway Creek (Fitzpatrick et al., 2009). Vegetation of the upper natural marsh consists mainly of homogeneous stands of reed canarygrass (*Phalaris arundinacea* L.), whereas the lower natural marsh and restored marsh have more diverse communities of both native and non-native grasses and forbs and scattered tussocks of small trees.

The hydrology of the restored marsh is managed through manipulation of a stop-log structure at the farthest upstream section (Cell A) of the restored marsh. The inlet was constructed to divert water into the restored marsh during periods of high flow, when the creek stage is >0.2 m. Diversion occurs only during strong storms or spring thaw, which results in a seasonal pulse of water, sediment, and nutrients to the restored marsh. Nutrients and sediments carried in base flow never reach the restored marsh. There is no outlet from the restored marsh, and to remove excess deposited sediment, occasionally a private landscaper mechanically removes soil from the restored marsh at no cost to the Fish and Wildlife Service. This removal of soil benefits both the marsh and the landscaper. The removal of accumulated soil prevents the restored marsh from filling in, and the landscaper profits from the sale of nutrient-rich soil.

Water Quality of Halfway and Sand Lake Coulee creeks

Continuous-record stream-flow and water-quality gaging stations were installed at two locations in Halfway Creek, at County Trunk Highway (CTH) ZN and the inlet of the diversion channel into Cell A of the restored marsh, and at one location on Sand Lake Coulee Creek, at CTH OT, in February 2004 (Fig. 1). Ice conditions at the gaging station on Sand Lake Coulee Creek prevented sample collection during the February 2004 runoff. The first samples collected at this site were on 7 Apr. 2004. A fourth station was added in February 2005 at the outlet of Halfway Creek to the Mississippi River, at the downstream end of the natural marsh (at CTH Z). Data were collected through December 2006 at all stations except the Sand Lake Coulee Creek station, which had data collection through September 2006. Each gaging station was equipped with Campbell Scientific CR10 dataloggers to record data at 15-min intervals during base flow and 5-min intervals during storm events. Stream stage was measured with a Sutron Accubar Model 5600 pressure sensor. Water samples were collected with an ISCO Model 2700 refrigerated water-quality sampler. Automated sample collection and data storage were controlled at each gage by the datalogger. The gaging station on Halfway Creek at CTH ZN served as a base station that was connected to the other gages by Campbell RF400 radios. A COM210 modem at the base station was used to transmit real-time data for all four gages to the USGS office in Middleton, WI. Stage measurement recorded by the pressure transducer system was converted to a record of stream flow based on a stage-flow relationship developed from manual flow measurements and a record of the stream cross-section at the location of the gage. Precipitation data were based on observations collected at the La Crosse Municipal Airport, the nearest recording station of the National Weather Service, which was 5 km south of the study area.

We characterized concentrations during base flow by routine, non-event, point samples collected manually at biweekly intervals during the period March through November and monthly during the winter months of December through February. Discrete storm-event samples were collected with refrigerated ISCO 2700R automated samplers, which were triggered in response to stage changes caused by runoff events. A stage increase or decrease of approximately 6 cm triggered the ISCO samplers to collect a sample. Samples were chilled in the refrigerator to 4°C and collected within 24 h of the end of a storm event.

All samples were analyzed at the USGS Upper Midwest Environmental Sciences Center water quality laboratory, La Crosse, WI. A representative subset of the collected samples from each storm was analyzed for nutrient and sediment concentrations to sufficiently characterize the discharge–concentration relationship for each storm. Sample size depended on the storm and ranged from 2 to 40 samples collected, with at least half of the collected samples being analyzed. Smaller events had a higher percentage of collected samples analyzed. In the laboratory, samples were split for appropriate analyses using a Decca 10-port splitter. Whole water samples were analyzed for total N (TN), total P (TP), and suspended sediment concentration (SSC). Water filtered through a 0.45-μm glass-fiber filter was analyzed for NO_3^-–N, soluble reactive P (SRP), and NH_4^+–N. Nitrate concentration was determined using the automated Cd reduction method (American Public Health Association, 2005). Ammonium was determined using the automated phenate method (American Public Health Association, 2005). Total N was determined using persulfate digestion followed by the automated Cd reduction method (American Public Health Association, 2005). Soluble reactive P was determined using the ascorbic acid method (American Public Health Association, 2005). Total P was determined using persulfate digestion followed by the ascorbic acid method (American Public Health Association, 2005). Suspended sediment concentration was determined using Test Method B of ASTM Method D3977-97 (ASTM, 1997).

Load Estimation

Nutrient and sediment concentration and stream discharge data for Halfway Creek at CTH ZN and the diversion channel (February 2004–December 2006), in Sand Lake Coulee Creek (April 2004–September 2006), and in Halfway Creek at CTH Z (February 2005–December 2006) were used to compute the daily load for all the analyzed constituents at each of the gaging stations. The USGS Graphical Constituent Loading Analyses System (GCLAS) (Koltun et al., 2006) was used to analyze and compute the time-series water quality loads. The GCLAS dynamically links measured concentration and transport curves by integrating the concentration values over the discharge hydrograph. Standard errors of the load estimates cannot be made with GCLAS; however, it is considered to be an acceptable method as long as sufficient data (100–200 samples yr^{-1}) are collected to illustrate the changes in water quality (Robertson, 2003).

To estimate yearly loads, daily loads calculated with the GCLAS method were summed across the entire annual period of record. The installation of the monitoring station at CTH Z (Z) did not occur until February 2005, causing us to estimate loads in 2004 based on discharge at CTH ZN (ZN). We regressed discharge at ZN against discharge at Z from 2005, 2006, and 2007 (model

$R^2 = 0.63$, $P < 0.0001$). We used the regression model to calculate discharge at Z for all 3 yr of the study. The calculated discharge at Z was then regressed against the nutrient and sediment data from Z in 2005 and 2006 (SSC model $R^2 = 0.53$, TP model $R^2 = 0.46$, SRP model $R^2 = 0.42$, TN model $R^2 = 0.55$, NO_3^-–N model $R^2 = 0.12$, and NH_4^+–N model $R^2 = 0.57$; $P < 0.0001$ for all models). The resulting regression models were used to calculate the nutrient and sediment loads at Z in 2004 based on the calculated discharge at Z.

