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Rejuvenating the largest municipal treatment wetland in Florida

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Abstract

The Orlando Easterly Wetland (OEW), one of the largest constructed wetlands for the treatment of wastewater in Florida, started operation in 1987 for the reduction of nutrient loads in tertiary treated domestic wastewater produced by the City of Orlando. The wetland has performed better than design expectations, but phosphorus removal effectiveness experienced some seasonal declines beginning with the winter of 1999. Subsequent studies indicated that the OEW treatment capacity was hindered by inefficient phosphorus removal in the upstream cells of one of three flow trains. Therefore, rejuvenating management activities were initiated on these cells in 2002. The management included the removal of plants and organic top sediments, site grading in the interior of the cells, construction of baffles and islands, and re-vegetation. This study evaluates the improvement in hydraulic and phosphorus removal performance realized from the wetland modifications. Improvement of hydraulic performance was evaluated based on tracer tests, and improvement of phosphorus removal performance was evaluated based on episodic spatially distributed water samples as well as model prediction. The results showed that both the hydraulic efficiency and the phosphorus removal effectiveness of the rejuvenated wetland were significantly increased. However, the wetland has likely re-entered a start-up phase and long-term observation will be necessary to determine eventual steady-state conditions.

Keywords: Treatment wetland; Rejuvenation; Residence time distribution (RTD); Hydraulic efficiency; Phosphorus removal effectiveness; k–C^* model

1. Introduction

Treatment wetland ecosystems have proven to be an economic and effective water pollution control technology since the early 1970s (Kadlec and Knight, 1996). Currently, about 400 wetland ecosystems are operated in North America for treating wastewater from a variety of sources, including municipal wastewater, industrial wastewater, storm water runoff, and agricultural runoff (Knight et al., 1994; Finney, 2000). Of these treatment wetland ecosystems, about sixty per-

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cent are free-water surface treatment wetlands (Finney, 2000).

Treatment mechanisms in wetlands involve naturally abiotic and biotic processes, and their interactions, which can all be mediated by local climate, hydrologic processes and hydraulic processes. Inherently dynamic characteristics of these processes and interactions will eventually result in varying treatment effectiveness of wetlands. The treatment effectiveness may also decrease as the operational period increases (i.e. wetland age).

The Orlando Easterly Wetland (OEW) is one of the largest constructed wetlands for the treatment of wastewater in Florida and was constructed primarily for reducing nutrient loads in tertiary treated domestic wastewater produced by the City of Orlando before discharge into the St. Johns River. The OEW has performed better than design expectations since operation started in 1987, but after 13 years of receiving treated domestic wastewater effluent, phosphorus removal performance began to decline seasonally (City of Orlando, 2002; Black and Wise, 2003). In 2002, several wetland modifications were implemented in a portion of the OEW to maintain effective phosphorus treatment, and to ensure the longevity of the wetland for expected increases in phosphorus loading rates in the coming decade. The wetland rejuvenation program included removal of accrued sediment in the inflow region, re-grading, construction of baffles and islands, and re-vegetation. This study evaluates the improvement in hydraulic and phosphorus removal performance realized from these wetland modifications.

Because the OEW is one of the oldest large free-water surface treatment wetlands in North America (Finney, 2000), it is also one of the first to face the issue of treatment effectiveness decrease. Evaluation of the above wetland modifications and their corresponding effect on phosphorus removal performance may therefore be helpful for the management of other free-water surface treatment wetlands.

1.1. Site description

The OEW is a 490 ha municipal wastewater treatment wetland located in Christmas, Florida, approximately 3 km west of the St. Johns River, and 20 km east of the City of Orlando. The OEW was constructed on pastureland and began to receive advanced treatment wastewater discharged from the City of Orlando's Iron Bridge Regional Water Reclamation Facility in July 1987. Its primary function is to reduce total nitrogen (TN) and total phosphorus (TP) in treated water before discharging to the St. Johns River, and its secondary functions are to serve as wildlife habitat and a public amenity (park).

The wetland has a cell-network structure, which is comprised of 17 cells separated by berms and a lake (Fig. 1). Cells are connected through weir–culvert pairs constructed in the berms. Flow between cells can be controlled by adjusting the level of the rectangular weirs. The annual inlet flow rate is currently about 76,000 m^3/d (20 million gal/d, or mgd), and it has been permitted to receive up to 130,000 m^3/d (35 mgd). The OEW may be divided into three parallel flow trains (north, central and south) based on flow paths, or into five strata based on nutrient polishing functions and vegetative communities. Influent wastewater is typically discharged equally into the three flow trains by a three-way splitter box at the wetland inlet. Stratum 1 (cells 1, 2, 11, and 12), stratum 2 (cells 3–6) and stratum 3 (cells 8–10) are deep mash habitats composed mainly of cattails (*Typha* sp.) and bulrush (*Scirpus* sp.). These three strata were primarily designed for the bulk removal of nutrients from the wastewater. Stratum 4 (cells 13–15) has a mixed community of marsh vegetation and was designed for further nutrient polishing and wildlife habitat. Stratum 5 (cells 16a, 16b, and 17) was designed as a hardwood swamp for wildlife habitat; however, permanent inundation has led to a dominance of *Typha* sp.

Water quality is monitored daily at the wetland inflow and outflow locations, and monitored monthly at 10 sample stations located internal to the wetland. All internal stations, except one in the lake, are co-located with a weir (Fig. 1). Solute concentrations measured at the sample stations were assumed to represent the outlet concentrations of strata. For example, the concentration measured at location 8-X was assumed to represent the outlet concentration of stratum 3 at the north flow train, including weirs 7-X, 8-X and 8-Y. Water quality data from these sample stations were used to estimate the phosphorus profile in the wetland and the pollutant removal effectiveness of strata (i.e. Kadlec and Newman, 1992; Black and Wise, 2003; PBS&J 1990–1998; City of Orlando, 1999–2002).

