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Nutrient removal from eutrophic lake water by wetland filtration

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Abstract

Lake Apopka is a large (125 km^2), shallow (mean depth 1.6 m) lake in Florida, USA. The lake was made hypereutrophic by phosphorus loading from floodplain farms and has high levels of nutrients, phytoplankton (Chl *a* 80 μg l^{-1}), and suspended matter. The restoration plan developed by the St. Johns River Water Management District encompasses the biomanipulation concept in which the critical step for large shallow lakes is increasing the transparency of the water to allow the re-establishment of submerged macrophytes. Restoration includes operation of a treatment wetland, reduction in external P loading, harvest of fish, fluctuation of lake levels, and littoral planting. The District constructed a 2-km^2 pilot-scale treatment wetland to test nutrient-removal and hydraulic performance. Lake water was recirculated for 29 months, and the removal of suspended solids and particle-bound nutrients was assessed. Hydraulic loading rate varied from 6.5 to 65 m year^{-1} with a mean hydraulic residence time of about 7 days. The inflow contained 40–180 mg l^{-1} TSS, 80–380 μg l^{-1} TP (mostly particulate organic), and 3–9 mg l^{-1} TN (mostly dissolved and particulate organic). Overall, particulate matter was removed (> 90%) by the wetland, and soluble organic compounds were unaffected. Soluble inorganic compounds such as nitrate, ammonia, and soluble reactive phosphate (SRP) were low in the lake water but increased during passage through the wetland. Particulate matter at the outlet was enriched in both N (2-fold) and P (5-fold) compared to particles in the inflow. Mass removal efficiencies were 89–99 (TSS), 30–67 (TP), and 30–52% (TN), but efficiency fell when hydraulic short-circuiting occurred. First-order removal coefficients were 107 (TSS), 63 m year^{-1} (TP) and 98 m year^{-1} (particulate N). Areal particulate removal rates were 5.4 g dry matter m^{-2} day^{-1}, 0.18 g PON m^{-2} day^{-1}, and 0.006 g POP m^{-2} day^{-1}. The ratio of N:P removal was 28:1. Total sedimentation rate was 0.4 mm day^{-1} of very light matter (4.4 g dw l^{-1}). About 40% of the dry matter and nitrogen removed and about 80% of the phosphorus was found in the new sediments. Relative to the inflow of lake water, evapotranspiration (4.3%), seepage (2.6%), and rainfall (2.8%) were low. Major problems were initial leaching of SRP, but not ammonia, from native organic soils and vegetation when this former farmland was flooded; hydraulic short-circuiting via former drainage ditches; and low inflows under drought conditions. After 6 months SRP release declined, and initial SRP leaching could be prevented with soil treatment. Hydraulic short-circuiting occurred only after modifications were made. Low gravity flows were augmented

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with pumped inflows. With these improvements P-removal should increase from the measured 0.48 to at least 3 g P m^{-2} $year^{-1}$. Based on the pilot project results, the first phase of an improved 14-km^2 wetland filter has been constructed. This project should accelerate improvements in the water quality of Lake Apopka and, ultimately, create a new, large wildlife-rich marsh.

Keywords: Wetland; Treatment; Nutrient; Phosphorus; Nitrogen; Restoration; Removal; Lake Apopka; Florida; USA

1. Introduction

Lake Apopka is a large (125 km^2), shallow (mean depth 1.6 m), hypereutrophic lake located near Orlando in central Florida, USA. Stormwater discharge since the 1940s from 80 km^2 farm lands located on drained floodplain marshes increased areal phosphorus (P) loading 7-fold and was the primary cause of eutrophication (Battoe et al., 1999; Lowe et al., 1999; Schelske et al., 2000). The originally abundant submersed macrophytic vegetation was displaced by persistent blooms of predominantly cyanobacteria and disappeared from Lake Apopka by the early 1950s. The lake water is characterized by high concentrations of chlorophyll *a*, suspended solids, and particulate nitrogen and phosphorus (Table 1). Re-establishment of submersed plants is hindered by shading both from dense algae and from flocculent surficial sediments resuspended by wind action and by benthic feeding by fish.

A 14-km^2 treatment wetland (marsh flow-way) being constructed on former farmlands is part of the restoration plan developed for Lake Apopka by the St. Johns River Water Management District (District) (Lowe et al., 1989, 1992; Battoe et al., 1999). The primary goal of the marsh flow-way project is the removal of algae, resuspended sediments, and particle-bound phosphorus from Lake Apopka. Model results predict that removal of P from the lake water will accelerate the recovery of Lake Apopka after P loading from the watershed is controlled (Lowe et al., 1992). Other benefits of the wetland filter project include reduction in nutrient transport to downstream lakes and restoration of wetland habitat.

Table 1
Median values over the 29-month operational period for selected water chemistry variables in the inflow from Lake Apopka

Variable	Median ($mg\,l^{-1}$)
TSS	76
Chl *a*	0.078
NH_4^+–N	0.017
NO_x–N	0.011
DON	1.68
PON	2.96
TN	4.60
SRP	0.006
DOP	0.002
POP	0.163
TP	0.173

In addition to wetland treatment, the District's comprehensive restoration plan for Lake Apopka includes acquisition of the floodplain farms to reduce external P loading, harvest of gizzard shad (*Dorosoma cepedianum*) to remove P and reduce internal P cycling, and planting and fluctuation of lake levels to promote regrowth of littoral vegetation (Conrow et al., 1993; Battoe et al., 1999). Because of the large size of the lake, a successful restoration must be self-sustaining. A critical step in restoration is to increase transparency of the lake water to allow regrowth of submersed, rooted macrophytes.

Increasingly, natural or constructed wetlands are used for removal of pollutants from wastewater or for treatment of stormwater runoff from agricultural lands and other non-point sources (Kadlec and Knight, 1996; Mitsch et al., 2000). In the North American Treatment Wetlands Database (v. 2.0; CH2M Hill, 1998), 95% of 337 treatment wetland systems were less than 0.9 km^2 in size, and 95% of 198 systems had design flows less than 0.16 m^3 s^{-1}. Very few large treatment wetland projects have been operated. One important exception is restoration of the Everglades (Florida, USA) where stormwater will be treated for phosphorus removal in large constructed wet-

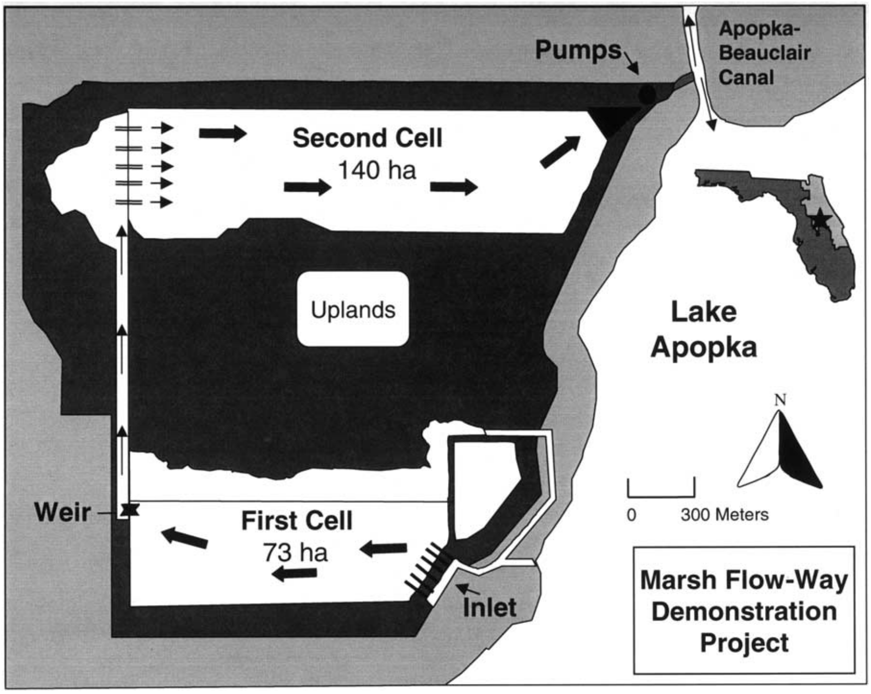

Fig. 1. Schematic of marsh flow-way demonstration project at Lake Apopka.

lands. In the Everglades Nutrient Removal Project, a 15-km^2 prototype wetland was operated for 5 years (Nungesser and Chimney, 2000), and the final stormwater treatment system will consist of six wetlands which vary from 10 to 67 km^2 in size (Goforth, 2000).

