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The Water Quality Consequences of Restoring Wetland Hydrology to a Large Agricultural Watershed in the Southeastern Coastal Plain

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Abstract

To ameliorate local and coastal eutrophication, management agencies are increasingly turning to wetland restoration. A large portion of restoration is occurring in areas that were drained for agriculture. To recover wetland function these areas must be reflooded and disturbances to soils, including high nutrient content due to past fertilizer use, loss of organic matter and soil compaction, must be reversed. Here, we quantified nitrogen (N) and phosphorus (P) retention and transformation in a unique large-scale (440 ha) restored wetland in the North Carolina coastal plain, the Timberlake Restoration Project (TLRP). For 2 years following restoration, we quantified water and nutrient budgets for this former agricultural field. We anticipated that TLRP would export high concentrations of inorganic P immediately following reflooding, while retaining or transforming inorganic N. In the first 2 years after a return to the precipitation and wind-driven hydrology, TLRP retained or transformed 97% of NO_3–N, 32% of TDN, 25% of NH_4–N, and 53% of soluble reactive phosphorus (SRP) delivered from inflows and precipitation, while exporting 20% more dissolved organic nitrogen (DON), and 13% more total P (inorganic, organic, and particulate P) than inputs. Areal mass retention rates of N and P at TLRP were low compared to other restored wetlands; however, the site efficiently retained pulses of fertilizer NO_3–N derived from an upstream farm. This capacity for retaining N pulses indicates that the potential nutrient removal capacity of TLRP is much higher than measured annual rates. Our results illustrate the importance of considering both organic and inorganic forms of N and P when assessing the benefits of wetland restoration. We suggest that for wetland restoration to be an efficient tool in the amelioration of coastal eutrophication a better understanding of the coupled movement of the various forms of N and P is necessary.

Key words: wetland; restoration; nitrogen; phosphorus; retention; mitigation.

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Introduction

Nutrient loading to freshwater and coastal ecosystems is occurring as a result of fossil fuel combustion, human waste disposal, and agriculture.

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Human activities have roughly doubled the annual global rates of nitrogen fixation (N), whereas the phosphorus (P) accumulation within terrestrial soils and aquatic sediments has increased at least by 75% over pre-industrial levels (Vitousek and others 1997; Bennett and others 2001). The resulting elevated nutrient loading to rivers and coastal waters creates harmful algal blooms and areas of regional hypoxia (Diaz and Rosenberg 2008). Because natural wetlands have a high and long-term capacity to retain and transform N and P from surface and subsurface runoff (Johnston 1991; Zedler 2003), wetlands play an important role in ameliorating nutrient loading to coastal ecosystems (Verhoeven and others 2006). Wetlands have the ability to retain and remove N through microbial uptake and transforming reactive nitrogen into inert gaseous forms (N_2) through denitrification (Weller and others 1994; Mitsch and Gosselink 2007). Phosphorus retention in wetlands occurs mostly through sedimentation, mineral adsorption, and plant uptake (Reddy and others 1999). These natural ecosystem services, however, have been lost from many watersheds worldwide due to the draining and destruction of approximately half the global wetland area (Zedler and Kercher 2005).

To regain services provided by wetlands, management agencies are increasingly turning to wetland restoration (Mitsch and others 2001; Zedler 2003). The primary goal of wetland restoration is to reinstate pre-disturbance hydrology to degraded landscapes to provide wetland habitat, flood abatement, carbon sequestration, and nutrient retention (Verhoeven and others 2006). In the US a variety of conservation and mitigation programs promote wetland restoration (Zedler 2003), and are largely responsible for the recent gains in total wetland area (Dahl 2006). Much of the restoration is occurring in former agricultural lands, and consists of small wetlands (<1 ha) at the downstream end of tile drainage or irrigation ditches (Woltemade 2000). Although we have learned much from studies conducted in large natural and restored wetlands such as the Everglades in Florida (Kadlec 2006; Richardson 2008), Caernarvon Diversion in Louisiana (Mitsch and others 2005), and Delaware Bay (Mitsch and Gosselink 2007), to date the majority (95%) of restored wetlands that have been studied have been smaller than 1 ha (Wagner and others 2008). We still lack a clear understanding of the recovery trajectory and function of restored wetlands of intermediate to large sizes (10–1000 ha).

Wetland restoration in former agricultural areas must reverse numerous accumulated disturbances to soils, including high nutrient content as a legacy of past fertilizer use, loss of organic matter, soil compaction, and disruption of natural soil profiles (Bruland and Richardson 2005; Duff and others 2009). Restoration practices seek to reverse the impacts of these disturbances by reinstating wetland hydrology and planting obligate and facultative wetland plants (Zedler 2003). Restoring wetland hydrology to a former agricultural field will have immediate consequences for N and P cycling (Mitsch and Gosselink 2007). Nitrification, denitrification, nitrogen mineralization, and biotic assimilation are the major processes determining the fate of N in wetland ecosystems (Reddy and DeLaune 2008). Nitrogen mineralization and nitrification occur under aerobic conditions, whereas denitrification occurs under anaerobic conditions and depends on the availability of NO_3^- and labile carbon (Davidson and Swank 1986). Dry, aerobic conditions increase N mineralization and nitrification rates in soils (Venterink and others 2002), whereas moist, anaerobic conditions lead to high denitrification rates (Davidson and Swank 1986). Because denitrification leads to removal of NO_3^- from the ecosystem, wetland restoration projects implemented for water quality improvement seek to maximize denitrification rates.

The conditions that promote denitrification, however, can also facilitate the release of P from soils and sediments. The fate of P in wetlands is primarily controlled by geochemical sorption and biotic assimilation (Walbridge 1991). Adsorption of inorganic P onto non-crystalline Al and Fe oxides is the dominant mechanism of long-term P removal in wetlands on acidic soils (Richardson 1985). Flooding can lead to the release of Fe-bound P, due to the reduction of Fe(III) to Fe(II) under anoxic conditions (Reddy and others 1999). Release of inorganic P could be a serious problem in former agricultural fields due to the presence of legacy fertilizer P (Pant and Reddy 2003). Several recent studies of wetland restoration in former agricultural areas have reported release of inorganic P after re-flooding (Van Dijk and others 2004; Aldous and others 2005, 2007). Because maximum rates of denitrification require flooded, anoxic sediments while anoxic sediments release legacy P, it is difficult to maximize retention or transformation of both nutrients simultaneously.

This study sought to quantify N and P retention and transformation in a large-scale (440 ha) restored wetland in the North Carolina coastal plain, the Timberlake Restoration Project (TLRP). The large size of this site makes it unusual compared to the vast majority of wetland restoration projects that have been implemented or studied in the past.

For 2 years following reflooding, we quantified water and nutrient budgets for this former agricultural watershed. We assessed the water quality consequences of the TLRP project in two ways: first, we compared the annual areal rates of N and P retention and export relative to other published studies of restored wetlands; second, we collected samples from the drainage of an agricultural field immediately upstream of the restoration project and used the concentrations together with historic data on pre-restoration water export from TLRP to estimate N and P export under active agricultural management. We predicted that this large restored wetland would: (1) efficiently retain NO_3–N and NH_4–N, relative to its prior, cultivated state and in comparison to previously studied restored wetlands; (2) export large quantities of inorganic P immediately following reflooding; and (3) by year 2 would efficiently retain P relative to pre-restoration conditions.

Methods

Albemarle Peninsula

The Albemarle Peninsula, in the northern coastal province of NC, extends 5000 km^2, with 2700 km^2 under 1-m elevation (Poulter and Halpin 2008). The peninsula is surrounded by the Albemarle, Croatan, and Pamlico sounds on the northern, eastern, and southern ends, respectively. The climate in this region is considered ''humid subtropical.'' Mean annual precipitation is 1330 mm/y (from Plymouth Weather Station, Washington County, NC), and mean annual temperature is 16.6°C.

The region was historically dominated by peat bogs covered with pocosin plant communities, characterized by evergreen shrub-scrub vegetation in the understory and pine in the canopy (Richardson 2003). Pocosin wetlands originally covered 907,933 ha of the coastal plain, but by 1979 only 281,000 ha remained (Richardson 1983). Much of the landscape was deforested at the beginning of the 20th century; and in the 1970–1980s large canals and ditches were built to facilitate agriculture (Carter 1975). Much of the region remains in agriculture, even though 80% of the area requires active pumping to allow agriculture (Neely 2008).

Site Description

Timberlake Restoration Project (TLRP), located in the Albemarle Peninsula in Tyrrell County NC (35°54′22″ N 76°09′25″ W), is part of Great Dismal Swamp Mitigation Bank, LLC [for description of mitigation banking industry in NC see BenDor and others (2009)]. The primary goal of this bank is ''to establish self-sustaining, functioning aquatic systems to replace the functions and acreage of wetlands and other aquatic resources anticipated to be adversely affected'' (Army Corps of Engineers 1997). TLRP is a 1704.2-ha private compensatory mitigation bank comprised of: 420 ha of mature forested wetland that was never in agricultural production, 787 ha of forested wetland (PA), 57.2 ha of drained shrub-scrub, and 440 ha of former agricultural fields undergoing stream and wetland restoration (RW, Figure 1). The site drains to the Little Alligator River (3 km away from the site), which drains into the Alligator River and the Albemarle Sound. The elevation gradient at the site ranges from −0.4 to 5.1 m above sea level (Needham 2006). There are five soil series in the site: Ponzer muck, Hyde loam, Roper muck, Weeksville silt loam, and Pungo muck [very poorly drained hydric soils, USDA SSURGO Database 2005 (Needham 2006)].