Retention by the total marsh complex was the difference between inputs (CTH ZN, inflow into the diversion channel [DC], and Sand Lake Coulee Creek [SLCC] loads) and outputs (outflow at Z). We calculated the retention efficiency (%) during 2004, 2005, and 2006 using

$$\text{Retention efficiency} = \frac{\text{Load}_{\text{ZN}} + \text{Load}_{\text{SLCC}} + \text{Load}_{\text{DC}} - \text{Load}_{\text{Z}}}{\text{Load}_{\text{ZN}} + \text{Load}_{\text{SLCC}} + \text{Load}_{\text{DC}}} 100 \qquad [1]$$

Retention by the natural marsh was the difference between the inputs (CTH ZN and Sand Lake Coulee Creek) and outputs (outflow at Z). Because there is no outlet from the restored marsh, we assumed that all the material entering it was retained.

Soil Characteristics and Marsh Complex Denitrification and Nitrification

In 2004, multiple sediment cores (10-cm diameter, 10 cm deep) were collected at 34 random points in Cell A (28 dry, six in standing water), 10 each in Cells B and C, and five points each in the upper and lower natural marsh; in Halfway Creek, four samples each were collected at the inflow to the restored marsh, below the restored marsh at ZN, near the outfall to Lake Onalaska at Z, and Sand Lake Coulee Creek. A greater sampling effort was exerted in Cell A to enable better characterization of the primary flood-deposited mass of sediment and nutrients. Four cores were used for experimental estimation of the N and C limitation of denitrification using the acetylene block technique (Groffman et al., 1999; Richardson et al., 2004). Rates of nitrification were estimated on a second set of cores using the nitrapyrin technique, where the addition of nitrapyrin to sediment slurries inhibits nitrification, resulting in an increase in NH_4^+–N (Hall, 1984; Strauss and Lamberti, 2001; Strauss et al., 2004).

To evaluate temporal variations in denitrification, denitrification enzyme activity, and nitrification, we sampled sediment or soil at 25 sites in Cell A (20 dry, five in standing water) and five sites in the lower natural marsh in June and August during 2005. These areas had contrasting sediment characteristics that could affect N-cycling processes in the natural and restored marshes. We also measured soil NO_3^-–N in cores through KCl extraction followed by colorimetric analysis using the automated Cd reduction method (American Public Health Association, 2005).

In 2004, limitation experiments were conducted to determine the factors limiting denitrification rates. Slurries with added labile C (12 mg glucose-C L^{-1}, final concentration), NO_3^-–N (14 mg NO_3^-–N L^{-1}, final concentration), or a combination of both C and N (made from soil from the top 5 cm of the test core and mixed with filtered Halfway Creek water) were compared with slurries with no additions of C or N (ambient denitrification). All treatments contained a final concentration of 100 mg chloramphenicol L^{-1} to inhibit bacterial protein synthesis. Deviation of rates of denitrification in the C- and N-amended slurries from ambient denitrification indicated the degree of C or N limitation. These experiments also provided an indication of the capacity of marsh soils and stream sediments to denitrify newly loaded NO_3^-–N deposited during storm flows. Finally, the addition of both C and N is an estimation of the maximum enzymatic capacity or denitrification enzyme activity of soils and sediments and is useful for comparisons across sites (Groffman et al., 1999).

To estimate NO_3^-–N loss in the top 5 cm of soil from Cell A of the restored marsh, from the upper natural marsh, and from the lower natural marsh, we summed the mean nitrification and denitrification rates obtained from analyses of the 5-cm soil cores. We extrapolated the values to the entire area of each individual section (i.e., Cell A of the restored marsh, the upper natural marsh, and the lower natural marsh). To determine the loss in the natural marsh, we summed the extrapolated values from the upper natural marsh and the lower natural marsh. We assumed that nitrification and denitrification were coupled because denitrification was limited by NO_3^-–N and not C, and NO_3^-–N produced by nitrification on the soil surface can diffuse into the saturated soil and be available for denitrification (Pinay et al., 2002). The rate estimates used were from the 2004 and 2005 surveys. Because the upper natural marsh was sampled only in 2004, only estimates from that year were used.

Soil Excavation

During the summer of 2005, soil was excavated from Cell A and redeposited in upland sites by a commercial landscaper. Most of the material was removed from the top 0.5 m and focused on the area at the base of the diversion channel. Approximately 3513 ± 480 Mg were removed. We took soil cores in the area before excavation to measure C and N, which were determined with a Variomax CN analyzer (Elementar). Soil cores were taken from Cell A earlier in the summer and sediment acid-fluoride-extractable P was determined following the method of Olsen and Sommers (1982).

Statistical Analysis

Data were analyzed for assumptions of normality and homogeneity of variances. Any data not meeting these requirements were typically natural-log transformed before analysis, except when otherwise reported. We ran one-way ANOVAs to determine if there were any significant differences in the denitrification and nitrification rates among the different sampling locations. Data from 2004 and 2005 were combined for Cell A and the lower natural marsh.

For the temporal data analysis, we first conducted two-way ANOVAs to determine if there were significant differences among sampling locations (lower natural marsh and wet and dry areas in Cell A) and among sampling periods (August 2004, April 2005, and June 2005) and to determine if there were significant interactions between sampling location and sampling period. Because we had significant interactions between sampling locations and sampling periods for the two-way ANOVAs, we grouped the data by time and location and ran one-way ANOVAs to determine if there were significant differences in rates by a combination of sampling time and location.

For the denitrification limitation experiment, data were square-root transformed and were analyzed with a one-way ANOVA to determine if either of the treatments or the combination of the treatments had an effect on the denitrification rate. Significance of differences among treatment means for all ANOVAs was tested using the Tukey method. All statistical analyses were conducted with SAS software (SAS Institute, 1990).

