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NITROGEN DYNAMICS IN SEDIMENT DURING WATER LEVEL MANIPULATION ON THE UPPER MISSISSIPPI RIVER

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ABSTRACT

Nitrogen (N) has been linked to increasing eutrophication in the Gulf of Mexico and as a result there is increased interest in managing and improving water quality in the Mississippi River system. Water level reductions, or 'drawdowns', are being used more frequently in large river impoundments to improve vegetation growth and sediment compaction. We selected two areas of the Upper Mississippi River system (Navigation Pool 8 and Swan Lake) to examine the effects of water level drawdown on N dynamics. Navigation Pool 8 experienced summer drawdowns in 2001 and 2002. Certain areas of Swan Lake have been drawn down annually since the early 1970s where as other areas have remained inundated. In the 2002 Pool 8 study we determined the effects of sediment drying and rewetting resulting from water level drawdown on (1) patterns of sediment nitrification and denitrification and (2) concentrations of sediment and surface water total N (TN), nitrate, and ammonium (NH_4^+). In 2001, we only examined sediment NH_4^+ and TN. In the Swan Lake study, we determined the long-term effects of water level drawdowns on concentrations of sediment NH_4^+ and TN in sediments that dried annually and those that remained inundated. Sediment NH_4^+ decreased significantly in the Pool 8 studies during periods of desiccation, although there were no consistent trends in nitrification and denitrification or a reduction in total sediment N. Ammonium in sediments that have dried annually in Swan Lake appeared lower but was not significantly different from sediments that remain wet. The reduction in sediment NH_4^+ in parts of Pool 8 was likely a result of increased plant growth and N assimilation, which is then redeposited back to the sediment surface upon plant senescence. Similarly, the Swan Lake study suggested that drawdowns do not result in long term reduction in sediment N. Water level drawdowns may actually reduce water retention time and river-floodplain connectivity, while promoting significant accumulation of organic N. These results indicate that water level drawdowns are probably not an effective means of removing N from the Upper Mississippi River system. Copyright © 2006 John Wiley & Sons, Ltd.

KEY WORDS: drawdown; Upper Mississippi River; sediment nitrogen; denitrification; nitrification; nitrogen cycling

INTRODUCTION

The Mississippi River supplies municipal drinking water and supports diverse recreational, agricultural and industrial uses. In addition, the Mississippi River is the primary source of biologically available nutrients transported to the northern region of the Gulf of Mexico (Goolsby and Battaglin, 2001). These nutrients, primarily nitrogen (N), have been linked to increasing eutrophication in the Gulf of Mexico (Burkart and James, 1999) and the development of a seasonal hypoxic zone (Rabalais *et al.*, 2001). As a result of these issues, there is increased interest in managing and improving water quality in the Mississippi River system.

Water level manipulations, or 'drawdowns', are a common river management tool used to stimulate macrophyte production to increase habitat for ducks and other wildlife, but may also prove useful in reducing N loads in the Mississippi River. Water level reduction exposes normally water-saturated sediment to the atmosphere, oxidizing the usually anaerobic sediments. Desired responses include sediment compaction, increased water clarity, seed germination, increased plant growth and ultimately establishment of rooted macrophyte beds.

Rooted plants can contribute significantly to N cycling (Caffrey and Kemp, 1992; Fischer and Claflin, 1995; Eriksson and Weisner, 1999) by oxygenating sediments and increasing the interweaving of oxic and anoxic

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sediment microzones (Kleeberg and Heidenreich, 2004), potentially promoting coupled nitrification and denitrification. Plants also increase the redox potential in sediments (Havens, 1997; Wigand *et al.*, 1997; Sundby *et al.*, 2003) and provide the surrounding sediments with a wide range of labile and refractory organic compounds critical for microbial respiration.

Water level manipulation can also affect the microbial communities that govern N cycling (for a detailed description see Baldwin and Mitchell, 2000). The activity of bacteria responsible for N cycle processes is linked to oxygen and carbon (C) availability, and temperature fluctuations (Terry and Nelson, 1975; Nowicki *et al.*, 1999; Kadlec and Reddy, 2001) that are influenced by the presence of overlying water and duration of inundation. Saturated sediments are often anaerobic and devoid of nitrate (NO_3^-) because of reduced nitrification (aerobic oxidation of ammonium [NH_4^+] to NO_3^-). When sediment is exposed to the atmosphere, the increased availability of oxygen may stimulate nitrification while the newly produced NO_3^- may stimulate denitrification in anaerobic microsites (Mitchell and Baldwin, 1999). This mechanism (i.e. coupled nitrification-denitrification) has been observed in the Upper Mississippi River (UMR; Richardson *et al.*, 2004), in estuaries (Kemp *et al.*, 1990; Rysgaard *et al.*, 1993; Nowicki *et al.*, 1999; Risgaard-Peterson, 2003), lakes (Rysgaard *et al.*, 1993), and cropland soils (Minzoni *et al.*, 1988). It follows that aeration of anaerobic sediments through drying should promote more N loss from coupled nitrification-denitrification than during normal pool management.

Denitrification rate is also likely to increase when nitrate-rich sediments are rewetted and return to an anaerobic state. Sediment drying and rewetting is commonly reported to stimulate denitrification and nitrate loss from wetlands (Reddy *et al.*, 1989; Caffrey and Kemp, 1992), wet forests and grasslands (Groffman and Tiedje, 1988; Fierer and Schimel, 2002), croplands (Reddy and Patrick, 1984), lakes (Christensen and Sorensen, 1986; Qiu and McComb, 1996), and the tropical floodplains (Kern *et al.*, 1996; Kreibich and Kern, 2003). However, the effect of a riverine water level fluctuation on N-cycling processes (i.e. nitrification and denitrification) has not been assessed on temperate large river impoundments.

The lock and dam system was recently (2001 and 2002) used to drawdown water levels in UMR Pool 8 (near La Crosse, Wisconsin) to simulate a more natural summer low-water flow regime and promote macrophyte recolonization and growth. Because drawdowns are being used more frequently to manage large rivers and impoundments to improve vegetation and reduce downstream transport of sediments, evaluation of such a manipulation on the UMR for the affect on nitrogen was timely. Goals of this research were to determine the effects of sediment drying and rewetting resulting from water level drawdown on (1) patterns of sediment nitrification and denitrification; (2) concentrations of sediment and surface water total N (TN), NO_3^-, and NH_4^+; and (3) the potential long-term effect of annual drawdown on sediment NH_4^+ and TN. The hypotheses tested were that (1) sediment drying would increase nitrification rates (and therefore decrease sediment NH_4^+ while increasing sediment NO_3^-) relative to non-dried conditions, (2) sediment rewetting would increase denitrification and decrease sediment NO_3^- relative to the dry condition, and (3) sediment drying/rewetting would result in a detectable reduction in sediment TN.

MATERIALS AND METHODS

Upper Mississippi river

The UMR (Figure 1A), as defined here, is the segment of the Mississippi River north of Cairo, Illinois. The UMR is part of a segment of the Mississippi River that consists of a series of navigation pools delimited by 26 locks and dams (north of St. Louis, Missouri). Pools contain four main habitat types: a main channel, side channels, an impounded area in the lower portion of each pool, and many interconnected backwater areas (Strauss *et al.*, 2004). Each aquatic habitat has unique geomorphology and hydrology resulting in different sediment characteristics and vegetation distribution, and also explains a large portion of variation in N-cycling processes during normal pool management (Richardson *et al.*, 2004; Strauss *et al.*, 2004).

Backwaters and impounded habitats in the UMR are typically higher in sediment organic C, TN, and NH_4^+, and lower in surface water NO_3^- compared to main channel and side channel areas (Strauss *et al.*, 2004). Sediments in backwater and impounded areas of the UMR are typically anoxic, highly organic, relatively low in nitrate (NO_3^-, mean $\pm$ standard error $= 0.10 \pm 0.01$ mg N/L) and high in ammonium (exchangeable NH_4^+: 5.5 ± 0.2 mg N/L sediment; pore water NH_4^+: 2.6 ± 0.1 mg N/L).