The focus of this study is the 440-ha former agricultural field (RW), which is being restored into a large connected tract of riverine wetlands. A critical feature of this site, and its surrounding region, is that the hydrology of the rivers, streams, and hydrologically connected wetlands is freshwater and bi-directional: water flows are combinations of groundwater, rainfall runoff, and wind-driven tides (Poulter and others 2008). Flows at the site can move either upstream or downstream depending largely on rainfall and wind direction. To allow agriculture in this area, farmers often construct one-direction pump stations at the downstream end of properties. These pump stations actively lower the water table upstream and thus drain water from precipitation, and serve as dams by blocking upstream-moving water from wind tides. Restoring hydrology to such areas is done by removing the pump stations, which allows the water table to rise and the site to be re-inundated by tides.

After the last harvest in August 2004, major land movements began to re-establish the hydrology to the pre-agriculture state. Earth movement included filling 53 km (33 miles) of ''vee'' ditches (approximately 90–100 cm deep), plugging sections of the main canal (approximately 3 m deep), and creating a focused zone of preferential flow by connecting the lowest elevation areas across the site (Figure 1). Restoration also included planting 750,000 live saplings from eight species of native trees: *Taxodium distichum*, *Nyssa sylvatica var. biflora*, *Nyssa aquatica*,

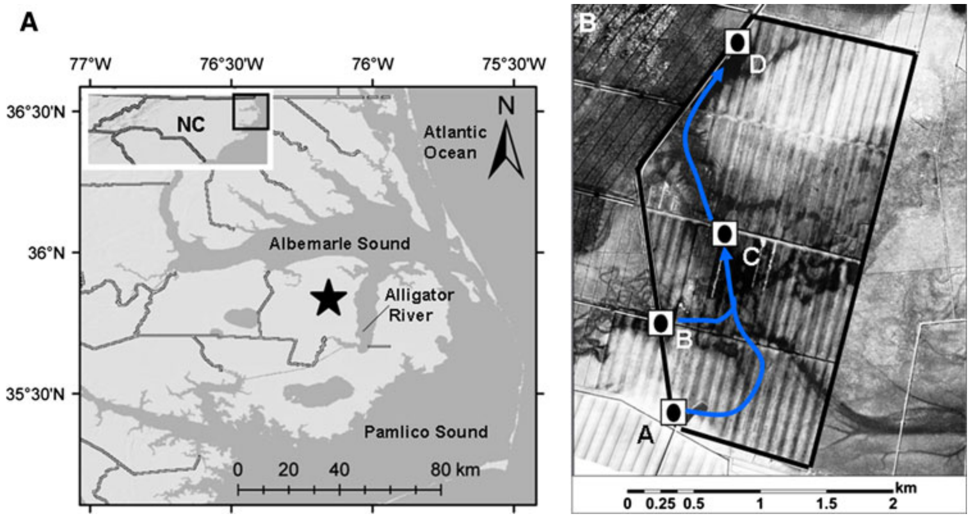

Figure 1. **A** Location of Timberlake Restoration Project (TLRP) in North Carolina. **B** Lidar image of Timberlake Restoration Project and adjacent farm taken on November 18, 2008. RW (sideways pentagon) is the main focus area of the study. Sites *A* (Ag input), *B* (Inflow), and *D* (Outflow) were instrumented with flow meters, automated samplers and were sampled weekly. Water from forested wetland enters the site through *B*.

Fraxinus pennsylvanica, *Salix nigra*, *Chamaecyparis thyoides*, *Quercus nigra*, *Quercus michauxii*, *Quercus phellos*, and *Quercus falcate var. pagodafolia* (Needham 2006). In February of 2007, RW was connected hydrologically to 420 ha of restored forest and the downstream gate-pump system was disabled, reinstating the precipitation and wind tide driven hydrologic regime. In August of 2007, a pump was installed in the upstream end of RW to allow the upstream 2424 ha farm to pump excess water during storm events onto the site.

Hydrologic Events

During the 2-year study period there were three large events that affected hydrologic and nutrient patters at the site. Coastal NC experienced a severe to extreme drought through most of the study period. The average Palmer Drought Severity Index was −1.92 over the 2 years declining from a high of 2.42 in February 2007 to a low of −4.37 in January 2008 (State Climate Office of NC 2009). Associated with the drought was a large (16,000 ha) wildfire in the nearby (30 km) Pocosin Lakes Wildlife Refuge from 2 June to 4 August 2008. The wildfire caused deposition of ashes on the site and created smoke plumes that affected cities as far as Raleigh, NC (320 km away, USFW 2009). Fires in the area are common when the Palmer drought severity index is below −1 (Poulter and others 2006). In addition to the natural drought, in 2008 we conducted two drawdown experiments (4–18 of February 2008 and 18 August to 2 September 2008). During the drawdown experiments, we used the pump on the downstream end of RW to decrease water depth by 1 m across approximately 10 ha for 2 weeks. The effect of the drawdown experiments can be seen as decreased water depths in the outflow (Figure 2A). Because of the long residence time of water within the site, the water depth in the inflow was not affected (Figure 2A). Because the nutrient fluxes during the drawdown experiments did not alter our conclusions, we did not include them in our annual nutrient budget. Changes in water chemistry during the drawdown experiments will be discussed elsewhere (Ardón and others unpublished data).

Water Budget

We measured water depth and velocity in the main inflow and outflow to RW from 25 February 2007 to 26 February 2009. The main inflow and outflow are through 3 m diameter aluminum culverts (Figure 1). Water depth and velocity were recorded every 15 min using acoustic Doppler area velocity meters (2150 Area Velocity Meter, Teledyne ISCO, Lincoln, NE, USA). We used water depth, velocity, and the known area of the culverts to estimate discharge at each site. The pump in the upstream end of the site was also instrumented with an acoustic Doppler area velocity meter (as above) to record water depth and velocity during pumping events, which enter the site through a 0.65 m diameter pipe. Because of the large volume of water that is mobilized during pumping events, the

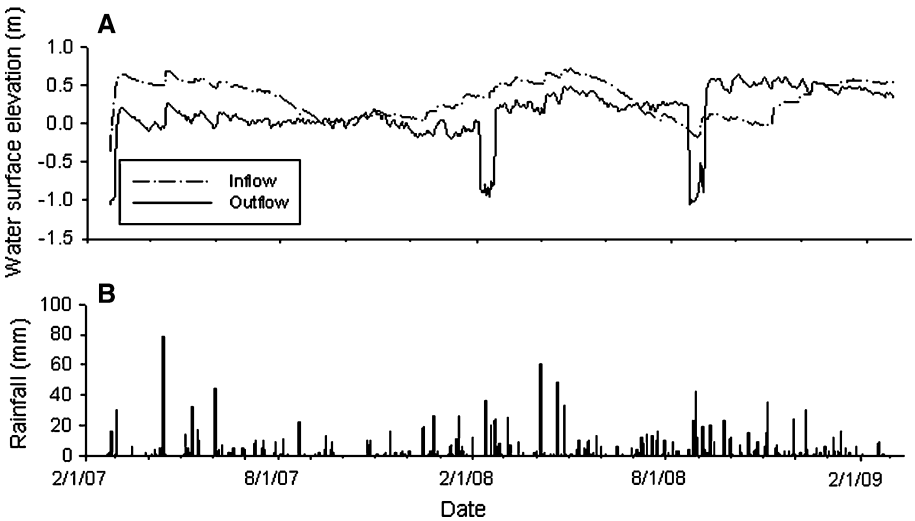

Figure 2. A Water surface elevation (in m above datum NGVD 29) in the Inflow and Outflow from the site for the 2 years of the study. **B** Daily rainfall for the site over the 2 years.

flow meter at this site was programmed to record every minute when the pump was turned on. Bidirectional flow due to wind tides and slow velocity made velocity measurements difficult and at times unreliable. We filtered data that were clearly outliers (that is, more than one order of magnitude greater than the highest velocity we recorded using a handheld flow meter) and interpolated between data points using weighted averages. We discarded and interpolated values for 12% of the Inflow data and 14% of the Outflow data. The average time period for which we had to interpolate values was 4.8 h, although this was primarily during low flows when fluxes were lowest. The longest time period between which we had to interpolate velocity values was 1.9 days. Rainfall was recorded at three sites using three tipping bucket rain gauges (0.01 inch rain tip gauge, Teledyne ISCO, Lincoln, NE, USA, and Infinities USA, Port Orange, FL, USA).

We estimated evapotranspiration for the second year of the study using a modified Penman–Monteith equation using water vapor, temperature, and sun-light measurements made on site during August to September 2008 (Stoy and others 2006). Based on daily estimates from that time period we calculated an average daily evapotranspiration rate, and then extrapolated to the duration of the growing season (200 days). Because we did not have the same empirical measurements available for year one, we used estimates of evapotranspiration measured by pan evaporation by the North Carolina Climate Center in the Plymouth Station (51 km away).