Results

Stream Discharge

Discharge of Halfway Creek (at CTH Z) was variable during the 3 yr of this study as a result of fluctuations in yearly precipitation (Table 2). Discharge was greatest in 2004 when 105 cm of precipitation fell, which is well above the average yearly precipitation of 82 cm. This followed a dry year in 2003 when only 57 cm of precipitation fell. Although precipitation levels were similar in 2005 and 2006 (approximately 76 cm yr^{-1}), discharge was greater in 2005. In 2006, flow into the restored marsh was minimal (Table 2). Discharge from storm and base flow during storm events accounted for 25, 16, and 29% (2004, 2005, and 2006, respectively) of the annual discharge at CTH ZN, and it accounted for 21 and 15% of total flows at CTH Z in 2005 and 2006, respectively.

Halfway Creek was the main source of water for the natural marsh. Sand Lake Coulee Creek contributed only 3% of the annual discharge into the natural marsh. By subtracting annual creek discharge values from the monitoring stations at CTH ZN and Sand Lake Coulee Creek from the annual creek discharge value at CTH Z, we estimated that the groundwater contribution of the natural marsh was 2.2 and 2.7×10^{-3} km^3 in 2005 and 2006, respectively.

Table 2. Yearly discharges for Halfway Creek (HWC) from February 2004 through December 2006 and Sand Lake Coulee Creek (SLCC) from February 2004 through September 2006.

Site	Discharge		
	2004	2005	2006
	———— $km^3\ yr^{-1}$ ————		
HWC into restored marsh	3.23×10^{-4}	1.22×10^{-4}	6.09×10^{-6}
HWC at County Highway ZN	1.61×10^{-2}	1.07×10^{-2}	8.89×10^{-3}
SLCC at County Highway OT	9.77×10^{-4}	4.31×10^{-4}	2.39×10^{-4}
HWC at County Highway Z	1.58×10^{-2}	1.33×10^{-2}	1.18×10^{-2}

Sediment and Nutrient Loads and Retention

The marsh complex retained the majority of the sediment (range 63–86%) that flowed into the complex during the study period (Fig. 2). The retention efficiency of the complex generally increased as discharge increased. In 2004, the year of greatest sediment retention, >8500 Mg of sediment was retained by the marsh complex. This was 15 times the 540 Mg of sediment that was retained in 2006, the year of lowest retention. Deposition in 2006, however, was still 63% of the total load carried by Halfway Creek and Sand Lake Coulee Creek. Due to limited creek discharge into the restored marsh, the majority of the retention occurred in the natural marsh. During the entire study, sediment retention of the restored marsh was 749 Mg of sediment, which was only 6.8% of the total sediment retention by the marsh complex. Meanwhile, the natural marsh retained 10,200 Mg of sediment.

The marsh complex retained N and P but at rates much lower than the rates of sediment retention. Total P, like sediment, was retained at a greater rate during high-discharge years. The greatest retention occurred in 2004 when >5700 kg of TP was deposited in the marsh complex, and the lowest retention occurred in 2006 when only 530 kg of TP was deposited (Fig. 3). Meanwhile, the marsh complex exported SRP, especially during increased discharge (Table 3). Total N was retained throughout the study, with 2005 having the greatest retention (Fig. 4). In 2004, the least amount of TN was retained even though that was the year when the most N was transported through the marsh complex. The natural marsh was a source of TN in 2004, exporting 1447 kg of TN to Lake Onalaska (Fig. 4). The complex still retained TN, however, because the restored marsh removed 2021 kg of TN from Halfway Creek. From 2004 to 2006, the marsh complex also retained NO_3^-–N but was a source of NH_4^+–N (Table 3). During 2004, the year of greatest NH_4^+–N and SRP export, the natural marsh contributed an additional 3275 kg of NH_4^+–N and 1278 kg of SRP to Lake Onalaska.

Marsh Complex Denitrification and Nitrification

Cell A had the highest ambient denitrification rates, which were significantly greater than the rates of the lower natural marsh ($F_{[7,\ 132]} = 4.23$, $P = 0.0003$; Fig. 5A). Denitrification enzyme activity (Fig. 5B) and nitrification rates (Fig. 5C) were not significantly different among areas.

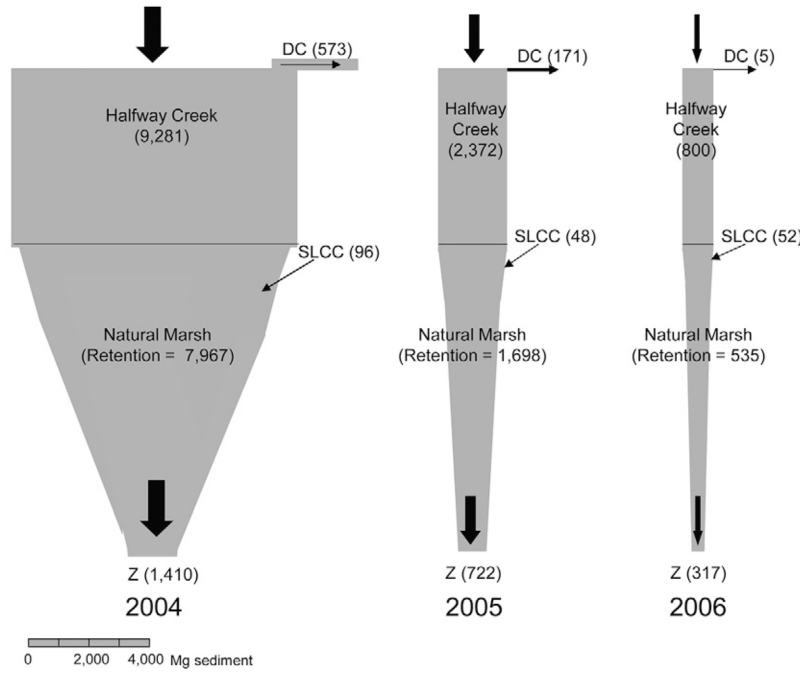

Fig. 2. Suspended sediment budget (in Mg) for Halfway Creek Marsh Complex in February through December 2004, January through December 2005, and January through December 2006. Marsh inlets include Halfway Creek, Sand Lake Coulee Creek (SLCC), and the diversion channel into the restored marsh (DC). Outflow is at Halfway Creek at County Highway Z (Z). Black arrows represent the direction of flow. Width of the gray bars and black arrows indicate sediment load.