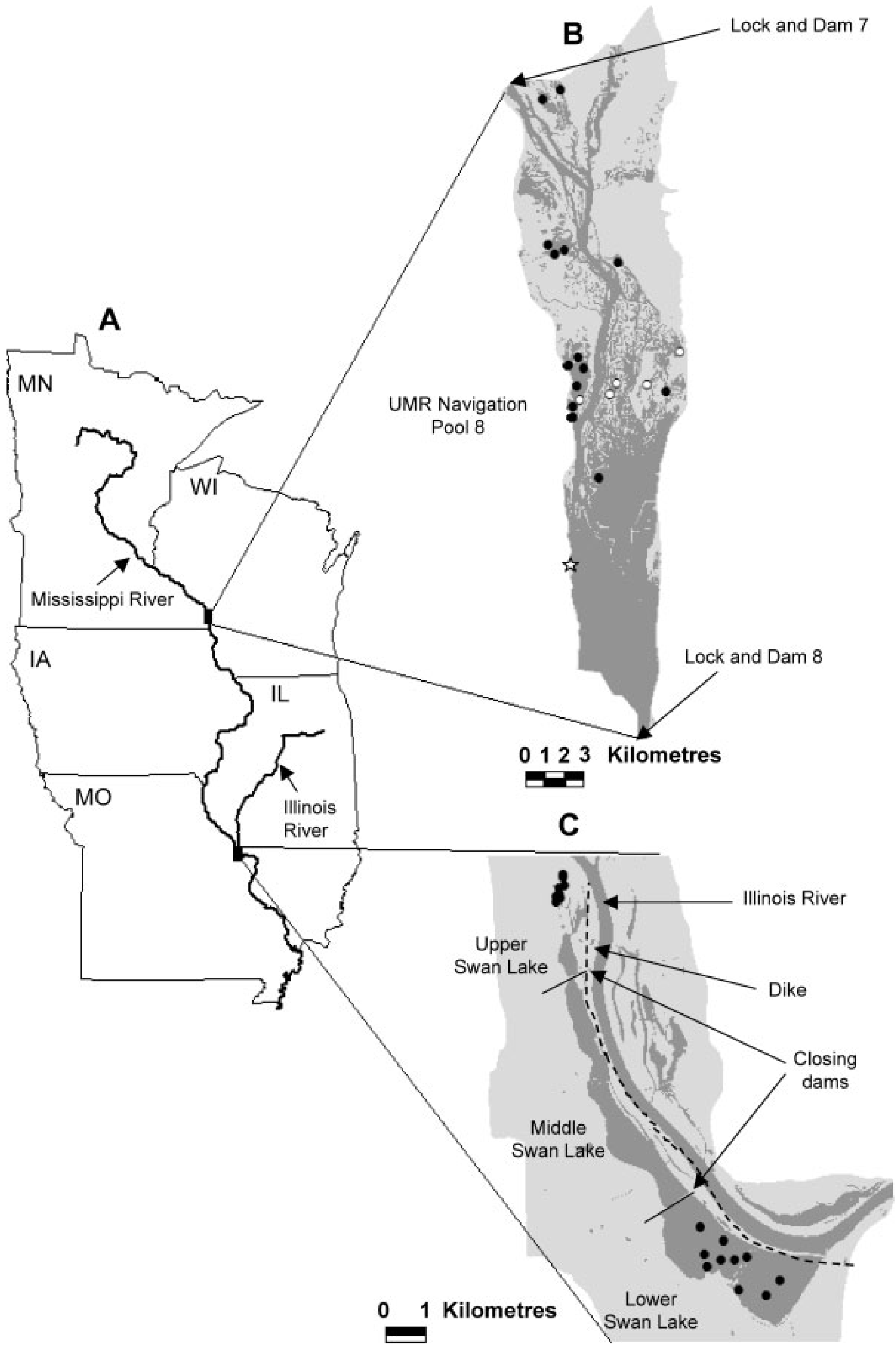

Figure 1. (A) Study locations within the Upper Mississippi River system. (B) Mississippi River Navigation Pool 8. Circles are sites sampled in 2001: filled circles are sites that remained wet and empty circles are sites that dried during the drawdown. The star in the southern end of Pool 8 is the study location for the 2002 drawdown sites. (C) Swan Lake, adjacent to the Illinois River. The 10 points in upper Swan Lake have dried annually for 30 years, and the 10 points in lower Swan Lake have never dried

The UMR dam system can facilitate drawdowns in the navigation pools by controlling water levels and flow. In summers 2001 and 2002, the water level of Pool 8 was reduced slowly by 0.46 m through modification of discharge at Lock and Dam 8 (Figure 1B). After the water depth in the pool reached the drawdown target level, it was maintained at that level for most of the growing season after which the water level was brought back to normal depth. The impounded habitat, a shallow area created in the downstream end of each pool by the lock and dam system, experienced the greatest change in water level during the drawdowns.

Swan Lake is a backwater lake parallel to the Illinois River, Illinois, 7 km upstream of its confluence with the Mississippi River (Figure 1C). The lake is isolated from the Illinois River during normal flow by a dike constructed along the eastern side of the lake. The lake itself is divided by closing dams into three sections: upper, middle, and lower. Upper Swan Lake has been drawn down annually since the early 1970s, middle Swan Lake has been drawn-down annually since 1999, and lower Swan Lake has never been drawn down. The drawdowns in Swan Lake are designed to stimulate food production and provide habitat for migrating waterfowl. During the Swan Lake drawdowns, surface water is removed almost entirely from the upper basin, although sediments do remain fairly saturated. The upper basin exhibits dense plant growth throughout the previously submersed areas during the drawdowns.

To address the effect of water level manipulation on N dynamics in the UMR we examined data from three studies: (1) in 2001, we sampled 18 backwater sites in Pool 8, five of which dried during the drawdown and were then re-inundated; (2) in 2002, we measured sediment N dynamics in paired sites (shoreline and inundated depth) in the impounded habitat in Pool 8; and (3) in 2004, we sampled sediments at Swan Lake, to determine the long-term effect of annual drawdowns on sediment N.

Site descriptions

In 2001, we sampled 18 backwater sites throughout Pool 8 once each in spring, summer, and fall (Figure 1B). The water level in Pool 8 was reduced on 30 June and the drawdown persisted until 15 September for a total of 71 d. We completed spring sampling in mid-May, summer sampling in late July, and fall sampling in early October. Sediment at 5 of these 18 sites was exposed 20 d before summer sampling (because of drawdown conditions) and rewetted 20 d before fall sampling. Sediment type at these backwater locations consisted primarily of silt and clay. These sites usually contained less than 35% macrophyte cover consisting of *Saggitaria latifolia*, *Ceratophylum demersum*, *Vallisneria americana*, and *Nymphaea odorata*. During the drawdown, macrophyte coverage, primarily *N. odorata*, increased to 75–100%.

In 2002, we selected a study site in the shallow impounded area of Navigation Pool 8 (Figure 1B). In the 123 d study we sampled 10 paired sites extending into the river from the land-water interface (shore). Each pair of sites was 10 m apart with two sites (1 = shallow/impacted and 2 = deeper/reference). All 20 sites were inundated prior to the drawdown. The 10 shallow sites were dewatered on 1 July 2002 (study day 25), and the 10 deeper sites remained inundated for the duration of the drawdown (85 d). On 24 September (day 110), the water level increased and the impacted sites were rewetted. We sampled all sites nine times in summer: two times before the drawdown (days 0 and 11), five times during the drawdown (days 26, 39, 53, 72 and 81), and two times after the termination of the drawdown (days 116 and 123) when all 20 sites were again inundated. The sediment type in this impounded area was a mixture of silt and clay and was typically unconsolidated with less than 35% coverage of aquatic macrophytes, primarily *Saggitaria latifolia.* Once surface water was removed and the duration of exposure increased, sediments consolidated (i.e. one could walk on top of formerly soft sediments) and macrophyte coverage increased to near 100%. In addition to *S. latifolia*, the macrophyte community composition increased in species richness to include among others *Nymphaea odorata, Nelumbo lutea,* and *Leersia oryzoides*.

In 2004, we collected sediment samples from 10 sites in each of the upper and lower basins of Swan Lake to determine sediment TN and exchangeable NH_4^+ concentrations. At the time of collection the drawdown in the upper basin had terminated and all the sites were inundated.

Sample collection and analysis

We collected intact sediment cores (7.62 cm diameter × 5 cm sediment depth) and overlying surface water at each site. Samples were transported to the laboratory on ice and stored at 4°C until analysed for surface and sediment porewater N concentrations and for denitrification and nitrification rates. Physical/chemical characteristics of the sediment and surface water also were monitored *in situ* at all sites throughout the study. Variables measured included surface water pH, temperature, conductivity and dissolved oxygen in the surface water (YSI 600XL multi-parameter probe) and sediment pH and temperature (Beckman Φ11 pH meter).

