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Management scenario evaluation for a large treatment wetland using a spatio-temporal phosphorus transport and cycling model

Rajendra Paudel, Joong-Hyuk Min, James W. Jawitz*

Soil and Water Science Department, University of Florida, Gainesville, FL 32611, USA

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ABSTRACT

This paper describes the development of a two-dimensional, spatially distributed model to simulate coupled hydrologic and phosphorus (P) biogeochemical processes in a 147-ha cell of a 1544-ha stormwater treatment wetland designed to help protect the greater Everglades, FL, USA. The model was used to assess the effects of a suite of feasible management alternatives on the long-term ability of the wetland to sustain total P (TP) removal. The spatial and temporal dynamics of TP retention were simulated under historical (1995–2000) conditions, and under assumptions of removal of short-circuiting channels and ditches, changes in external hydraulic and TP loading, and long-term (>20 years) impacts on soil and water column TP dynamics under current and reduced load conditions. Internal hydrology and transport processes were calibrated against measured tracer concentrations, and subsequently validated against outflow discharge and spatial chloride concentration data. Cycling of P was simulated as first-order uptake and release, with different uptake coefficients for open water/sparse submerged aquatic vegetation (SAV) areas (0.2 day^{-1}) and dense SAV areas (0.4 day^{-1}), and a much lower, uniform release coefficient (1.97×10^{-4} day^{-1}). The calibration and validation of the P model showed good agreement with field measurements of water column TP concentrations measured at the wetland outlet (calibration RMSE = 10.5 $\mu g\,L^{-1}$; validation RMSE = 15.6 $\mu g\,L^{-1}$). Under simulated conditions of preferential channels eliminated, average annual TP treatment effectiveness increased by 25%. When inflow TP loads were assumed to be eliminated after 6 years of loading, the release of accumulated soil P sustained predicted annual average outlet concentrations above 6.7 $\mu g\,L^{-1}$ for 18 years, decreasing at a rate of 0.16 $\mu g\,L^{-1}\,yr^{-1}$. Sensitivity analyses indicate that the most critical model input factors include flow resistance parameters, initial soil TP content, and P cycling parameters compared to initial water level, initial TP concentration in water column, ET and transport parameters.

1. Introduction

Over the past several decades, the Florida Everglades have been adversely impacted due to altered hydrologic regimes and elevated nutrient conditions. Specifically, agricultural drainage waters discharged into the northern Everglades have been enriched in phosphorus (P) compared to the historic rainfall-driven inputs, which has contributed to eutrophication and posed a significant threat to biodiversity and function in the Everglades (Chimney and Goforth, 2001; McCormick et al., 2002). To reduce the P in agricultural runoff before entering the Everglades, the 1994 Everglades Forever Act (EFA; Section 373.4592, Florida Statutes) provided direction to the South Florida Water Management District (SFWMD) to construct six large-scale treatment wetlands, known as Stormwater Treatment Areas (STAs). These STAs now comprise 18,200 ha and have been effectively maintaining discharge concentrations of total P (TP) well below the current legally mandated standard of 50 $\mu g\,L^{-1}$ (Pietro et al., 2009).

Because the P removal performance of treatment wetlands is affected by a combination of management and natural conditions, including vegetation type/distribution, soil conditions, and water chemistry (Kadlec and Wallace, 2008), it is often challenging to maintain or optimize the effectiveness of these wetlands indefinitely. A wide variety of models have been developed to help wetland managers understand these interactions and to predict their effects on fate and removal of P in wetland environments. An empirical mass balance approach based on input-output analysis is the simplest model used to describe P retention in various wetland systems (Kadlec and Newman, 1992; Walker 1995; Wong and Geiger, 1997; Moustafa, 1998; Black and Wise, 2003; Kadlec and Wallace, 2008). This approach has been often used to estimate the size and predict the performance of the treatment wetland by

* Corresponding author. Tel.: +1 352 392 1951x203.
E-mail address: jawitz@ufl.edu (J.W. Jawitz).