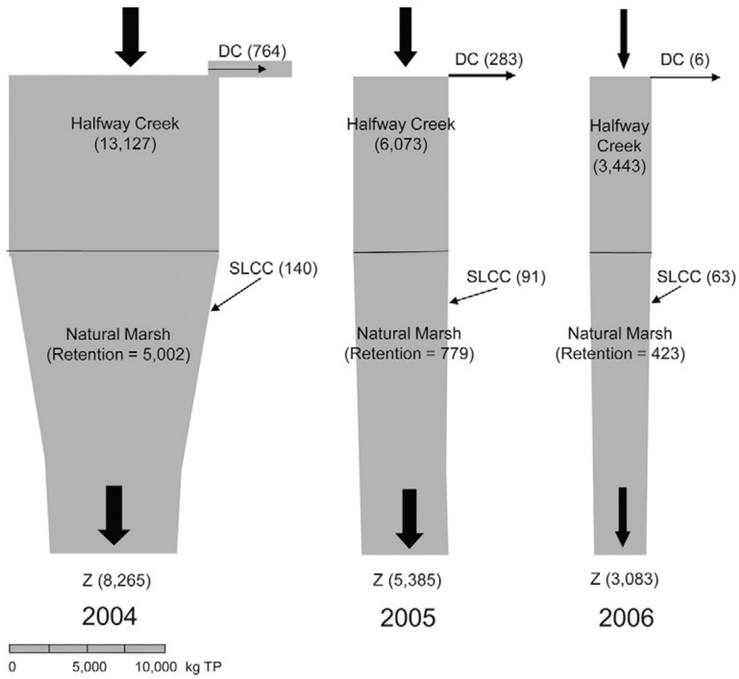

Fig. 3. Total P budget (in kg) for Halfway Creek Marsh Complex in February through December 2004, January through December 2005, and January through December 2006. Marsh inlets include Halfway Creek, Sand Lake Coulee Creek (SLCC), and the diversion channel into the restored marsh (DC). Outflow is at Halfway Creek at County Highway Z (Z). Black arrows represent the direction of flow. Width of the gray bars and black arrows indicate P load.

Table 3. Nitrate, NH_4^+, and soluble reactive P inflow loads from Halfway Creek (HWC) and Sand Lake Coulee Creek (SLCC) and outflow loads and retention efficiencies for Halfway Creek Marsh Complex from February 2004 through September 2006.

Nutrient	2004 Feb.–Dec.	2005 Jan.–Dec.	2006 Jan.–Sept.
Dissolved inorganic N, kg yr^{-1}			
Inflow to restored marsh	621	287	6
Inflow to natural marsh from HWC	26,937	23,956	18,838
Inflow to natural marsh from SLCC	35	112	153
Outflow from natural marsh	27,353	21,231	18,273
Retention in marsh complex, %	1	13	4
NO_3^-, kg yr^{-1}			
Inflow to restored marsh	390	100	5
Inflow to natural marsh from HWC	24,282	21,643	18,127
Inflow to natural marsh from SLCC		58	125
Outflow from natural marsh	21,388	17,655	17,468
Retention in marsh complex, %	13	19	4
NH_4^+, kg yr^{-1}			
Inflow to restored marsh	231	187	1
Inflow to natural marsh from HWC	2655	2313	711
Inflow to natural marsh from SLCC	35	54	28
Outflow from natural marsh	5965	3576	805
Retention in marsh complex, %	−104	−40	−9
Soluble reactive P, kg yr^{-1}			
Inflow to restored marsh	90	75	1
Inflow to natural marsh from HWC	1867	1927	846
Inflow to natural marsh from SLCC	18	24	9
Outflow from natural marsh	3163	2184	875
Retention in marsh complex, %	−60	−8	−2

Ambient denitrification rates around 50 mg N m^{-2} h^{-1} were observed in the dry areas of Cell A in April and June 2005 and in the wet area of Cell A in June 2005. These rates were significantly higher than the rates observed at all sites in August 2004, in the lower natural marsh during April and June 2005, and in the wet area of Cell A in April 2005 ($F_{[8,90]} = 43.71$, $P < 0.0001$). Denitrification enzyme activity was 140 ± 26 mg N m^{-2} h^{-1} in the wet area in Cell A during April 2005. This rate was significantly higher than the rates observed at all sites during the August sampling period ($F_{[8, 90]} = 7.99$, $P < 0.0001$). The nitrification rate was greatest in the wet area in Cell A in April 2005 at 8.6 ± 2.4 mg N m^{-2} h^{-1} ($F_{[8, 90]} = 3.59$, $P < 0.0001$). In 2005, soil NO_3^-–N concentrations were not significantly different among sites or between time periods. In April, the soil NO_3^-–N concentration was 3.09 ± 1.27 g NO_3^-–N L^{-1} in the dry area of Cell A and undetectable in the lower natural marsh and the wet area of Cell A, whereas in June, the soil NO_3^-–N concentration was undetectable in the lower natural marsh but was 0.92 ± 0.12 g NO_3^-–N L^{-1} in the dry area and 1.13 ± 0.32 g NO_3^-–N L^{-1} in the wet area of Cell A.

Denitrification was limited more by the availability of NO_3^-–N than C (Fig. 6). Denitrification in sediment from the lower natural marsh, Halfway Creek at CTH Z, and all three sites in the restored marsh increased with the addition of NO_3^-–N (Fig. 6A–6C, 6E, and 6F). Nitrate limitation was especially strong in sediments from the lower natural marsh (Fig. 6E) and Halfway Creek at Z (Fig. 6F). Stream sediments showed little change in ambient denitrification with the addition of N, except for Halfway Creek at Z. Carbon limitation occurred only in Halfway Creek at the diversion channel (Fig. 6G) and at ZN (Fig. 6I). Nitrogen and C colimitation occurred at all sites, except in the upper natural marsh (Fig. 6D) and Sand Lake Coulee Creek (Fig. 6H). Colimitation of N and C was strongest in the lower natural marsh (Fig. 6E) and lowest in the upper natural marsh (Fig. 6D).