The District tested a 2.1-km^2, pilot-scale wetland filter (Marsh Flow-Way demonstration project) to examine the capacity of a wetland system to remove suspended sediments and particulate nutrients from Lake Apopka. The demonstration project was designed, constructed, and operated as a field experiment to evaluate the relationships between nutrient removal, flow, and loading; the hydraulic behavior of a wetland filter; and the development and management of vegetation.

We report here the performance of the Marsh Flow-Way demonstration project for nutrient removal during a 29-month operational cycle from start-up to a drawdown for sediment consolidation. Our focus is on retention by the wetland of particulate matter and nitrogen and phosphorus species from lake water. We also consider the results from operation of the demonstration project that had greatest significance for design and operation of the full-scale treatment wetland.

2. Methods

2.1. Project site and operation

The Marsh Flow-Way demonstration project was located on the NW shore of Lake Apopka along the Apopka-Beauclair Canal at the southern end of the full-scale flow-way site (Fig. 1). Most of this area was sawgrass (*Cladium jamaicense*) marsh before it was drained for farming in the 1940s. Drainage resulted in the oxidation and subsidence of these highly organic muck soils. Soil surface elevations over most of the demonstration project site ranged from 0.5 to 0.75 m (first cell) and from 1.1 to 1.4 m (second cell) below current mean lake level (20.27 m NGVD).

The demonstration project consisted of two wetland cells operated in series. Water flowed from Lake Apopka through inlet culverts or, during some periods, was pumped into the first wetland cell (0.73 km^2, 180 acres). An outflow weir from the first cell discharged to a south-to-north channel. This channel terminated in 27 inlet culverts for the second wetland cell (1.4 km^2, 340 acres). Pumps discharged from the second cell to the lake (Fig. 1). Because the first cell provided the best model for size and operation of cells in the full-scale flow-way, we focused on performance data from this area. Operation started in November 1990 and was suspended for several months in 1991 and 1993 for experimental planting and maintenance. The project was drained starting in February 1994 to test drawdown as a technique to consolidate newly-deposited sediments. The first cell of the demonstration project operated during 29 months (877 days) from start-up until drawdown at varying flows and retention efficiencies.

Native soils on the project site are classified as euic hyperthermic Typic Medifibrist (Olila et al., 1997). The two wetland cells were in different land uses immediately prior to project construction. The first cell had been fallow for approximately 2 years. *Panicum dichotomiflorum* and *Ludwigia* spp. dominated the drier western half of the first cell. The poorly drained eastern half was shallow emergent marsh with *Typha latifolia*, *Polygonum punctatum*, *Pontederia cordata*, *Sagittaria lancifolia*, and other wetland species. *Eupatorium capilifolium* covered most of the second cell, which was dry and recently had been farmed (Stenberg et al., 1997). After construction, most of the wetland area was allowed to revegetate naturally, although an experimental planting project tested the success of various wetland plants and planting techniques in three 5-acre plots. The hydrophytic community that developed in the first cell was dominated by *T. latifolia*. *Hydrocotyl ranunculoides*, *P. cordata*, *S. lancifolia*, *Alternanthera philoxeroides*, *Salix caroliniana*, and *Ludwigia peruviana* were important subdominants. This vegetation formed extensive floating mats in the first cell starting in 1992, especially in the eastern half. Area-weighted mean plant biomass (live and standing dead, above and below ground) in the first cell was 1.0 kg dw m^{-2} when the site was first flooded in November 1990 and 1.5 kg dw m^{-2} after drawdown in March 1994 (Stenberg et al., 1997).

Starting in November 1992, experimental changes designed to improve flow patterns were made to the configuration of the first wetland cell. Most importantly, a terminal collector ditch was excavated along the west end of the cell to route flows to the single outlet weir. This modification had immediate, inadvertent, and undesirable effects of increased channelization of flow through remnants of the agricultural drainage system and through deeper areas created by natural topography. This channelization was especially problematic because remnant drainage ditches and canals were oriented with the axis of flow.

2.2. Sample collection and analyses

Water flows were measured daily with mechanical flow meters in the intake culverts for the first wetland cell and in the discharge pipes of the pumps that drained the second cell. Inlet pumps that were used during some periods instead of gravity flow to move water into the first cell also were metered. Precipitation was measured continuously at nine sites around the project area and estimated as an area-weighted mean. Meteorological conditions were monitored continuously on site and used to estimate evapotranspiration with a modified Penman method (Lamoreux, 1962). Water budgets were used to estimate flow out of the first cell and into the second cell. Seepage into the wetland cells was quantified during drawdown periods when hydrostatic pressure differential across the perimeter cell levees was maximal. Seepage into or out of the cells during operation was then estimated from water level measurements.

Water was sampled at the inflow and outflow from each wetland cell for chemical and physical analyses. Sampling frequency was twice weekly during most periods of operation. Water samples were taken with a horizontal Van Dorn water bottle at 0.5 m or one-half the depth when depth was less than 1 m. Oxygen was measured in situ with a calibrated probe (YSI Model 57). Water temperature was determined with a thermister probe (Hydrolab Surveyor). Nutrient species were determined using standard automated techniques, with filtration (Gelman Supor 450) used to separate total and dissolved forms. Total suspended solids were measured gravimetrically on a Whatman GF/C filter, and chlorophyll was determined spectrophotometrically after filtration (Whatman GF/C) and extraction into 90% acetone. See Table 2 for chemical and physical variables measured and methods. The following water chemistry variables were derived from measured values: particulate organic P (POP), total P (TP) minus dissolved TP; dissolved organic P (DOP), dissolved TP minus SRP; total N (TN), total Kjeldahl N plus NO_x–N; particulate organic N (PON), total Kjeldahl N minus dissolved Kjeldahl N; dissolved organic N (DON), dissolved Kjeldahl N minus NH_4–N.

Newly formed sediment was collected in the wetland using a 1.2-m long (12-mm inside diameter) borosilicate glass tube. The tube was carefully lowered through the highly flocculent sediment to the original soil surface, plugged, and withdrawn. The depth of sediment in the tube was measured, and the sediment was poured into a container. In some cases, the lower one-half of the sediment in the tube was sampled separately from the top half. Sediment samples were analyzed as water samples because of the low solids content. Sites for sediment sampling were regularly rather than randomly distributed in the wetland because of access difficulty.

Samples of above-ground biomass of common plant species in the wetland and of standing-dead vegetation were taken on two occasions (August 1991, January 1992) for N and P analyses ($n = 1$–8 depending on species). Composite vegetation samples ($n = 9$) also were taken for N and P analyses to represent less-common species. Composite samples of below-ground biomass (root mats, $n = 10$) were taken in August 1994. Vegetation samples were dried (70 °C), and N and P were analyzed by Kjeldahl digestion followed by assay of SRP and NH_4 (Table 2). These nutrient analyses were used together with biomass measurements (Stenberg et al., 1997) to estimate nutrient pools in vegetation.

Table 2
Chemical and physical variables measured and analytical methods

Variable	Abbreviation	Method
Water temperature	T	EPA 170.1
Dissolved oxygen	O_2	EPA 360.1
Chlorophyll *a* (uncorrected)	Chl *a*	EPA 10200 H (not corrected for pheopigments)
Total suspended solids	TSS	EPA 160.2
Total phosphorus	TP	EPA 365.4
Dissolved total phosphorus	–	EPA 365.4
Soluble reactive phosphorus	SRP	EPA 365.1
Total Kjeldahl nitrogen	–	EPA 351.2
Dissolved total Kjeldahl nitrogen	–	EPA 351.2
Nitrate + nitrite	NO_x	EPA 353.2
Ammonium	NH_4^+	EPA 350.1
Chloride	Cl^-	EPA 300.0

EPA: Kopp and McKee (1983). –, not applicable.

2.3. Data treatment

Twice-weekly observations of water chemistry variables were linearly interpolated to daily values. Daily values for constituents were multiplied by daily values for flow to calculate daily fluxes at each sampling point. Mass fluxes and flow-weighted average concentrations were calculated over consecutive 15-day periods for reporting and analysis. We used a first-order area-based model (Kadlec and Knight, 1996) to characterize the process of nutrient removal in the first cell of the wetland.