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modeling them as plug flow reactors in which nutrient load undergoes first-order removal under steady-state conditions. Models of these types assume non-dispersive, unidirectional, steady-state plug flow. Real wetlands do not behave as ideal plug flow reactors but exhibit conditions of dispersive flow caused by the presence of heterogeneous vegetation (Wörman and Kronnäs, 2005), and also are often subject to transient, non-steady inputs. Therefore, these models may not be adequate to predict wetland performance for conditions such as varying flow regime and altered vegetation type/density because the removal rate constant is not independent of model input conditions such as mass loading rate and residence time distribution (Kadlec, 2000; Wang and Jawitz, 2006). Walker and Kadlec (2005) developed a dynamic model for STAs (DMSTA) to simulate transient flow conditions and event-driven performance in treatment wetlands. DMSTA simulates daily water and mass balances in a user-defined series of treatment cells, each with specified hydrodynamics and P cycling parameters, which were determined from several wetland systems in Everglades. A maximum of six treatment cells can be linked in series or parallel compartments, and each cell is further divided in a series of continuous tank reactors to reflect the residence time distribution.

Alternatively, mechanistic, compartmental models that include interacting submodels for hydrology and P have been adopted by several researchers (e.g., Mitsch and Reeder, 1991; Christensen et al., 1994; Wang and Mitsch, 2000). However, in these models, surface-water flow through the wetland is described as simply a water mass balance, and thus predicting the effects of spatial variability arising from irregular locations of flow structures, irregular geometry, and heterogeneity in the system is limited. In addition, simplified P dynamics (described by a single settling rate coefficient) have been linked to wetland-scale two-dimensional (2-D) dynamic flow models (Tsanis et al., 1998; Raghunathan et al., 2001) to simulate water column TP behavior. Some attempts have been made to develop spatially explicit, mechanistic P models coupled with hydrodynamic models within the context of Everglades restoration (HydroQual, Inc., 1997; Fitz and Sklar, 1999). For example, a spatially distributed, transient Wetland Water Quality Model (WWQM) was developed based on mass transport and kinetic equations of nutrients in water, sediment, and emergent vegetation to simulate the P removal processes, particularly in STAs (HydroQual, Inc., 1997). The complexity of the biological–chemical processes included in this model (over 200 parameters) resulted in calibration difficulty and the model has not been adopted for management purposes.

The spatial variability in wetland plant communities, bottom elevation, and location of flow structures can influence the spatial distribution of P uptake and release within the wetland. Also, P transport and cycling processes are strong functions of inflow loadings, which are often dynamic in a hydrologically managed system such as STAs. Many management decisions may require spatial and temporal information, such as identifying high-P accumulation areas for potential cattail expansion, spatial depth regime to identify the suitability for desired plant communities, and long-term predictions to evaluate sustainability of wetlands. The utility of a model for management support to optimize P removal potential in a STA may largely depend on the ability to predict the spatio-temporal response of the system to changes in internal characteristics as well as external loads. Thus, there is a need to develop a spatially distributed model with the capability to integrate high resolution spatial input data and predict the dynamic response of the system under a variety of management conditions.

A recently proposed approach for treatment wetland modeling is based on coupling flow and transport models that can embrace both temporal and spatial heterogeneity with flexible reaction algorithms (Jawitz et al., 2008). This paper describes the application

