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Nitrogen dynamics of a large-scale constructed wetland used to remove excess nitrogen from eutrophic lake water

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ABSTRACT

Nitrogen loss from landscapes affects downstream freshwater and marine systems. Constructed wetlands, which are ecologically-engineered systems, can mitigate nitrogen loss from various landscape types. We examined the efficacy of a large-scale constructed wetland (the marsh flow-way at Lake Apopka) to remove particulate nitrogen and other nitrogen species from eutrophic lake water. During the first seven years of operation, the hydraulic loading rate to the flow-way was high (mean ± SD; 34 ± 10 m yr^{-1}). Inflow concentrations of total nitrogen (TN) were also high and variable (mean ± SD; 4 ± 1 mg L^{-1}). Both hydraulic loading rate and concentration of N species were affected by variations in lake level. Inflow nitrogen was mostly in particulate forms (particulate organic nitrogen [PON] was 58% of TN). Subsequently, the dominant biogeochemical mechanism for nitrogen removal was sedimentation of PON. Average annual removal rates were 56 g PON m^{-2} yr^{-1}, while TN and total Kjeldahl nitrogen removal rates were similar, about 30 g N m^{-2} yr^{-1}. Nitrogen removal was seasonally dependent, with greatest removal rates occurring during cool periods (October through May). Although the marsh flow-way showed substantial removal of particulate N, the wetland released dissolved inorganic N fractions. Release of ammonium (NH_4) averaged 21 g N m^{-2} yr^{-1}. Release of NH_4 was greatest during warm periods (June through September), which was probably related to increased water temperature, associated increased decomposition, and decreased dissolved oxygen content of wetland waters. Long-term first order rate constants (k) for removal were also high. Annual k values for PON were 85 m yr^{-1}. Managing the marsh flow-way to maximize particulate N removal in the long-term, while mitigating the release of inorganic N during warm periods, will require dynamic wetland management.

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1. Introduction

Discharge of excess nitrogen (N) affects both fresh and marine water quality (Echols and Vitousek, 2012; Howarth et al., 2012). For example, surface and subsurface water runoff from agricultural lands can contain excessive amounts of nutrients and pesticides (Sharpley, 1999; Power, 2010). At a larger scale, N loss from watersheds can contribute to algal blooms, hypoxia, and ecological damage to receiving aquatic systems (Howarth et al., 2011). Excess ammonia, in the unionized form, can also be toxic to fish and invertebrates at concentrations greater than 0.2 mg N L^{-1} (Kadlec and Wallace, 2009).

For many years, management practices within watersheds have tried to (1) control N loss at sources and (2) mitigate N losses downstream by restoring and enhancing landscape hydrology and increasing landscape nutrient storage. In combination with source control practices, ecologically engineered systems can mitigate N losses downstream. Ecologically-engineered systems to mitigate N loss and enhance water and ecosystem quality include natural wetlands (Kadlec, 2009), constructed wetlands (Braskerud et al., 2005; Blankenberg et al., 2006; Maynard et al., 2009), ponds and reservoirs (David et al., 2006; Hunt et al., 2006; Collins et al., 2010), denitrification walls (Schipper et al., 2010), wastewater applications to pasture and croplands (Healy et al., 2007), and within-river

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restoration (Toth, 1993; Bernhardt et al., 2005). Approaches can differ due to the type of N loss scenario, to site-specific characteristics like climate, topography, soils, hydrology, and to the availability of resources. However, one common characteristic across practices is that they adopt a process approach using sound ecological principles.

Wetlands contain a unique assemblage of water, vegetation, soils, and microbes because of their landscape position (Mitsch and Gosselink, 1993). These ecosystem components contribute to wetlands typically being net sinks for N, phosphorus (P) and carbon (C) (Brenner et al., 2001; Kadlec, 2005b; Poach et al., 2007; Kayranli et al., 2010). However, both natural and constructed wetlands can also be sources of N, P and C. For example, systems can release C and nutrients upon reflooding after water level drawdown (Olila et al., 1997; Pant and Reddy, 2001). The net effect of whether a system is typically a sink or a source of nutrients often depends upon the dominant biogeochemical processes, the environmental conditions that influence process rates, and the associated timescale. If wetland managers know the dominant biogeochemical processes that govern both retention and release, they can manage their system to provide suitable conditions for desired processes. For example, they can manipulate water level, flow, depth, and duration to provide aerobic conditions for nitrification and anaerobic conditions to facilitate denitrification. In practice, this could include a very shallow or intermittently flooded emergent marsh or wet prairie, followed by a deeper marsh zone.

The transformation and translocation of N in wetlands can occur via various processes, which include nitrification-denitrification, mineralization, sedimentation, resuspension, diffusion, sorption, assimilation, and volatilization (Craft, 1997; White and Reddy, 2003; Ardón et al., 2010; DeMeester and de Richter, 2010; Kadlec, 2010). Environmental factors that influence process rates include but are not limited to, temperature, pH, soil organic matter, water depth, water flow rate, alkalinity, dissolved oxygen, concentration gradients, and biota (Picard et al., 2005; Kadlec, 2010; Toth, 2010). There are numerous studies that report nitrification/denitrification dynamics and N budgets within wetlands from around the world (Kadlec, 2005a; Reinhardt et al., 2006; Hernandez and Mitsch, 2007; Kadlec et al., 2010; Watson et al., 2010; Tao et al., 2012; Martin et al., 2013). To our knowledge, few studies report the effectiveness of constructed wetlands to remove particulate N (Braskerud, 2002). We address this gap of knowledge with our paper. In addition, we demonstrate that particulate removal can be the primary N-removal mechanism for constructed wetlands treating eutrophic lake water. We also describe other N transformations that occur as water passes through the large-scale constructed wetland (the marsh flow-way) at Lake Apopka, FL.

The overall restoration program for Lake Apopka, implemented by the St. Johns River Water Management District, focuses on cost-effective methods to reduce external nutrient loading and remove nutrients already in the lake (Dean and Lowe, 1998; Hoge et al., 2003). A component of this programmatic approach is to use the marsh flow-way to remove nutrients from lake water. The marsh flow-way was designed to accelerate Lake Apopka's response to reduced loading by removing the nutrient inventory already in the lake (Lowe et al., 1992; Coveney et al., 2002). The primary goal was to remove phosphorus (P), but the effectiveness for N removal is an important benefit.

The marsh flow-way differs in some ways from conventional surface-flow constructed wetland systems. The aim of the system is to reduce the inventory of nutrients in the lake rather than attain specific outflow criteria. Therefore, it is more important that the system removes N at a high rate, rather than achieving high percent removal. The system also has varying hydraulic inflows and varying inflow organic N concentrations. This variability is a function of the rise and fall of lake water levels and the associated variation in lake nutrient concentrations. In addition, the native organic soils at the marsh flow-way have a legacy of soil-stored nutrients due to past agricultural land practices (Dunne et al., 2012), which also has management implications for the flow-way.