We measured sediment TN and total C (TC) using an Elementar VarioMax CN analyser and evaluated total organic carbon (TOC) using the following equation: (TC value of a previously dried sample [48 h, 105°C]—TC value of a previously ashed sample [6 h, 500°C]). We measured sediment moisture content and bulk density following Håkanson and Jansson (1983). Sediment pore water was centrifuged (1921× g, 4°C, 12 min) from a sediment subsample (10–25 cm^3) of one core from each site using the KCl extraction method (Caffrey and Kemp, 1992). Surface water and sediment pore water NO_3^- and NH_4^+ concentrations (KCl extractions) were determined using the automated cadmium reduction and phenate methods, respectively (American Public Health Association, 1998,). All water chemistries were analysed using a Bran + Luebbe continuous flow autoanalyser.

We determined denitrification in sediment slurries using a modification of the acetylene block technique (Sorensen, 1978; described by Richardson *et al.*, 2004). Slurries for denitrification estimates contained 20 mL sediment, 20 mL surface water, and 5 mL chloramphenicol solution (final concentration = 100-mg/L), and were incubated anaerobically. In addition, we determined denitrification enzyme activity (DEA) using the same technique as for denitrification except the slurries were incubated with 5 mL of DEA solution (final concentration = 100-mg/L chloramphenicol, 12-mg/L glucose-C, and 14-mg/L potassium nitrate as N). Denitrification enzyme activity is a measurement of denitrification potential because of unlimited substrate (N and C) availability, whereas unamended denitrification only quantifies the rate with ambient substrate availability. Acetylene was added to the headspace of sealed incubation jars and jars were incubated at ambient temperatures in the dark on shaker tables. Nitrous oxide concentrations from denitrification and DEA analyses were determined on a Hewlett Packard 5890 Gas Chromatograph equipped with a ^{63}Ni electron capture device. Denitrification and DEA rates were calculated following the equations in Groffman *et al.* (1999).

Nitrification rates were determined using a modification of the nitrapyrin method (as described in Strauss *et al.*, 2004). Two slurries for each sediment sample were incubated aerobically for 3 days in flasks on shaker tables. One flask contained 25 mL sediment, 81 mL surface water, and 20 μL nitrapyrin/dimethyl-sulfoxide solution (final nitrapyrin concentration = 10-mg/L) which inhibits NH_4^+ oxidation. The other flask contained 25 mL sediment, 81 mL surface water, and 20 μL dimethyl-sulfoxide. Ammonium concentrations from subsamples of the slurries extracted with KCl were measured at the start and end of the incubations, and the gross nitrification rate was determined from the difference in NH_4^+-N between the two flasks (Strauss and Lamberti, 2000).

Denitrification in these sediments is limited by NO_3^- availability and tightly coupled with nitrification (Richardson *et al.*, 2004). Because acetylene-based denitrification assays also inhibit nitrification (Hynes and Knowles, 1978), they do not accurately reflect ambient denitrification rates. Therefore, we calculated N loss by way of denitrification based on an estimated denitrification rate (EDR) for each sampling interval (i.e. the time between each sampling event), similar to that described in Richardson *et al.* (2004). In that study, we calculated EDR as the lower value between (a) DEA rate and (b) unamended denitrification plus nitrification rate. This calculation was based on three assumptions: (1) denitrification was not limited by carbon, (2) oxygen in the sediments was distributed heterogeneously to facilitate concurrent nitrification and denitrification, and (3) denitrification was limited by NO_3^- availability. These assumptions were shown valid for the UMR system (Richardson *et al.*, 2004). However, the validity of the third assumption is in question for our 2002 study because we detected significant changes in the concentrations of NO_3^- in sediments during the dry period. To account for changing sediment NO_3^-, we subtracted the observed change in NO_3^- concentration in our N loss equation (below). In addition, the unamended denitrification rate plus nitrification rate was always lower than the DEA rate in 2002, so DEA was not considered in our calculation of N loss. Therefore, N loss by way of denitrification (L) for each interval between sampling events in 2002 was calculated with the equation:

$$L_{ij} = \{[(D_i + D_j)/2] + [(N_i + N_j)/2]\} * T_{ij} - (O_j + O_i)$$

where, D = un-amended denitrification rate (mg N/g/d), N = nitrification rate (mg N/g/d), O = sediment nitrate concentration (mg N/g), T_{ij} = time of interval between sampling events i and j (days), $i,j, \ldots q$ = Sampling day (first day of experiment = day 1). Total N loss during the 2002 drawdown from denitrification was calculated as the sum of the L_{ij} for each sampling interval during the drawdown ($L_{ij} + L_{jk}$, etc.). The N loss calculated is reported on a per g dry weight basis instead of on an aerial basis to minimize the error associated with sediment compaction and expansion.

We determined plant biomass and plant TN once during the 2002 study. On day 72 we collected the aboveground vegetation at 5 of the 10 impacted sites using a 533-cm^2 quadrat. To determine biomass, entire vegetation samples were dried (48 h, 105°C) and weighed. Total C and N contents were determined from subsamples of the dried plant material that were ground to a coarse powder using a Wiley Mill with a 420-um screen and analysed with an Elementar varioMax CN analyser.

Statistical analysis

We assessed the effects of the 2001 drawdown using a repeated measures analysis of variance (ANOVA) model comparing sediment and plant metrics in dewatered to continuously inundated sites over the three sampling events. For the 2002 Pool 8 study, we conducted tests of hypotheses using a repeated measures mixed model ANOVA with a Tukey's *post hoc* test. The data set contained two site types (impacted and reference), and the sample dates were divided into three time categories (before, during and after impact). We first examined the data set for spatial and temporal autocorrelation. We did not detect temporal correlation, and spatial correlation was detected only in sediment TN. We then adjusted our model for sediment TN by grouping the observations with the same covariance parameters (impacted and reference). If we found a significant difference between dry and wet sites and a trend within a time category, we then looked for specific differences between collection dates two (just before the drawdown), seven (the last dry day samples were collected), and eight (the first day samples were collected after the impacted sites were rewetted). With the exception of sediment TN, all variables had a non-normal distribution and were log transformed ($\log_{10}$) to gain normality. The effect of Swan Lake drawdowns (comparing upper and lower sections) was determined using *t*-tests. All statistical analyses were completed with software by the SAS Institute, Inc., Cary, NC, Version 8.

RESULTS

Pool 8-2001

There was approximately a 50% reduction in sediment exchangeable NH_4^+ and TN concentrations at the sites that dried (impacted), and this decrease persisted after the sites were rewetted (Figures 2A–2B). There was little change in sediment NH_4^+ and an increase in TN at the reference sites. Despite the seemingly large reductions in sediment N at the impacted sites, these concentrations were not significantly different ($p > 0.1$).

Pool 8-2002

Concentrations of N in surface water were lower at the impacted sites than the reference sites before and after the drawdown (Figure 3). Sediment moisture content decreased significantly at the impacted sites during the drawdown ($p = 0.0058$, data not shown).

Before the drawdown, concentrations of sediment inorganic N were not different among the reference and impacted sites. Sediment NO_3^- concentrations were similar at impacted and reference sites before the drawdown, and increased at the impacted sites as sediments dried to levels significantly greater than the reference sites ($p < 0.0001$; Figure 4A). Rain on sample day 53 may have briefly created anaerobic conditions and a subsequent loss of NO_3^- from denitrification, or caused a temporary dilution of the sediment NO_3^- concentrations. During the rewetting phase, NO_3^- concentrations declined in the impacted sites to levels similar to those in the reference sites ($p = 0.0822$). Exchangeable NH_4^+ concentrations were highest at the beginning of the study at all sites and generally declined throughout the study, with concentrations significantly lower at the impacted sites during the drying phase (day 39; $p = 0.0001$; Figure 4B). The decreased exchangeable NH_4^+ concentrations persisted after rewetting (Day 116; $p < 0.0001$; Figure 4B). Sediment TN and TOC levels were initially higher in the impacted sediments and this relation did not change as a result of the drawdown (Figure 4C; Table I), and exhibited a net increase throughout the study at both locations.