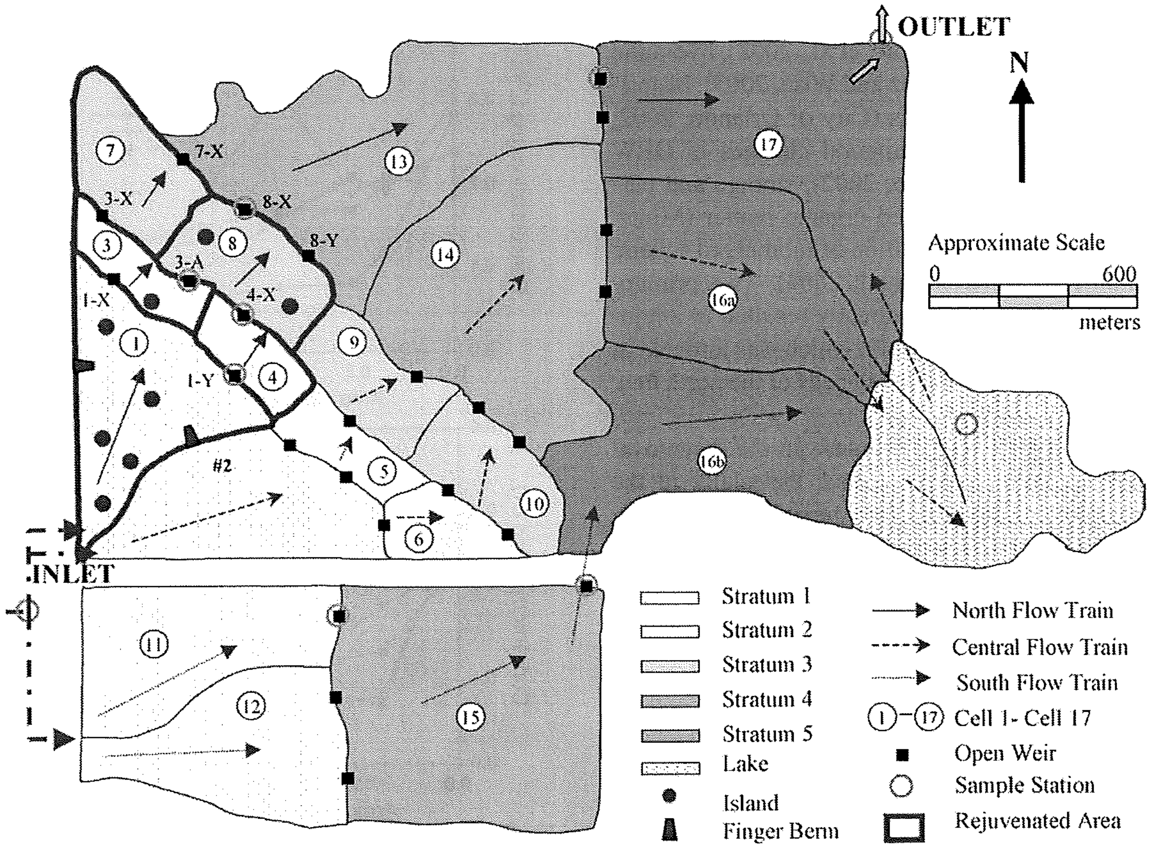

Fig. 1. Plan view of the Orlando Easterly Wetlands. The rejuvenated area of the north flow train is outlined in bold.

2. Wetland performance history

The OEW was designed primarily for polishing wastewater via the removal of total nitrogen (TN) and total phosphorus (TP). Nitrogen removal performance has been consistent over time with discharged TN concentrations from the wetland lower than 1.0 mg/L since it came online. This value is well below the permit level of 2.31 mg/L set by the Florida Department of Environmental Protection (FDEP). Total P discharge concentrations have also been below FDEP permitted levels (0.2 mg/L), but discharged P levels increased during the winter of 1999. This seasonal P spike has since been recurrent, with concentrations between 0.1 and 0.2 mg/L (Fig. 2). Although these spikes were within permitted levels, their occurrence indicated that phosphorus removal effectiveness has declined since 1999. A series of studies was initiated to investigate the P spikes, and to assess the P removal capacity of the wet-

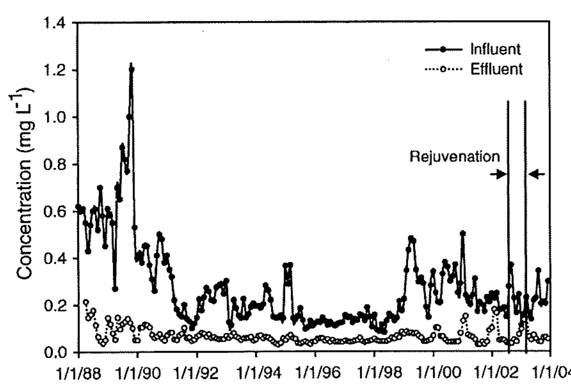

Fig. 2. Monthly average TP concentration at the OEW inlet and outlet from 1988 to 2003. [Data sources: 1988–1989 from Kadlec and Newman (1992), 1990–1998 from PBS&J (1990–1998), 1999–2003 from City of Orlando, 1999–2003].

land. These studies included P spatial gradients and distribution in the wetland (White et al., 2002), hydraulic efficiency analysis (Martinez and Wise, 2003), historical monitoring data analysis (City of Orlando, 2002; Black and Wise, 2003), temporal changes in OEW vegetation (City of Orlando, 2002), storage and partitioning of soil and accreted organic matter (Miner, 2001), and mesocosm evaluation of submerged aquatic vegetation (DB Environmental, 2004). Two primary findings from these studies directly resulted in a rejuvenation management program, which was initiated in August of 2002 for the front-end cells of the north flow train.

The first finding was a decrease in the P removal effectiveness through the wetland, especially in the front-end cells of the north flow train. The historical-average (1988-1998) TP data from internal sampling locations showed concentrations decreasing along the flow path through the wetland, but data from 1999 to 2002 showed higher concentrations throughout the wetland, although the concentration at the outlet had not significantly changed (Fig. 3). This trend was more pronounced in the north flow train (Fig. 3A) than in the south flow train (Fig. 3B). The elevated P concentrations in the front-end cells of the north flow train indicated ineffective P removal in these cells. Episodic spatially distributed water samples collected in April 2001 also showed select locations in the front-end cells of the north flow train with TP concentrations above the influent TP concentration (White et al., 2002). Such findings implied that some portions of the wetland had become TP sources rather than sinks due to increased P recycling from the sediment.

The second finding was that a significant shift of vegetative communities had occurred in the wetland over the past several years, including an increased cover of terrestrial species in front-end cells and a proliferation of floating aquatic vegetation (FAV) in back-end cells (DB Environmental, 2004). The coverage of terrestrial vegetation increased from 8% in 1998 to 48% in 2002, indicating that the front-end cells were under the processes of establishing a terrestrial ecosystem. The proliferation of FAV in the back-end cells resulted from enhanced downstream P concentrations. Decomposition of FAV and subsequent release of P stored in their tissue after the FAV senesce in winter was proposed as the reason for seasonal P spikes (DB Environmental, 2004).

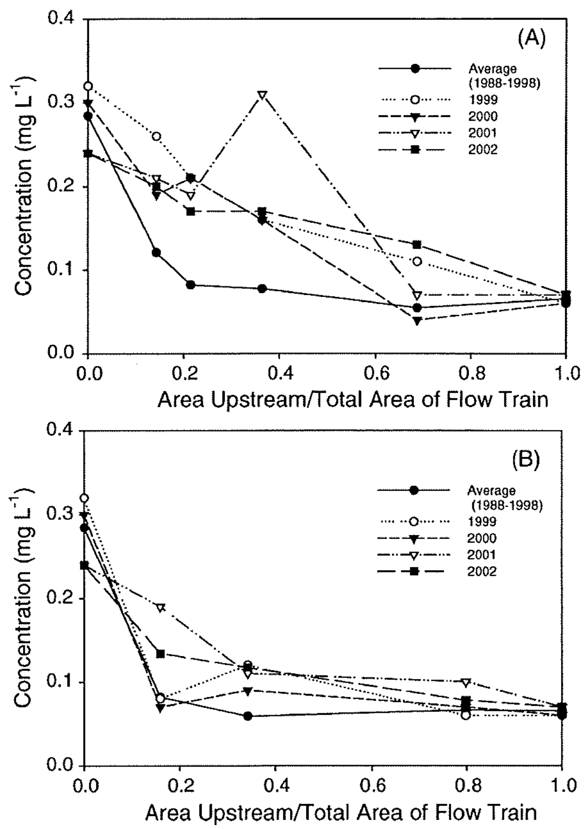

Fig. 3. Total phosphorus profile through the northern (A) and southern (B) flow trains.