To estimate the annual water budget we used a simple water balance model:

$$\Delta S = P + S_{\mathrm{i}} - \mathrm{ET} - S_{\mathrm{o}}$$

where ΔS is the change in volume of water storage, P is precipitation, S_{i} is surface water inputs, ET is evapotranspiration, and S_{o} is surface water outputs. Groundwater inputs and outputs were not included into the water budget calculations because the site is underlain by an impervious mineral horizon layer (30–50 cm below the soil surface) that serves as a hydrologic barrier.

To estimate uncertainty in the water budget we used a root square sum method to estimate the propagation of error through calculations (Genereux and others 2005). The uncertainty in the total input and output was estimated as the sum of the squared uncertainties in each of the parameters. Based on the differences between the three rain gauges, we estimated that our error in rainfall measurements was 11%, which is within the range of error in rainfall measurements reported in the literature (5–15%, Winter 1981). Based on our interpolation methods for the flow velocity data, we assumed a 15% error in the surface water discharge measurements. For evapotranspiration, we used 20% error as the highest error reported using similar methods (Stoy and others 2006).

Surface Water Chemistry

Two water samples were collected weekly from the main Inflow and Outflow, one sample was filtered immediately (GF/F Whatman filters, 0.7 µm) and the other was left unfiltered for total nutrient digestions. Samples were refrigerated and maintained at 4°C until analyzed (usually within 3 weeks of collection). Soluble reactive phosphorus

(SRP) and total phosphorus (TP) were measured using the ascorbic acid and molybdenum blue method (APHA 1998) with a Lachat QuickChem automated system (Lachat QuikChem 8000, Lachat Instruments, Milwaukee WI). Total phosphorus and total N were analyzed on unfiltered samples after persulfate digestion and measured as SRP and NO_3–N, respectively (Koroleff 1983). NH_4–N was measured using the phenate method on a Lachat QuickChem automated system. For both NH_4 and SRP our lowest standard was 0.005 mg/l; any concentration below that we assigned a value of 0.0025 mg/l. NO_3–N was measured using a Dionex ICS-2000 ion chromatograph with an AS-18 column (Dionex Corporation, Sunnyvale, California, USA). Our lowest standard was 0.001 mg/l; concentrations below that we report as 0.0005 mg/l. Total dissolved nitrogen (TDN) concentrations were measured on a Shimadzu TOC-V total carbon analyzer with a TNM-1 nitrogen module (Shimadzu Scientific Instruments, Columbia, Maryland, USA). Our lowest standard for TDN was 0.1 mg/l; any concentration under that we report as 0.05 mg/l. Digestions for total N showed that our TDN measurements were capturing the majority of N in surface water, so we decided to focus on TDN measurements. We calculated dissolved organic nitrogen concentrations as: TDN − (NO_3–N + NH_4–N). Because a large percentage of P exits the site as particulate P, we focused our monitoring on TP and SRP. To estimate uncertainty in analytical procedures we analyzed every 10th sample in duplicate for all nutrient analyses. Water pH, dissolved oxygen, conductivity, and temperature were measured using a handheld device (YSI Multiprobe Model 560, Yellow Springs, Ohio, USA) every time we collected water samples.

Beginning on 2 September 2007, bulk deposition samples were collected weekly using a 30 cm diameter funnel connected to a 6L pre-acid washed HDPE bottle (Likens and Bormann 1995). The bottle was changed weekly. The small opening of the funnel was covered with a table tennis ball to prevent particles from entering the bottle. If particles were observed inside the bottle, the sample was discarded. The samples were filtered on site and analyzed as above. From February to August 2007 we used average rainfall N concentrations from two nearby National Atmospheric Deposition Program sites (Carteret County, NC and Bertie County, NC). Because the NADP sites do not report organic N (DON), we used relationships between dissolved inorganic N (DIN) and DON ($r^2 = 0.60$, $P < 0.05$), for rain collected at the site, to estimate DON inputs for February to August 2007. For that same period, for SRP and TP in rainfall we used average monthly concentrations from 2008 collected at the site.

In addition to weekly grab samples we conducted high frequency sampling using automated samplers (ISCO 6712, Teledyne ISCO, Ohio) during rainfall events. Storm-triggered samples were collected from the automated samplers within a week, refrigerated, filtered (within 1 week) in the laboratory and analyzed as above (within 3 weeks). We collected and analyzed samples during 10 rainfall events over the 2 years. Samples were collected every 2–12 h to characterize the rise and falling limb after rainfall events.

Nutrient Budget

We used the concentrations from our weekly water samples and daily discharge to estimate nutrient inputs and exports. For dates between samples, we used the average of the beginning and ending concentration values times discharge during the time period (Likens and Bormann 1995). We used rainfall chemistry from our samples, or NADP data, and the rainfall amount measured in the site to determine rainfall inputs. We used the concentrations and the discharge from the high-frequency, storm-triggered sampling to estimate nutrient fluxes during the ten storm events we sampled. We included the storm fluxes in the annual estimate of exports.

Statistical Analyses

To examine differences in water chemistry among sites and years we used analysis of variance (ANOVA) on log-transformed data to meet assumptions of normality. We used the bootstrap technique to estimate 95% confidence intervals around our annual nutrient fluxes (Efron 1982; Jordan and others 2003). Because annual fluxes can be dominated by a few weeks with high fluxes, with the bootstrapping technique we could determine the consequence of randomly including or excluding weekly fluxes from the annual estimate. We conducted 1000 iterations of our dataset by randomly selecting data points (with replacement so they could be selected again) from the original dataset. If the 95% confidence intervals around the fluxes do not overlap, then the fluxes are significantly different at the $P < 0.05$ level. We used linear regression to examine the relationship between rainfall and export of SRP, TP, NH_4–N, and TDN for the 10 storms sampled.

Evaluating Water Quality Benefits

We evaluated the water quality benefits of RW in two ways. First, we compared mass and percentage reduction in nutrient exports observed at RW to values in the literature from restored or constructed wetlands in former agricultural fields. Second, we compared nutrient exports after restoration to estimates of nutrient export from the site pre-restoration and under agricultural practices. We used monthly pumping rates from the downstream pump that drained the site for agriculture for the year 1993, which received similar rainfall (1200 mm) to what we measured in 2008 (1000 mm). We used the water chemistry information for 2008 from the upstream farm as an estimate of the nutrient content of water that would have left RW under active agricultural management. The estimate is based on the assumption that the active farm is applying fertilizer at similar rates to what would have been applied at RW. To determine the validity of the comparison we obtained the USDA-FSA reports for 1993 for TLRP and 2008 for the adjacent farm. The reports show that in 1993, 30% of the site was planted in corn and 70% was in soybean. In 2008, 40% of the upstream farm was in corn and 60% in soybean. It is possible that the actual nutrient export from the site in 1993 was lower than our estimates because of the lower percentage of corn planted, but we think the comparison is a useful way of qualitatively evaluating the consequences of wetland restoration.

Results

Water Budget

Inputs

The site was flooded on February 27, 2007 resulting in an immediate 1 m increase in water depth at both inflow and outflow (Figure 2). As mentioned in the ''Methods'' section, coastal NC experienced a severe to extreme drought through most of the study period which affected water levels. Water depth and discharge varied during the study period (Figure 2), with water depth at the inflow showing seasonal fluctuations and the highest depths occurring in the spring of each year (Figure 2A). The median discharge was 0 m^3/day (Figure 3A). Most of the water entered the site as rainfall (74% in year 1 and 97% in year 2, Table 1), which was higher the second year. One of the rain collectors malfunctioned in October 2008, so for the last 5 months of the study we used the data from two tipping bucket collectors. In year 1, RW received 25% of the inflow from the hydrologically connected restored forest. In year 2, however, the flow from the restored forest was negative, likely due to recharge following the severe drought in the area. In year 2, the site received water pumped from the upstream farm (Ag input), accounting for approximately 4% of the total inputs that year (Table 1). The pump was activated for a total of 19 days through March and April 2008 (Figure 3A).

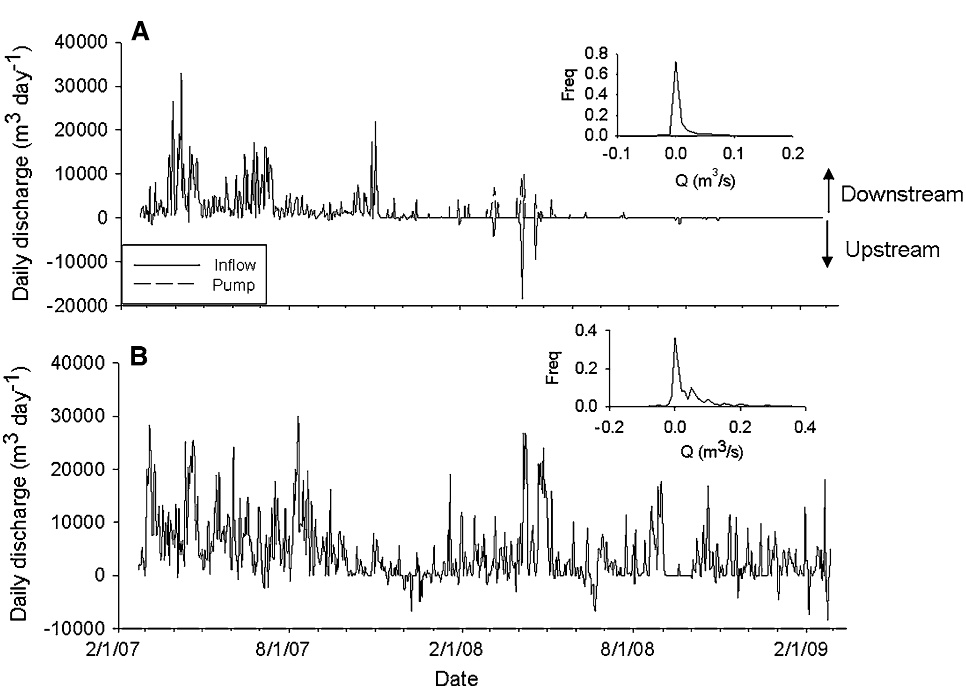

Figure 3. **A** Discharge at the inflow and the pump station at the upstream end of the site, *inset* is the frequency distribution of average daily discharge (m^3/s) over the 2 years. *Dashed line* denotes pumping events from upstream farm (Ag Inputs Table 1). **B** Discharge at the outflow for the 2 years of the study period, *inset* is the frequency distribution of average daily discharge (m^3/s). Positive numbers reflect downstream flow and negative numbers reflect upstream flow due to wind tides or rain events.