Denitrification rates were initially significantly lower at the impacted sites before the drawdown (days 0, 11, and 26; $p < 0.05$), increasing significantly during the drawdown ($p < 0.05$; days 53, 72, and 81; Figure 5A and

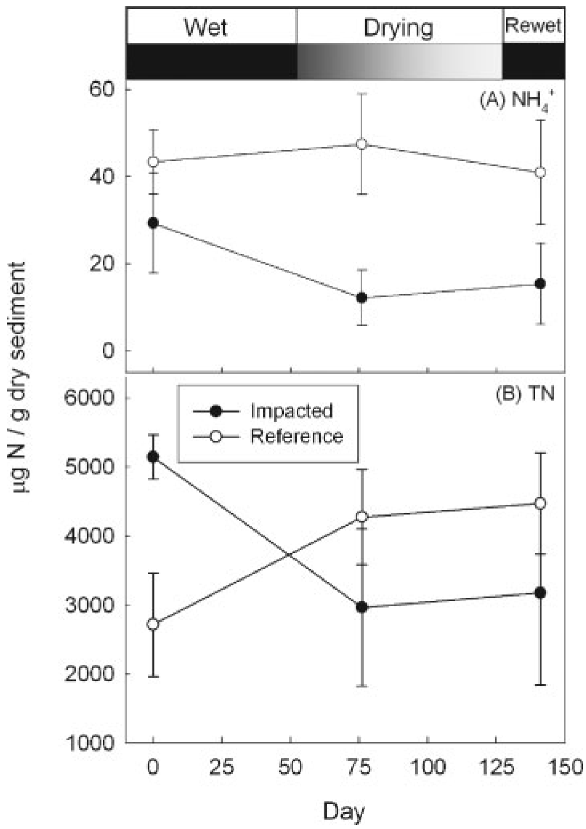

Figure 2. Mean sediment nitrogen (N) concentrations (±standard error) from Pool 8 of the Upper Mississippi River in the 2001 study: (A) exchangeable ammonium (NH_4^+), (B) total N (TN). Impacted = sediments dried during the drawdown ($n = 5$); Reference = sediment was not in the area that dried during the drawdown ($n = 13$)

were positively correlated with sediment NO_3^- concentrations ($r^2 = 0.688$; $p < 0.0001$). In general DEA rates declined at both the impacted and reference sites throughout the study although more so at the impacted sites (Figure 5B and were significantly lower at the impacted sites in the latter half of the drawdown ($p < 0.05$ on days 53, 72, and 81). Nitrification rates also declined at all sites throughout the study, although at a lesser rate at the impacted sites, and were slightly higher at impacted sites after rewetting (Figure 5C). While nitrification rates were relatively low, they were sufficient to account for the slight increase in sediment nitrate concentrations observed at the impacted sites.

Examination of an N mass balance indicates a loss at the impacted sites (197 μg/g) and a gain at the reference sites (143 μg/g) in sediment TN after drying; however, both locations exhibited a slight net gain (impacted = 367 μg/g; reference = 419 μg/g) after rewetting (Table I; Figure 6A). Total gross N loss during the drawdown at the impacted sites was only 10% of the total sediment N before the drawdown and approximately 6% of the N loss was from denitrification. In comparison, N loss from denitrification at the reference sites was about 18% of the total sediment N during the drawdown. There was greater gross N loss from EDR at the reference sites than the impacted sites ($p = 0.0003$; Figure 6B). Overall, there was a total gross gain of N at both sites, although less at the impacted sites, however the difference was not significant ($p = 0.28$; Figure 6C). Plants assimilated about 37-mg N/g plant material in 2002, well exceeding the TN we measured in the top 5 cm of sediment. Plant N was not measured at the reference sites.

Swan Lake

Sediments collected from the annually drawn down upper Swan Lake contained significantly higher TN when compared to those collected from the lower basin ($p = 0.0021$; Figure 7A). Conversely, differences in sediment

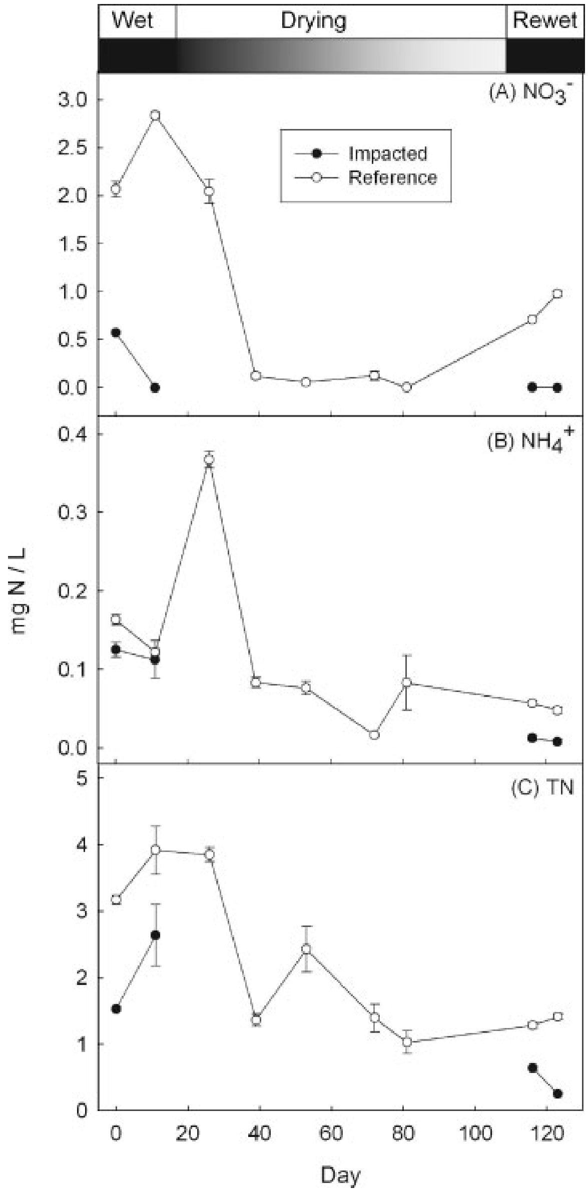

Figure 3. Mean surface water nitrogen (N) concentrations (±standard error) from Pool 8 of the Upper Mississippi River in the 2002 study: (A) nitrate (NO_3^-), (B) ammonium (NH_4^+), (C) total N (TN). Shading bar at the top designates the start and finish of the drawdown. Impacted = sediments dried during the drawdown; Reference = sediment was not in the area that dried during the drawdown

exchangeable NH_4^+ concentrations were moderately different between the two basins ($p = 0.11$), with higher concentrations in lower Swan Lake (Figure 7B).

DISCUSSION

N cycling dynamics

We expected the exposure and rewetting of sediments to result in reduced sediment N because of increased oxygen penetration to sediments, nitrification, and subsequent denitrification. Other studies on sediment drying and rewetting have commonly found some or all of these results (e.g. Minzoni *et al.*, 1988; Reddy *et al.*, 1989; De Groot

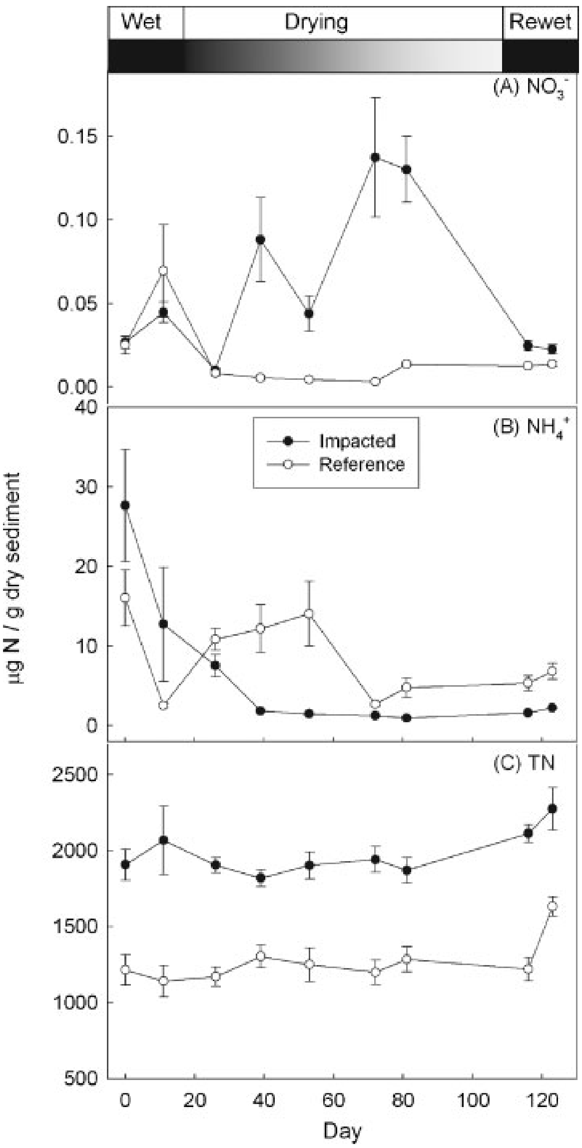

Figure 4. Mean sediment nitrogen (N) concentrations (±standard error) from Pool 8 of the Upper Mississippi River in the 2002 study: (A) nitrate (NO_3^-), (B) exchangeable ammonium (NH_4^+), and (C) total N (TN). Shading bar at the top designates the start and finish of the drawdown. Impacted = sediments dried during the drawdown; Reference = sediment was not in the area that dried during the drawdown

and Van Wijck, 1993; Baldwin and Mitchell, 2000; James *et al.*, 2004). We saw a distinct reduction of sediment NH_4^+ at all study locations with sediment drying, coincidental with increases in plant biomass and with slight increases in sediment NO_3^- at the impacted sites at Pool 8 in 2002. Although the exposure of sediment to air creates better conditions for NO_3^- production, it also stimulates plant growth that may reduce levels of available NH_4^+ for nitrification (Kaye and Hart, 1997). Despite the significant reduction in sediment NH_4^+ at the Pool 8 study locations, we did not observe a significant reduction in TN. In fact, in Pool 8 in 2002 and Swan Lake, TN levels exceeded those present before the disturbances.