3. Results

3.1. Hydrology, temperature, O_2

At the areal hydraulic loading rates (HLR) tested, suspended solids and particulate nutrients in the lakewater inflow were removed primarily in the first wetland cell of the project. Because little particulate load reached the second cell, little

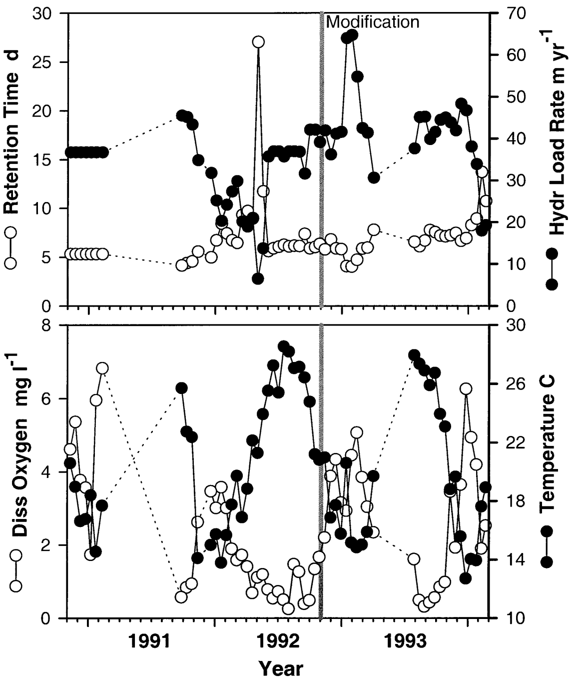

Fig. 2. (Top) Areal hydraulic loading rate (HLR) and theoretical hydraulic residence time (HRT) for the first wetland cell. (Bottom) Flow-weighted temperature and dissolved oxygen at the outlet of the first cell. Vertical line indicates beginning of hydraulic modifications to the project. Dotted lines indicate periods when flow-through operation was suspended for planting (mid 1991) and for maintenance (late 1991 and 1993).

material was removed there. Furthermore, HLR values for the first cell were equivalent to rates projected for cells in the full-scale flow-way. We focus here on performance data from the first wetland cell because this area provided the best model for cells in the full-scale flow-way.

Inflow varied from 0.15 to 1.5 $m^3\ s^{-1}$, equivalent to HLR in the first wetland cell of about 6.5–65 m $year^{-1}$ (Fig. 2). The theoretical hydraulic retention time (HRT) for the first cell ranged between 4 and 27 days but was less than 8 days during most of the period (Fig. 2). Initially, water inflow to the project was by gravity through culverts. Because of a decline in the water level in Lake Apopka, gravity flows declined starting in October 1991 (Fig. 2). We changed to pumped inflows in May 1992 to increase flow rates and used either culvert or pumped inflow in later periods. As density of vegetation and thickness of new sediments increased through time, mean water depth in the first cell increased from 0.5 to 0.9 m at HLR of 35–45 m $year^{-1}$. We limited HLR in the later period of operation to avoid further increase in depth because of concern for the health of the emergent vegetation and because water elevation at the inlet approached the design limit.

Flow of lake water through the wetland dominated the water budget. Monthly precipitation and evapotranspiration for the first cell averaged only 2.8 and 4.3% of lakewater inflow, respectively. Water was both gained and lost through seepage depending on relative water surface elevations within and outside the wetland. In total, net seepage was 0.2% of lakewater inflow. Monthly seepage (in or out) averaged 2.6% of lakewater inflow. Comparison of mass fluxes of the conservative ion Cl^- at the inlet and outlet of the first cell provided a check on water budget calculations for the wetland. In 53 of the 58 2-week periods tabulated, retention of Cl^- varied between -10 and $+10\%$, close to the expected value (0%).

Water temperature at the outflow from the first cell varied seasonally and ranged between about 13 and 29 °C. Dissolved O_2 varied inversely with temperature (Fig. 2), probably because of combined effects of lower solubility of O_2 and greater biochemical oxygen demand in the wetland at higher temperatures. Oxygen never was undetectable for sustained periods in the outflow from the first cell, but levels fell below 0.5 mg $O_2\ l^{-1}$ in late summer (Fig. 2).

3.2. Input–Output comparisons

Total suspended solids (TSS) ranged from about 35 to 190 mg l^{-1} in the inflow (Fig. 3). While the wetland was operated in the initial configuration, outflow concentrations varied from about 0 to 11 mg TSS l^{-1}. In November 1992, a terminal collector ditch was completed to route outflows to the single weir and improve flow patterns. Instead, this modification had immediate and undesirable effects of increased channelization of flow through remnant agricultural ditches. Particle removal rates decreased, and TSS in the outflow increased several-fold (Fig. 3). Several further modifications to the project were made after this point in attempts to stop the channelized flow that had been created. Inflow concentrations of TSS and particulate nutrient fractions were higher in 1993 compared to previous years (Figs. 3–5). A severe storm in March 1993 may have resulted in higher than normal internal nutrient loading through wind-driven disturbance to sediments.

TN in the inflow from Lake Apopka ranged from about 3 to 9 mg N l^{-1} (Fig. 4). Levels of dissolved inorganic N were very low. PON averaged 63% of the TN in the inflow, and DON made up almost all the remainder (Fig. 4). From 2 to 4 mg TN l^{-1} was present in the outflow during the operational period prior to experimental modifications in November 1992. The net change in nitrogen forms in the wetland was dominated by removal of PON (Fig. 4). As was the case with TSS (above), modifications to the project in November 1992 were immediately reflected in higher levels of particulate organic N and, therefore, TN at the outflow (Fig. 4). Atmospheric deposition of N was negligible compared to lakewater inflow. Median deposition (dry and wet) of N measured for Lake Apopka from 1989 through 1992 was less than 1 mg TN m^{-2} day^{-1}, compared to a median loading to the wetland of 475 mg TN m^{-2} day^{-1} in lake water.

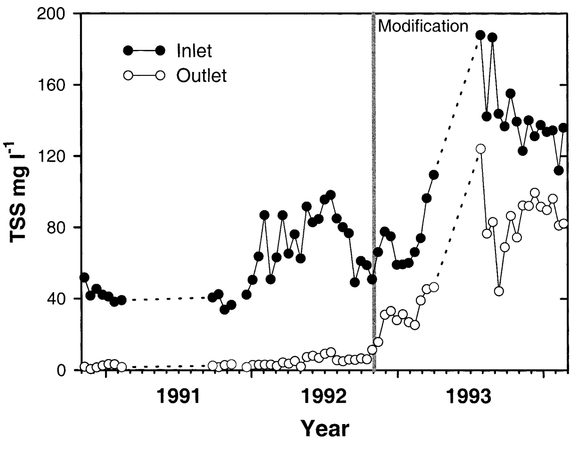

Fig. 3. Flow-weighted concentrations of total suspended solids (TSS) at the inlet and the outlet of the first wetland cell. Vertical line and dotted lines are explained in Fig. 2.

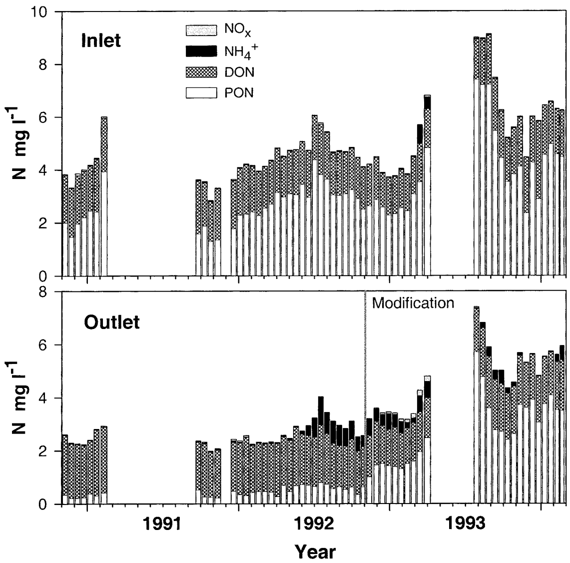

Fig. 4. Flow-weighted concentrations of nitrogen species at the inlet and the outlet of the first cell. DON, dissolved organic nitrogen; PON, particulate organic nitrogen. Vertical line indicates beginning of hydraulic modifications to the project. Missing data indicate periods when flow-through operation was suspended for planting (mid 1991) and for maintenance (late 1991 and 1993).