Exports

Fluctuations in water depth from the Outflow were influenced by both precipitation and wind tides (Figure 2A). As mentioned earlier, in 2008 we conducted two drawdown experiments (4–18 of February 2008 and 18 August to 2 September 2008), which decreased water depths in the outflow (Figure 2A). Discharge from the outflow was higher than the inflow, particularly in year 2 (Figure 3). The median discharge at the outflow was also 0 m^3/s (Figure 3). Evapotranspiration losses accounted for approximately 65% of rainfall inputs in both years (Table 1). These estimates agree with previous reports that evapotranspiration returns approximately 70% of rainfall inputs in wetlands in the region (Skaggs and others 1980; Richardson 1983), providing confidence in our model estimates.

Table 1. Water Inputs and Exports for Timberlake Farm

	Year 1	Year 2
Inputs		
Rainfall	3.06 ± 0.34	3.69 ± 0.41
Inflow	1.04 ± 0.16	−0.05 ± −0.01
Ag input	0	0.15 ± 0.02
Total inputs	4.1 ± 0.37	3.78 ± 0.41
Exports		
Outflow	2.01 ± 0.30	1.09 ± 0.16
Evapotranspiration	1.97 ± 0.39	2.41 ± 0.48
Total export	3.99 ± 0.50	3.50 ± 0.51
Difference	0.11 ± 0.62	0.29 ± 0.65
(inputs – exports)	(3%)	(8%)

Year 1 was from 26 Feb 2007 to 25 Feb 2008, and year 2 from 26 Feb 2008 to 25 Feb 2009. See text for uncertainty estimate for each parameter. All values in $\times 10^6$ m^3*. Number in parentheses in difference cell is the % difference between inputs and exports. Ag input water pumped from upstream farm.*

Water Balance

The close agreement between all inputs and exports (3% year 1 and 8% in year 2, Table 1) supports our assumption that there is no net loss or gain from groundwater. The assumption of minimal exchange with groundwater was also validated by the differences in soil water chemistry above and below the impervious mineral horizon (Ardón, unpublished data).

Water Chemistry

Inputs

Concentrations of total dissolved nitrogen inputs in surface water were dominated by NH_4–N and

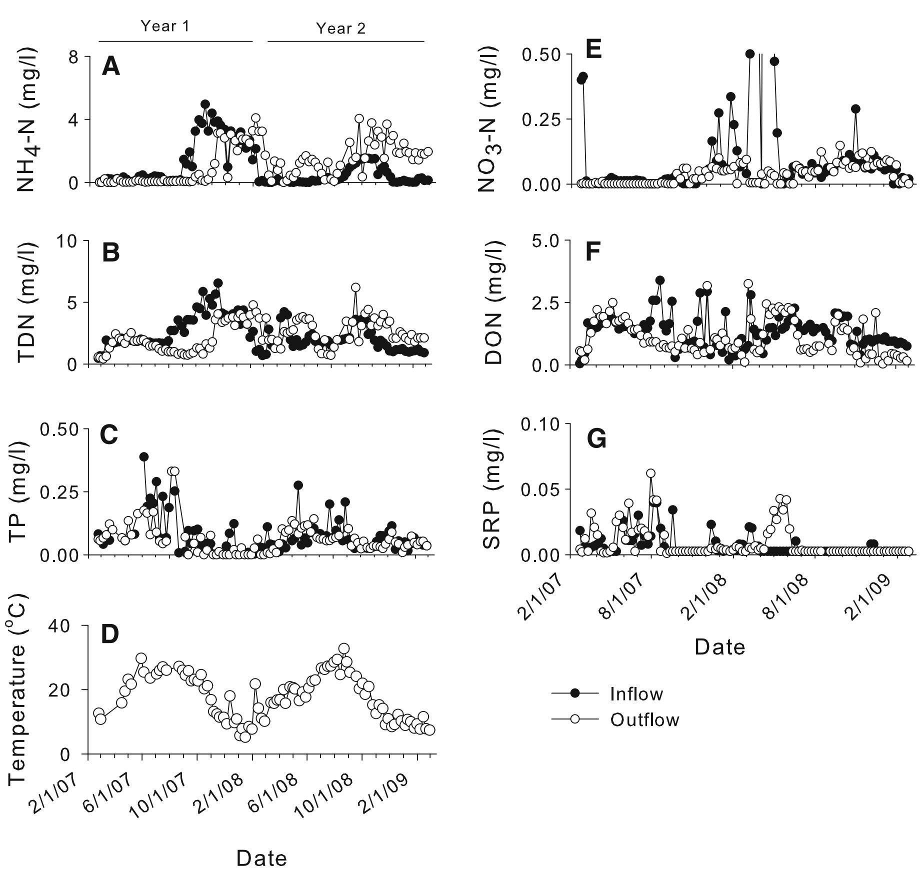

Figure 4. Concentrations (mg/l) in surface water of: **A** NH_4–N, **B** TDN, **C** TP, **E** NO_3–N, **f** DON, and **G** SRP at the inflow (*black circles*) and outflow (*white circles*) over the 2 years of the study. **D** Water temperature at the outflow over the 2 years. Arrow in **E** denotes a pumping event from upstream farm, which is shown in more detail in Figure 6A.

Table 2. Annual Mean and Range (Min and Max) Nitrogen Concentrations (mg/l) in Inputs and Exports from Timberlake Restoration Project

Inputs	NH_4–N	NO_3–N	DON	TDN	NH_4:NO_3	DIN:DON
Year 1						
Rainfall	0.27 (0.01–0.78)	0.73 (0.001–1.8)	0.25 (0.01–0.55)	1.24 (0.31–2.38)	0.37	4
Inflow	1.38 (0.016–4.96)	0.06 (0.001–0.41)	1.34 (0.04–3.38)	2.78 (0.46–6.53)	23	1.1
Outflow	0.88 (0.001–4.09)	0.02 (0.001–0.01)	1.12 (0.19–3.17)	2.02 (0.41–4.77)	44	0.8
Year 2						
Rainfall	0.43 (0.0025–1.79)	0.48 (0.08–1.85)	0.35 (0.007–2.02)	1.18 (0.14–3.25)	0.9	2.6
Inflow	0.39 (0.001–1.55)	0.30 (0.001–3.3)	1.21 (0.26–2.26)	1.90 (0.68–4.21)	1.3	0.5
Ag input	0.51 (0.0025–1.07)	5.28 (0.001–9.82)	0.65 (0.01–2.01)	5.63 (0.69–11.37)	0.10	8.5
Outflow	1.21 (0.001–4.0)	0.04 (0.001–0.14)	1.03 (0.03–3.24)	2.31 (0.40–6.18)	30.25	1.2

Table 3. Annual Mean and Range (Min and Max) Phosphorus Concentrations (mg/l) in Inputs and Exports from Timberlake Restoration Project

Inputs	SRP	TP	SRP:TP
Year 1			
Rainfall	0.01 (0.0025–0.03)	0.023 (0.0025–0.049)	0.43
Inflow	0.009 (0.0025–0.039)	0.079 (0.0025–0.38)	0.11
Outflow	0.01 (0.001–0.062)	0.065 (0.0025–0.33)	0.16
Year 2			
Rainfall	0.023 (0.0025–0.1)	0.037 (0.0025–0.17)	0.62
Inflow	0.005 (0.0025–0.021)	0.066 (0.01–0.27)	0.08
Ag input	0.016 (0.0025–0.086)	0.021 (0.0025–0.25)	0.50
Outflow	0.01 (0.0025–0.062)	0.064 (0.0025–0.33)	0.12

DON (Figure 4; Table 2). NH_4–N concentrations in the Inflow were higher in year 1 than in year 2 ($F_{1,104} = 20.2$, $P < 0.001$). Inputs from the upstream farm (Ag input) in year 2, however, were dominated by NO_3–N (Table 2). Ammonium and TDN increased during fall and winter (Figure 4A, B). The highest NO_3^- concentrations (>2 mg NO_3–N l^{-1}) in surface water inputs occurred during March and April 2008, which coincided with pumping events from the upstream farm and fertilizer application (Figure 4E). P concentrations in surface water were dominated by particulate and organic forms, with SRP tending to be less than 15% of TP (Table 3; Figure 4C, G). SRP concentrations in surface water were higher in year 1 than in year 2 ($F_{1,105} = 22.65$, $P < 0.001$, Table 3). Starting in the summer of year 2, TP and SRP concentrations in rainfall increased, but high variation led to no significant difference between years (SRP $F_{1,62} = 2.05$, $P = 0.15$, TP $F_{1,62} = 2.5$, $P = 0.11$; Figure 5C, G). The highest SRP concentration in rainfall (0.17 mg l^{-1}) was on 26 June 2008, which is within the time frame of the previously mentioned large fire in nearby Pocosin Lakes Wildlife Refuge.