The primary source of sediment N at the impacted and reference sites is probably organic N from senescing plant material, although there appeared to be less gain of sediment N at the impacted sites. While the impacted sites gain organic N from the senescing plant material, the reference sites are influenced by senescing plant material as well as

Table I. Mean concentrations (mg/g dry sediment, ±standard error) of total N (TN), total organic C (TOC), and C:N (molar) in sediment from Navigation Pool 8 of the Upper Mississippi River in summer 2002

	Impacted		Reference	
	Before	After	Before	After
TN	1.91 (0.10)	2.27 (0.14)	1.21 (0.10)	1.63 (0.06)
TOC	19.15 (1.28)	25.98 (2.22)	11.48 (1.01)	14.23 (0.81)
C:N	10.8	13.3	11.7	10.2

Impacted = sediments dried during the drawdown; Reference = sediment was not in the area that dried during the drawdown.
'Before' samples were collected 15 days ($n = 10$) before the drawdown; 'After' samples were collected 17 days ($n = 10$) after the termination of the drawdown.

transfer of N into the sediments from the water column. This may account for the small difference in gross N gain between the sites.

These findings suggest drawdowns may be an important mechanism for temporarily lowering sediment NH_4^+, especially for sediments that typically contain high levels of NH_4^+ like those in the UMR (Strauss *et al.*, 2004). Reduction in sediment NH_4^+ can result from increased nitrification rate and/or assimilation by plants and microorganisms. James *et al.* (2004) observed a reduction in sediment NH_4^+ in an *in vitro* experiment where UMR sediments were dried and rewetted under controlled conditions, and attributed the loss of NH_4^+ to increased nitrification, which ultimately resulted in an 18% loss of sediment TN. Our laboratory assays did not show increased nitrification at the impacted sites in either year in Pool 8, or a net loss of sediment TN. A temporary loss of NH_4^+ due to plant uptake was assumed. In 2001 and 2002, we observed an increase in plant biomass after sediment drying at all study locations coincident with the loss of sediment NH_4^+. James *et al.* (2004) observed lower plant biomass on partially dewatered (60%) and rewetted substrates, compared to inundated sediments. Their experimental design (isolated sediment cores, 8.5 cm deep × 7.6 cm width), however, eliminated the potential movement of NH_4^+ from lateral and underlying sediments. We consistently observed plant roots extending beyond these depths in sediments throughout Pool 8, suggesting that isolating sediments may have caused the reduced plant growth in their study. Others have found plant uptake can significantly reduce sediment N (Caffrey and Kemp, 1992; James *et al.*, 2001; Clarke, 2002) and soil N (Tufekcioglu *et al.*, 2003). However, plant senescence and subsequent microbial processing will release plant-bound C and N back to the top sediment surface, resulting in a net gain of N (Weisner *et al.*, 1994; Nowicki *et al.*, 1999; Kleeberg and Heidenreich, 2004). Uptake of NH_4^+ through plant assimilation may be significant in our studies, but it appears to be only a temporary mechanism for N loss from sediments in the UMR.

According to Patrick and Wyatt (1964) and Qiu and McComb (1996), increased plant biomass and rooting also creates aerated zones and macropores in anoxic sediments, promoting NH_4^+ oxidation (nitrification). However, we did not observe significantly elevated rates of nitrification in the Pool 8 studies. In 2002, we observed an increase in NO_3^- likely due to nitrification; however the levels of NO_3^- are a very small proportion of the NH_4^+ loss suggesting that nitrification did not play a significant role. Nitrifying bacteria are sensitive to desiccation, and many studies have shown drying to cause a significant reduction in microbial biomass (De Groot and Van Wijck, 1993; Qiu and McComb, 1996; Baldwin and Mitchell, 2000). In addition, Zaman and Chang (2004) observed a significant lag time before nitrifying bacteria began processing sediment NH_4^+ (>30 day of sediment aeration). Nitrifying bacteria are also poor competitors for NH_4^+ and can be out-competed by the increased uptake of NH_4^+ of heterotrophic bacteria and aquatic macrophytes (Kaye and Hart, 1997; Bodelier *et al.*, 1998; Strauss and Lamberti, 2000; Strauss *et al.*, 2002) that result from aerated conditions during a drawdown. We postulate that during drawdown conditions there may be a delayed response before significant nitrification occurs (approximately 4 weeks), caused by a desiccation-induced reduction in the population of nitrifying bacteria. After an initial lag period and the nitrifiers respond to the newly aerated conditions, competition with plants increases, and the nitrifiers may become NH_4^+ limited.

Under normal river management, there is a significant pool of sediment NH_4^+ generated by mineralization (conversion of organic N to NH_4^+). During water saturated conditions nitrification and denitrification are coupled

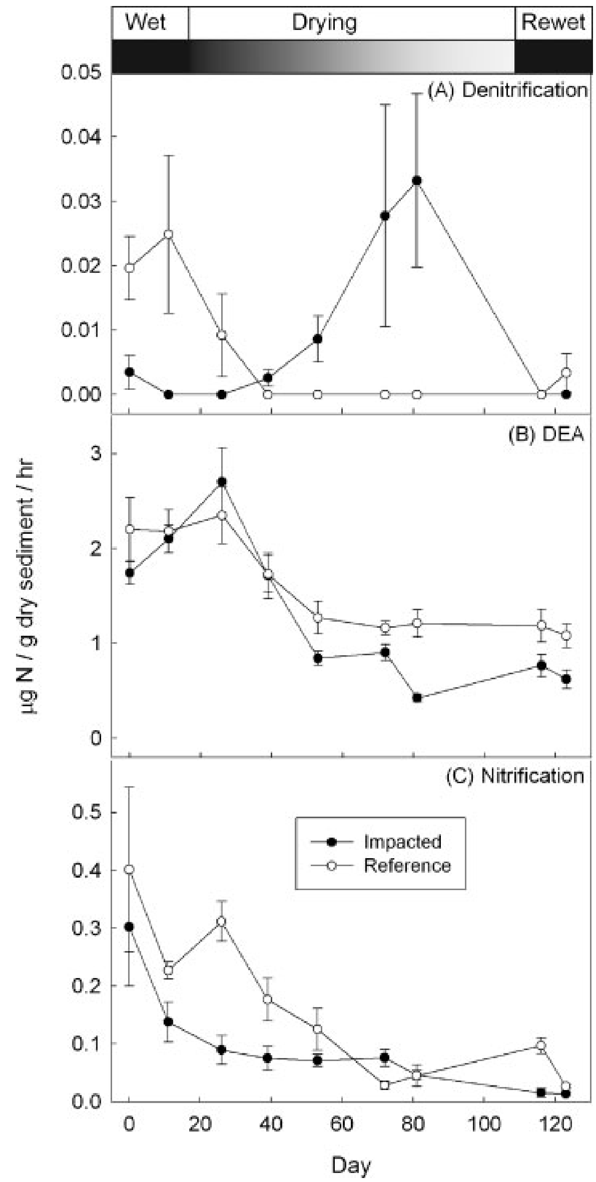

Figure 5. Mean sediment nitrogen (N) processing rates (±standard error) from Pool 8 of the Upper Mississippi River in the 2002 study: (A) denitrification, (B) denitrification enzyme activity (DEA), and (C) nitrification. Shading bar at the top designates the start and finish of the drawdown. Impacted = sediments dried during the drawdown; Reference = sediment was not in the area that dried during the drawdown

(Richardson *et al.*, 2004) which results in low levels of NO_3^- in the sediments (Figure 8A). During drawdown conditions, high plant assimilation, increased nitrification, and possibly a slight reduction in mineralization (Zaman and Chang, 2004) significantly reduces the sediment NH_4^+ pool (Figure 8B). In addition, the oxic conditions reduce denitrification in much of the upper sediment layers, as shown by the elevated NO_3^- in the dried sediments. Our assays showed significantly increasing denitrification rates as sediments dried, probably in response to the available NO_3^-. During incubation, sediments are anaerobic and the denitrification rate is directly related to sediment NO_3^- concentrations. During the drawdown, exposed sediments were aerobic inhibiting denitrification (except in anoxic microsites) while accumulating NO_3^-. When rewetted, plant senescence and decomposition increased the organic N pool. The return to anaerobic conditions stimulated denitrification that quickly depleted the available NO_3^- in the