The front-end cells of the northern flow train were also found by Martinez and Wise (2003) to have inefficient hydraulic performance. Persson and Wittgren (2003) identified ten design aspects that can affect the hydraulic performance of a treatment pond: (1) profile, such as bottom slope; (2) berms; (3) islands; (4) depth; (5) length–width ratio; (6) flow meandering; (7) wetland form or shape; (8) baffles; (9) inlet and outlet location; and (10) vegetation type and density. Practical cost considerations constrained factors such as the wetland shape and inlet and outlet location to remain as originally constructed. However, many of the other above-cited factors were amenable to modification for the improvement of hydraulic performance. For example, the construction of islands and berms would promote more tortuous flow paths, reducing the probability of short-circuiting directly from the inflow location to the outflow, and increasing the effective length–width ratio of the wetland. Also, site grading in the interior of the wetland cells would remove relict agricultural drainage ditches that might act as preferential flow paths and contribute to treatment short-circuiting.

Taken in their entirety, these findings indicated that the OEW treatment capacity was hindered by ineffective P removal in the front-end cells of the north flow train, which was in turn a combined result of encroaching terrestrial vegetation, low hydraulic efficiency, and increased P recycling from the sediments (City of Orlando, 2002). Therefore, restoration of these cells was chosen as the first step for rejuvenating the wetland. Muck removal followed by re-vegetation was chosen as the rejuvenation management program, in favor of other alternatives such as dry-down, dry-down followed by vegetation burning, dry-down followed by vegetation disking, and dry-down followed by a surficial application of alum (DB Environmental, 2004). Even though muck removal was more costly than the other options listed above, it was the only management alternative available that could address all three causal factors simultaneously for the front-end cells of the north flow train (City of Orlando, 2002).

3. Rejuvenation management

Rejuvenation management activities (muck removal followed by re-vegetation) were initiated in May of 2002. The front-end cells of the north flow train (Cells 1, 3, 4, 7, 8 and portions of Cell 13; total area of 59 ha) were drained and then mechanically scraped. The resulting vegetation, sediments, and organic debris generated approximately 130,000 m^3 of waste that was disposed of on-site as an upland soil amendment. The cell interiors were then mechanically graded, and spoil islands were constructed in Cells 1, 3, and 8 to help alleviate short-circuiting within the treatments cells, as well as provide wildlife habitat.

In Cell 1, five islands and two finger berms were constructed (Fig. 1). The diameter of each of these five islands is approximately 25 m, and the finger berms extend into the wetland about 30 m from the wetland boundary berm, with widths of 20–25 m. The islands and finger berms represent about 2% of the area of Cell 1. The northwest island in Cell 1, along with the island in Cell 3 and the two islands in Cell 8, were placed in topographic depressions where hydraulic short-circuiting was expected to occur. The island in Cell 3 is 70 m × 25 m, representing approximately 3% of the area of this cell. The south and north islands in Cell 8 are 35 m × 25 m and 50 m × 25 m, respectively, representing approximately 1.8% of the area of this cell. The construction of the four southern-most islands and finger berms in Cell 1 was based on the "bowling pin" approach. It was expected that the island and finger berm placement pattern would cause the water to take a tortuous path, reducing the likelihood of short-circuiting directly from the inflow location to the outflow of Cell 1.

Finally, over 160,000 giant bulrush (*Scirpus californicus*) were planted in the rejuvenated area. It was expected that the bulrush would create a dense monotypic stand and completely fill in the cells within 1–2 years (City of Orlando, 2002). The rejuvenated Cell 1 came online in January 2003, with Cells 3, 4, 7, 8 and 13 in May of 2003.

4. Effectiveness of the rejuvenation

The effectiveness of these activities for the improvement of hydraulic performance was evaluated through tracer tests. Pre-rejuvenation hydraulic performance was determined from tracer tests conducted in 2001 by Martinez and Wise (2003). Post-rejuvenation tracer tests are described in this work. The effectiveness of the rejuvenation for the improvement of phosphorus removal performance was evaluated through episodic spatially distributed water sampling as well as model prediction.

4.1. Hydraulic performance evaluation

4.1.1. Hydraulic efficiency

The hydraulic performance of the wetland cells was quantified in terms of hydraulic efficiency, η (Thackston et al., 1987; Martinez and Wise, 2003):

$$\eta = \frac{\tau}{t_N} \tag{1}$$

where τ is the mean residence time (or actual residence time) in a cell, determined from tracer tests, and t_N the nominal residence time, defined as:

$$t_N = \frac{V}{Q} \tag{2}$$

where V is the wetland cell (or stratum) volume [L^3], and $\bar{Q}$ the average flow rate entering and exiting the cell (or stratum) [L^3T^{-1}] (USEPA, 2000).

4.1.2. Post-rejuvenation tracer test

Two tracers, one cationic (lithium, Li^+) and one anionic (bromide, Br^-), were used in this study. The tracer solution injected into the wetland influent was created by mixing 300 kg of 99.7% KBr (2520 mol of Br^-) and 250 kg of 40% LiCl (2350 mol of Li^+) in a 3790 L polypropylene tank. After mixing for several hours, the tracer solution was injected over a period of 30 min into the north flow train from the splitter box at the wetland inlet on October 7, 2003. Water samples were collected at each open weir of the rejuvenated cells using 24-bottle automatic samplers. Initial sampling intervals were 2–8 h, depending on distance from the wetland inlet, and sample collection continued until tracer concentrations in samples from Cells 7 and 8 were reduced to background levels. The background concentrations for Br^- and Li^+ were 0.13 mg/L and below detection limit (<0.005 mg/L), respectively. Water samples were shipped to the University of Florida Soil and Water Science Department Laboratory for analysis. Lithium was analyzed by atomic absorption spectrometry (AA), and bromide was analyzed by high-pressure liquid chromatography (HPLC) using the method described by Martinez and Wise (2003).