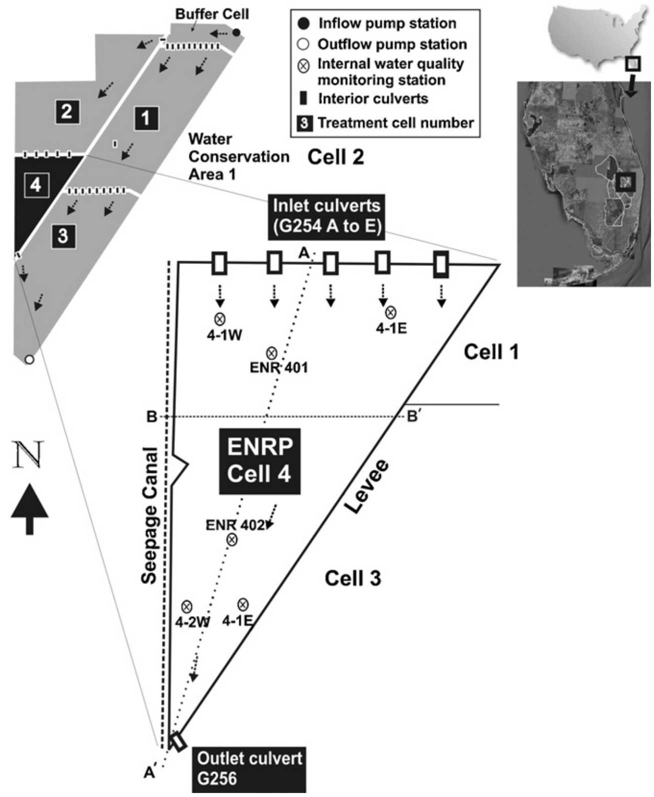

Fig. 1. Location and plan view of study area, Cell 4 of ENR Project.

of an integrated hydrologic, transport/reaction model to simulate spatial and temporal variation of TP dynamics in response to the variety of forcing functions in a stormwater treatment wetland of the northern Everglades. The coupled movement of overland flow and groundwater flow are simulated using the 2-D, Regional Simulation Model (RSM), originally designed to simulate regional flows across south Florida (SFWMD, 2005b). The transport/reaction model (RSMWQ, James and Jawitz, 2007) is internally coupled with RSM to provide hydrologic information (i.e., depth, velocity field) needed to accurately simulate the transport processes. RSMWQ uses flexible, user-defined reaction algorithms so that the modeler can incorporate physically based descriptions of biogeochemical cycling in aquatic systems. The paper first describes the model development, calibration and validation with extensive field data from Cell 4 of the Everglades Nutrient Removal (ENR) Project, one of the most-studied treatment wetlands in the Everglades (Dierberg et al., 2005; Reddy et al., 2006). Then, the model is implemented to investigate the short- and long-term effects on spatio-temporal TP behavior in soil and the water column under various feasible management alternatives such as removing short-circuiting channels/ditches, and changes in external hydraulic and TP loadings.

2. Site description

Cell 4 is one of the treatment cells of Stormwater Treatment Area 1West (STA 1W, formerly known as the ENR Project), which is located in central Palm Beach County, along the northwestern boundary of Water Conservation Area 1 (WCA-1) and on the eastern boundary of the Everglades Agricultural Area (EAA) in south Florida (26°38′N, 80°25′W, Fig. 1). Cell 4 is a 147 ha marsh dominated by submerged aquatic vegetation (SAV), and had proven to be the most effective of the four large ENR cells for P removal (DBEL, 2002). The ENR Project was initiated in August 1994 as a buffer to