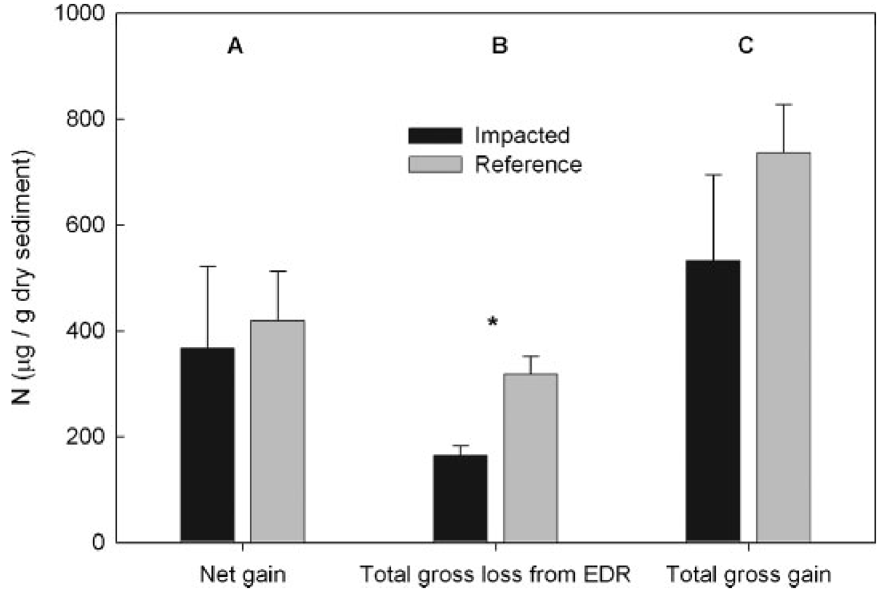

Figure 6. Mean ($n = 10$) nitrogen (N) gain and loss (±standard error) in Pool 8 of the Upper Mississippi River in 2002. (A) Net gain is the overall change in total sediment N throughout the study, (B) total gross N loss from estimated denitrification rates (EDR) throughout the study, and (C) total gross gain in sediment total N throughout the study (net gain + gross loss). Asterisk (*) = $p < 0.05$. Impacted = sediments dried during the drawdown; Reference = sediment was not in the area that dried during the drawdown

sediment (Figure 8C). Had we sampled on or near the day the surface water returned to our impacted sites, we might have observed a temporary increase in denitrification rates.

Our field studies suggest nitrification is not a key player in removal of NH_4^+ from UMR sediments. Another possible explanation for observed losses in NH_4^+ could be ammonia (NH_3) volatilization; however, the sediment pH ranges we observed in the 2001 and 2002 field studies (6.8–7.2) were well below those that would result in

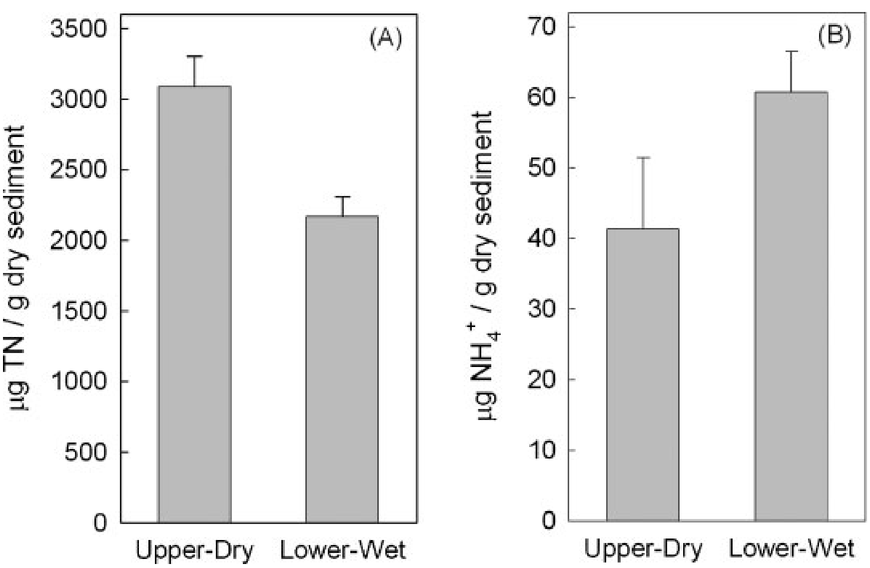

Figure 7. (A) Total nitrogen (TN) and (B) exchangeable ammonium (NH_4^+) in sediment at Swan Lake, Illinois, in 2004. 'Upper' samples are from the upper basin of Swan Lake ($n = 10$) that experiences annual drawdowns; 'Lower' samples are from the lower basin of Swan Lake ($n = 10$) that has never been drawn down. All points are mean ± standard error

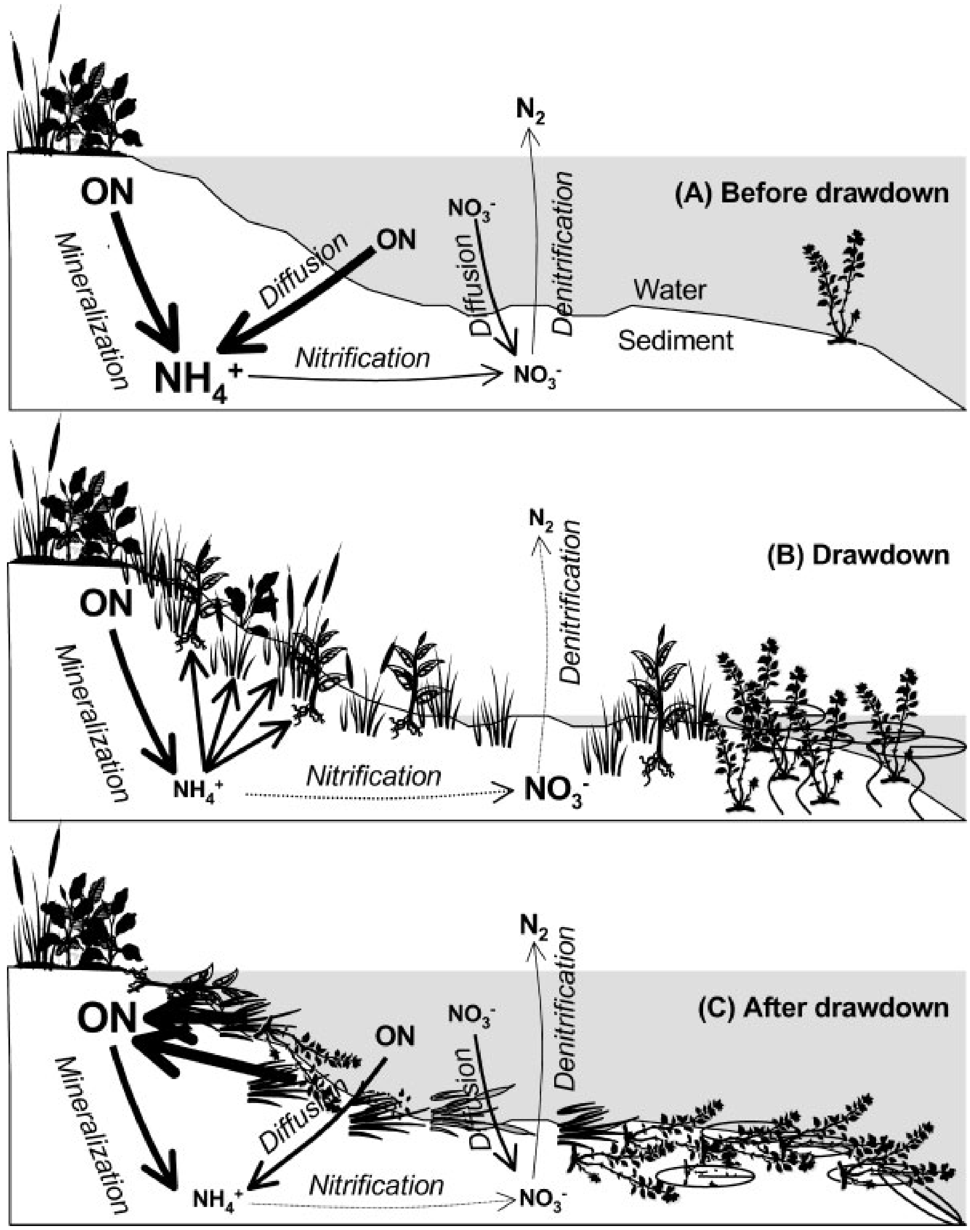

Figure 8. Conceptual model of nitrogen (N) cycling in Upper Mississippi River Pool 8 in 2002: (A) saturated conditions before water level drawdown, (B) during water level drawdown, and (C) after rewetting. Under normal Pool management (A), there is a significant pool of sediment ammonium (NH_4^+), primarily generated from mineralization of organic nitrogen (ON); nitrification and denitrification are coupled resulting in very low levels of sediment nitrate (NO_3^-). During drawdown conditions (B), plant assimilation, initial increases in nitrification, and potentially a slowing of mineralization significantly reduces the sediment NH_4^+ pool, whereas nitrification and denitrification are uncoupled resulting in a build up of sediment NO_3^-. Upon rewetting (C), plant senescence and decomposition increase the organic N pool, but the anaerobic conditions and low NH_4^+ in the sediment continues to inhibit nitrification. Anaerobic conditions also stimulate denitrification and subsequent reduction in sediment NO_3^-. Arrow line thickness and text size correspond to the relative concentrations of N in each pool