4.1.3. Water flow rate

The average flow entering the OEW during the 2001 and 2003 tracer tests was 0.791 m^3/s (18.1 mgd) and 0.795 m^3/s (18.2 mgd). The inflow to the OEW is divided approximately equally among the three flow trains at a splitter box at the wetland inlet. Flow rates over the weirs in the OEW were calculated using the Kindsvater and Carter (1959) equation, which was developed for measuring flow in open channels:

$$Q = 0.554\left(1 - 0.035\frac{h}{P}\right) \times (b + 0.0025)\sqrt{g}(h + 0.001)^{3/2} \qquad (3)$$

where h is the measured head above weir crest in the approach channel (m), P the depth from the top of weir crest to channel bottom (m), b the measured weir crest width (m), and g the acceleration due to gravity (9.81 m/s^2). Water elevations above weir crest were recorded every 2 h using pressure transducers.

4.1.4. Cell volume

Cell bottom elevations were surveyed by the City of Orlando after muck removal and site grading. Cell volumes were calculated based on measured bathymetry and the mean water level recorded during the observation period. Water surface elevation was assumed to be a single value throughout each cell, as controlled by the outlet weir. The backwater profile, calculated using the method of French (1985) and Kadlec and Knight (1996), was found to contribute less than 2% additional volume in each cell and was thus not considered here.

4.1.5. Residence time distribution (RTD) analysis

The RTD of a wetland is the probability density function for the residence time of water parcels in the wetland. It was originally developed from chemical reactor theory (Danckwerts, 1953), and detailed by Kadlec and Knight (1996) for application in wetland studies. The RTD may be measured from the response of an impulse release of an inert tracer. The RTD function, $f(t)$ [T^{-1}], for a tracer released at $t=0$ may be expressed:

$$f(t) = \frac{Q(t)C(t)}{\int_0^\infty Q(t)C(t)\,dt} = \frac{Q(t)C(t)}{M_0} = \frac{M(t)}{M_0} \qquad (4)$$

where $C(t)$ [ML^{-3}] is the tracer concentration at the measurement location (here, the cell weirs); $M(t)$ [MT^{-1}] the mass flow rate; and M_0 the total tracer mass recovered at the weir [M]. For a cell with multiple outlet weirs, the RTD was determined from the lumped composite tracer response from the outlet weirs, as described by Martinez and Wise (2003).

The gamma distribution is among the functions commonly employed to describe RTDs (Nauman and Buffham, 1983; Jawitz, 2004):

$$f(t, N, \tau) = \frac{N}{\tau\Gamma(N)}\left(\frac{Nt}{\tau}\right)^{N-1} \exp\left(-\frac{Nt}{\tau}\right), \quad N > 0 \qquad (5)$$

where N is a real number, τ the mean residence time and $\Gamma(N)$ the gamma function:

$$\Gamma(N) = \int_0^\infty x^{N-1}e^{-x}\,dx \qquad (6)$$

If N is constrained to integer values, Eq. (5) reduces to the tanks-in-series (TIS) model of Kadlec and Knight (1996).

Field-measured breakthrough curves and RTDs, which are typically characterized by an early peak followed by a long, dispersed tail, have been shown to be well represented by the superposition of unimodal models such as the advection–dispersion equation, the TIS model, or probability density functions such as the lognormal and gamma distributions (Jawitz et al., 2003). The tracer data presented here were well represented by a two-path TIS model, with the resulting RTD expressed:

$$g(t) = \phi f(t, N_1, \tau_1) + (1 - \phi) f(t, N_2, \tau_2) \quad (7)$$

where the subscripts 1 and 2 refer to the two flow paths and ϕ the fractional contribution of the first path.

The ith moment, m_i, of the RTD is defined:

$$m_i = \int_0^\infty t^i f(t)\,\mathrm{d}t \quad (8)$$

Jawitz (2004) differentiated between the moments of complete distributions, which, as described by Eq. (8), are representative of the entire range of a distribution, and the moments of truncated (or incomplete) distributions where data are unavailable above or below some truncation point. Because of analytical detection limits and temporal constraints on field data collection, measured RTDs are by definition incomplete, usually exhibiting what is termed upper truncation where late-time, or tail, data are unavailable (Jawitz, 2004). Estimates of RTD complete moments may be obtained by considering the complete moments to be the sum of the moments of the measured data and the missing data in the distribution tail. The moments of the measured data were determined by numerical integration according to Eq. (8), but with limits of integration of $[0, t_{\max}]$. The missing concentration data in the RTD tail were assumed to follow an exponential decline (Sater and Leverspiel, 1966; Curl and McMillan, 1966; Martinez and Wise, 2003) in the interval $[t_{\max}, +\infty)$, and the flow rate was assumed to be approximately constant during this period. The moments of this portion of the RTD were determined from the truncated moment expression for the exponential distribution presented by Jawitz (2004).

The mean residence time in a cell was calculated from the difference between the first moments of the RTD at the cell outlet and inlet (subscripts out and in, respectively):

$$\tau = m_{1,\text{out}} - m_{1,\text{in}} \quad (9)$$

4.2. Phosphorus removal performance evaluation

The effectiveness of management activities for the improvement of wetland phosphorus removal performance can only be reliably evaluated through long-term phosphorus monitoring. In practice, such time frames should be longer than the length of "start-up" period for the wetland. During the start-up period, the removal of phosphorus is mainly achieved through the adsorption of P on fresh bottom soils as well as by the rapid vegetation expansion after initial planting (Kadlec and Knight, 1996). The operational history of the OEW suggests that the start-up period may be between 2 and 4 years. Kadlec and Newman (1992) reported that the vegetation density at the inlet zones for *Typha* sp. and *Scirpus* sp. increased approximately four-fold during the first two operation years, but appeared to stabilize in 1990. Likewise, the wetland uptake rate constant increased from 2.5 to 8 m/year during the first year of operation in 1987, fluctuated within a range of 5–10 m/year until 1991, and from 1991 to 1997 decreased and appeared to stabilize within a range from 3 to 5 m/year (Kadlec and Newman, 1992; Black and Wise, 2003).

4.2.1. Water quality sampling

Arithmetic averaging of monthly water quality samples provided annual average TP concentration profiles through the northern flow train at the wetland inlet and weirs 1-Y, 3-A, 4-X and 8-X (City of Orlando, 2002). Spatially distributed water samples were also collected at 31 sample stations in the interior of the OEW during two sampling episodes, one before rejuvenation management activities (April 2001) and one after (December 2003). The interior water quality sampling station locations were recorded with the global positioning system and samples were collected using a peristaltic pump with Nalgene tubing attached to a rod that was extended approximately 2 m away from the airboat and submerged under the water surface. Three purge volumes of surface water were flushed through the tubing before sample collection. Samples for TP were collected and preserved with concentrated H_2SO_4

(one drop per 20 ml) and placed on ice. Water samples were shipped to the University of Florida Soil and Water Science Department Laboratory for TP analysis using automated colorimetric techniques (Method 365.1, USEPA, 1993) after acid digestion.