Flux of dissolved inorganic N species, especially ammonium, out of the wetland was greater than flux in, although NH_4–N concentrations at the outflow usually (75%) remained below 0.5 mg l^{-1}. Low net release of inorganic N from the soils occurred immediately after inundation (Fig. 4). After 13 months of operation, net release of primarily NH_4 and some NO_x (NO_2+NO_3) increased. Release of NH_4 was especially high during periods of high temperature and low O_2 (Figs. 2 and 4). The fact that release of inorganic N species occurred later, rather than earlier, in the operation of the wetland indicated that mineralization of sedimented material likely was responsible. DON passed through the wetland unattenuated, or was somewhat augmented, and made up most of the N at the outflow prior to November 1992. The mean rate of release of DON was 3.7 mg N m^{-2} day^{-1}.

Total phosphorus (TP) in the inflow from Lake Apopka ranged from about 0.08 to 0.38 mg l^{-1} (Fig. 5). Particulate organic phosphorus (POP) typically made up more than 90% of the TP, and soluble organic and inorganic forms of P were present in only low concentrations. As with nitrogen, atmospheric deposition was a negligible source of P to the wetland. Median deposition (dry and wet) of P at Lake Apopka from 1989 through 1994 was about 0.07 mg TP m^{-2} day^{-1} (Stites et al., 2001), compared to a median inflow to the wetland in lake water of about 18 mg TP m^{-2} day^{-1}.

During the initial period of operation, TP concentrations at the outflow from the first cell were higher than inflow values (Fig. 5). These elevated P values were caused almost entirely by release of soluble reactive phosphorus (SRP), which masked the simultaneous removal from the inflowing water of particulate P (Fig. 5). The immediate release of SRP upon inundation of the first cell indicated that the primary sources were SRP pools in soils and the death and decomposition of a portion of the initial plant cover rather than mineralization of sedimented POP from the lake water.

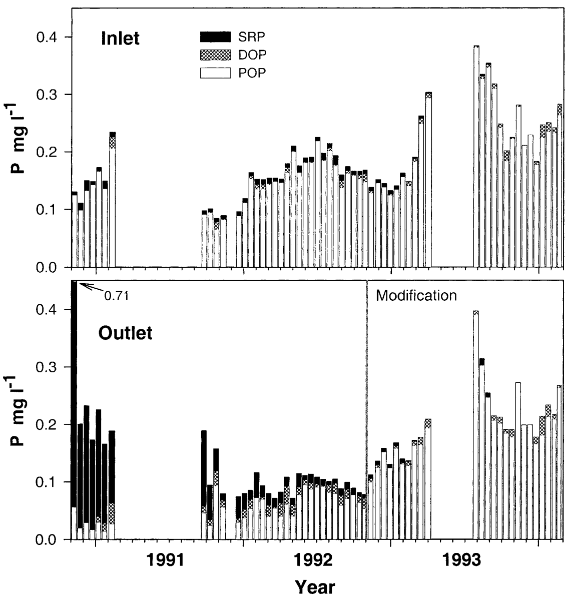

Fig. 5. Flow-weighted concentrations of phosphorus species at the inlet and the outlet of the first cell. SRP, soluble reactive phosphorus; DOP, dissolved organic phosphorus; POP, particulate organic phosphorus. Vertical line and missing data are explained in Fig. 4.

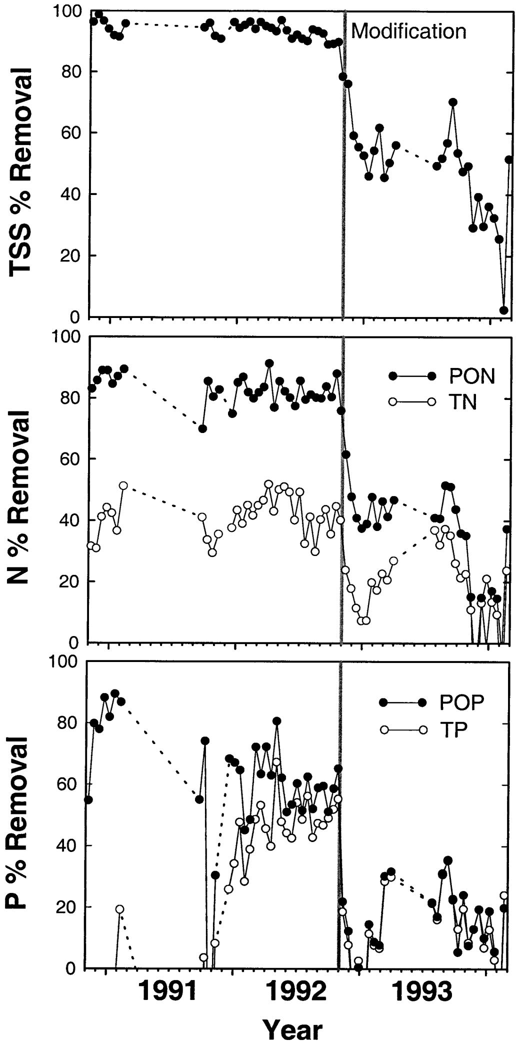

Fig. 6. Percent mass removal in the first cell of total suspended solids (TSS), particulate organic nitrogen (PON) and total nitrogen (TN), and particulate organic phosphorus (POP) and total phosphorus (TP). Periods when the first cell showed net release of P (negative retention) are not shown. Vertical line and dotted lines are explained in Fig. 2.

Because inflowing lake water contained almost no SRP (Fig. 5), the increase in SRP from the inflow to outflow was a useful measure of the rate of release. SRP was released immediately from soils in the first cell at a net rate of 10–20 mg P m^{-2} day^{-1}. The wetland effectively retained P after decline in these initial rates of release (Fig. 6). Release rates declined to approximately 1 mg

P m^{-2} day^{-1} by the end of 1991 after 6 months of flow-through operation and ultimately fell to zero (Fig. 7). In the first cell, cumulative release of SRP calculated by mass balance was about 3.0 g P m^{-2} during initial operation. Cumulative SRP release in the second cell reached a similar level (2.5 g P m^{-2}). However, a different pattern of SRP release was evident in the second cell, where P release occurred over a 20-month period of operation (Fig. 7). In contrast to SRP, dissolved organic P (DOP) was released in small and variable amounts in the wetland throughout the period of operation (Fig. 5), with a mean release rate in the first cell of 0.47 mg P m^{-2} day^{-1}.

The wetland effectively removed suspended particulate matter from the start of operation. Mass removal efficiencies for TSS generally were > 90% for the period prior to hydraulic modification (Fig. 6). After the advent of hydraulic short-circuiting in November 1992, TSS removal efficiencies dropped to 40–60% or lower (Fig. 6). Similarly, removal of PON in the first cell varied between 75 and 90% initially (Fig. 6). During the modification period, PON removal dropped first to around 45% and finally to less than 20%. Removal efficiencies calculated for TN were substantially lower than those for PON (Fig. 6). This difference resulted mainly from the high levels of DON in the inflow from Lake Apopka (Fig. 4) which were unattenuated or even augmented as water passed through the wetland.

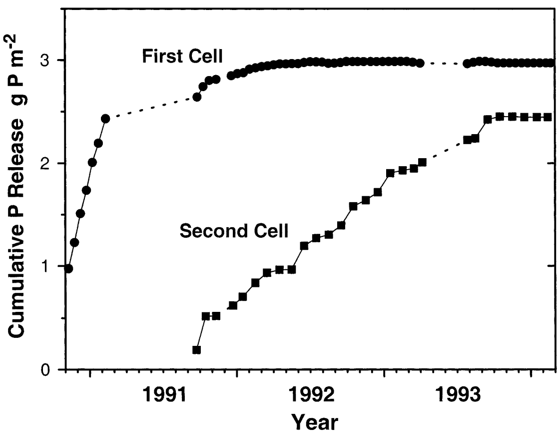

Fig. 7. Cumulative release of soluble reactive phosphorus (SRP, measured as mass difference from inlet to outlet) in the first and second cells during flow-through operation. Values for the first cell were calculated from interpolated daily concentrations and measured daily flows at the inflow and outflow. Values for the second cell were estimated from median monthly concentrations at the inflow and outflow and total monthly flows. SRP in drainage water from the site during project construction and when operation was suspended for planting (mid 1991) was not quantified.