significant volatilization (pH $>$ 8.0). Mineralization is probably a main contributor to the high sediment NH_4^+ concentrations during 'normal' saturated conditions. Desiccation could be inhibiting mineralization, yet many studies have shown mineralization is positively related to sediment drying (Reddy and Patrick, 1975; De Groot and Van Wijck, 1993; Venterink *et al.*, 2002; Olfs *et al.*, 2004). Zaman and Chang (2004) suggest, however, that the optimal soil moisture for soil mineralization is wet with good aeration. During drawdowns it is common for sediments to become desiccated and cracked. Although we observed high plant biomass, which may have caused the reduction in sediment NH_4^+, it is possible that some of the loss may have been caused by a slowing of mineralization. It is unclear which NH_4^+ pathway is dominant in the absence of plants and nitrification.

Plant dynamics are likely the key to understanding patterns in sediment TN concentrations. Essentially, plants are working as N ‘pumps’, extracting N from the subsurface and depositing it on the sediment surface (Kleeberg and Heidenreich, 2004). As a result, the increased plant growth and the altered sediment N cycling produced from water level reductions may not cause significant changes in sediment TN in high N systems, such as the UMR.

Management implications

These results suggest that drawdown conditions may alter microbial processes and N cycling but do not significantly reduce TN concentrations in sediments and could actually increase sediment TN. Although there is some potential for N transformation (e.g., N immobilization), this transformation appears to simply recycle N. Drying and rewetting of sediments has been shown to effectively increase sediment N removal in other systems, such as wetlands (Qiu and McComb, 1996), lakes (Scholz *et al.*, 2002), and soils (Fierer and Schimel, 2002), however, repeated drying and submergence over more than 30 years at Swan Lake did not result in any detectable long-term N-reduction. Upper Mississippi River sediments contain N concentrations at least two times higher than observed in other studies, and it is consequently more difficult to detect significant reductions in sediment N.

Nutrient and water quality issues have existed in the Mississippi River basin since the 1950s (Rabalais *et al.*, 1996), and now there may not be any single management technique that will correct local or downstream problems (e.g. excessive algal growth or marine hypoxia). Some studies have recommended biomass harvesting in wetlands to reduce sediment nutrients (Koerselman *et al.*, 1990; Clarke and Baldwin, 2002) and De Groot and Van Wijck (1993) suggest that burning fields after harvesting prevents mineralization of organic N. Presently, N fluxes conservatively through the UMR, with the main channel generally acting as a conduit for N and with little net loss. The sediment of backwater areas are typically anoxic containing high levels of organic material and are optimal environments for sediment N loss (via denitrification). In another study (Richardson *et al.*, 2004), we showed that areas disconnected from the main channel only receive delivery of high NO_3^- surface water during spring floods. During the rest of the year, denitrification in backwaters is typically limited by low sediment NO_3^- (Johnston *et al.*, 2001; Richardson *et al.*, 2004), and we suggest that increasing connectivity between main flow and the more biogeochemically active backwaters may increase N loss. Others have suggested that increasing water retention time (Jossette *et al.*, 1999) and river connectivity to backwater and wetland habitats are ecologically important (Tockner *et al.*, 2000) and can be used as a mechanism for reducing N loads (Mitsch *et al.*, 2001; Hey, 2002). During drawdown conditions, there is even less water flowing over biogeochemically active backwater sediments, reducing river connectivity with areas that hold the greatest capacity to remove N from the system. Denitrification is inhibited in areas affected by drawdowns and sediment drying, further reducing the potential for sediment N removal. In addition, present UMR management (i.e. flow redirection to main channels) results in reduced retention time and rapid export downstream. In effect, annual water level drawdowns may reduce water retention time and river-floodplain connectivity, while promoting significant accumulation of organic N (and possibly mineralization) in backwater and impounded areas. This may result in greater surficial accumulation of sediment N (via translocation from lower sediments) than would under the present river management (no drawdowns).

In our study, we demonstrated that water level drawdowns can effect N transformations; however assimilation by rooted vegetation may be a more significant, but temporary loss of N. Fall senescence will return organic N back to the sediment for mineralization or flux downstream. Most importantly, N loss from sediments is reduced in areas affected by drawdowns because of aerated conditions. Further research is needed to determine the role of plants in the mass balance of N in river ecosystems. Water re-direction into biogeochemically active backwaters will likely prove to be a more effective N removal strategy than water level reductions on the UMR. Given the already high concentrations of N in the UMR, there is a need for management strategies to improve water quality (i.e. those that decrease N loads) before the water reaches the Mississippi River.

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REFERENCES

APHA (American Public Health Association). 1998. *Standard Methods for the Examination of Water and Wastewater*. (20th edn). American Public Health Association: Washington, D.C.

Baldwin DS, Mitchell AM. 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: a synthesis. *Regulated Rivers: Research & Management* **16**: 457–467.

Bodelier PLE, Duyts H, Blom CWPM, Laanbroek HJ. 1998. Interactions between nitrifying and denitrifying bacteria in gnotobiotic microcosms planted with the emergent macrophyte Glyceria maxima. *FEMS Microbiology Ecology* **25**(1): 63–78.

Burkart MR, James DE. 1999. Agricultural-nitrogen contributions to hypoxia in the Gulf of Mexico. *Journal of Environmental Quality* **28**(3): 850–859.

Caffrey JM, Kemp WM. 1992. Influence of the submersed plant, *Potamogeton perfoliatus*, on nitrogen cycling in estuarine sediments. *Limnology and Oceanography* **37**(7): 1483–1495.

Christensen PB, Sorensen J. 1986. Temporal variation of denitrification activity in plant-covered, littoral sediment from Lake Hampen, Denmark. *Applied and Environmental Microbiology* **51**(6): 1174–1179.

Clarke SJ. 2002. Vegetation growth in rivers: influences upon sediment and nutrient dynamics. *Progress in Physical Geography* **26**(2): 159–172.

Clarke E, Baldwin AH. 2002. Responses of wetland plants to ammonia and water level. *Ecological Engineering* **18**(3): 257–264.

De Groot CJ, Van Wijck C. 1993. The impact of desiccation of a freshwater marsh (Garcines Nord, Camargue, France) on sediment-water-vegetation interactions. *Hydrobiologia* **252**: 83–94.

Eriksson PG, Weisner SEB. 1999. An experimental study on effects of submersed macrophytes on nitrification and denitrification in ammonium rich aquatic sediments. *Limnology and Oceanography* **44**(8): 1993–1999.

Fierer N, Schimel JP. 2002. Effects of drying-rewetting frequency on soil carbon and nitrogen transformations. *Soil Biology and Biochemistry* **34**: 777–787.

Fischer JR, Claflin TO. 1995. Declines in aquatic vegetation in Navigation Pool No. 8, Upper Mississippi River between 1975 and 1991. *Regulated Rivers: Research & Management* **11**: 157–165.

Goolsby DA, Battaglin WA. 2001. Long-term changes in concentrations and flux of nitrogen in the Mississippi River Basin, USA. *Hydrological Processes* **15**: 1209–1226.

Groffman PM, Tiedje JM. 1988. Denitrification hysteresis during wetting and drying cycles in soil. *Soil Science Society of America* **52**: 1626–1629.

Groffman PM, Holland EA, Myrold DD, Rogertson GP, Zou X. 1999. Denitrification. In *Standard Soil Methods for Long-Term Ecological Research*, Robertson GP, Coleman DC, Bladsoe CS, Sollins P (eds). Oxford University Press: New York.