4.2.2. Phosphorus removal performance model

The phosphorus removal performance was predicted using the steady-state first-order removal model (or k–C^* model) described by Kadlec and Knight (1996):

$$\frac{C_{\mathrm{out}} - C^*}{C_{\mathrm{in}} - C^*} = \exp\left(-\frac{k_{\mathrm{u}} A}{Q}\right) = \exp\left(-\frac{k_{\mathrm{u}} t_N}{d}\right) \quad (10a)$$

where C_{in} and C_{out} are observed concentrations [ML^{-3}] at the wetland inlet and outlet, respectively, C^* is the background concentration [ML^{-3}], k_{u} the areal uptake rate constant [LT^{-1}], A the wetland surface area [L^2], and d the average depth of the wetland [L]. Note that by considering the actual surface area of the wetland, this formulation of the k–C^* model is expressed in terms of the nominal residence time, defined by Eq. (2). However, for wetlands with hydraulic efficiency less than one, the mean residence time is less than the nominal time ($\tau = \eta t_{\mathrm{N}}$) and the effective wetland area, A_{e} is less than the actual surface area ($A_{\mathrm{e}} = \eta A$). Thus, when considering the actual residence time, the k–C^* model should be expressed as follows:

$$\frac{C_{\mathrm{out}} - C^*}{C_{\mathrm{in}} - C^*} = \exp\left(-\frac{k'_{\mathrm{u}} A_{\mathrm{e}}}{Q}\right) = \exp\left(-\frac{k'_{\mathrm{u}} \tau}{d}\right) \quad (10b)$$

where k'_{u} is the areal uptake rate constant for the effective wetland area.

The k–C^* model can be expanded beyond considering only the mean residence time by directly incorporating the entire RTD (Kadlec and Knight, 1996):

$$\frac{C_{\mathrm{out}} - C^*}{C_{\mathrm{in}} - C^*} = \int_0^{\infty} f(t) \exp\left(-\frac{k''_{\mathrm{u}} t}{d}\right) \mathrm{d}t \quad (11)$$

which, for a TIS RTD, can be expressed simply as (Kadlec and Knight, 1996):

$$\frac{C_{\mathrm{out}} - C^*}{C_{\mathrm{in}} - C^*} = \frac{1}{[1 + k''_{\mathrm{u}} \tau/(dN)]^N} \quad (12)$$

Note that consideration of the entire distribution of residence times requires the introduction of a third definition of uptake rate constant, k''_{u}.

Differentiating between k_{u}, k'_{u}, and k''_{u} is important for accurately evaluating the performance of treatment wetlands because the value of the areal uptake rate constant determined from fitting C_{in} and C_{out} data will depend on which formulation of the k–C^* model is used, with $k_{\mathrm{u}} \le k'_{\mathrm{u}} \le k''_{\mathrm{u}}$. For example, Black and Wise (2003) applied Eq. (10a) to the 1998 TP data from location 1-Y of the OEW and determined $k_{\mathrm{u}} = 18$ m/year. However, application of Eq. (10b) to the same data results in $k'_{\mathrm{u}} = 57$ m/year (using Cell 1 $\eta = 0.32$; Martinez and Wise, 2003). Finally, application of Eq. (12) to these data results in $k''_{\mathrm{u}} = 147.4$ m/year ($\tau = 0.74$ and $N = 6.35$, Martinez and Wise, 2003). Therefore, it is suggested that interpretation of literature-reported uptake constants should be made with caution, with special consideration of which formulation of the k–C^* model was used to determine these values. Note that reported areal uptake rate constants are often k_{u} because the actual wetland area is easily measured.

In this study, C_{in} was the concentration at the OEW inlet while C_{out} was the concentration at a given stratum outlet, and the reaction uptake rate constant was therefore an apparent parameter that specified P removal performance over the wetland area between these locations. This area is denoted hereafter as In-S$_i$ where i is the stratum number. For example, the area from the wetland inlet to stratum 1, equivalent to cell 1, was labeled In-S$_1$. The corresponding P removal performance analysis is referred to here as upstream analysis. Upstream analysis using Eqs. (10a), (10b) and (12) enables estimation of areal uptake rate constants when C_{in} and C_{out} are known, or inverse estimation of the maximum value of C_{in} that can be effectively treated when the other parameters are specified. These three k–C^* model formulations respectively consider the nominal residence time, the actual mean residence time (and by extension, hydraulic efficiencies less than one), and both the mean and spread about the mean of the actual RTD. The latter formulation is more comprehensive and was therefore used here, such that areal uptake rate constants reported here are k''_{u}. However, the use of Eq. (12) for estimating wetland P removal performance at flow rates other than that at which τ and N were determined is limited because the RTD, and consequently the RTD parameters τ and N, may be a function of flow rate (Eq. (4)).

Eq. (12) may be generalized to explicitly incorporate flow rate through consideration of the wetland RTD dimensionless variance, σ_θ^2, defined as (Kadlec and Knight, 1996, p. 243):

$$\sigma_\theta^2 = \frac{\sigma^2}{\tau^2} = \frac{1}{N} \tag{13}$$

where σ^2 is the RTD variance. Note that this definition of dimensionless variance is equivalent to the square of the RTD coefficient of variation, which is commonly used to describe the relative variability between data sets (Ott and Longnecker, 2001, p. 93). If the wetland RTD dimensionless variance is assumed to be constant, the wetland mean residence time and RTD variance measured at a reference flow rate can be related to the corresponding values measured at a different flow rate:

$$\tau = \frac{Q_r}{Q}\tau_r \tag{14a}$$

$$\sigma^2 = \frac{Q_r^2}{Q^2}\sigma_r^2 \tag{14b}$$

where the subscript r refers to the reference case. The assumption of constant RTD dimensionless variance is appropriate when wetland geometry and hydraulic efficiency are approximately constant, and flow changes do not cause significant changes in hydraulic characteristics of the wetland such as wetland volume. This assumption is likely reasonable throughout most of the OEW operating history, except 1997. The average flow rate with 95% confidence interval from 1990 to 2000 without 1997 was $0.655 \pm 0.064\,m^3/s$. In 1997, a hydraulic capacity test was conducted in the northern flow train, resulting in a flow rate about three times higher than the historical average. This high flow rate may have inundated regions that normally were isolated from the primary flow paths (Black and Wise, 2003). Furthermore, the hydraulic characteristics of the wetland are not expected to change dramatically as a function of flow in future operations because the maximum flow rate that the OEW is permitted to receive ($1.53\,m^3/s$) is less than double the current flow rate (about $0.8\,m^3/s$). Note that this assumption is equivalent to that commonly applied in wetland hydraulic studies (e.g., Arceivala, 1981; Kadlec and Knight, 1996, p. 254; Nameche and Vasel, 1998) of constant dispersion number using the plug flow with dispersion (PFD) model.

Combination of Eqs. (7), (12) and (14) results in the following k–C^* model for a two-path TIS RTD that can be applied under different flow rates:

$$\frac{C_{out} - C^*}{C_{in} - C^*} = \frac{\phi_r}{(1 + (k_u'' Q_r \tau_{1,r}/Q\, dN_{1,r}))^{N_{1,r}}} + \frac{1 - \phi_r}{(1 + (k_u'' Q_r \tau_{2,r}/Q\, dN_{2,r}))^{N_{2,r}}} \tag{15}$$

Note that when observed data are fit with Eq. (10), k_u is a function of wetland flow rate (Kadlec, 2000; Black and Wise, 2003); however, when Eq. (15) is used, k_u'' is the independent of flow rate because the entire RTD was used to obtain this relation (see Eq. (11)) (Carleton, 2002).