Soil and Nutrient Removal

In 2005, 14.2 Mg of N were permanently removed from the restored marsh by a combination of soil excavation (39% of N removal) and denitrification (61% of N removal). Ambient denitrification resulted in a loss of 62.1 kg N d^{-1} in the top 5 cm of soil from Cell A. When accounting for the additional denitrification stimulated by nitrification-derived NO_3^-–N, 71.3 kg N d^{-1} was lost. If we assume a growing season of 120 d, biogeochemical loss of NO_3^-–N would be about 8.6 Mg yr^{-1} in the top 5 cm of soil throughout Cell A of the restored marsh. In the natural marsh, 89.9 kg N d^{-1} would be denitrified in the top 5 cm of soil. The combination of ambient denitrification and coupled nitrification–denitrification would result in 17.75 Mg of NO_3^-–N lost from the natural marsh during the growing season. When the area around the base of the diversion channel was excavated and 3.6 Mg of sediment was removed, approximately 64.8 Mg of soil C, 5.6 Mg of soil N, and 2.7 Mg of soil P were removed.

Discussion

Sediment and Nutrient Retention

The Halfway Creek Marsh Complex was an effective sediment trap for loads delivered by Halfway Creek and Sand Lake Coulee Creek. The marsh complex had an average retention rate of 30 Mg sediment ha^{-1} yr^{-1}, which was 75% of the load carried by the two streams and greater than the reported average retention rate of 7.8 Mg sediment ha^{-1} yr^{-1} for 25 wetlands in Pennsylvania (Wardrop and Brooks, 1998) but within the range reported for other wetlands in the United States (range 0.18–78.4 Mg sediment ha^{-1} yr^{-1}; Johnston, 1991). Our estimated retention of 3500 Mg yr^{-1} was less than the overbank load of 4500 Mg sediment yr^{-1} measured by Fitzpatrick et al. (2007), who sampled sediment cores extensively throughout the marsh complex.

Because of low connectivity between Halfway Creek and the natural and restored marshes during base flow, the marsh complex was not as effective as other wetlands in removing N, but it did effectively remove P. The marsh complex removed 10% of the TN load at a rate of 26 kg N ha^{-1} yr^{-1} (range 4.7–51.5 kg N ha^{-1} yr^{-1}), which is <8% of the average value of 387 kg N ha^{-1} yr^{-1} reported for surface wetlands (Kadlec and Knight, 1996) and less than the estimated rate of 290 kg NO_3^-–N ha^{-1} yr^{-1} for wetlands in the Mississippi River basin (Mitsch et al., 2005). Meanwhile, the marsh complex removed 26% of the inflow TP at a rate of 20 kg P ha^{-1} yr^{-1}, which is less than the average for surface-flow wetlands (62 kg P ha^{-1} yr^{-1}; Kadlec and Knight, 1996) but more than the range of 8 to 15 kg P ha^{-1} yr^{-1} reported for nutrient-enriched wetlands (Craft and Schubauer-Berigan, 2006). The marsh complex removal rates were similar to rates reported in other restored agricultural wetlands. Jordan et al. (2003) reported a yearly retention range of −11 to 45 kg N ha^{-1} yr^{-1} and −11 to 59 kg P ha^{-1} yr^{-1} for a wetland in Maryland, and Ardon et al. (2010) reported a range of 2.1 to 9 kg N ha^{-1} yr^{-1} and −0.5 to 0.2 kg P ha^{-1} yr^{-1} for a wetland in North Carolina.

Nitrate retention was less in the Halfway Creek marsh complex than in other restored wetlands. On average, 12% of inflow NO_3^-–N was removed from the marsh complex, while 52% of NO_3^-–N was removed from a restored marsh in Maryland (Jordan et al., 2003), 76% of NO_3^-–N was removed from a wetland complex in California (Mayer, 2005), and approximately 45% of NO_3^-–N was estimated to be removed by other restored wetlands in the Upper Mississippi River basin (Mitsch et al., 2005). Only 20% of the total NO_3^-–N flowing through the marsh complex during the year was transported during high-discharge events. The majority of NO_3^-–N transport occurred under base flow conditions when Halfway Creek was confined to its stream bed. During this time, little water was exchanged with the rest of the natural marsh and no water flowed into the restored marsh. Denitrification rates in Halfway Creek were low and there was little opportunity for biotic uptake in the sparsely vegetated creek. This was in contrast to the wetlands in Maryland (Jordan et al., 2003) and California (Mayer, 2005), where water was allowed to flow throughout the entire marsh and was not confined to a channel, enabling greater denitrification and uptake rates than in Halfway Creek.

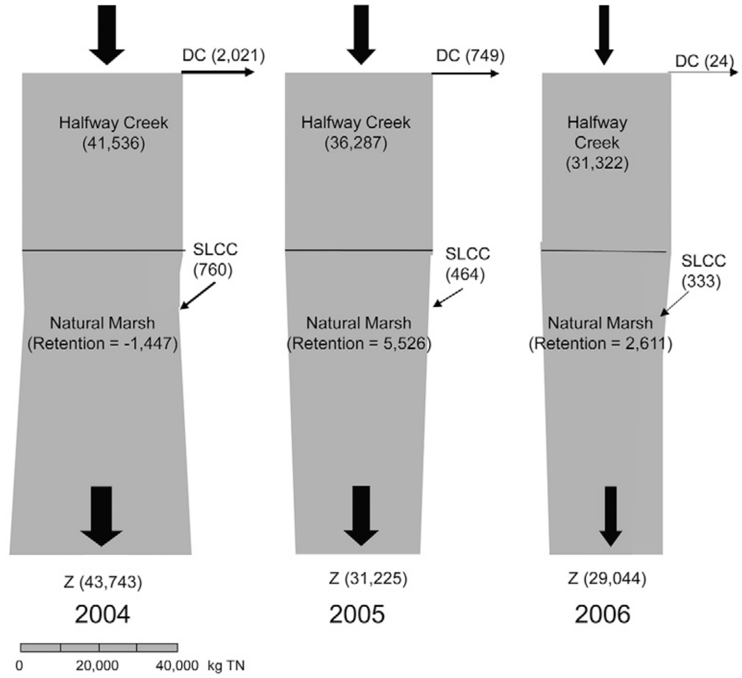

Fig. 4. Total N budget (in kg) for Halfway Creek Marsh Complex in February through December 2004, January through December 2005, and January through December 2006. Marsh inlets include Halfway Creek, Sand Lake Coulee Creek (SLCC) and the diversion channel into the restored marsh (DC). Outflow is at Halfway Creek at County Highway Z (Z). Black arrows represent the direction of flow. Width of the gray bars and black arrows indicate N load.