Håkanson L, Jansson M. 1983. *Principles of Lake Sedimentology*. Springer-Verlag: New York.

Havens KJ. 1997. The effect of vegetation of soil redox within a seasonally flooded forested system. *Wetlands* **17**(2): 237–242.

Hey DL. 2002. Nitrogen farming: harvesting a different crop. *Restoration Ecology* **10**(1): 1–10.

Hynes RK, Knowles R. 1978. Inhibition by acetylene of ammonia oxidation in *Nitrosomonas europea*. *FEMS Microbiology Newsletter* **4**: 319–321.

James WF, Barko JW, Eakin HL, Helsel DR. 2001. Changes in sediment characteristics following drawdown of Big Muskego Lake, Wisconsin. *Archives fur Hydrobiolgy* **151**(3): 459–474.

James WF, Barko JW, Eakin HL. 2004. Impacts of sediment dewatering and rehydration on sediment nitrogen concentration and macrophyte growth. *Canadian Journal of Fisheries and Aquatic Sciences* **61**: 538–546.

Johnston CA, Bridgham SD, Schubauer-Berigan JP. 2001. Nutrient dynamics in relation to geomorphology of riverine wetlands. *Soil Science Society of America Journal* **65**: 557–577.

Jossette G, Leporcq B, Sanchez N, Philippon X. 1999. Biogeochemical mass-balances (C,N,P,Si) in three large reservoirs of the Seine Basin (France). *Biogeochemistry* **47**(2): 119–146.

Kadlec RH, Reddy KR. 2001. Temperature effects in treatment wetlands. *Water Environment Research* **73**(5): 543–557.

Kaye JP, Hart SC. 1997. Competition for nitrogen between plants and soil microorganisms. *Trends in Ecology & Evolution* **12**(4): 139–143.

Kemp WM, Sampou P, Caffery J, Mayer M. 1990. Ammonium recycling bersus denitrification in Chesapeake Bay sediments. *Limnology and Oceanography* **35**(7): 1545–1563.

Kern J, Darwich A, Furch K, Junk WJ. 1996. Seasonal denitrification in flooded and exposed sediments from the Amazon floodplain at Lago Camaleao. *Microbial Ecology* **32**(1): 47–57.

Kleeberg A, Heidenreich M. 2004. Release of nitrogen and phosphorus from macrophyte stands of summer dried out sediments of a eutrophic reservoir. *Archives fur Hydrobiology* **159**(1): 115–136.

Koerselman W, Bakker SA, Blom M. 1990. Nitrogen, phosphorus and potassium budgets for two small fens surrounded by heavily fertilized pastures. *Journal of Ecology* **78**: 428–442.

Kreibich H, Kern J. 2003. Nitrogen fixation and denitrification in a floodplain forest near Manaus, Brazil. *Hydrological Processes* **17**: 1431–1441.

Minzoni P, Bonetto C, Golterman H. 1988. The nitrogen cycle in shallow water sediment systems of rice fields. Part 1: the denitrification process. *Hydrobiologia* **159**: 189–202.

Mitchell AM, Baldwin DS. 1999. The effects of sediment desiccation on the potential for nitrification, denitrification, and methanogenesis in an Australian reservoir. *Hydrobiologia* **39**: 11–23.

Mitsch WJ, Day JW, Gilliam JW, Groffman PM, Hey DL, Randall GW, Wang NM. 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: strategies to counter a persistent ecological problem. *Bioscience* **51**(5): 373–388.

Nowicki BL, Requintina E, VanKeuren D, Portnoy J. 1999. The role of sediment denitrification in reducing groundwater-derived nitrate inputs to Nauset Marsh estuary, Cape Cod, Massachusetts. *Estuaries* **22**(2A): 245–259.

Olfs HW, Neu A, Werner W. 2004. Soil N transformations after application of N-15-labeled biomass in incubation experiments with repeated soil drying and rewetting. *Journal of Plant Nutrition and Soil Science* **167**(2): 147–152.

Patrick WH, Wyatt R. 1964. Soil nitrogen loss as a result of alternate submergence and drying. *Soil Science Society of American Journal* **28**: 647–653.

Qiu S, McComb AJ. 1996. Drying-induced stimulation of ammonium release and nitrification in reflooded lake sediment. *Marine and Freshwater Research* **47**: 531–536.

Rabalais NN, Wiseman WJ, Turner RE, SenGupta BK, Dortch Q. 1996. Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf. *Estuaries* **19**(2B): 386–407.

Rabalais NN, Turner RE, Wiseman WJ. 2001. Hypoxia in the Gulf of Mexico. *Journal of Environmental Quality* **30**(2): 320–329.

Reddy KR, Patrick WH. 1975. Effect of alternate aerobic and anaerobic conditions on redox potential, organic matter decomposition and nitrogen loss in a flooded soil. *Soil Biology and Biochemistry* **7**: 87–94.

Reddy KR, Patrick WH. 1984. Nitrogen transformations and loss in flooded soils and sediments. *Critical Reviews in Environmental Control* **13**(4): 273–309.

Reddy KR, Patrick WH, Lindau CW. 1989. Nitrification-denitrification at the plant root-sediment interface in wetlands. *Limnology and Oceanography* **34**: 1004–1013.

Richardson WB, Strauss EA, Bartsch LA, Monroe EM, Cavanaugh JC, Vingum L, Soballe DM. 2004. Denitrification in the Upper Mississippi River: rates, controls, and contribution to nitrate flux. *Canadian Journal of Fisheries and Aquatic Sciences* **61**: 1102–1112.

Risgaard-Peterson N. 2003. Coupled nitrification-denitrification in autotrophic and heterotrophic estuarine sediments: on the influence of benthic microalgae. *Limnology and Oceanography* **48**(1): 93–105.

Rysgaard S, Risgaard-Peterson N, Nielsen LP, Revsbech NP. 1993. Nitrification and denitrification in lake and estuarine sediments measured by the 15N dilution technique and isotope pairing. *Applied and Environmental Microbiology* **59**(7): 2093–2098.

Scholz O, Gawne B, Ebner B, Ellis I. 2002. The effects of drying and re-flooding on nutrient availability in ephemeral deflation basin lakes in western New South Wales, Australia. *River Research and Applications* **18**(2): 185–196.

Sorensen J. 1978. Denitrification rates in marine sediments as measured by the acetylene inhibition technique. *Applied Environmental Microbiology* **36**: 139–143.

Strauss EA, Lamberti GA. 2000. Regulation of nitrification in aquatic sediments by organic carbon. *Limnology and Oceanography* **45**(8): 1854–1859.

Strauss EA, Mitchell NL, Lamberti GA. 2002. Factors regulating nitrification in aquatic sediments: effects of organic carbon, nitrogen availability, and pH. *Canadian Journal of Fisheries and Aquatic Sciences*. **59**(3): 554–563.

Strauss EA, Richardson WB, Bartsch LA, Cavanaugh JC, Bruesewitz DA, Imker HJ, Heinz JA, Soballe DM. 2004. Nitrification in the Upper Mississippi River: patterns, controls and contribution to the nitrate budget. *Journal of the North American Benthological Society* **23**(1): 1–14.

Sundby B, Vale C, Caetano M, Luther GW III. 2003. Redox chemistry in the root zone of a salt marsh sediment in the Tagus Estuary, Portugal. *Aquatic Geochemistry* **9**: 257–271.

Terry RE, Nelson DW. 1975. Factors influencing nitrate transformations in sediments. *Journal of Environmental Quality* **4**(4): 549–554.

Tockner K, Milard F, Ward JV. 2000. An extension of the flood pulse concept. *Hydrological Processes* **14**: 2861–2883.

Tufekcioglu A, Raich JW, Isenhart TM, Schultz RC. 2003. Biomass, carbon and nitrogen dynamics of multi-species riparian buffers within an agricultural watershed in Iowa, USA. *Agroforestry Systems*. **57**(3): 187–198.

Venterink HO, Davidsson TE, Kiehl K, Leonardson L. 2002. Impact of drying and re-wetting on N, P and K dynamics in a wetland soil. *Plant and Soil* **243**(1): 119–130.

Weisner SEB, Eriksson PG, Graneli W, Leonardson L. 1994. Influence of macrophytes on nitrate removal in wetlands. *Ambio* **23**(6): 363–366.

Wigand C, Stevenson JC, Cornwell JC. 1997. Effects of different submersed macrophytes on sediment biogeochemistry. *Aquatic Botany* **56**(3–4): 233–244.

Zaman M, Chang SX. 2004. Substrate type, temperature, and moisture content affect gross and net N mineralization and nitrification rates in agroforestry systems. *Biology and Fertility of Soils* **39**: 269–279.