5. Results and discussion

5.1. Hydraulic performance improvement

The tracer response curves obtained for Br^- and Li^+ were virtually identical at all sampling locations, suggesting that both ionic tracers were nonreactive. Sample RTDs for both tracers at location 3-X are presented in Fig. 4D. Note that for Li^+, equivalent concentrations (the product of observed concentration, μM/L, and the injected mole ratio of Br^- and Li^+) are presented. Because of the similarity of the Br^- and Li^+ RTDs, only Li^+ was used for RTD analysis in the remainder of this study.

Average flow rate, mass recovered, and the RTD first moment for each weir are listed for both the 2001 and 2003 tracer studies in Table 1. The relative contributions of the exponentially extrapolated tail extensions to the RTD moments are also reported in Table 1. These results demonstrate that, as described by Jawitz (2004), the relative differences between incomplete moments calculated from observed data only ($t=0$ to t_{max}) and complete moments calculated from both observed data and tail extension ($t=t_{max}$ to ∞) increase with moment order. For example, the highest relative differences for the zeroth, first and second moments in the two tracer tests were 7.67%, 12.2%, and 39.7%, respectively.

The two-path TIS parameters determined from fitting Eq. (7) to the measured In-S RTDs obtained from the 2001 and 2003 tracer tests are listed in Table 2.

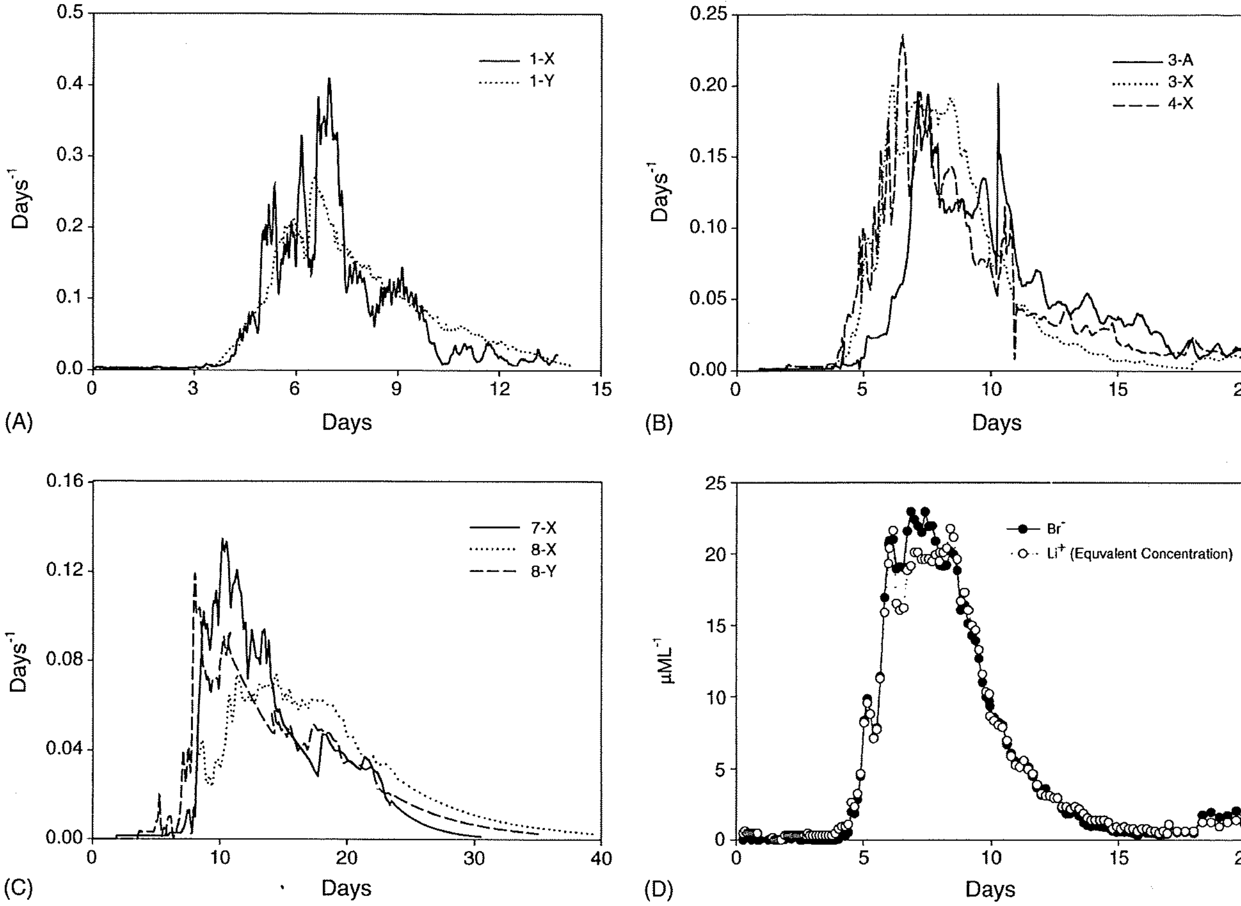

Fig. 4. Tracer residence time distributions measured in 2003 at (A) stratum 1, (B) stratum 2, and (C) stratum 3. Also shown with (D) 2003 Br^- and Li^+ tracer response curves measured at sample station 3-X. RTDs from Strata 1 and 2 in the figure do not include extrapolated tails, while those from stratum 3 include exponentially extrapolated tail extensions starting at $t = 25$ d.

Cell volume, nominal retention time and hydraulic efficiencies calculated from Eq. (1) are listed in Table 3 for both the pre- and post-rejuvenation tracer studies. The tracer results demonstrate a marked increase in hydraulic efficiency for these five cells following the wetland rejuvenation activities. The average hydraulic efficiency increased from 0.34 in 2001 to 0.74 in 2003.

5.2. Phosphorus removal performance improvement

Pre- and post-rejuvenation TP concentration spatial distributions in the front-end cells of the northern flow train are presented in Fig. 5. The average TP concentrations for the 31 sampling stations decreased from 0.463 mg/L (2001) to 0.048 mg/L (2003), a reduction of 90%. A paired t-test determined that the post-rejuvenation surface water TP concentrations were significantly lower with $p < 0.001$. Note that the observed reduction in surface water TP concentrations was achieved under similar TP loads: inflow TP to the OEW during the water quality sampling events were 0.20 mg/L in 2001 and 0.30 mg/L in 2003 with corresponding flow rates of 0.610 m^3/s (13.9 mgd) and 0.618 m^3/s (14.1 mgd), respectively (City of Orlando, 2003).