Exports

Concentrations of total N exports in surface water were dominated by NH_4–N and DON (Table 2; Figure 4). The NH_4:NO_3 ratios increased between the Inflow and the Outflow in both years (Table 2). The DIN:DON ratio decreased in both years, the biggest change was in year 2 between the Ag input (8.51) and the outflow (1.75, Table 2). Surface water concentrations of NH_4–N ($F_{1,104} = 10.1$, $P < 0.001$), NO_3–N ($F_{1,105} = 25.3$, $P < 0.0001$), and TDN ($F_{1,105} = 8.98$, $P < 0.001$), increased from year 1 to year 2 (Table 2). The highest NH_4–N concentrations occurred during fall and winter of each year (Figure 4A). NO_3–N concentrations tended to increase in the summer and fall of 2008, but declined again in the winter of 2008–2009 (Figure 4E). SRP concentrations decreased between the 2 years ($F_{1,104} = 2.76$, $P < 0.01$, Table 3; Figure 4G). SRP was high during the first 6 months, declined during the fall and winter of 2007, and then increased again in spring and summer of 2008 (Figure 4G). This increase in SRP concentrations at the outflow in the spring of 2008 did not coincide with an increase in SRP concentrations at the inflow. We did not find any relationship between

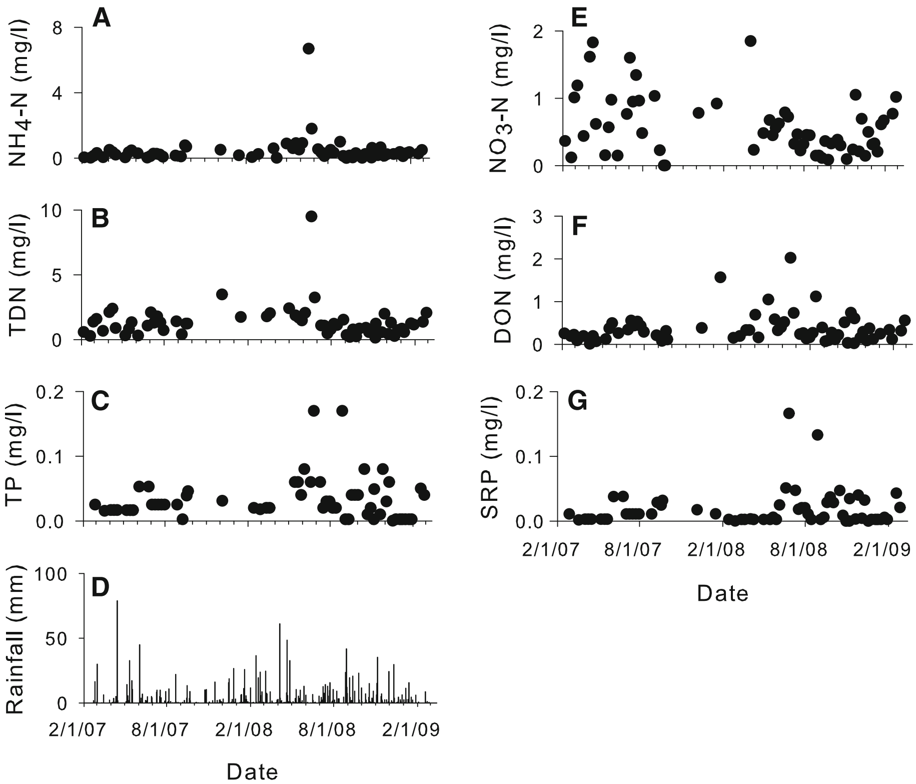

Figure 5. Concentrations (mg/l) in rain water of **A** NH_4–N, **B** TDN, **C** TP, **E** NO_3–N, **F** DON, and **G** SRP. **D** Daily rainfall (mm) over the 2 years.

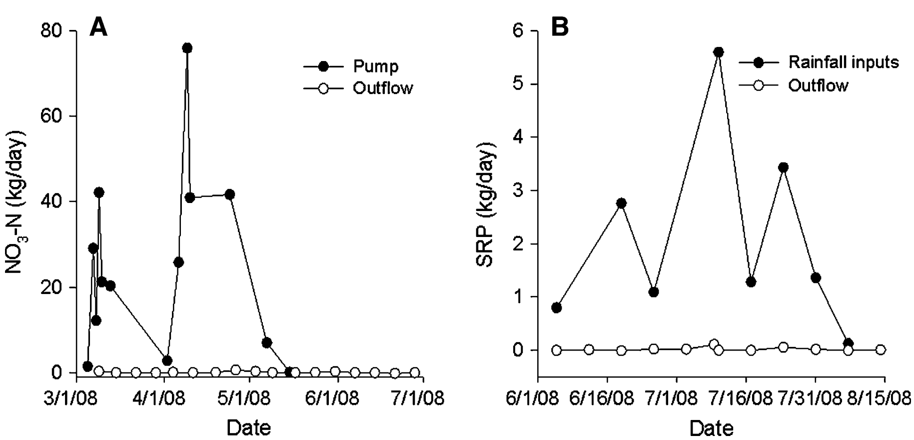

Figure 6. **A** NO_3–N flux (kg/day) during pumping events in March and April 2008 at from the upstream farm (Ag input in Table 1, *black circles*) and the outflow (*white circles*). **B** Rainfall inputs (*black circles*) and export (*white circles*, kg/day) of SRP during the summer of 2008.

nutrient concentrations and discharge for either the weekly sampling or the storm samples. Average surface water temperature was 17°C (range 5–32°C, Figure 4D), specific conductivity was 6.2 mS/cm (range 0.05–10 mS/cm), dissolved oxygen was 50.9% (range 3–112%), pH was 4.24 (range 3.4–6.8), and oxidation–reduction potential was 243.9 mV (range −205 to 468 mV).

Nutrient Budget

Inputs

Surface water N inputs in year 1 were dominated by NH_4–N and DON, whereas in year 2 inputs were dominated by NO_3–N (Table 4). In year 2, there were large inputs of NO_3–N from the upstream farm (Ag, Table 4; Figure 6A). Rainfall inputs were a large percentage (45–90%) of total inputs for all nutrients. Nutrient inputs from rainfall were larger than surface water inputs for all nutrients in year 2 (Table 4), with the difference being largest for TP and SRP. There were large inputs of SRP from rainfall during the summer of 2008 (Figure 6B).

Exports

Nitrogen (N) exports were dominated by NH_4–N and DON both years (Table 4). Exports of NH_4–N almost doubled from year 1 to 2 but the difference

Table 4. Nutrient Inputs and Exports (mean kg/y ± error estimate) for RW for 2 Years

	NH_4–N	NO_3–N	DON	TDN	SRP	TP
Year 1						
Inputs						
Rainfall	596.6 ± 65.6	1139.7 ± 125.4	414.3 ± 45.6	2058.8 ± 226.5	25.84 ± 2.8	44.7 ± 4.9
Inflow	849.5 ± 220.6	17.7 ± 4.3	1555.5 ± 260.1	2414.6 ± 399.9	11.75 ± 1.9	96.3 ± 21.4
Total inputs	1446.2 ± 230.2	1157.5 ± 125.4	1969.8 ± 264.1	4473.4 ± 458.2	37.59 ± 3.4	141.0 ± 22.0
Export	684.1 ± 193.0	16.8 ± 3.5	2431.1 ± 363.2	3224.8 ± 470.7	30.6 ± 5.8	206.6 ± 50.1
Difference (inputs − exports)	762.0 ± 300.4	1140.7 ± 125.4	−461.3 ± 449	1248.7 ± 656.8	6.99 ± 6.7	−65.7 ± 54.6
Year 2						
Inputs						
Rainfall	1044.7 ± 114.9	1028.1 ± 113.1	869.5 ± 95.6	3004.4 ± 330.5	56.27 ± 6.2	91.9 ± 10.1
Inflow	−7.0 ± 3.5	−95.8 ± 63.0	−43.1 ± 6.6	−132.1 ± 91.5	−0.39 ± 0.2	−2.2 ± 1.3
Ag input	63.8 ± 26.8	331.1 ± 100	66.5 ± 46.6	620.2 ± 603.0	0.99 ± 0.3	3.7 ± 1.7
Total inputs	1101.5 ± 118.1	1263.4 ± 163.6	893.0 ± 106.3	3492.5 ± 693.7	56.87 ± 6.2	93.4 ± 10.3
Export	1221.3 ± 278.1	43.0 ± 8.2	991.7 ± 312.1	2167.0 ± 436.0	12.13 ± 2.6	58.7 ± 15.4
Difference	−119.8 ± 302.1	1220.5 ± 163.8	−98.7 ± 320.6	1325.2 ± 819.6	44.7 ± 6.7	34.6 ± 18.5
Combined						
Inputs						
Rainfall	1641.3 ± 180.5	2167.8 ± 168.8	1283.8 ± 106.0	5063.2 ± 400.6	82.1 ± 6.8	136.6 ± 11.2
Inflow	842.5 ± 126.4	−78.1 ± 63.1	1512.4 ± 260.2	2282.5 ± 409.4	11.4 ± 1.9	94.1 ± 23.7
Ag input	63.8 ± 9.6	331.1 ± 100.1	66.5 ± 46.0	620.2 ± 603	1.0 ± 0.3	3.7 ± 1.7
Total inputs	2547 ± 220	2420.9 ± 206.2	2862.7 ± 284.7	7965.9 ± 831.7	94.5 ± 7.1	234.4 ± 52.4
Export	1905 ± 285.8	59.7 ± 8.9	3667.7 ± 478.8	5391.8 ± 641.1	43.5 ± 6.4	279.3 ± 41.9
Difference	642.3 ± 426.0	2361.5 ± 206.4	−560 ± 557.9	2574.1 ± 1050	51.7 ± 9.5	−31.1 ± 58.6