Mass removal efficiencies for POP varied between about 50 and 90% prior to hydraulic modifications (Fig. 6). Removal efficiencies calculated for TP were much lower than those based on POP because of release of SRP from soils and vegetation when the wetland first was inundated (Fig. 5). Removal of TP was negative (outlet mass > inlet mass) for the first 6 months of operation because of high rates of release from soils, but efficiency for TP increased thereafter as release rates declined (Fig. 6). During the modification period removal of phosphorus declined greatly, and efficiency based both on TP and POP ranged from 5 to about 30% (Fig. 6).

The fact that removal efficiency for TSS was greater than efficiencies for both PON and POP (Fig. 6) indicated that particulate matter at the outlet was enriched in both N and P compared to particulate matter at the inlet. Average N and P concentrations in particulate matter increased by factors 2.4 × and 5.4 ×, respectively, as water moved through the wetland. For the period of effective particulate removal (from start of operation through October 1992), flow-weighted average concentrations of nitrogen in particulate matter increased from 44 mg N g dw^{-1} at the inflow to 108 mg N g dw^{-1} at the outflow. Similarly, concentrations of phosphorus in particulate matter at the inflow and outflow were 2.4 and 13 mg P g dw^{-1}. These increases could have been caused by preferential removal in the wetland of particles with low N and P content, by release from the wetland of particles with high N and P content, or by scavenging of dissolved N and P by particles that traversed the wetland.

3.3. Mass balances for dry matter, N, and P

The first wetland cell operated approximately 29 months (877 days) from start-up until drawdown at varying flows and removal efficiencies.

Table 3
Net mass removal (inflow minus outflow) for phosphorus and nitrogen species in the first wetland cell during the 29-month (877 days) operational period

Species	Mass (10^6 g)
Phosphorus	
POP	4.00
SRP	−2.17
DOP	−0.30
Total P	1.53
Nitrogen	
PON	113
NH_4^+	−14.8
NO_x	−0.9
DON	−2.4
Total N	95.1

Positive values indicate removal, and negative values indicate release. See Section 2 for abbreviations. Release of SRP occurred from soils and vegetation immediately after initial flooding.

About 16 months of operation took place prior to hydraulic modifications. During the subsequent 13 months, flow was largely short-circuited, and particulate and nutrient removal efficiencies were low (Fig. 6). Over the entire period net removal of suspended matter totaled 3.5×10^9 g dw.

Removal of particulate P in the first cell over the 29-month period amounted to 4.0×10^6 g P. However, about 2.2×10^6 g SRP was lost initially from the soils (Fig. 7, Table 3). Considering both this initial release of SRP and a lesser release of DOP, net retention of P in the first cell was about 1.5×10^6 g (Table 3). Removal of particulate N in the first cell over the 29-month period was 11.3×10^7 g (Table 3). In comparison with removal of PON, minor amounts of soluble organic and inorganic N species were released. Net retention of N in the first cell was about 9.5×10^7 g (Table 3).

We compared major fluxes (particulate removal and soluble release) of dry matter, P, and N with major pools (accumulation of new sediment and vegetative biomass) in a partial budget for the wetland during the 29-month period (Table 4). Removal of dry matter from lake water in the wetland totaled 4750 g m^{-2}, and about 43% of this material was recovered in the accumulated sediments. Some dissolved organic carbon was released from the wetland, although this flux was not quantified. Carbon lost from the sediments as CH_4 and CO_2 could have accounted for much of the 2730 g dw m^{-2} lost. Schipper and Reddy (1994) found CH_4 release to account for ~70% of combined CH_4 and CO_2 release at five wetland sites in Florida. They measured 1.7 g C m^{-2} d^{-1} CH_4 release at a single site in the first cell (Schipper and Reddy, 1994). If this rate were assumed to be constant and representative, then about 2100 g C m^{-2} (1.7 g C m^{-2} day^{-1} × 877 day/0.7) would have been lost as CH_4 and CO_2 from the first cell during 29 months.

Removal of particulate N from lake water totaled 155 g N m^{-2}, and N was recovered in new sediments at a similar percentage (42%) as the bulk dry matter (Table 4). Release of soluble N species from the wetland was equivalent to about 16% of the mass of particulate N removed. Nitrogen recovered in plant biomass amounted to about 20% of particulate removal. Additional loss of N through denitrification probably occurred.

Table 4
Major fluxes and pools for dry matter, nitrogen and phosphorus on an areal basis in the first wetland cell for the 29-month operational period

	Dry matter (g m^{-2})	Nitrogen (g m^{-2})	Phosphorus (g m^{-2})
Particulate removal	4750	155	5.48
Sediment accumulation[1]	2020 (43%)	65.3 (42%)	4.62 (84%)
Soluble release	–	24.8 (16%)	3.39 (62%)
Vegetative biomass[2]	1540 (NA)	31.0 (20%)	2.81 (51%)

[1] Median sediment accumulation 33 cm.
[2] Above and below-ground, living and dead.
Values in parentheses are percent of particulate mass removal. –, not quantified; NA, not applicable.

However, concentrations of NO_3 were low in the inflow (Table 1), and supply of additional NO_3 through nitrification may have been limited by low O_2 levels during the warm season (Fig. 2).

Particulate P removal from lake water in the first cell totaled about 5.5 g P m^{-2} (Table 4). About 4.6 g P m^{-2}, or 84%, was recovered in the accumulated new sediments. An additional 3.4 g P m^{-2} was released as soluble P species, and 2.8 g P m^{-2} was recovered in vegetative biomass (Table 4). The sum of the P recovered in the wetland sediment and vegetation plus the soluble P lost from the discharge exceeded the P removed from the inflow by a factor of 2. This inequality demonstrated the importance of mobilization of pre-existing pools of P from soil (and, to a lesser extent, vegetation) in the overall P budget for the wetland.

3.4. First-order model for nutrient removal

We used the first-order area-based '$k-C^*$' model (Kadlec and Knight, 1996) to characterize the process of nutrient removal in the first cell of the wetland. We applied the model to data for TSS, TP, POP, and PON. We did not attempt to use this simple model to characterize removal of TN, since about 37% of TN in the inflow consisted of DON, NH_4^+, and NO_x (Table 1). These dissolved N species generally were unchanged or were augmented in the wetland (Fig. 4, Table 3). The model was applied to 33, 15-day (approximately) averaging periods during the time of effective particulate removal (from start of operation through October 1992). The $k-C^*$ model can be represented as:

$$C_o = C^* + (C_i - C^*)\exp(-kA/Q),$$

with C_o = flow-weighted mean concentration in the outflow, g m^{-3}; C^* = background concentration in the wetland, g m^{-3}; C_i = flow-weighted mean concentration in the inflow, g m^{-3}; k = first-order areal rate constant for removal, m $year^{-1}$; A = area of the wetland, m^2; Q = water flow, m^3 $year^{-1}$.

This model describes the long-term average removal of a substance in a wetland under ideal plug-flow conditions. Necessary assumptions of the model concern primarily hydrologic characteristics of the wetland and the temporal distribution of loading. The primary hydrologic requirement, that rainfall balance evapotranspiration, was met because both these components were small in relation to flow, and seepage was negligible. In all but two of the 26 averaging periods, outlet flow was within 10% of inlet flow. The other primary requirement is that flow-weighted mean concentrations at the inlet not differ appreciably from time-averaged mean concentrations. In fact, for the nutrients modeled, these two means differed by less than 5% in 25 of the 26 averaging periods. For TSS, differences were less than 5% in 24 of 26 periods.

The first-order areal rate constant k describes the decrease in concentration of a constituent in the wetland from the level at the inlet towards a background level C^* set by internal processes, rainfall, or other factors (Kadlec, 1999). Because the background concentration C^* frequently is assumed to be zero, we derived one set of rate constants under this same assumption. We also estimated background concentrations from our data and derived another set of k values using these background concentrations. Kadlec (2000) used simulated data to demonstrate weaknesses in the first-order model. In particular, parameters assumed in the model to be constants actually varied with inlet loading. These problems were less severe when non-zero values were used for C^*. Because we currently lack information about internal mixing in the wetland, we assumed ideal plug flow. This conservative assumption provided a lower bound on k values.