Although N and P were retained in the marsh complex, there was net export of NH_4^+–N and SRP. Export was especially large during the spring snowmelt, when the entire marsh complex was inundated. Halfway Creek and Sand Lake Coulee Creek flowed directly into the marsh complex, yet the majority of the natural marsh and restored marsh were dry most of the year. During the dry, oxic periods, plant decomposition and the mineralization of organic N and P can rapidly occur, causing an excess pool of available NH_4^+–N and SRP (Kleeberg and Heidenreich, 2004). If the soil was aerobic, some of the NH_4^+–N was oxidized to NO_3^-–N and some of the SRP was precipitated with Fe and immobilized (Reddy et al., 1999). During the growing season, a portion of the NH_4^+–N and SRP was assimilated into more plant biomass. Plant decomposition still occurs during the winter but at a reduced rate (Kadlec and Reddy, 2001). When the snow melted, NH_4^+–N and SRP still present in the soil were released into the water column and transported from the marsh complex (Corstanje and Reddy, 2004). In spring 2004, the nutrient export from the natural marsh was high, probably due to nutrient buildup during the previous year when the precipitation total was 25 cm below average, resulting in minimal flooding and flushing of the marsh complex. Dunne et al. (2010) observed increased P release from deep marsh soils as the number of days since inundation increased.

Marsh Complex Denitrification

Although discharge into the marsh complex was greater in 2004 than 2005, denitrification rates and denitrification enzyme activity were significantly less in 2004. Denitrification rates were limited by NO_3^-–N availability in all areas of the marsh complex except the upper natural marsh and in Halfway Creek

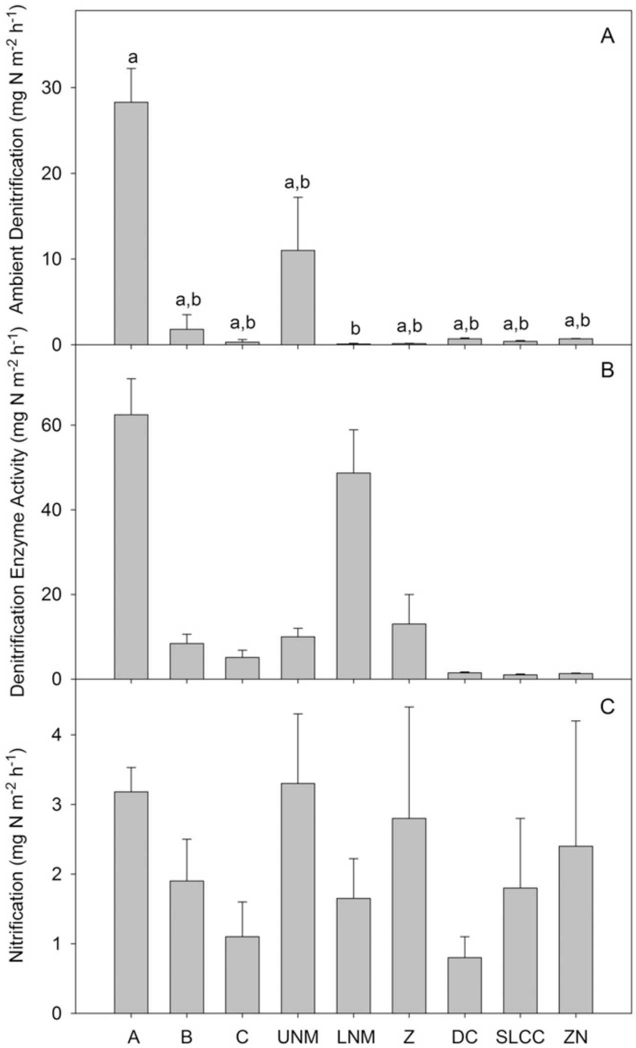

Fig. 5. Average (±1 SE) rates of (A) ambient denitrification, (B) denitrification enzyme activity, and (C) nitrification in sediment or soil from the restored marsh (Cells A, B, and C), the upper natural marsh (UNM), lower natural marsh (LNM), Halfway Creek at County Highway Z (Z), the diversion channel (DC) into the restored marsh, Sand Lake Coulee Creek (SLCC), and Halfway Creek at County Highway ZN (ZN) in 2004 and 2005. Different letters indicate significant differences among sites. There were no significant differences in denitrification enzyme activity or nitrification rates among the sites.

above the lower natural marsh. In 2005, NO_3^-–N did not appear to limit denitrification in Cell A. Soil NO_3^-–N was detected in the dry area of Cell A during both sampling periods and in the wet area during the June sampling period when the soil was dry. Nitrate was not detected in the wet soil of the lower natural marsh during either sampling period. The absence of NO_3^-–N corresponded with a significantly smaller denitrification rate in the lower natural marsh during both time periods and in the wet area of Cell A during the early sampling period.