The objectives of our manuscript were to: (1) determine the efficacy of the marsh flow-way to remove N, specifically particulate nitrogen, from incoming eutrophic lake water; (2) examine the different N species in lake water and how they change via wetland treatment; and (3) investigate the effect of environmental factors like temperature, pH, dissolved oxygen and seasonality on N transformation and removal.

2. Materials and methods

2.1. Site description and maintenance

The marsh flow-way is located along the northwest shore of Lake Apopka (Fig. 1). We described site details in a previous publication (Dunne et al., 2012). Briefly, the marsh flow-way consists of four independently operated treatment cells (Fig. 1). Lake water enters each cell through an array of culverts. Water passes through each cell and flows out over riser boards and through culverts. All treated water collects in a pump basin. Electric pumps return water back to the lake. About $80 \pm 15\%$ (mean $\pm$ SD) of the treated water returns to the lake, while the remainder flows downstream (north) through the Apopka-Beauclair Canal (Fig. 1).

Site maintenance was undertaken between years 2007 and 2009. Between March 2007 and January 2008, we drew down B cell water levels, mowed vegetation removed sediment from lateral ditches, reflooded and used alum injection post reflooding for a few months. During B cell maintenance, C cells did not treat lake water for about three months. The C cells also underwent maintenance between November 2008 and October 2009. During this period, cells were drawn down, vegetation mowed removed sediment from lateral ditches, cells reflooded with lake water and alum was injected post the reflood period. We also constructed small finger dikes in C2 to mitigate a hydraulic short circuit.

2.2. Sample collection

During the operating period covered herein (November 2003 through December 2010), weekly water samples were collected from the main inflow and outflow of the system. The main inflow is a feeder canal from the lake and the main outflow is located in the pump basin. At both locations, a Van Dorn horizontal water sampler (Wildco, Yulee, FL) was placed to mid-water depth and water samples were collected. Samples were filtered (if required) in the field to 0.45 μm and preserved (if required). Samples were placed on ice and stored at 4 °C prior to laboratory analyses. During the maintenance periods, treatment flows were stopped, and water samples were not collected from most of the marsh flow-way.

2.3. Laboratory analyses and nitrogen forms

We measured total Kjeldahl N (TKN) on preserved (H_2SO_4) water samples using a Perstorp Autoanalyzer, according to EPA method 351.2. Ammonium was also measured in acid preserved water samples using a LaChat Quickchem AE flow injection analyzer (EPA method 350.1). Both nitrate and nitrite (NO_x) were

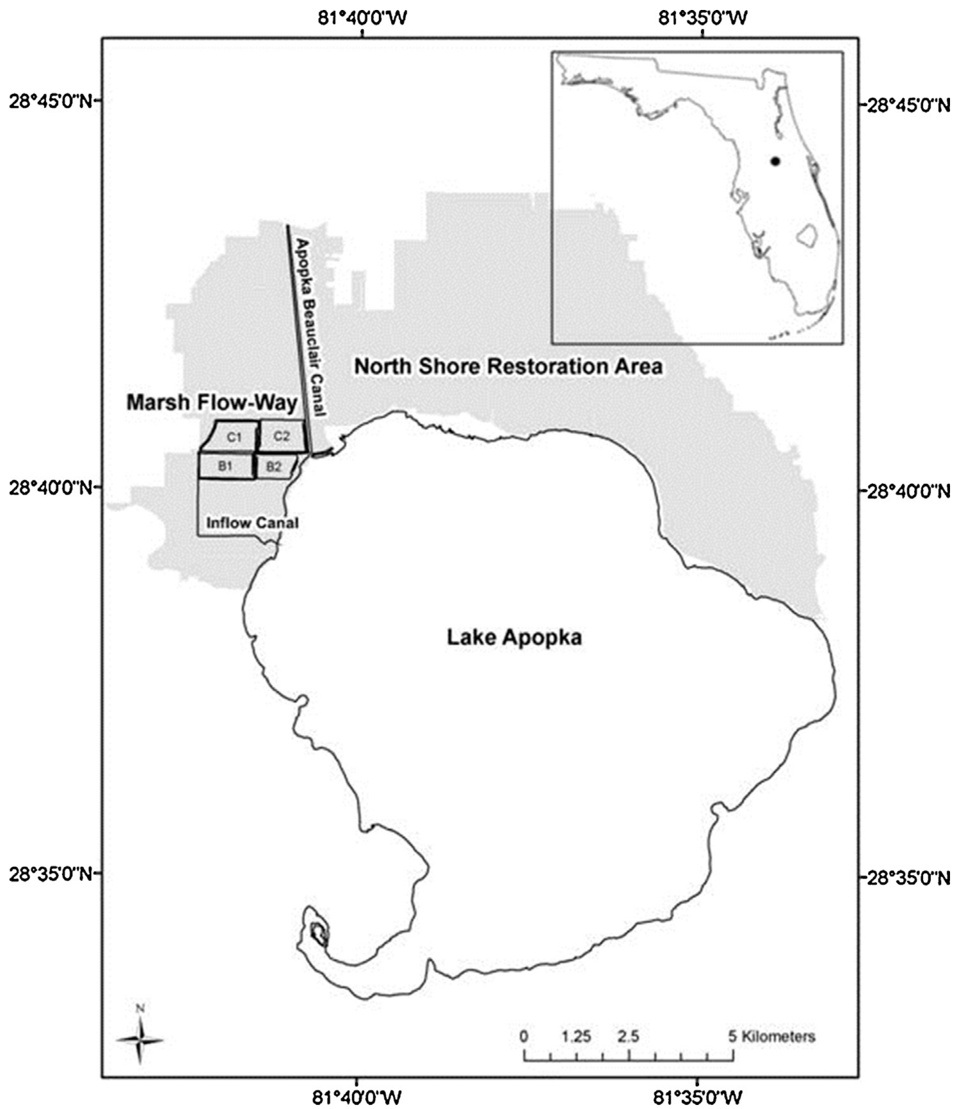

Fig. 1. Inset map shows location of Lake Apopka Basin (highlighted in black) in north central Florida. Map of the marsh flow-way, which is located on the northwest shore of Lake Apopka. The four treatment cells illustrated as B1, B2, C1 and C2. The main inflow canal from the lake is south of the cells. Outflows can return to the lake or flow downstream via the Apopka-Beauclair Canal. The area highlighted in grey, north of the lake is the North Shore Restoration Area.

measured on a chilled water sample using a Seal QuAAtro segmented flow analyzer according to EPA method 353.2. Other nitrogen forms were calculated and not analytically measured. These included total nitrogen (TN, the sum of TKN and NO_x), particulate organic N (PON, the difference between TKN-total and TKN-dissolved), and dissolved organic N (DON, the difference between TKN-dissolved and NH_4). Dissolved inorganic nitrogen (DIN) was calculated as the sum of NH_4 and NO_x.