We estimated background concentrations for nutrient fractions in the wetland from the lowest concentrations observed at the outflow. A lower boundary for concentration of each fraction could be described as a curvilinear function of temperature (e.g. Fig. 8). We used an approximation of the Arrhenius equation (Kadlec and Knight, 1996; Chapra, 1997) to fit a line by eye that intersected one or two points and formed a lower bound for the remaining observations (Fig. 8). Equations for these lines were used to estimate C^* values, as follows: $C^*(TSS) = (0.5)^*(1.25) \wedge (T-18.98)$ mg l^{-1}, $C^*(TP) = (0.0720)^*(1.025) \wedge (T-21.29)$ mg

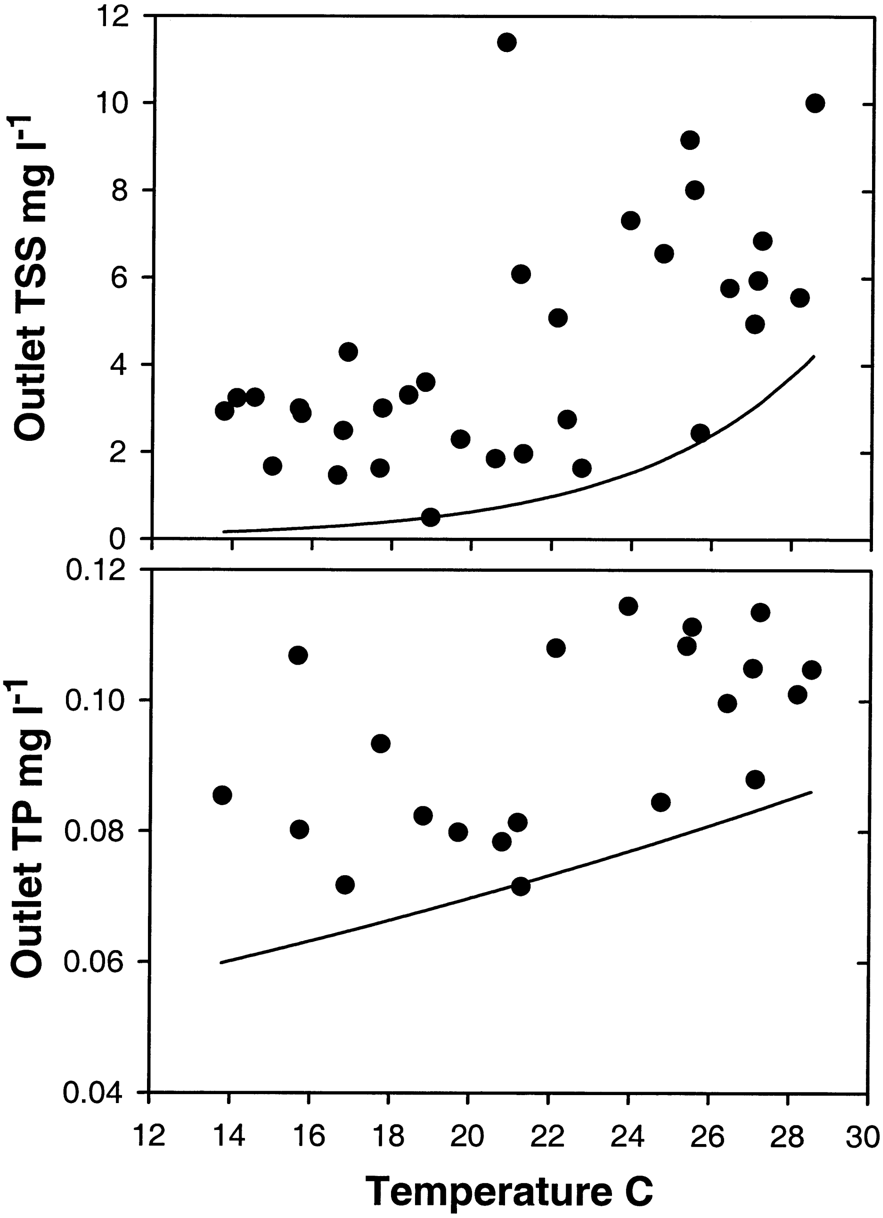

Fig. 8. Relationship of flow-weighted concentration of total suspended solids (TSS, top) and of total phosphorus (TP, bottom) with water temperature at the outlet from the first cell. Line is Arrhenius equation used to estimate background concentration C^*. Line was fit by eye as a lower bound for observed values.

P l^{-1}, $C^*(\mathrm{POP}) = (0.0243)^*(1.165) \wedge (T - 22.75)$ mg P l^{-1}, and $C^*(\mathrm{PON}) = (0.252)^*(1.15) \wedge (T - 22.38)$ mg N l^{-1}, where T = temperature in Celsius.

The apparent areal rate constants calculated for removal of TSS increased almost linearly with hydraulic loading rate up to a HLR of about 30 m year^{-1} (Fig. 9). Because outflow concentrations of TSS were low, calculated k values were similar whether C^* was assumed to be 0 or was estimated from T. The positive relationship between k and HLR could indicate that reduction in TSS required only a fraction of the wetland area at low HLR. At low HLR, calculation of k from inlet and outlet TSS concentrations and the entire wetland area would tend to underestimate k.

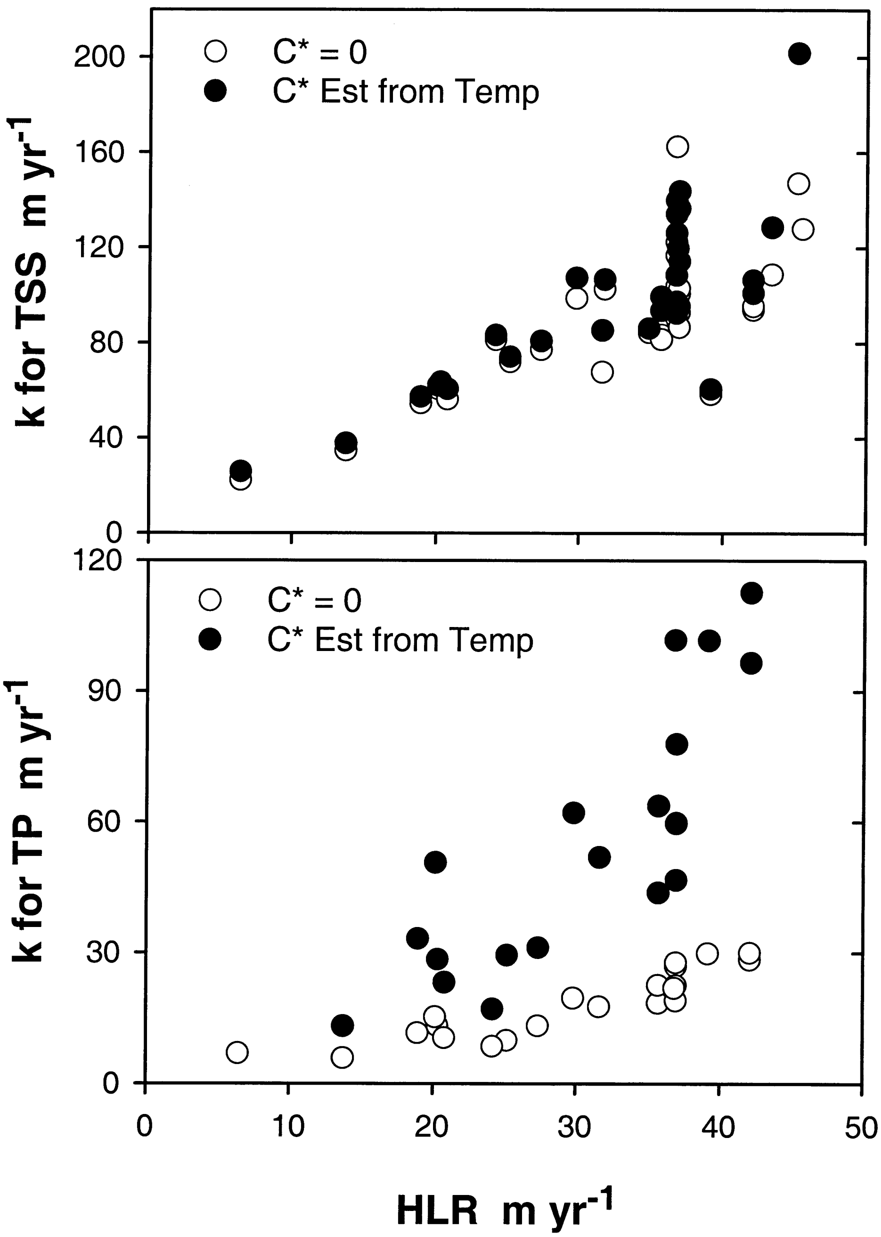

Fig. 9. Relationship between areal first-order rate constants k for removal of total suspended solids (TSS, top) and total phosphorus (TP, bottom) and areal hydraulic loading rate for the first wetland cell. Rate constants were calculated both for background concentrations (C^*) = 0 and for C^* estimated from temperature (see text). Data are from start of operation (TSS) or from December 1991 (TP) through October 1992.