Year 1 was from 26 February 2007 to 25 February 2008, and year 2 from 26 February 2008 to 25 February 2009. Errors represent 1 standard deviation based on bootrstrapping technique. Positive values in Difference indicate the site was a net sink, whereas negative values indicate the site was a net source.

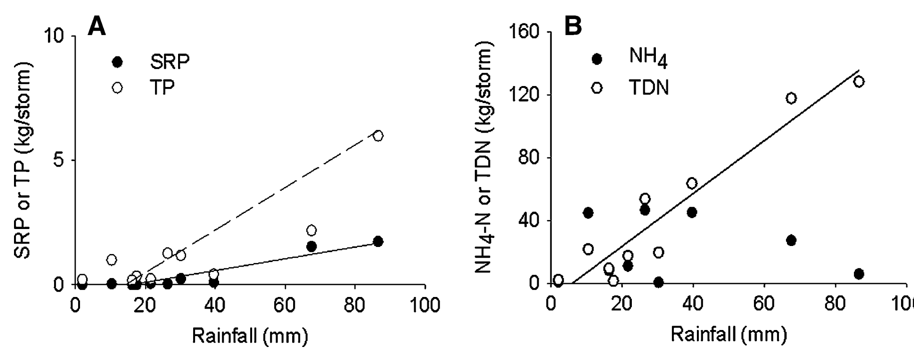

Figure 7. **A** Relationship between rainfall and cumulative SRP or TP flux (kg/storm) during 10 storm events over the 2 years. *Dashed line* represents regression for TP ($r^2 = 0.74$, $P < 0.001$) and solid line is the regression for SRP ($r^2 = 0.68$, $P = 0.01$). **B** Relationship between rainfall and TDN or NH_4–N flux (kg/storm). *Solid line* represents regression for TDN ($r^2 = 0.90$, $P < 0.001$), relationship for NH_4–N was not significant.

was not significant ($P > 0.05$, Table 4). Exports of TP and SRP declined by almost half between years 1 and 2 ($P < 0.05$ in both cases, Table 4). There was a strong correlation between rainfall amount and cumulative discharge ($r^2 = 0.87$, data not shown) for the 10 rain events that we sampled over the 2 years. Rain events increased the amount of cumulative P exported both as SRP ($r^2 = 0.87$, $P < 0.001$) and TP ($r^2 = 0.68$, $P < 0.01$, Figure 7A). TDN export also increased ($r^2 = 0.90$, $P < 0.001$), whereas NH_4–N export declined with increasing rainfall (Figure 7B).

The range of discharge covered by our weekly sampling (−0.6 to 1.03 m^3/s) was similar to the range of discharge from our storm sampling (−0.54 to 1.08 m^3/s). Even though we only sampled 3% of the rainfall events, our storm sampling included the two largest rain events (April 15, 2007, 78 mm/

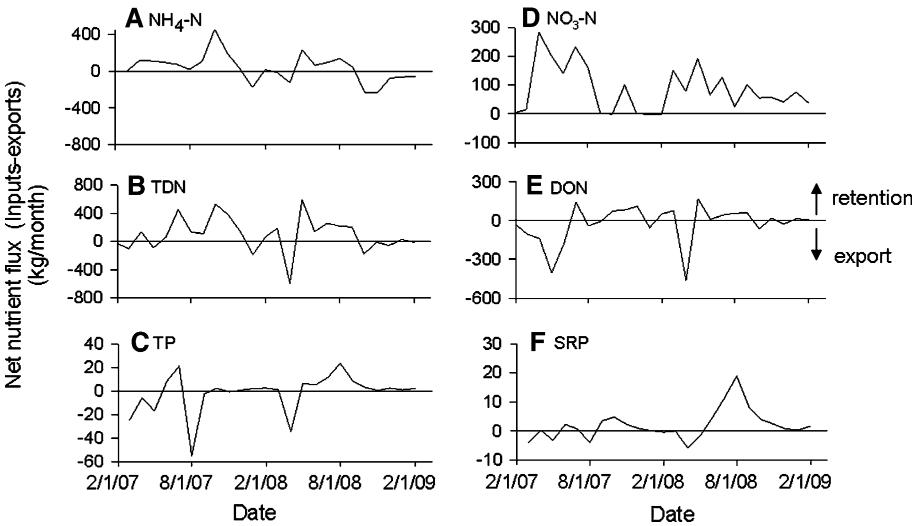

Figure 8. Monthly net nutrient flux (inputs minus exports in kg/month) for **A** NH_4–N, **B** TDN, **C** TP, **D** NO_3–N, **E** DON, and **F** SRP. Positive values denote retention or transformation (inputs > exports) and negative values denote release (inputs < exports).

day, and April 20, 2008 57 mm/day). Thus, we are confident that our sampling covered the range of flows that occurred during the 2 years. We included the storm data in our estimates of nutrient export even though the cumulative storm export did not account for more than 10% of the annual exports for any of the nutrients (Figure 7).

Nutrient Flux Balance

RW changed from a sink of NH_4–N (762 ± 300 kg NH_4–N/y) in year 1 to a modest source in year 2 (−119 ± 302 kg NH_4–N/y, Table 4). Inputs of NO_3–N and TDN were higher than exports in both years (Table 4), but the differences were only significant for NO_3–N ($P < 0.05$). Exports were higher than inputs for DON in both years (net export of 461 ± 449 kg DON/y in year 1, and 98.7 ± 320 kg DON/y in year 2, Table 4). The watershed was retentive of SRP in both years ($P < 0.05$), although changing from a source to a sink of TP from year 1 to year 2 (Table 4).

To look at seasonal patterns in nutrient fluxes we estimated net nutrient flux as inputs minus exports on a monthly basis (Figure 8). NH_4–N was mostly released during the fall and winter of both years (Figure 8A). Differences between inputs and exports for NO_3–N were largest during spring and summer of both years (Figure 8D). Net fluxes of TDN were similar to patterns for NH_4–N (Figure 8B). DON tended to be released in the early spring (Figure 8E). We observed large retention of SRP in the summer of 2008 (Figure 8F).

Evaluating Water Quality Benefits

Mass retention rates of N and P at RW were very low compared to other restored and constructed wetlands receiving unregulated flows (Table 5). In contrast, the percentage of N and P inputs retained were similar to literature values (Table 5).

Based on our estimates of pre-restoration fluxes, restoring RW reduced the export of NO_3–N to the estuary (Figure 9). We estimate that NO_3–N export to the Albemarle Sound would have been an order of magnitude higher under agriculture (1.6 kg ha^{-1} y^{-1}) than what we measured over the first 2 years since restoration (0.15 kg ha^{-1} y^{-1}). Our estimate of pre-restoration export of TDN (3.2 kg ha^{-1} y^{-1}) was lower than the measured export over the 2 years (7.3 kg ha^{-1} y^{-1}). DON export was also higher after restoration (3.8 kg ha^{-1} y^{-1}) than pre-restoration (0.8 kg ha^{-1} y^{-1}). The masses exported after restoration of SRP (0.05 kg ha^{-1} y^{-1}) and TP (0.6 kg ha^{-1} y^{-1}) were also higher than our estimates of SRP (0.01 kg ha^{-1} y^{-1}) and TP (0.07 kg ha^{-1} y^{-1}) losses under agriculture (Figure 9).