2.4. Calculations, data analyses and statistics

We used data from normal operating periods, that is, periods when the marsh flow-way was treating Lake Apopka water. The hydraulic loading rate (HLR) and the nominal hydraulic residence time were calculated as described by Dunne et al. (2012). To calculate daily loads into and out of the system, daily measured and estimated flows were multiplied by daily concentrations. Daily concentrations were determined by linear interpolation between weekly measurements. Mass removal rates, percent mass removal and mass areal removal rates were also calculated. In addition, we calculated monthly flow-weighted concentrations by dividing the monthly mass totals by monthly flow volume totals into and out of the system.

To characterize long-term TN and PON removal, we calculated the rate constant (k) using a combined removal model. Kadlec and Wallace (2009) define this model as:

$$\frac{C_o - C^*}{C_i - C^*} = \frac{1}{(1 + (k/Pq))^P}$$

where, C_o is the outlet concentration, mg L^{-1}, C_i the inlet concentration, mg L^{-1}, C^* the background concentration, mg L^{-1}, k the modified first-order constant, m yr^{-1}, P the number of tanks in series, q the hydraulic loading rate, m yr^{-1}.

We used a background TN concentration (C^*) of 1.5 mg L^{-1} and three tanks in series (Kadlec and Wallace, 2009). For PON, we also used three tanks in series. From observing our PON data, it seemed that a C^* concentration of 0.05 mg L^{-1} was appropriate, as outflow concentrations of PON tended towards a low concentration limit of about 0.05 mg N L^{-1} (see Fig. 4). We used this value when estimating the rate constant (k) for long-term removal. This

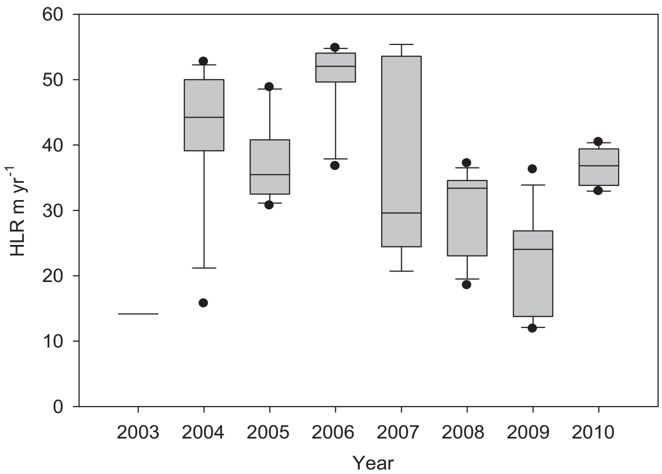

Fig. 2. Yearly box plots of the hydraulic loading rate (HLR) to the marsh flow-way. The annual box plots were generated from monthly median values. The boundary of the box closest to zero is the 25th percentile, median is the line within the box, and the boundary of the box farthest from zero, is the 75th percentile. Error bars above and below the box indicate the 90th and 10th percentiles. Solid circles are data outliers. The operating period for the flow-way was from November 23, 2003 through December 31, 2010.

concentration is one-half the minimum PON outflow concentration that we observed during the seven years of operation.

We found weak temperature dependence for TN k values ($r = -0.2$; $p = 0.03$; $n = 83$; data not shown) and no temperature dependence for PON k values. Due to the lack of strong temperature dependence with k, we do not present temperature dependent k values.

In many places of the text and tables, we report summary statistics like mean, standard deviation (SD), coefficient of variation (CV), medians, quartiles, and correlations. Fig. 2 summarizes annual data using a range of summary statistics illustrated in box plot format. See figure legend for further detail. Other figures include linear regressions and correlations (Pearson product moment correlation) that report R^2 and r-values, respectively. We also show the statistical significance of these relationships at the $p < 0.05$ level. We used Minitab (Version 16) for all statistical analyses.

3. Results and discussion

3.1. Hydraulic loading rate and residence times

We reported the HLRs from November 2003 through March 2007 in detail previously (Dunne et al., 2012). Briefly, median annual HLR generally increased from 2003 through 2006 (Fig. 2). During 2007, the HLR decreased below 30 m yr^{-1} and was quite variable. Both the decrease and the associated variability in 2007 were mostly due to maintenance of the B cells between March and December. Maintenance included a range of activities: water level drawdown, vegetation mowing, lateral ditch cleaning, and reflooding. We also used alum injection for about two to three months to mitigate within-cell soluble P release upon reflooding. We hypothesize that the release was from historically P laden soils along with the decomposition of senescing vegetation (Dunne et al., 2012).

For years 2008 through 2010, the HLR to the flow-way was typically less than previous years, with annual medians ranging between 22 and 35 m yr^{-1}. During 2009, we also undertook maintenance in the C cells due to the development of hydraulic short-circuits – the legacy of initial site conditions incompletely resolved during construction. During this maintenance period, only the B cells treated incoming lake water.

Relative to flows, which varied because one or more cells were offline, nominal residence times changed little from year to year; values ranged between 3 and 5 days (Table 1). Annual median water depths were also stable, ranging between 40 cm (2010) up to 57 cm (2006). Deepest water depths generally corresponded with high flow years. Inflow volume of Lake Apopka water to the flow-way was the dominant hydrologic loading variable. It was, on average, 30 times greater than rainfall inputs (Table 1).

3.2. Concentrations

Inflow concentrations of TN, TKN and PON were typically greater than outflow concentrations, with greatest differences occurring between PON inflow and PON outflow (Fig. 3). This indicates greater retention of PON relative to the two other fractions. Particulate organic N concentrations were slightly less than TN and TKN concentrations, with the latter two being similar. Both inflow and outflow concentrations of TN, TKN, and PON increased sharply between 2008 and 2009. During this period, water levels in the lake were extremely low; Lake Apopka lost up to 52% of its volume (Coveney, 2009). We found an inverse relationship between inflow TN concentrations and HLR to the marsh flow-way (Spearman rank correlation $r = -0.25$; $p = 0.02$). This corresponded to higher in-lake nutrient concentrations during low lake water levels. During these periods, it becomes difficult, due to infrastructure, to operate at high HLR to the marsh flow-way.

For the seven years of operation, monthly median inflow TN concentrations ranged between 2.7 mg L^{-1} and 8.1 mg L^{-1}, whereas outflow concentrations ranged between 2.0 mg L^{-1} and 6.4 mg L^{-1}. Dissolved organic N concentrations and NH_4 concentrations were typically greater in outflow waters than inflow waters (Fig. 3). This

Table 1
Annual hydrologic parameters for the marsh flow-way constructed wetland at Lake Apopka, FL. Water depth and residence times are yearly medians calculated from daily values. Rainfall and evapotranspiration (ET) are yearly totals, calculated from daily values. Inflow and outflow are also yearly totals. Flow totals (m^3) were calculated on a daily basis, summed for the year, and then divided by the treatment cell area (276 ha).