Table 5
Median values (and ranges) for first-order areal rate constants (k) for removal of particulate and total nutrients in the first cell calculated both with $C^* = 0$ and with C^* values estimated from temperature

	k, $C^* = 0$ (m year^{-1})	k, estimated C^* (m year^{-1})
TSS	97 (59–163)	107 (61–202)
PON	64 (42–89)	98 (47–125)
POP	33 (14–83)	50 (15–108)
TP	23 (18–30)	63 (44–113)

Medians were tabulated using 2-week periods, which had hydraulic loading >29 m year^{-1} and which were prior to the start of short-circuiting. n varied from 12 to 24.

However, underestimation of actual C^* levels, which is possible when C^* is predicted from the lowest observed outflow concentrations, also would result in underestimation of k at low HLR (Kadlec, 2000). At HLR greater than 30 m year^{-1}, apparent rate constants for TSS removal were variable with no real trend (Fig. 9). We conclude that these k values at higher HLR should most accurately represent the capacity of the wetland for TSS removal. The relation between areal rate constants calculated for removal of particulate organic nitrogen and HLR was very similar, with increasing k values up to a HLR of about 35 m year^{-1} (data not shown).

Areal rate constants calculated for removal of TP were related to HLR in a similar manner as removal of TSS (Fig. 9). As expected, rate constants calculated with $C^* = 0$ were lower than those calculated with C^* estimated from T. Rate constants for removal of TP increased with HLR throughout the range of hydraulic loading. Values did not tend to level off at higher HLR and still may underestimate the actual removal capacity of the wetland at high HLR. As for TSS (above), both an increase in the wetland area involved in phosphorus removal and underestimation of C^* could cause the observed patterns. The relation between the rate constant for removal of POP and HLR was similar to that of TP (data not shown).

We used k values calculated for periods with high HLR (>29 m year^{-1}) to characterize the capacity of the wetland for nutrient removal (Table 5). Whether C^* was assumed to be 0 or to vary with temperature, median k values for TSS were about 100 m year^{-1} (Table 5). Median values of k for PON were 64 and 98 m year^{-1}, with C^* taken as 0 or estimated, respectively. Similarly, median values of k for POP were 33 and 50 m year^{-1}. Values of k calculated for TP removal, 23 and 63 m year^{-1}, respectively, were close to those for removal of POP. This similarity was expected, because POP made up most of TP and was the P species removed in the wetland (Fig. 5). Coveney et al. (2001) used another approach to derive k for TP removal from this same dataset by global fitting of the $k - C^*$ model to observed outflow concentrations. That result, $k = 55$ m year^{-1}, was close to the median value for k (estimated C^*) at high HLR (Table 5).

3.5. Sediment accumulation

A median of 33 cm flocculent sediment ($n = 48$ sites) accumulated in the first wetland cell over the 29-month period from start of operation until drawdown. For 20 samples from different depths in these new sediments, median TSS was 4.4 g dw l^{-1}, median N concentration (total Kjeldahl-N) was 140 mg N l^{-1}, and median P concentration was 11 mg TP l^{-1}. Reasonably good relationships were found between sediment dry weight and Kjeldahl-N ($R^2 = 0.67$) and TP ($R^2 = 0.74$). The slopes of linear regressions of N and P on TSS gave mean concentrations of N and P in sediments of 3.4 and 0.25%, respectively. Because the newly deposited sediment did not consolidate as rapidly as it accreted, we tested drawdown of the wetland over 120 days to increase sediment bulk density. A detailed evaluation of the drawdown will be presented elsewhere. Briefly, drawdown increased median bulk density of the new sediments over 7-fold to 36 g dw l^{-1}, and values as high as 96 g dw l^{-1} were recorded. Maximal consolidation of sediments in the field was reached within about 65 days after the start of de-watering.

4. Discussion

Compared to surface-flow wetlands used to treat wastewater, the Lake Apopka Marsh Flow-Way demonstration project operated at lower inflow concentrations of nutrients, especially P (Table 1), but at higher HLR (Kadlec and Knight, 1996). The marsh flow-way also differs from most treatment wetlands in other respects. First, treated water recirculates back to the lake. Design flow through the full-scale (14-km^2) wetland will be about 14 m^3 s^{-1} which will treat the entire lake volume twice per year. Second, the wetland will operate at relatively low mass removal efficiencies for P in order to maximize the rate of P removal. Finally, mainly particulate P will be removed; more than 90% of the phosphorus in Lake Apopka water generally is particle-bound. Sedimentation of particles from lake water and accretion of sediment and plant detritus into new wetland soil are the mechanisms whereby phosphorus will be removed and sequestered in the wetland.

The wetland was capable of removal of TSS and TP from Lake Apopka water at or above target efficiencies of 85 and 30%, respectively (Fig. 6), which confirmed earlier projections (Lowe et al., 1989, 1992). For the 10-month period of operation after SRP leaching but prior to short-circuiting, mass removal efficiency for TP was 30–67%. These values were similar to P removal reported from other high-flow wetlands receiving ambient surface water inflows. Mitsch et al. (1995) measured 53–96% removal of P over 3 years in two wetlands receiving Des Plaines River water (TP 0.11–0.18 mg l^{-1}). For two wetlands receiving inflow from the Olentangy River (TP $\sim$ 0.17 mg l^{-1}), Nairn and Mitsch (2000) measured 58 and 62% removal of TP over 2 years.

The goal of the Marsh Flow-Way project is to maximize power (P removed per unit time) and capacity (permanent P storage) rather than efficiency (fraction of P removed in a single pass). Because efficiency declines asymptotically with increased load (Nichols, 1983), maximum power will be achieved at low efficiencies (Lowe et al., 1989). Consequently, we expect to operate at P loading rates that exceed the assimilative capacity (Richardson et al., 1997) of the wetland. With high P loading rates, we expect that effluent P concentrations will be above background and that the wetland vegetation will be characteristic of eutrophic, impacted sites.

The actual net removal rate for P in the first wetland cell was 0.48 g m^{-2} year^{-1} during the period of operation prior to short-circuiting. Two factors contributed to this low net removal rate: the leaching of SRP from soils immediately after inundation (Table 3), and low gravity flow through the wetland prior to the use of inflow pumps (Fig. 2). For comparison, we estimated the hypothetical P removal rate that could have been achieved in the absence of these problems. Use of inlet pumping to maintain at least 37 m year^{-1} HLR would have increased the net removal rate for P in the wetland to almost 1 g m^{-2} year^{-1}. Prevention of the initial leaching of SRP would have been even more significant. Net P removal from lake water in the absence of this SRP release and with inlet pumping could have exceeded 3 g m^{-2} year^{-1}.

Coveney et al. (2001) estimated maximal potential P removal rates for the project by extrapolation to higher HLR using the $k - C^*$ model. Although fit between observed and predicted data was not uniformly good, they predicted that P loading rates in the range 10–15 g P m^{-2} year^{-1} should give maximal P removal rates averaging about 4 g P m^{-2} year^{-1}. This optimal range in P loading would correspond to a range in HLR of 60–90 m year^{-1}, about twice the typical HLR in the demonstration project. The HLR necessary to maximize P removal rate in the full-scale project is high but should be feasible based on experience with other treatment wetlands (Kadlec, 1999). The predicted maximal P removal rate (4 g P m^{-2} year^{-1}) was similar to the mean rate (5.4 g P m^{-2} year^{-1}) observed in wetlands receiving river water with mean TP 0.17 mg P l^{-1} and with particle-bound P the predominant fraction (Nairn and Mitsch, 2000).