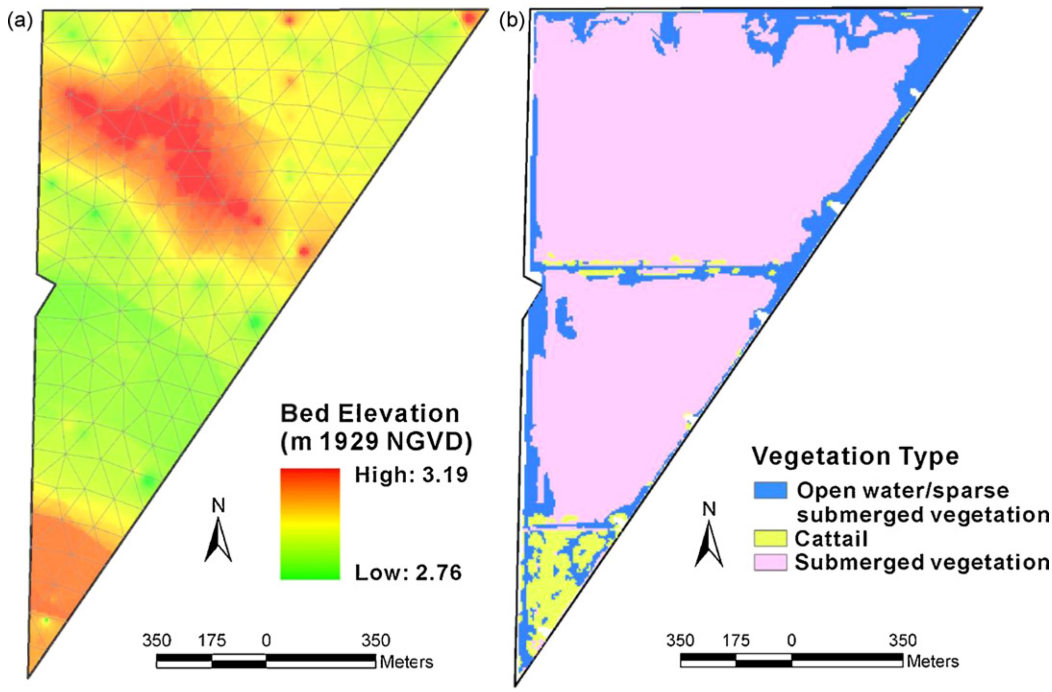

Fig. 2. Cell 4 of the ENR Project (a) model mesh and bathymetry and (b) vegetation type and coverage. The mean bed elevation of raster cells within each mesh element was considered in the model.

reduce concentrations of nutrients such as P from EAA drainage waters before entering the adjacent natural wetlands (Chimney et al., 2000). Cell 4 was flooded during the study period (1995–2000); thus the dominant mechanism for flow and transport was overland flow. The primary inflow to Cell 4 consisted of surface-water from Cell 2 through G254 (5 culverts), with outflow through G256 (5 culverts) (Fig. 1). A seepage return canal adjacent to the western levee was used to collect the groundwater seepage (approximately 11% of outflow discharge, Nungesser and Chimney, 2006) from the wetland. As the ENR Project was built on former agricultural land, abandoned agricultural ditches and borrow canals remained in the wetland (Guardo and Tomasello, 1995; Dierberg et al., 2005). The open-water borrow canals oriented parallel to the primary flow direction along the levees were up to 1.2 m deeper than the average Cell 4 ground elevation, creating preferential flow conditions (Dierberg et al., 2005).

The study area was actively maintained with SAV and periphyton community dominated by coontail (*Ceratophyllum demersum*) and southern naiad (*Najas quadalupensis*) with less abundant pondweed (*Potamogeton illinoensis*) and cattail (*Typha sp.*) in small patches along the east–west canal in the middle of the wetland and near the outlet structure (Chimney et al., 2000, Fig. 2b). Although there was temporal variation in the vegetation community during the model application period (1995–2000), *Najas* remained the dominant vegetation, followed by *Ceratophyllum*, usually found in the inflow zone (DBEL, 2002). For detailed information about Cell 4 characteristics and operational histories, the reader is referred to Chimney et al. (2000) and Chimney and Goforth (2006).

3. Methods

3.1. Modeling framework

The P transport and cycling model was developed based on the framework of RSMWQ (James and Jawitz, 2007) which was internally embedded with the Hydrologic Simulation Engine (HSE) of RSM developed by the SFWMD. The RSMWQ is a linked-library model that runs with HSE which provides hydrologic information that is needed to simulate P transport mechanisms. RSMWQ is constructed in two parts: (1) transport (e.g., advection and dispersion) of soluble nutrients, and (2) a flexible biogeochemical reaction module. Information about topography, mesh, and run time step is controlled by the hydrologic model.

3.1.1. Hydrodynamic model

A physically based, spatially distributed flow dynamics model of Cell 4 was developed based on the framework of HSE, which simulates the coupled movement and distribution of overland and groundwater flow. In addition, HSE also simulates hydraulic structures, canal networks, well pumping, levee seepage, and other operational rules and conditions unique to the Everglades treatment wetlands. In HSE, governing equations that describe the physical processes of fluid flow are based on the depth-averaged Saint-Venant equation. The acceleration terms of the momentum equations are neglected for the diffusion flow assumption, which has been successfully applied for regional conditions in South Florida (Lal, 1998, 2000). In RSM, the weighted implicit finite-volume method is employed in which the continuity equation is expressed in integral form over an arbitrary control volume that satisfies strict mass balance because of conservative properties (Lal, 1998). Earlier studies have demonstrated the potential application of HSE for the shallow flow conditions across south Florida at the basin scale (approximately 10^4 km^2, Lal et al., 2005), and a relatively small area of ridge and slough landscape in central WCA-3A (6 km^2, Min et al., 2010). For detailed description of RSM, the reader is referred to SFWMD (2005b,c).