Parameter	Years							
	2003[a]	2004	2005	2006	2007	2008	2009	2010
Water depth, m	0.47	0.52	0.56	0.57	0.50	0.45	0.52	0.40
Residence time, days	8	3	4	3	3	4	5	3
Inputs								
Rainfall, m	0.01	1.2	1.26	1.02	1.03	0.96	1.15	0.88
Inflow, m	1.6	41.5	36.7	49.4	25.1	30.2	21.4	36.7
Outputs								
ET, m	0.06	1.11	1.11	1.11	1.13	1.13	1.13	1.18
Outflow, m	1.6	41.6	40.1	49.3	25.0	30.1	21.5	36.4

[a] 2003 is a partial year (November 23 through December 31). All other years are full calendar years.

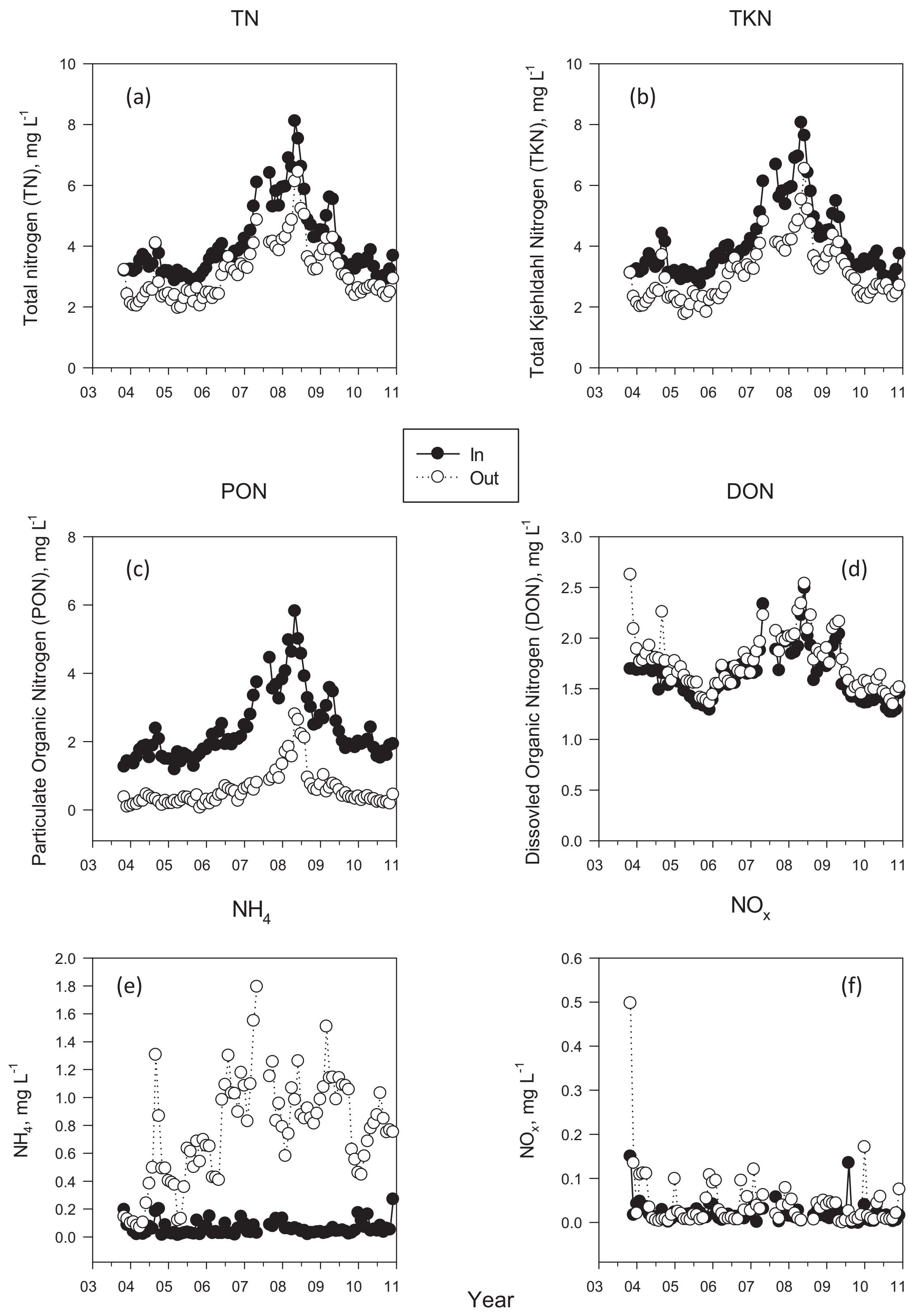

Fig. 3. Plots of flow-weighted inlet and outlet concentrations of (a) total nitrogen (TN), (b) total Kjeldahl nitrogen (TKN), (c) particulate organic nitrogen (PON), (d) dissolved organic nitrogen (DON), (e) ammonium (NH_4), and (f) nitrate + nitrite (NO_x) through time. Monitoring period was from November 2003 (03) through December 2010 (10). Solid circles represent monthly inlet concentrations and open circles represent outlet concentrations.

indicates a net release of dissolved organic and inorganic N forms through time. Similar to changes in TN, TKN, and PON; concentrations of DON also increased between 2008 and 2009.

Ammonium (NH_4) concentrations in outflows increased during summer periods (Fig. 3). This seasonal pattern of release suggests that the biological transformation rate of organic nitrogen to NH_4 (ammonification) increased during warmer summer periods. Ammonification can occur via microbial breakdown of organic tissues containing amino acids (Kadlec and Knight, 1996) with optimum pH ranges between 6.5 and 8.5 (Reddy and Patrick, 1984). Other contributing factors include dissolved oxygen and water temperature, which we discuss later.

Since NH_3 can be toxic to biological organisms, we also calculated NH_3 concentrations based upon NH_4 concentrations, pH and temperature in marsh flow-way treated waters (data not shown). For the seven years of operation, NH_3 concentrations were well below 0.2 mg L^{-1}. For most years, monthly median values were below 0.01 mg NH_3 L^{-1} and undoubtedly were lower in localized mixing zones at the outflow where wetland treated water mixed with lake water.

Inflow and outflow NO_x concentrations were typically similar, and concentrations were typically low (<0.2 mg N L^{-1}). However, during winter periods there was a small peak in NO_x concentrations (Fig. 3). This was probably related to decreases in winter water temperature and an increase in DO concentrations relative to warmer summer periods (see later section for further discussion).

Inflow and outflow concentrations of TN, TKN, and DON were positively related ($R^2 > 0.72$; $p < 0.001$; Fig. 4). Inflow and outflow concentrations of PON were also related, but in a non-linear fashion. For example, at inflow PON concentrations of about 4 mg N L^{-1}, outflow concentrations increased abruptly (Fig. 4). No relationships existed between inflow and outflow concentrations for inorganic fractions (NH_4 and NO_x).