Discussion

Our mass balance approach showed that in the first 2 years after reinstating the precipitation and wind-driven hydrology, RW was a net sink for NO_3–N and SRP. A portion of this ''sink'' was offset by the production and export of less available DON and particulate and organic P. Contrary to our predictions areal N retention rates within TLRP were low relative to published studies (Table 5), yet the restored wetland was capable of completely eliminating fertilizer derived NO_3^- delivered to the site from the adjacent agricultural field. Following restoration the site exported both SRP and TP at annual rates 2–8× higher than estimates of pre-restoration fluxes from the site under agriculture. Early estimates of the effectiveness of this large-

Table 5. Literature Values of N and P Removal Rates (g m^{-2} y^{-1} and % of influx) for Restored or Constructed Wetlands

Wetland	Wetland size	N (g N m^{-2} y^{-1})	% influx	P (g P m^{-2} y^{-1})	% influx	Reference
Caernarvon, LI, USA	26,000	46	39–92	–	–	Mitsch and others (2005)
Everglades Marsh, Fl USA[a]	8000	10 - 11	75–450	0.4-0.6	87-133	Craft and Richardson (1993)
Everglades Nutrient Removal Project, FL, USA	1545	10.8	26	0.94	79	Moustafa (1999)
Houghton Lake, MI, USA	700	12.8	97	5.14	94	Kadlec (2009b)
Timberlake Restoration Project, NC, USA	440	0.21–0.9	8–28	−0.05–0.02	−14–22	This study
Lake Apopka, Fl, USA	200	–	–	0.48	30–67	Coveney and others (2002)
Created river diversion wetland, OH USA	3	32	30	4.5	50	Fink and Mitsch (2007)
Palustrine wetlands, WA, USA	1.5–2	ND	ND	0.4–3	7.5–82	Reinelt and Horner (1995)
Restored agricultural wetland, MD	1.3	−1.1–4.5	−8.4–38	−0.28–2.8	−11–59	Jordan and others (2003)
Agricultural wetlands, OH, USA	1.2	39[b]	40.2	6.2	56.2	Fink and Mitsch (2004)
Created river wetlands, OH, USA	1	58–66	30	5.2–5.6	50	Mitsch and others (1998)
Constructed agricultural wetlands	0.4–0.16	14.2–23.2	23–44	0.06–0.1	40–79	Kovacic and others (2006)
Lake Bloomington, Il						
Constructed wetlands, Norway	0.35–0.09	50–285	3–15	26–71	21–41	Braskerud (2002a, b)
Constructed wetland, Victoria Australia	0.04	23	11	2.8	17	Raisin and others (1997)

[a] *Determined from N and P accumulation in sediments.*
[b] *Nitrate only.*

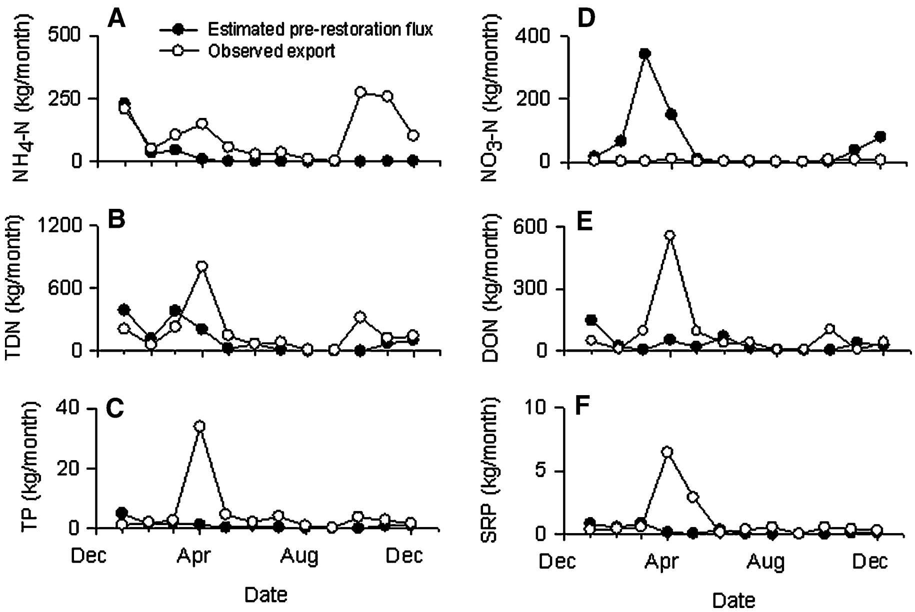

Figure 9. Estimated pre-restoration nutrient flux and observed nutrient flux after restoration from RW. See text for calculations of pre-restoration fluxes.

scale restoration project at improving water quality are thus mixed—the restoration provided small water quality benefits as typically measured (mass of nutrient retained per year), it increased export of SRP, TP, DON, and NH_4^+ from its pre-restoration agricultural state, but provided a significant benefit in protecting the estuary from large pulses of fertilizer NO_3^-. The unimpressive annual rates of retention and the impressive capacity to attenuate nutrient pulses both result from the long residence time of water in RW due to its large size relative to its watershed.

Water Quality Consequences—Nitrogen

In the first 2 years following hydrologic reconnection, the site retained or transformed 97% of the NO_3–N that entered as deposition or as inflow from the upstream farm. We estimate that under continuing agricultural management, the site would have exported 10× higher fluxes of NO_3–N than we observed. The efficiency of this wetland in reducing NO_3–N export was expected and agrees with previous reports of low export from created wetlands in NC (Hunt and others 1999; Poe and others 2003).

Although RW proved very effective at removing NO_3–N immediately following restoration, NO_3–N is not the dominant form of dissolved N supplied to the site. The site retained or transformed 32% of the total dissolved nitrogen that was received through precipitation and surface water. Although there was NH_4–N retention in year 1, in year 2 the RW site exported more NH_4–N than it received from surface water and precipitation, and NH_4–N exports were 6× higher than estimated for the site under prior agricultural management (Figure 9). Net export of NH_4–N had been previously reported for a small in-stream agricultural wetland in NC (Hunt and others 1999), and our results collectively suggest that nitrification is inhibited in the flooded acidic soils of this restored wetland relative to its prior, actively drained state.

Flooding the site increased the export of DON relative to our estimates under agricultural management (Figure 9) and the site was a net source of DON relative to inputs (Table 4). Reflooding increased the concentrations of both NH_4–N and DON relative to the agricultural field outflow. We thus estimate that on an annual basis 2× more TDN was exported from RW than would have been exported when it was actively managed for agriculture. When compared to its agricultural past, the initial effect of restoration has been to increase the export of N to the downstream estuary. Because much of this increased load is as DON, we anticipate that it will result in less N pollution than an equivalent mass of NO_3–N. Our preliminary estimates suggest that 10–25% of the DON is bioavailable (Ardón unpublished data), suggesting that DON export could offset some of the benefits of reducing NO_3–N exports (Wiegner and Seitzinger 2004).

The capacity of RW to remove NO_3–N appears to be much higher than our measured rates of N retention. RW retained only 0.6–2.8 kg of N ha^{-1} y^{-1}, primarily by retaining or transforming NO_3–N over the 2 years. This rate falls well below N retention reported in the literature for restored and treatment wetlands (Table 5), and is well below the sustainable removal rate of 100–400 kg N ha^{-1} y^{-1} suggested by Mitsch and others (2001). Yet, RW showed an impressive capacity for eliminating the NO_3–N pulse. Indeed a considerable proportion of the total NO_3–N retention in year 2 (26%) was due to the complete elimination of the extremely high NO_3–N concentrations delivered following fertilization of the upstream agricultural field in the spring of 2008 (Figure 6A). Denitrification potential rates measured across the site range from 0.68 to 2.75 mg N kg^{-1} day^{-1} (Morse 2010). These rates are low compared to rates measured in other natural and restored wetlands (average = 13.9 mg N kg^{-1} day^{-1}, range = 1–74 mg N kg^{-1} day^{-1}, Reddy and Delaune 2008). Low denitrification rates at RW might be due to low NO_3–N loading (Hunt and others 1999; Poe and others 2003). Our results suggest that this restored wetland could remove or retain more N if a greater proportion of inputs were delivered as NO_3–N, or were converted to NO_3^- through nitrification within the wetland. Other work at the site indicates that low nitrification rates may limit the amount of NO_3–N available for denitrification (Morse 2010). As the vegetation on the site matures we anticipate that plant-derived labile C and plant transport of oxygen to the sediments will increase the potential for coupled nitrification–denitrification within the site (Poe and others 2003). Active management approaches might use liming to raise sediment pH and hydrologic manipulation to increase oxygenation of surface waters to make conditions more favorable for nitrification within the wetland.

Water Quality Consequences—Phosphorus

We found seasonal export of both SRP and TP in the 2 years following restoration. During this time TLRP exported both SRP and TP at annual rates 2–8× that of our estimates of SRP and TP losses under agriculture (Figure 9). Changes in the SRP:TP ratio (Table 3), however, suggest that TLRP was effective at altering the form of P export to downstream ecosystems. The SRP:TP ratio of water exports was similar between both years, even after Ag inputs with a high SRP:TP ratio (0.5) and higher TP deposition rates in year 2 (year 1 = 10 mg P m^{-2} y^{-1}, year 2 = 20 mg P m^{-2} y^{-1}), probably due to the large wildfire in Pocosin Lakes Wildlife Refuge. These results suggest that although the site was a net source of TP over the 2 years, it altered the form of exports from inorganic P to particulate P.

The observed TP export is consistent with other studies that have reported P release after reflooding former agricultural fields from: peat soils in the Netherlands (Van Dijk and others 2004), sandy peat in Sweden (Venterink and others 2002), and soils from a restored lake fringe wetland in Oregon (Aldous and others 2005, 2007). Each of these studies attributed the P releases to decreases in redox and subsequent release of Fe-bound P (Aldous and others 2005), increases in pH of surface water (Van Dijk and others 2004), and release of microbial P (Venterink and others 2002). Each study measured P export for less than 1 year and suggested that P release is a temporary phenomenon. The results of our 2 year study suggest that P release may continue for several years following reflooding. We observed seasonal trends of SRP and TP export over the 2 years (Figure 8C, F). There was a net export of SRP (8 kg) and TP (76 kg) during the first 6 months after restoration (March to August 2007). The following spring, we again observed a net export of SRP (8 kg) and TP (24 kg) (February to May 2008). Our results showed that there is seasonality of P release, and although TP export declined from year 1 to 2, SRP export remained similar. In both years, the early spring release of SRP coincides with high temperatures and the maximum spatial extent of inundation—potentially allowing the release of P from soils that remained dry, or hydrologically disconnected for most of the preceding year.