In 2001, TP aqueous concentrations were greater than 0.1 mg/L after three strata. In 2003, after rejuvenation, TP aqueous concentrations were lower than 0.05 mg/L after only two strata. In addition, in 2001 TP concentrations of some samples were higher than the influent TP concentration, suggesting that some por-

Table 1
Average flow rate and RTD moments at weirs

Weir	Average flow rate, Q (L/s)	Mass recovered M_0[a] (kg)	m_1 (days)	Percentage difference of RTD moments with and without tail extension[c]				t_{max}
				m_0 (%)	m_1 (%)	m_2 (%)	Variance (%)[b]	(Days)
2001								
1-X	142.9	216.4	1.23	1.06	9.36	37.9	49.2	9
1-Y	37.1	57.9	1.91	0.44	3.24	15.1	23.4	13
3-A	65.6	93.2	2.53	1.21	6.91	28.4	44.6	13
3-X	74.9	103.3	2.61	0.63	3.15	14.0	27.2	13
4-X	39.3	55.0	4.17	2.86	12.2	39.7	57.4	16
7-X	60.9	92.2	5.05	1.13	5.03	19.1	32.2	23
8-X	91.6	134.5	4.03	0.0002	0.001	0.004	0.01	27
8-Y	25.6	32.2	6.13	1.53	8.41	32.0	47.4	29
2003								
1-X	81.3	5.82	7.44	2.18	2.70	7.95	34.2	14
1-Y	101.3	9.27	7.70	0.86	0.93	2.59	10.7	15
3-A	13.7	1.00	11.3	3.16	4.32	11.9	30.2	23
3-X	97.0	5.95	8.71	1.56	3.05	10.3	35.7	21
4-X	106.4	6.18	9.22	2.07	3.04	8.57	22.4	21
7-X	99.0	6.00	14.1	0.72	0.88	2.46	8.46	25
8-X	10.2	0.50	18.3	7.67	8.32	21.1	53.2	25
8-Y	113.5	6.16	15.4	3.85	5.42	14.8	35.5	25

[a] Injection mass was 250 kg of KBr for 2001 and 16.33 kg of Li^+ for 2003. Mass recovery (%) for stratum 1 (cell1), stratum 2 (cell 3 and cell 4) and stratum 3 (cell 7 and cell 8) was 109.7, 91.7, and 102.9%, respectively, for 2001; and 92.5, 87.1, and 96.5%, respectively, for 2003.
[b] Variance $= m_2 - m_1^2$.
[c] Percentage difference (%) $= 100(m_{i\text{ with extension}} - m_{i\text{ without extension}})/m_{i\text{ with extension}}$.

Table 2
Two-path TIS parameters for modeling RTDs at strata outlets

Location	d (m)	N_1	τ_1 (d)	N_2	τ_2 (d)	ϕ	R^{2a}
2001							
In-S_1	0.31	93.6	0.56	5.18	1.13	0.33	0.98
In-S_2	0.39	22.2	1.64	4.42	2.93	0.42	0.98
In-S_3	0.40	8.67	10.0	4.51	3.06	0.19	0.96
2003							
In-S_1	0.60	11.8	9.15	27.5	6.49	0.49	0.97
In-S_2	0.52	13.9	13.9	17.3	7.64	0.22	0.98
In-S_3	0.59	11.6	18.0	27.3	10.5	0.58	0.95

[a] R^2: the non-linear coefficient of determination.

tions of the wetland had become TP sources rather than sinks. After rejuvenation, there was no evidence of the existence of P source zones in these areas.

While the episodic spatially distributed water samples only provided snap shot information of the TP distribution in the wetland, the k–C^* model of Eq. (15) enabled prediction of the long-term average removal. Post-rejuvenation model predictions of P removal performance were compared to analogous pre-rejuvenation model fits for the steady-state period from 1990 to 1998 and the period from 1999 to 2002. Before 1990, the wetland was in a start-up phase, while

Table 3
Physical parameters and hydraulic efficiency of cells

	Cell volume ($\times 10^3$ m^3)		Nominal residence time (days)		Hydraulic efficiency	
	2001	2003	2001	2003	2001	2003
Cell 1	70.8	136.5	4.34	8.65	0.32	0.88
Cell 3	30.9	24.4	2.57	2.62	0.52	0.62
Cell 4	32.3	15.6	8.90	1.69	0.25	0.90
Cell 7	67.0	86.3	12.9	10.1	0.19	0.53
Cell 8	29.7	83.8	2.94	7.84	0.44	0.77

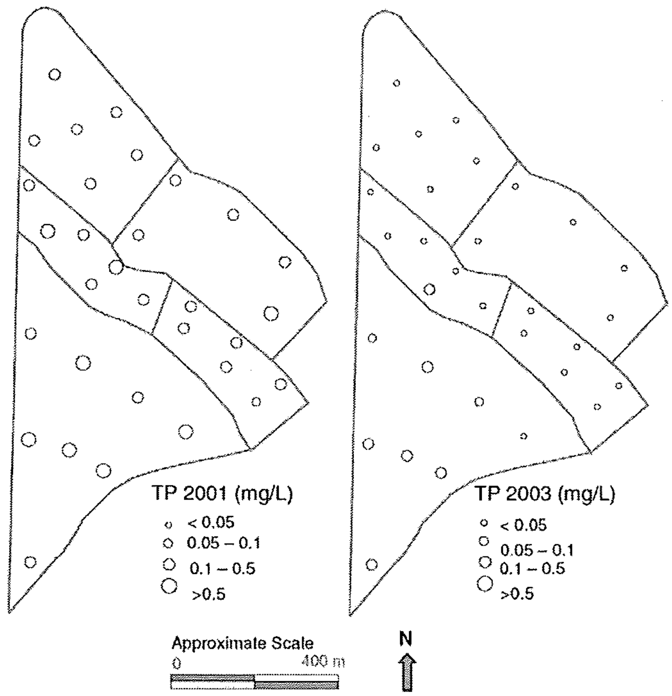

Fig. 5. Water column TP spatial distribution in the front-end cells of the northern flow train before (April 2001) and after (December 2003) rejuvenation.

the wetland treatment effectiveness was reduced during the period 1999–2002. Data from 1997 were excluded because of a hydraulic capacity test conducted that year.

The reference pre-rejuvenation hydraulic parameters were obtained from the 2001 tracer test (Table 2). Martinez and Wise (2003) concluded that, because no site grading was conducted in the interior of the wetland cells when the OEW was originally constructed, short-circuiting and dead zones were largely the result of historic land alterations, such as relict agricultural drainage ditches. Therefore, it was assumed here that the hydraulic parameters measured during the 2001 tracer test were generally representative of the entire pre-rejuvenation period. Post-rejuvenation predictions used the hydraulic parameters obtained from the 2003 tracer test and the pre-rejuvenation values for k_u'' and C^*. Note that this assumption implies that any predicted improvements in P removal performance were the result of improvements in hydraulic performance only. Organic sediment removal and re-vegetation were of course expected to affect k_u'' and C^* through the creation of new substrate sorption sites, changes in P uptake rates and capacities from different microflora and vegetation, and reduced P recycling from the new sediment substrate. However, these effects were expected to be manifested primarily during a start-up period following rejuvenation. Once treatment approaches steady-state conditions, k_u'' and C^* are expected to return to pre-management values. Nevertheless, the actual changes in steady-state k_u'' and C^* effected by the management activities will only become clear following several years of additional monitoring.