3.3. Mass areal loadings, removals and performance patterns

There were statistically significant, but much weaker (relative to concentration data), correlations between inlet mass areal loading rate (g N m^{-2} yr^{-1}) and outflow concentrations of TN, TKN, and PON ($r > 0.29$; $p < 0.01$) (Fig. 5). Representing loading rate against outflow concentration data is a standard approach to illustrating wetland performance data (Kadlec and Wallace, 2009). The lack of strong relationships is not surprising, as we do not operate the marsh flow-way to attain a certain outflow concentration; rather, we operate to maximize nutrient removal rate. No relationships between loading rate and outflow concentration existed for the inorganic fractions (NH_4 and NO_x).

Total N and TKN removal rates (g N m^{-2} yr^{-1}) were similar during the seven years of operation, with greatest removal rates occurring during 2007 and 2008 (Table 2); years that had both high HLR and high inflow concentrations. Greater loading rates provide greater opportunities for increased nutrient removal (Reddy et al., 1999). Generally, there is a similar pattern of storage for TN and TKN in wetland ecosystem components like water, detritus, soils, and biological organisms (Kadlec and Knight, 1996).

Annual PON removal rate was much greater than both TN and TKN removal rates because dissolved N fractions (DON, NH_4 and NO_x) were typically released from the marsh flow-way (Table 2), with the magnitude of retention/release being seasonally dependent. Thus the areal removal rates for TN and TKN decreased during the warm months (June through September) and increased during the cooler months (October through May) (Fig. 6a). Removal rates were variable across months; however, the magnitude of that variability was consistent through time. The coefficient of variation (CV) for both TN and TKN removal rates across months was about 55%. There was an increase in removal rate variability during the later warm months (August and September); CV values increased up to 85%.

The average percent mass removal for PON and TN during the seven years of operation was 78 ± 10% (mean ± SD) and 24 ± 9%, respectively. Total N percent removal decreased much more during warm months, relative to PON (Fig. 6b). During the warmer summer/fall months, TN percent removal was less than 20%. Associated with the summer decrease in percent removal, the CV increased up to 66%. However, the CV associated with PON percent removal, remained somewhat consistent, averaging 13%. This suggests that the process governing PON removal (i.e. sedimentation) was not affected by the time of year, whereas additional transformation processes like ammonification, nitrification-denitrification, and assimilation may have affected TN percent removal.

Our results indicate that the dominant mechanism for N removal within the marsh flow-way constructed wetland was sedimentation of PON. Annual removal rates were 31 ± 10 g TN m^{-2} yr^{-1} (mean ± SD). The dominance of PON in inflows-coupled with high HLRs-provided high inlet loading rates, which in turn, provided an opportunity for high removal rates. Short retention times (3–5 days) limited the transformation rate of settled particulates relative to the settling rate. Thus, it appears that the settling of particles overwhelmed competing biogeochemical processes such as decomposition and release of dissolved N fractions. Braskerud (2002) reported that various constructed wetlands in an agricultural landscape removed between 50 and 285 g TN m^{-2} yr^{-1}, most of which was retained via sedimentation. The high removal rates were stimulated by the high hydraulic loading rates. Reinhardt et al. (2006) undertook an isotope mass balance approach to quantify flux of N in a small constructed wetland. In contrast to the previous study, they found that denitrification accounted for 94% of the removal, while only 6% accumulated in sediments. The TN removal rate reported was 45 g m^{-2} yr^{-1}. Many other studies report that the dominant N removal pathway in wetlands is denitrifcation (Verhoeven et al., 2006; Batson et al., 2012; Kadlec, 2012) with both denitrification and sedimentation related to TN loading rate (Saunders and Kalff, 2001). The relative importance of one biogeochemical process to remove N, depends upon a range of components that include characteristics of the inflowing water (Kadlec, 2012), hydrologic characteristics of the site (Brinson et al., 1984), environmental factors like the O_2 content of wetland water and soil (Batson et al., 2012), along with wetland management regimes.

Annual areal removal rates of DON and NO_x were typically negligible or slightly negative indicating N release (Table 2). However, we did find that net production of NO_x was greater during cooler periods, as NO_x concentrations tended to increase in outflows (Fig. 3). This is not surprising due to the temperature-dependent solubility of oxygen (Kadlec and Reddy, 2001); DO concentrations tended to increase during cool periods and decrease in warm periods. In cool periods, monthly DO concentrations ranged between 3 and 5 mg O_2 L^{-1}, while during warm periods, monthly concentrations ranged between 1 and 2 mg L^{-1}. As the marsh flow-way typically released NH_4, there probably was an ample supply of ammonium for nitrification to occur during cool periods; however, the rate of nitrate production may have been limited during warm periods due to low DO concentrations.

Ammonium release rates (denoted as a negative) were greatest in 2006 and 2007 and least in 2004 (Table 2). As the marsh flow-way retained most of its nitrogen in particulate organic fractions (Table 2), there probably was an ample supply of organic nitrogen for ammonification/mineralization to occur; thereby, supplying NH_4 to the water column. Further, organic N mineralization