How Does the Restored Wetland Compare to other Wetlands?

Mass retention rates of N and P at RW were low compared to rates reported for other natural and restored wetlands (Table 5), but this low mass retention was a reflection of the low nutrient loading to the site. When we compare the percentage retained for N and P in RW, they are similar to what has been observed in much larger wetlands such as the Everglades Nutrient Removal Project (Table 5). Previous studies have found an asymptotic relationship between nutrient retention and nutrient loading rate (Mitsch and others 2005; Kadlec 2009a). The low annual mass retention of inputs suggests that RW is unexceptional, despite its large size. Yet we found that RW was unique in comparison to typical restored and constructed wetlands in its ability to retain pulsed N and P inputs. High stormflow inputs of NO_3^- pumped onto the site from an upstream farm and high P deposition delivered as ash from a drought-induced wildfire led to no measurable increases in the export of NO_3 or SRP (Figure 6). This capacity for retaining nutrient pulses indicates that the potential total nutrient removal capacity of RW is much higher than the annual rates we measured.

In contrast to previous studies on small restored wetlands, the large area, shallow water, and long residence time allowed RW to attenuate large pulses of nutrients without increasing export. In constructed and restored wetlands, where the retention time is on the order of days, single rain events can cause water to move through the wetland in a matter of hours, reducing the ability of the wetland to retain and transform nutrients. The few studies that have examined the importance of pulse events have shown decreased nutrient retention during times of increased water movement. For a 1 ha restored wetland in Ohio, Mitsch and others (2005) showed that SRP retention declined when a wetland received pulsed water inflow as opposed to more constant flow. A 1.2 ha wetland in Ohio also showed decreased N and P retention due to spring and early summer storms (Fink and Mitsch 2007). P export increased with high discharge, partly from mobilized sediments, due to tropical storms and hurricanes in a 3.3 ha in-stream wetland in the Coastal Plain of NC (Novak and others 2007).

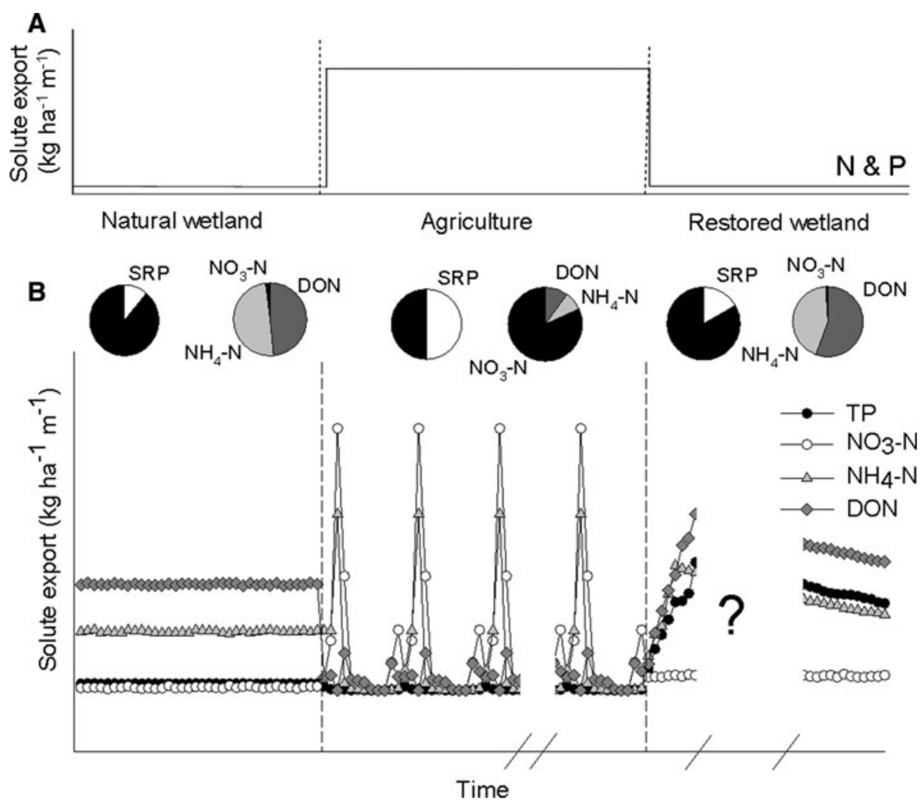

Figure 10. Conceptual model of the ''traditional'' restoration practice view of the recovery of nutrient export (kg ha^{-1} month^{-1}) from a natural wetland, converted to an agricultural field, and then restored to wetland conditions (**A**). It is generally assumed that natural wetlands retain and transform all species of N and P, these functions are lost when wetlands are drained and put into agriculture, and recovery of transformation of both nutrients will occur when wetlands are restored (**A**). Our results suggest that a more nuanced understanding of the biogeochemical transformations of N and P is needed to assess the costs and benefits of wetland restoration. It is well known that not all natural wetlands always function as sinks of nutrients (Mitsch and Gosselink 2007). Natural wetlands in the coastal plain of North Carolina historically retained NO_3–N and P, but released NH_4–N, DON, and TDN (Richardson 1983, **B**). During agriculture it is likely that these sites retained P but released all forms of N. Our results suggest that wetland restoration can lead to retention of NO_3–N and release of DON, NH_4–N, and TP (**B**). Pie charts indicate the relative forms of TP (SRP in white, and particulate plus organic P in *black*), and total N (NO_3–N in black, NH_4–N in light gray, and DON in dark gray) under the different land uses (**B**). We determined the relative concentrations of different forms of N and P and magnitude of fluxes based on our data (Natural wetland data are from Y1 from Inflow, agriculture data from Ag input, and restored wetland from Outflow). Because our goal was to do a qualitative comparison, not a quantitative one, we removed values from the *y*-axis. Breaks in the *x*-axis indicate years in agriculture, and unknown time required for the restored wetland to go back to its natural wetland state.

Raisin and others (1997) reported decreased N and P retention during storm events in a 450 m^2 constructed wetland in Australia.

Comparisons between our site and other wetlands should be interpreted with caution given that TLRP was under a drought during a portion of the study period. Low nutrient loading from the mature forested wetland in year 2, lower water export from the site during year 2, and the deposition of ash due to a large forest fire in Pocosin Wildlife Refuge, are all consequences of the drought that likely affected our results. We hypothesize that higher loading rates during a wetter year will likely increase the mass retention rates of N and P, but the percent retention of different solutes will be similar. Although the drought makes it challenging to generalize our results, global circulation models predict that droughts, punctuated by severe storms, are going to become more common (Allan and Soden 2008). Understanding how natural and restored wetland ecosystems respond to both droughts and storms will become more important under future scenarios of climate change.

Management Implications and Conclusions

Wetland restoration approaches for water quality improvements tend to look backwards, seeking the re-establishment of historical conditions. It is generally assumed that restoring wetland hydrology will result in high rates of N removal (primarily through denitrification) and P retention (sorption onto sediments and biotic uptake) that were lost when wetlands were drained. Our results, however, indicate that a more nuanced view of the recovery of biogeochemical function in restored wetlands is needed (Figure 10). Although flooded conditions can lead to the removal of NO_3–N, these same conditions can lead to export of NH_4–N, DON, and TP (Figure 10). Potential tradeoffs due to mitigation strategies have been called *diffuse pollution swapping*, where a measure introduced to reduce one pollutant can increase the release of a different pollutant (Stevens and Quinton 2009). Increases in NH_4–N can fuel algal blooms in local and downstream ecosystems; the impacts of increased DON and TP concentrations will depend on how bioavailable they are to biota (Wiegner and Seitzinger 2004). To fully assess the net gains from wetland restoration, an increased understanding of the various forms of N and P will be required. Recent calls for large-scale wetland restoration in the Mississippi basin (Mitsch and others 2001) and the Chesapeake Bay (Chesapeake Bay Program 2000) to reduce NO_3–N loads to coastal systems should consider the potential negative consequences of increased export of organic and particulate forms of N or P.

In the first 2 years after reinstating the precipitation and wind-driven hydrology, RW retained or transformed 97% of NO_3–N, 32% of TDN, 25% of NH_4–N, and 53% of SRP delivered from inflows and precipitation, while exporting 20% more DON and 13% more TP than inputs. One of the most important ecosystem services provided by this very large restored wetland is its capacity to attenuate and retain pulsed nutrient inputs. As precipitation regimes become more extreme and land-use change routes stormflows more efficiently into wetland and riverine ecosystems, a larger proportion of annual nutrient exports are expected to occur during peak flows (Royer and others 2006; Bernhardt and others 2008; Shields and others 2008). Our results suggest that for wetland restoration to be an efficient tool in the amelioration of coastal eutrophication under an accelerating hydrologic cycle, it will become increasingly important to restore systems with sufficient residence times to attenuate large pulses of nutrients delivered during stormflows.

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