Values for k for removal of TP in the demonstration project were high compared with the mean k (10 m year^{-1}) for a set of 92 treatment wetlands (Kadlec, 1999). The likely explanation was the dominance of particulate P in Lake

Apopka water contrasted with the dominance of dissolved P in most wastewaters. During initial operation of wetlands, removal of P can be inflated because of rapid storage of P on soil sorption sites and in growing vegetative biomass (Kadlec and Knight, 1996). However, this phenomenon did not operate in the demonstration project, where the soil was a source rather than a sink for P during initial operation (Fig. 7), and the changes in mean plant biomass were moderate (Stenberg et al., 1997).

Estimated background concentrations in the wetland (C^*) for TP lay between 0.06 and 0.09 mg P l^{-1} (Fig. 8) which were high compared with other systems (Kadlec, 1999). We expected this high C^* for TP because the wetland was constructed on P-rich organic soils and received hypereutrophic lake water. If C^* for TP remains high in the wetland as the mean TP in lake water declines through time, then C^* ultimately will limit the utility of the wetland for P removal. However, C^* also might decline as the wetland system becomes less enriched, and this effect would tend to stabilize P removal despite the lower inflow concentrations. In either case, the value of the treatment wetland lies primarily in P removal from lake water in the early stages of the restoration to accelerate the recovery of the lake (Lowe et al., 1992).

Costs of construction were approximately $1 million km^{-2} wetland treatment area for the demonstration project and $1.6 million km^{-2} for phase 1 of the full-scale project. This difference reflects in part the cost to provide infrastructure for greater operational flexibility in the full project with multiple independent cells. Our preliminary estimate of the operational cost of P removal for the full project is $0.02–0.03 g P^{-1}. This cost primarily represents inlet and outlet pumping which will be necessary to sustain high HLR. Although the operational goal for the wetland is to maximize rate rather than efficiency of P removal, efficiency will be important in determining the cost of P removal. The positive relationships between C^* for TP and for TSS and temperature (Fig. 8) indicate that seasonal variation in water flow may be one technique to control treatment cost. The operational cost of P removal would be lowest when water temperature is low (i.e. low C^*) and TP in the inflow is high, and water flow could be maximized during these periods.

The demonstration project provided important information for the design and operation of the full-scale project, especially with regard to hydraulic function, efficiency of nutrient removal, and characteristics of newly deposited sediment. HLR values in the range 35–45 m $year^{-1}$ (10–12 cm day^{-1}) resulted in the maximum desirable water depths at the inlet end of the densely-vegetated first cell. The water surface elevations observed in the demonstration wetland resulted in part from irregular soil surface elevations. However, the resistance to flow shown by vegetation and by new flocculent sediments was one reason why multiple cells in parallel were constructed in the full-scale project rather than a single large treatment wetland. These cells have a lower aspect ratio than the first cell in the demonstration project, and multiple cells allow independent operation and maintenance. Inlet and outlet pumps will be used to ensure high HLR.

Minimal channelization and short-circuiting were required in the demonstration project to meet or exceed target removal efficiencies for TSS and P. Channelization in the wetland was induced by hydraulic modifications that increased flow through remnants of the agricultural drainage system, through deep areas created by natural topographic variation, and between dense stands of vegetation. Actual hydraulic residence times and nutrient removal rates decreased as a result (Fig. 6). Remnant drainage ditches and canals were oriented along the axis of flow in the demonstration project, which made channelization especially problematic. Furthermore, single inlet and outlet locations were used, which may contribute to the development of channelized flow (Kadlec and Knight, 1996). In contrast, both inlet and outlet flows were distributed evenly over the ends of the cells in the full-scale design. Creation of deep areas perpendicular to flow in constructed wetlands was suggested to alleviate channelization through interception and spreading of flow paths (Kadlec and Knight, 1996; Kadlec, 2000). This technique was applied in the full-project area with regularly-spaced ditches oriented perpendicular to the flow axis.

De-watering through drawdown was a necessary and effective technique to consolidate new sediment in the pilot wetland. Phosphorus content of the sediments in the field remained about 0.25% of dry weight for at least the first 65 days of drawdown. The bulk density and phosphorus content of the sediment after drawdown confirmed earlier projections of sediment storage capacity for the full-scale flow-way (Lowe et al., 1992). The median density of P in the sediments of the demonstration project after 65 days of drawdown was about 0.1 g TP l^{-1} which exceeded the target of 0.05 g TP l^{-1} (Lowe et al., 1992).

Preliminary evaluations indicated that extension of the drawdown in the field beyond 65 days resulted in mineralization of sediment phosphorus and subsequent SRP release. After 120 days of drawdown, median TP was 0.18% of dry weight ($n = 7$). Laboratory studies showed a decrease in labile organic P and an increase in KCl-extractable P when sediment cores from the first cell were drained (Olila et al., 1997). In the laboratory, flux of SRP to the overlying water when drained cores were reflooded was greatly increased by longer draining. Because maximal consolidation in the field was observed by 65 days, drawdown of the full-scale project will be limited to 60 days or less to minimize mineralization and release of SRP. Wetland cells in the full-scale project were designed to facilitate drawdown through the incorporation of perpendicular drainage ditches. Sediment consolidation will be more quickly achieved and less variable than that measured in the demonstration project, which should result in bulk densities close to the highest values found in this initial field test.

Release of soluble phosphorus in the demonstration wetland occurred when the organic soils and initial vegetation first were inundated and again when the wetland was drawn down. The wetland was effective in retention of P after decline in the initially high rates of release from soils during the first 6 months of operation (Fig. 6). Subsequent SRP release in our system was similar to or less than initial SRP release (0.3–1.5 mg P m^{-2} day^{-1}) from a former agricultural site in the Florida Everglades during the first 4 months after flooding (Newman and Pietro, 2001).

Net SRP release from soils in the first cell derived by mass balance in our study did not differ significantly from measurements by D'Angelo and Reddy (1994a,b) using laboratory (intact cores) and in situ (diffusion sampler) techniques

Table 6
Flux of SRP and NH_4 from soils in the first wetland cell determined in this study by mass balance (outflow–inflow) and estimated by D'Angelo and Reddy (1994a,b) from in situ diffusion samplers and from laboratory measurements in intact soil cores

	Period	Flux rates (mg m^{-2} day^{-1}) Mean (S.D.)		
		Mass balance[1]	Diffusion samplers[2]	Intact cores[3]
SRP	Feb 1992	1.6	1.6 (1.1)	–
	Jul 1992	0.2	1.5 (1.3)	–
	Aug 1992	−0.3	–	3.0 (4.0)
	Dec 1992	0.0	0.4 (0.5)	–
NH_4–N	Feb 1992	0.7	3.2 (0.4)	–
	Jul 1992	82	8.3 (4.9)	–
	Aug 1992	76	–	115 (53)
	Dec 1992	58	6.0 (4.4)	–

[1] This study.
[2] D'Angelo and Reddy (1994a).
[3] D'Angelo and Reddy (1994b).
Calculation periods were 4 weeks for mass balances. Values from D'Angelo and Reddy (1994a,b) are means and S.D. for four sites in the first cell. –, not quantified.

(Table 6). In contrast with SRP, release of NH_4 calculated by mass balance greatly exceeded the fluxes calculated by D'Angelo and Reddy from in situ diffusion samplers (Table 6). Our release rates for NH_4 by mass balance were closer to fluxes that they measured in intact cores and support the conclusion (D'Angelo and Reddy, 1994b), at least for NH_4, that intact cores provided more realistic flux values than diffusion samplers.

We expect that maintenance of flooded fallow conditions prior to construction may reduce but will not eliminate SRP release from organic soils. Application of a soil amendment (alum residual from potable water treatment) prior to flooding to bind SRP (Ann et al., 2000) is one method that the District is using to control release of SRP from soils in the full-scale project. Other techniques will include recycling of initial flood waters through adjacent wetland cells and minimizing both the frequency and the duration of drawdowns. In the demonstration project, vegetation treatment (herbicide, burning) conducted in the first wetland cell during the drawdown probably contributed to release of SRP when the wetland was refilled.

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