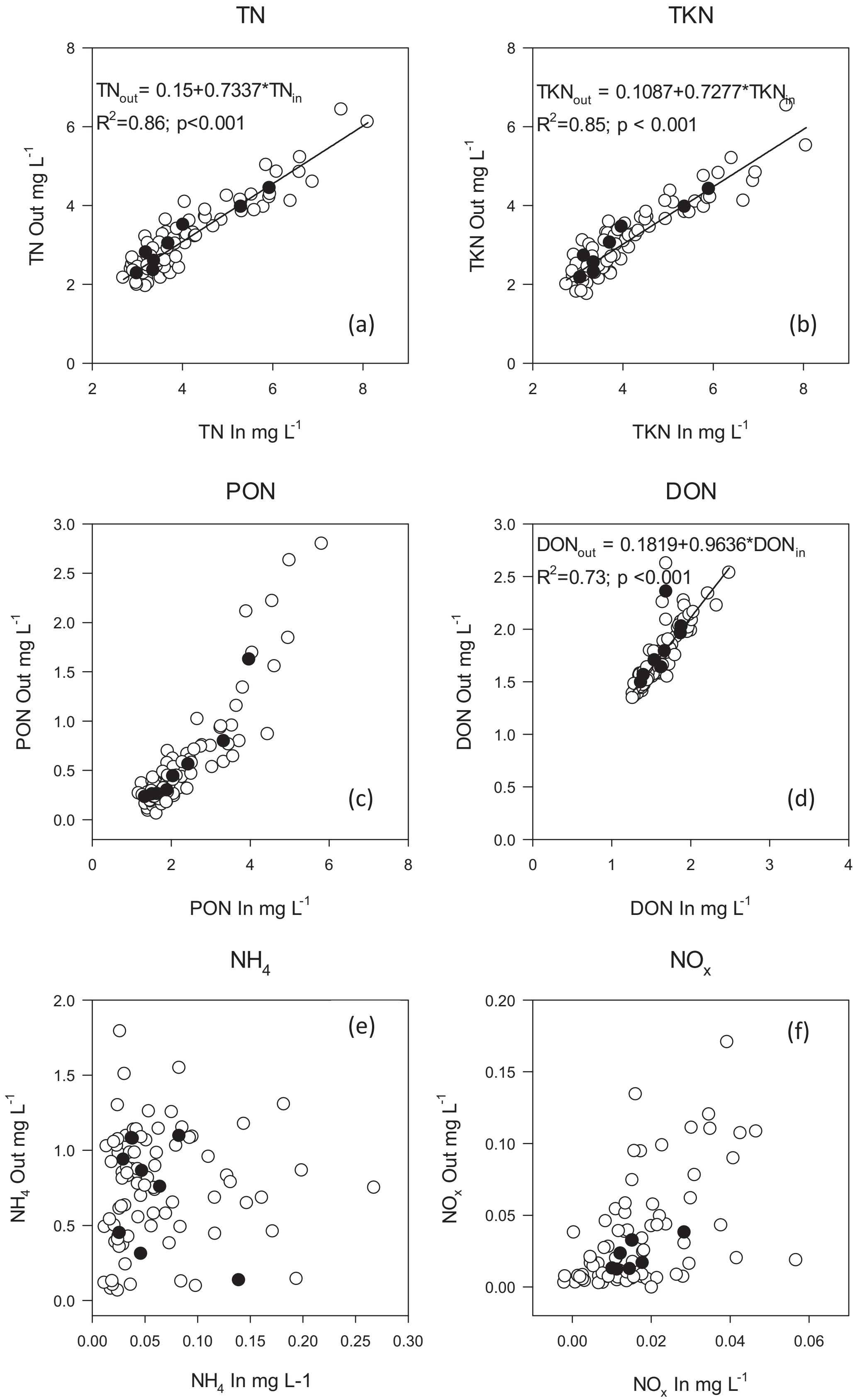

Fig. 4. Plots of (a) TN, (b) TKN, (c) PON, (d) DON, (e) NH_4, and (f) NO_x flow-weighted concentrations into and out of the marsh flow-way between November 2003 and December 2010. Open circles represent monthly medians and solid circles are yearly medians. If relationships between concentrations were significant ($p < 0.05$), we reported linear regression equations and goodness of fit (R^2).

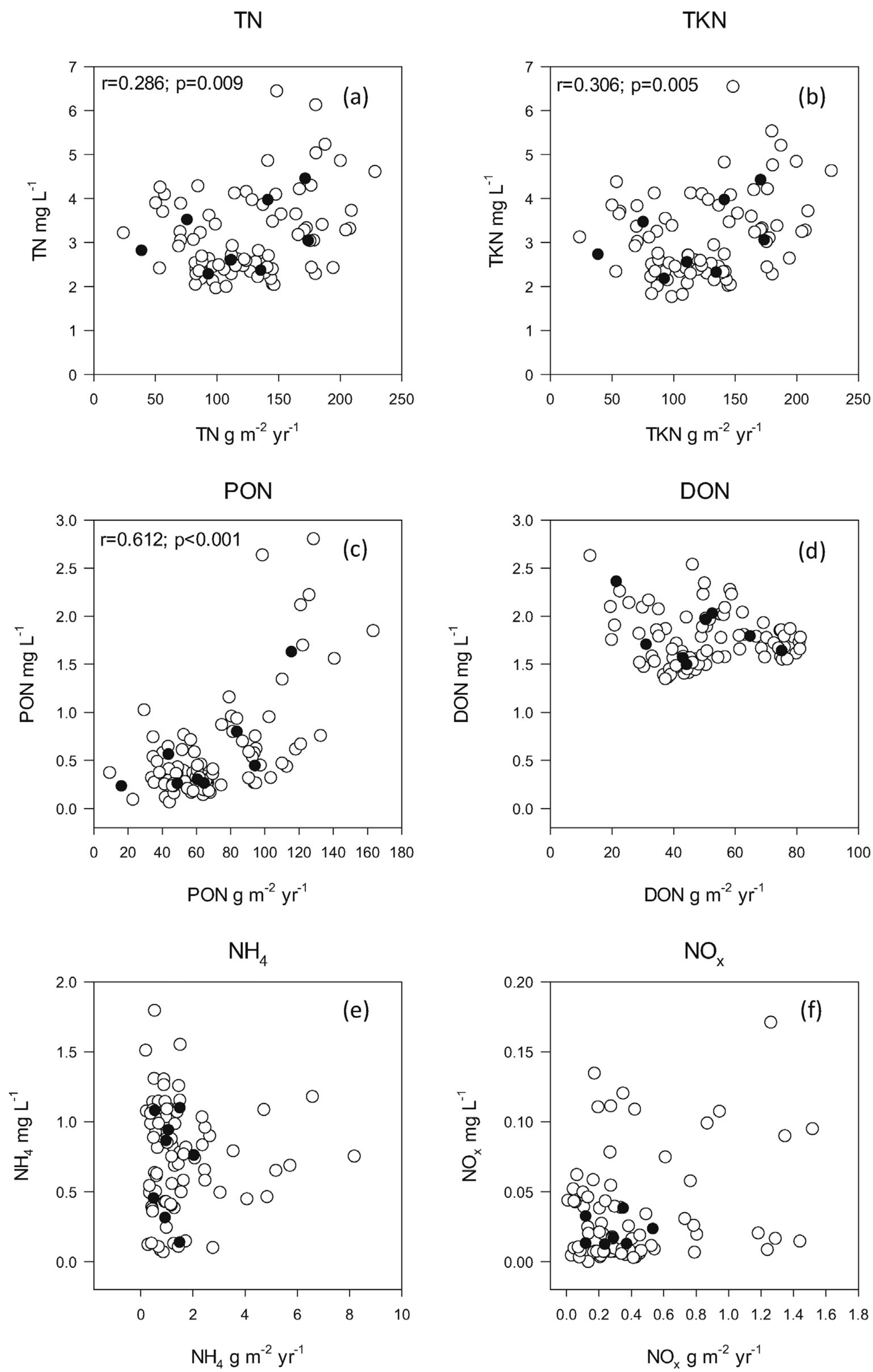

Fig. 5. Plots of (a) TN, (b) TKN, (c) PON, (d) DON, (e) NH_4, and (f) NO_x mass areal inlet loading rate against outlet flow-weighted concentrations. We show correlation coefficients (r) and their significance ($p < 0.05$). Open circle values are monthly medians and solid circles are yearly medians. Monitoring period was from November 2003 through December 2010.

can increase in wetland soils that are P loaded (White and Reddy, 2000), and the marsh flow-way has a P legacy in soils due to the history of farming. Ammonium release rates were seasonally dependent, with outflow concentrations typically increasing during warm periods, when temperature was high and DO was low (Figs. 3 and 7). During warm periods, water temperature ranged between 25 and 29 °C, whereas during cool months, outflow water temperature ranged between 15 and 20 °C. Average monthly water temperatures for the complete operating period was 22 ± 5 °C (mean ± SD). As water passed through the marsh flow-way, water temperature typically decreased by about 1–2 °C (data not shown).