The values of k_u'' and $C*$ used here were calibrated by fitting Eq. (15) to the observed TP interior profiles in the front-cells of the northern flow train from 1990 to 1998, excluding 1997 (data from City of Orlando, 2002), when the wetland treatment was approximately steady state. The average influent TP concentration from 1990 to 1998 was 0.213 mg/L with $\bar{Q} = 0.656\,m^3/s$ (13.9 mgd). From 1999 to 2002, the average influent TP concentration was 0.287 mg/L with $\bar{Q} = 0.784\,m^3/s$ (17.9 mgd). Post-rejuvenation model predictions were also conducted for a flow rate of $1.531\,m^3/s$ (35.0 mgd), the maximum flow rate that the OEW is permitted to receive (City of Orlando, 2002).

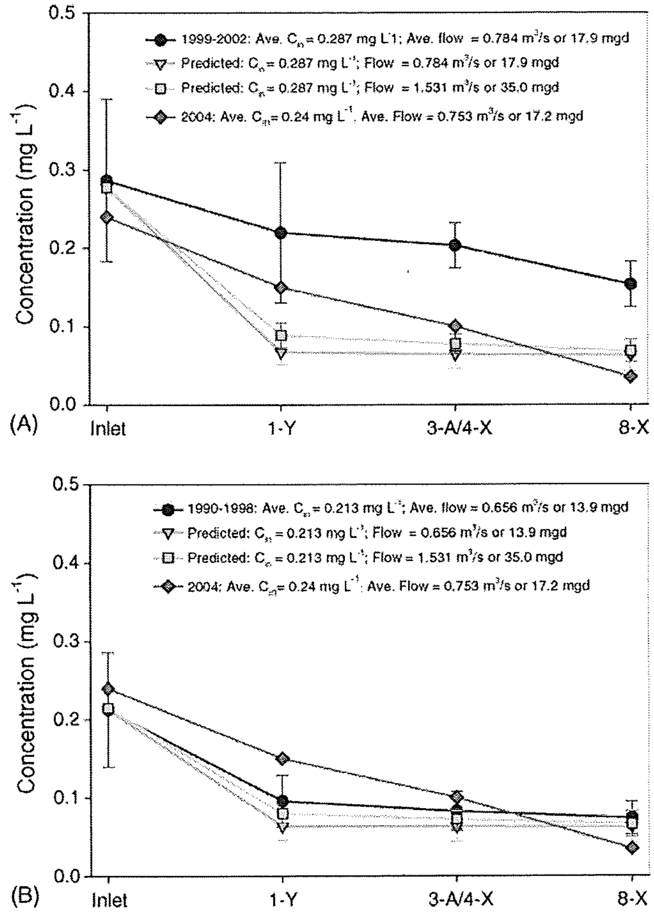

Fig. 6. Historical observed pre-rejuvenation and predicted post-rejuvenation TP profiles (mean with 95% confidence interval), and the profile of 2004. In (A), the historical observation was from 1999 to 2002, when the wetland treatment performance was impaired. In (B), the historical observation was from 1990 to 1998, when the wetland treatment performance was approximately steady-state.

Fitting and prediction were conducted using the nonlinear regression procedure PROC NLIN in SAS.

The k_u'' and C^* mean values, with 95% confidence interval, determined from fitting Eq. (15) to the pre-rejuvenation TP profiles were 127.9 ± 59.2 m/year and 0.062 ± 0.021 mg/L, respectively, with the nonlinear coefficient of determination $R^2 = 0.923$. The predicted post-rejuvenation water column TP profiles are compared in Fig. 6 to model fits for both the steady-state and impaired pre-rejuvenation periods. These model results predicted improved post-rejuvenation P removal performance compared to both pre-rejuvenation periods, significantly so for the 1999–2002 impaired period (Fig. 6A). As the wetland volume was assumed to be constant, increased flow rates resulted in decreased mean residence times and, therefore, decreased removal effectiveness. However, even at the highest permitted flow rate, post-rejuvenation performance was still predicted to be superior to either pre-rejuvenation period.

Accurate assessment of the effects of rejuvenation management on the wetland P removal performance will require long-term monitoring. However, TP profiles from 2004, the first full year following rejuvenation, are compared to both historic data and predicted performance in Fig. 6. The average inlet TP concentration for 2004 was 0.24 mg/L, which was comparable to the steady-state period value of 0.21 mg/L. The annual average TP profile from 2004 indicates measured concentrations at 1-Y and 3-A/4-X that were lower than those measured during the impaired period (Fig. 6A), but higher than those measured during the steady-state period (Fig. B). Fitting Eq. (11) to the 2004 TP profile resulted in $k''_u = 30$ m/year and $C^* = 0.04$ mg/L. These data indicate that the rejuvenation has resulted in improved P removal performance compared to the impaired period, but it seems that as of 2004 the rejuvenated wetland had not re-attained the performance of its historic steady-state period. This observation seems to conflict with the intuitive expectation that the rejuvenated area should have better P removal performance shortly after rejuvenation, because of the creation of new substrate sorption sites, reduced P recycling from the new sediment substrate, and high P uptake rates from rapid vegetation expansion after initial planting. In addition, the 2004 data seem counter to historic observations in the OEW, where the P removal performance in the start-up period before 1991 was better than that of the steady-state period 1991–1998 (Kadlec and Newman, 1992; Black and Wise, 2003). However, it should be noted that post-rejuvenation TP concentrations at the outlet of the rejuvenated area (8-X) were lower than even the pre-rejuvenation steady-state values. It is not clear at this time whether the wetland performance in 2004 was more representative of a post-rejuvenation steady-state or merely a transient start-up period.

It should be emphasized that it is difficult to predict long-term improvement of the P removal performance based on a short period of observation. Furthermore, it is difficult to quantitatively assess how much of the observed post-rejuvenation performance improvement was due to hydraulic performance improvement, re-vegetation or the removal of accrued sediments. Therefore, the predictions in Fig. 6 should be viewed as approximate estimates of P removal performance in a post-rejuvenation steady-state period. More precise assessment of the rejuvenation management on the wetland P removal performance will require long-term studies.

6. Summary

Muck removal followed by re-vegetation in the front-end cells of the OEW northern flow train was chosen as the wetland management strategy for improvement of P removal performance that had become hindered by the combined effects of encroaching terrestrial vegetation, low hydraulic efficiency, and increased P recycling from the sediments.

The hydraulic performance of the rejuvenated wetland was significantly improved. The average hydraulic efficiency of the rejuvenated cells increased from 0.34 to 0.74.

Both the episodic spatially distributed water samples and the long-term model prediction indicated that the phosphorus removal effectiveness in the rejuvenated area has been improved. However, the wetland has likely re-entered a start-up phase and long-term observation will be necessary to determine eventual steady-state conditions.

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