The method we used to measure denitrification may have contributed to the higher observed rates of denitrification in our dry sites. Burgin et al. (2010) measured the O_2 content in two sites in New York and found that the dry riparian site had a soil O_2 content of 20% throughout the year while the wet riparian site had a soil O_2 content of 0 to 20% depending on the water content of the soil. Because the two sites in our study that had NO_3^-–N were dry, the soil was probably aerobic, limiting in situ denitrification. In the acetylene-block method, O_2 is removed from the soil at the beginning of the incubation to prevent it from inhibiting denitrification, and chloramphenicol is added to prevent the production of additional denitrifying enzymes. When we subjected the soil to anoxic conditions, the available NO_3^-–N was quickly denitrified. The denitrification rates in the dry sites were similar to the denitrification enzyme activity values and were possibly potential denitrification rates instead of actual rates. Across sites, potential denitrification enzyme activity was high, suggesting that under ideal conditions (e.g., abundant soil NO_3^-–N, labile C, and anoxia), the marsh complex has denitrification hotspots and hot moments where the majority of the in situ denitrification occurs (McClain et al., 2003; Groffman et al., 2009).

Our average denitrification rate (8.94 ± 5.99 mg N m^{-2} h^{-1}) for the entire marsh complex was slightly greater than other wetlands, possibly due to our potential overestimation of denitrification rates in the dry areas of Cell A in 2005. Tomaszek et al. (1997) recorded a denitrification range of 1.12 to 11.98 mg N m^{-2} h^{-1} for a riparian wetland in Ohio, with greater rates observed in late May and early July. We recorded our greatest ambient denitrification and denitrification enzyme activity rates in early summer. Our average denitrification rate in 2004 was 2.88 ± 2.05 mg N m^{-2} h^{-1}, which is similar to the reported denitrification range of <0.28 to 3.64 mg N m^{-2} h^{-1} for eight riparian wetlands in New Jersey (Seitzinger, 1994).

Assessment of Marsh Complex Functioning

Management of the marsh complex by the U.S. Fish and Wildlife Service for wildlife benefit proved effective in removing significant amounts of transported sediments and nutrients from Halfway Creek, a tributary of the Upper Mississippi River. Because the restored marsh had an inlet but no outlet, NH_4^+–N and SRP could not be exported as in the natural marsh, and sediment and nutrients that were discharged into the restored marsh were retained. While initial emphasis was placed on assessing the retention capabilities of the restored marsh, the natural marsh proved more effective at removing sediment and nutrients. The natural marsh removed 60% of the sediment load and >10% of the P load but exported NH_4^+–N and SRP during high-discharge events. The restored marsh removed <10% of the creek sediment load and only 5% of the P load. Because the primary functions of the restored marsh were to increase habitat for migrating waterfowl and to capture sediment, the stop-log control structure allowed the flow to be diverted into the restored marsh only during high-discharge events. The volume of water entering the restored marsh amounted to <2% of the total discharge of Halfway Creek, so the majority of the retention occurred in the natural marsh.

Soil excavation was the most efficient way to remove soil and nutrients from the marsh complex. If we estimate that in 1 yr 600 Mg of sediment was deposited into the restored marsh (given the high-discharge conditions of 2004), then soil excavation removed 6 yr of sediment deposition (3600 Mg of sediment) and 3 yr of P (2.7 Mg of P) and N (5.6 Mg of N). This removal is especially important for P because mechanical removal is the only mechanism for permanent removal. Phosphorus burial provides long-term storage of P; however, the absorption capacity of the soil is finite (Reddy et al., 1999). By the physical removal of soil,

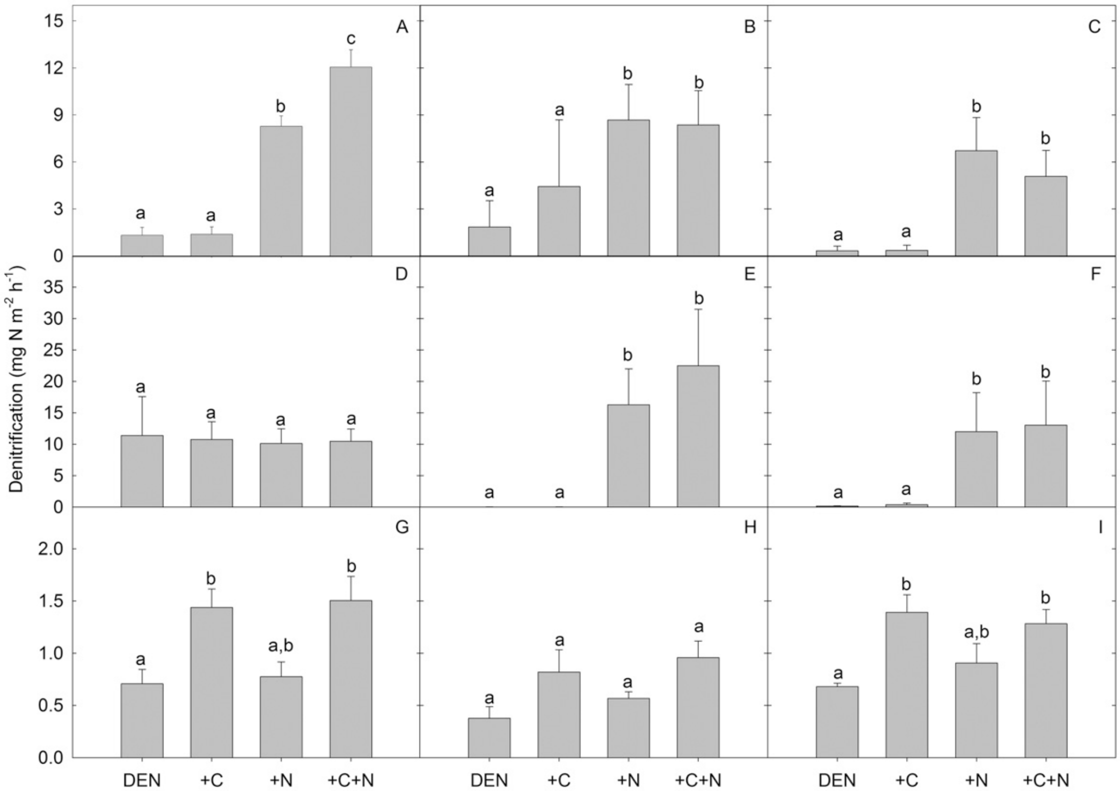

Fig. 6. Average (±1 SE) rates of denitrification (DEN) resulting from the addition of C (+C), N (+N), or both C and N (+C+N) to sediment or soil from (A) Cell A, (B) Cell B, and (C) Cell C of the restored marsh, (D) the upper natural marsh, (E) the lower natural marsh, (F) Halfway Creek at County Highway Z, (G) the diversion channel into the restored marsh, (H) Sand Lake Coulee Creek, (I) and Halfway Creek at County Highway ZN. Different letters indicate significant differences among treatments.

the marsh complex has the capacity to retain more P. Nitrogen can also be permanently removed through denitrification, and the restored marsh has the potential to remove 8.6 Mg N yr^{-1}.