Table 2
Yearly nitrogen fractions (total nitrogen [TN], total Kjeldahl nitrogen [TKN], particulate organic nitrogen [PON], dissolved organic nitrogen [DON], ammonium [NH_4], and nitrate + nitrite [NO_x]) that were removed or released from the marsh flow-way constructed wetland at Lake Apopka. Yearly values were calculated from monthly medians. Yearly medians precede values in brackets. Values in brackets are the first and third quartiles. Year 2003 is a partial year (November 23 through December 31). All other years are full calendar years through December 31, 2010. *Data was not available.

Year	TN	TKN	PON	DON	NH_4	NO_x
	$g\,m^{-2}\,yr^{-1}$					
2003	7 (*, *)	8 (*, *)	14 (*, *)	−5 (*, *)	0 (*, *)	−0.4 (*, *)
2004	37 (31, 54)	38 (30, 54)	56 (41, 57)	−3.4 (−4.7, −0.1)	−9 (−17, −3)	0 (−0.3, 0.1)
2005	23 (12, 36)	25 (13, 37)	41 (35, 47)	−3.4 (−5.4, −1.5)	−17 (−19, −10)	−0.5 (−1.5, 0)
2006	32 (20, 55)	35 (20, 55)	72 (62, 82)	−2 (−4.6, 0.6)	−39 (−48, −18)	−0.3 (−1.2, 0.1)
2007	39 (34, 44)	40 (34, 45)	63 (61, 92)	−0.7 (−3.4, 0.3)	−35 (−41, −22)	−0.3 (−0.4, 0.1)
2008	42 (28, 62)	42 (28, 61)	70 (47, 81)	−2.6 (−5.4, 0.4)	−21 (−24, −18)	−0.1 (−0.3, 0.1)
2009	12 (10, 21)	13 (10, 21)	36 (29, 39)	−2 (−3.5, −1.3)	−17 (−22, −13)	−0.2 (−0.4, −0.1)
2010	25 (19, 34)	28 (19, 37)	51 (41, 57)	−5.1 (−5.9, −2)	−21 (−25, −15)	−0.5 (−3.5, −0.1)

As outflow water temperature increased, the relative proportion of NH_4 to DIN concentrations increased exponentially (Fig. 8). Between water temperatures of 25 and 30 °C, NH_4 constituted more than 95% of DIN. Conversely, as water temperatures decreased, the relative proportion of NO_x increased; however, even at relatively low water temperatures (~10 °C) NO_x was never the dominant DIN fraction in marsh flow-way treated water.

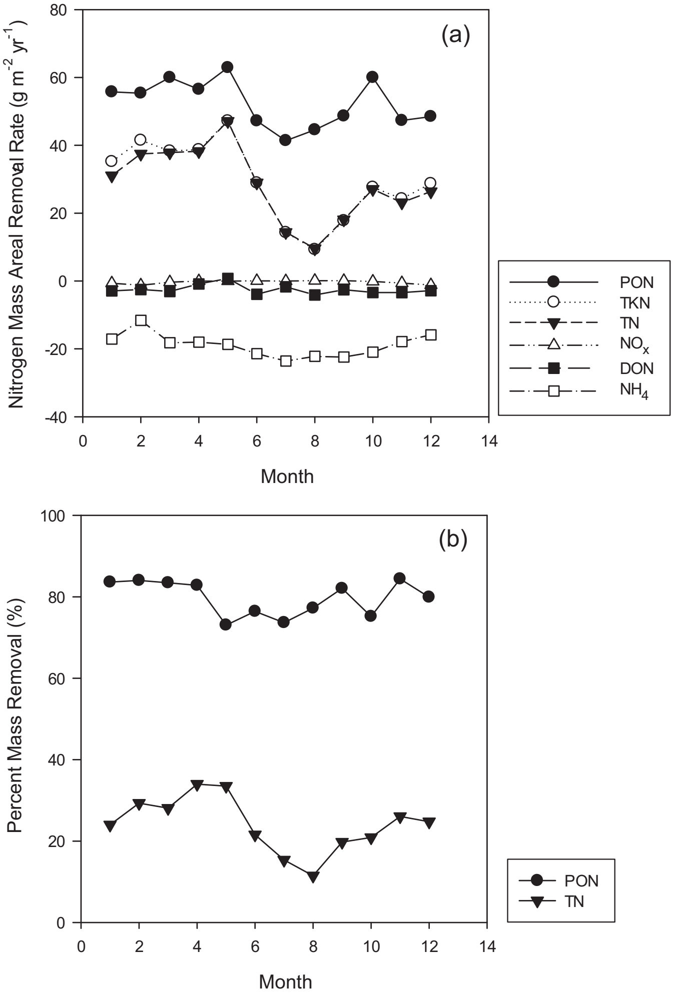

Fig. 6. (a) Plot of TN, TKN, PON, DON, NH_4, and NO_x monthly median mass areal removal rates ($g\,m^{-2}\,yr^{-1}$) for all months during the seven-year operating period (November 2003 through December 2010). (b) Percent mass removal of TN and PON during the same period. Values are also monthly medians.

3.4. *Rate constants*

For the period of marsh flow-way operation, annual *k* values for TN averaged $22 \pm 6\,m\,yr^{-1}$ (mean ± SD) and PON values were $85 \pm 23\,m\,yr^{-1}$ (mean ± SD). Variability among years was high due to greater *k* values occurring during the early years (2004–2006) of operation relative to the later years (2007–2010). This was probably associated with the system start-up phase during the first few years.

4. Implications for management

We manage the marsh flow-way, which is a free water surface wetland, to maximize nutrient removal rates. We do not operate and manage the system to achieve a specific outflow concentration criterion. This is a major management difference to many conventional constructed wetlands. Although empirical relationships between removal rates and loading may not be useful for wetland design (Kadlec and Wallace, 2009), we do find them important for operation and management. For example, at a given loading rate, how much removal might one expect? Since 2004, annual TN loading rates were high, with subsequent high removal rates (Fig. 9).

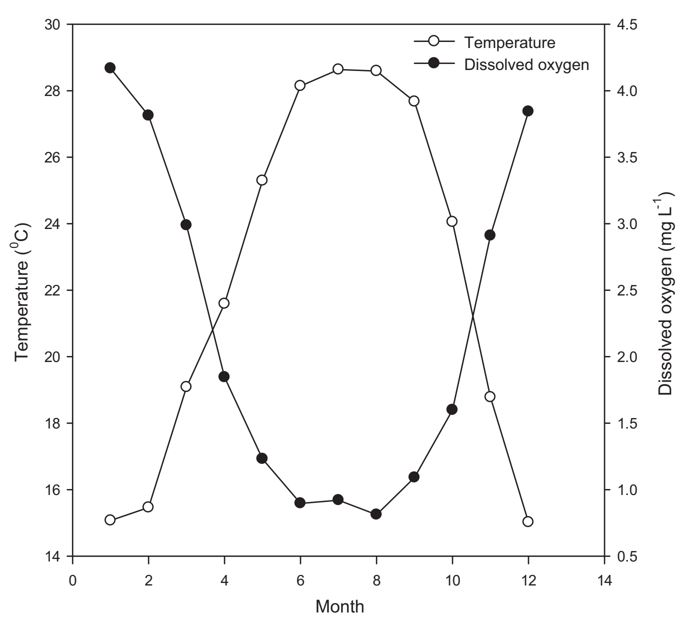

Fig. 7. Monthly median temperature (°C) and dissolved oxygen ($mg\,L^{-1}$) content of marsh flow-way treated water. We show all months during the seven-year operating period (November 2003 through December 2010).

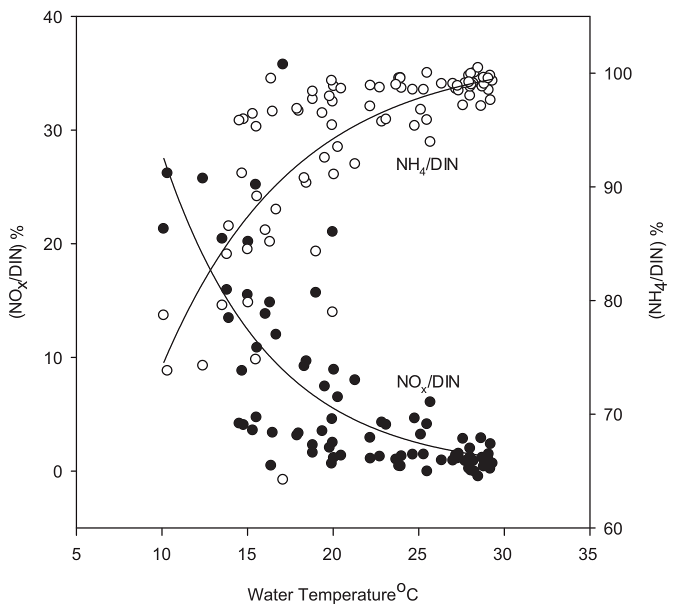

Fig. 8. Relationship between nitrate + nitrite (NO_x), and ammonium (NH_4) expressed as a percentage of dissolved inorganic nitrogen (DIN) and water temperature of marsh flow-way treated water. Values are monthly medians.

We generally found higher TN and PON removal during winter/spring periods relative to summer/fall periods. Using seasonal data in a dynamic operational context, we have increased cost-effectiveness for nutrient removal by shutting down cells that perform poorly during the summer. Further, increasing HLR during cooler periods, for example those periods when the system removes highest rates of TN and PON, may also contribute to increased N removal from eutrophic lake water.

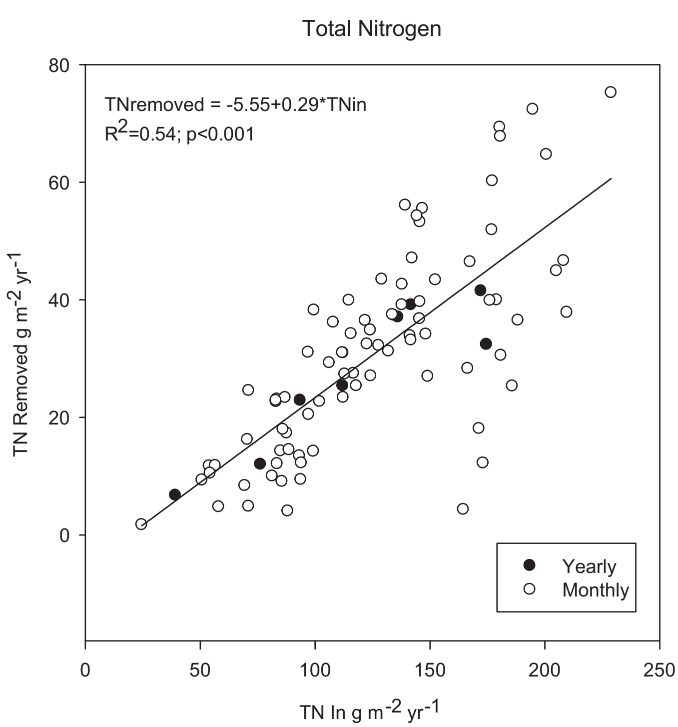

Fig. 9. Plot of total nitrogen (TN) inlet loading rate relative to areal removal rate ($g\,m^{-2}\,yr^{-1}$). Open circles represent monthly medians and solid circles are yearly medians.

Looking at constructed wetland performance and characteristics during the long-term by using multiple lines of evidence helps provide context and improve understanding of N removal performance and system characteristics at the operational wetland-scale. Tracking data (flows and water quality) through time and using these data to support dynamic management decisions was, and is, critical to sustaining present and future performance of a changing, ecologically-engineered system.

We expect lake nutrient concentrations to decline during the long-term because of reductions in watershed loading and through efforts to reduce nutrients in the lake. Lower concentrations of N in lake water have the potential to reduce loading rates; thereby, affecting constructed wetland removal rates. This presents both a challenge and an opportunity for wetland management. To respond to decreases in nutrient concentrations, we may consider seasonally increasing the HLR to sustain higher loadings and subsequently higher rates of nutrient removal.

5. Conclusions

Our results indicate that the marsh flow-way constructed wetland removed significant amounts of nitrogen, mostly particulate N, from eutrophic water. The dominant biogeochemical N removal process for this constructed wetland was sedimentation. Removal of N was affected by variation in lake water N concentrations, in addition to variable HLRs. Areal removal rates were linearly related to loading rates, and both TN and PON removal rates were greatest when incoming lake N concentrations and HLR were high. This increased the opportunity for N removal, which typically occurred during cooler periods.

There were significant relationships between inflow and outflow concentrations. The strongest relationships occurred between inflow/outflow concentrations of TN and PON fractions. Outflow concentrations of NH_4 were typically greater than inflow concentrations, with release being greatest during summer periods. This was probably a result of increased water temperature and a decrease in DO, with subsequent increases in ammonification.

The marsh flow-way, which is an ecologically-engineered system, continues to change through time. Rate constants for removal of TN and PON were greater in early years relative to later years. To sustain N removal performance into the future, dynamic management of the system is required, as it responds to both declining lake nutrient concentrations and within-system changes.

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