Management Implications

This study indicates that, in combination with natural wetland processes, management of the Halfway Creek Marsh Complex for wildlife benefit significantly reduced downstream movement of sediments and particle-associated nutrients, protecting the Upper Mississippi River from further nutrient and sediment loads. Hydrologic management of the marsh complex was a key factor in its retention capability. The restored marsh was an ideal place for high nutrient retention, but sediment and nutrients could only be introduced into Cell A during storm events. The majority of retention occurred in the natural marsh, which was more hydrologically connected to Halfway Creek than the restored marsh. During base flow, however, the majority of the water in Halfway Creek was confined to the channel and the natural marsh received little water, limiting sediment and nutrient retention.

Ultimately, greater sediment and nutrient retention would occur in the Halfway Creek Marsh Complex if more water was discharged into Cell A of the restored marsh throughout the year and if more water was discharged into the natural marsh during base flow. To effectively remove more sediment and nutrients, the inlet should be placed lower to allow base flow into the restored marsh. Once in the restored marsh, the captured material can be either temporarily stored through soil burial and biotic uptake or permanently removed through soil excavation and denitrification. The excavated soil, which is rich in nutrients, should be tested for pollutants, and if the soil quality is acceptable, it can be redeposited in the uplands where it originated. Ideally, excavation would be done in a way that does not alter the denitrifying capability of the wetland or interfere with water flow through the wetland.

While the restoration of riparian wetlands provides improved water quality and optimal wildlife habitat, there are some potential negative consequences. Increased denitrification leads to elevated emissions of the greenhouse gas N_2O, especially in wetlands that have significant inputs of NO_3^-–N (Verhoeven et al., 2006). Restored wetlands that were once agricultural land may actually be a source of nutrients (Ardon et al., 2010). In the Halfway Creek Marsh Complex, SRP and NH_4^+–N were released to Lake Onalaska during high-discharge events. The released nutrients are readily available for uptake by algae and metaphyton, potentially leading to algal blooms. In the backwater areas of the Upper Mississippi River, high nutrient concentrations have been associated with an increase in metaphyton biomass and hypoxia (Houser and Richardson, 2010). Thick layers of metaphyton limit light penetration and gas diffusion into the water column. The decrease in photosynthesis coupled with the increase in microbial respiration of organic matter result in less available O_2 for aquatic species.

These site-scale drawbacks are small compared with the larger ecological benefits obtained from riparian wetland restoration in the Mississippi River basin. While some bioavailable nutrients are released into the river, overall more nutrients and sediment are retained. Nutrient retention is important because it slows the eutrophication of the Mississippi River. Because the river NO_3^-–N concentration influences the size of the hypoxic zone in the Gulf of Mexico (Turner et al., 2008) and P loading has been linked with Gulf eutrophication (Sylvan et al., 2006), wetland nutrient retention in the tributaries has the potential to reduce the size of the Gulf hypoxic zone. The large retention of sediment is important to slow the degradation of backwater habitats, due in large part to filling. Backwaters in Pool 8 that are connected to tributaries have experienced a 20-fold increase in sedimentation compared with the rest of the pool (Belby, 2009). Large concentrations of suspended material limit light penetration in the water column, causing decreased production of submersed macrophytes, a key component of backwater habitats (Moore et al., 2010). Restored wetlands can be designed to allow periodic excavation of deposited sediment, which could be redeposited onto the upland landscape and provide a beneficial use of the nutrient-rich sediments. Also, wetlands designed to provide access for sediment excavation could allow sediment-associated contaminates to be removed if a catastrophic release of chemicals occurs in the watershed above the wetland.

This study provides a valuable template for consideration in future sediment and nutrient capture projects now planned along the Upper Mississippi River corridor. Conservation practices such as riparian buffers, wetland restoration, and wetland wildlife habitat management are becoming more common on agricultural lands, and it has been recently recognized that a greater understanding of the potential for these practices to reduce the nutrient inputs that drive the Gulf of Mexico hypoxia issue are needed (Brinson and Eckles, 2011; Fennessy and Craft, 2011). Halfway Creek Marsh Complex was restored primarily to provide wildlife habitat and trap sediment, but after restoration, nutrients were also retained. Thus, as more wetlands are restored or created in the Upper Mississippi River basin regardless of the restoration objectives, additional benefits will potentially occur. Also, existing wetlands need to be effectively managed to optimize sediment and nutrient retention. The combination of restoration projects and better management of existing wetlands may well lead to a more heterogeneous landscape where water quality is improved and biodiversity is increased (Moreno-Mateos and Comin, 2010).

As land-use changes associated with intensified corn-based biofuel and food production occur in the Upper Mississippi River basin, sediment and nutrient fluxes are expected to increase (Donner and Kucharik, 2008). To mitigate increased agricultural production without further compromising the ecological condition of the Mississippi River and the Gulf of Mexico, restoration and creation of more riparian wetlands would appear to be beneficial. The Halfway Creek Marsh Complex management strategy is an example of a sustainable restoration alternative to the expected negative water quality impacts to the river and Gulf ecosystems. Also, economic valuation of ecosystem services of wetlands has helped develop a water quality credit and trading market where the potential value of a restored wetland along the Mississippi River is US\$1035 ha^{-1} (Jenkins et al., 2010). Comprehensive studies like the Halfway Creek Marsh Complex project are rare and will significantly contribute to a better understanding of wetland management for use in water quality and trading markets.

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