3.1.2. Water quality model

The transport/reaction algorithm of RSMWQ was used to simulate 2-D transport and transformation of P in Cell 4. RSMWQ was designed to simulate water quality conditions (primarily P) and ecosystem responses to hydrologic and water management changes in wetland environments. RSMWQ includes a flexible biogeochemical module which allows the user to incorporate a wide range of biogeochemical process reactions between various wetland components (e.g., water, soil, macrophyte and plankton). RSMWQ simulates reactive transport of mobile materials in variable-depth water bodies, and numerically solves the advection-dispersion-reaction equation. RSMWQ can be applied to problems with irregular boundary geometries as it solves a finite element problem on an unstructured triangular mesh in which each mesh has a unique response function. The accuracy of the model was determined by comparing analytical solutions with simulations in one and two-dimensions with non-reactive and reactive transport (James and Jawitz, 2007). Here, the model is used to simulate 2-D reactive transport in a large field-scale wetland with variable depth flow. For detailed information about RSMWQ discretization, boundary conditions, and numerical solution algorithms, the reader is referred to James and Jawitz (2007).

3.2. Phosphorus cycling model

The P cycling model implemented here was based on the primary mechanisms regulating TP behavior in soil and the water column, and all TP removal processes were lumped as soil–water uptake and release to describe the exchange of TP between soil and the water column.

$$\frac{dS_{\mathrm{TP}}}{dt} = k_{\mathrm{u}} d\theta_{\mathrm{sw}} C_{\mathrm{TP}}^{\mathrm{SW}} - k_{\mathrm{r}} S_{\mathrm{TP}} \quad (1)$$

$$\frac{dC_{\mathrm{TP}}^{\mathrm{SW}}}{dt} = -k_{\mathrm{u}} C_{\mathrm{TP}}^{\mathrm{SW}} + \frac{k_{\mathrm{r}}}{d\theta_{\mathrm{sw}}} S_{\mathrm{TP}} \quad (2)$$

in which, S_{TP} = soil TP [M L^{-2}]; $C_{\mathrm{TP}}^{\mathrm{SW}}$ = water column TP [M L^{-3}]; k_{u} = first-order uptake rate constant [T^{-1}]; k_{r} = first-order release rate constant [T^{-1}]; d = depth of water column [L]; θ_{sw} = water content in water column (i.e., reduced volume due to vegetation) [L^3 L^{-3}].

The uptake/release terms are represented by volumetric first-order reaction rate constants which reflect the net effects of several combined physical and biochemical processes of P removal (Eqs. (1) and (2)). More detailed description of P cycling processes could include P forms such as soluble reactive P (SRP), particulate P (PP) and dissolved organic P (DOP). However, accurate simulation of processes associated with these state variables requires more types of data than are available for the STAs. Therefore, a simple P cycling model was selected with processes that were particularly supported with available data, in contrast to the more mechanistic approaches such as those of HydroQual, Inc. (1997) and Wang and Mitsch (2000). Further, by representing both uptake and release of P, we extend previous STA models that employed a single, net settling rate coefficient (Walker, 1995; Kadlec and Wallace, 2008).

3.3. Data sources

The bulk of the field measurement data (i.e., hydrological, meteorological and water quality) employed in this study were collected by SFWMD personnel and are publicly available on their online environmental database, DBHYDRO (http://my.sfwmd.gov/dbhydroplsql/show_dbkey_info.main_menu). Flow data were typically available as a daily average for each flow control structure. Chloride concentrations at internal monitoring stations (Fig. 1) were sampled bi-weekly/monthly using grab samples. Similarly, TP concentrations at inflow and outflow structures were collected weekly from time-proportioned composite samples using autosamplers. Spatial data for topography, vegetation, and soil TP were provided by SFWMD personnel. Rhodamine-WT tracer concentration data from Dierberg et al. (2005) were used to calibrate the hydrologic and transport components of the model. Wet deposition of TP, 10 μg L^{-1}, was based on an earlier study conducted at ENR site (Ahn and James, 2001).

3.4. Hydrologic and transport model simulations

Cell 4 was represented by a 2-D finite element mesh of 298 unstructured triangular elements (average area: 5100 m^2) and 192 nodes, generated in Groundwater Modeling System (GMS) v6.0. The mesh density was further refined in some specific areas to better represent the location of flow structures. The bathymetry was generated using inverse distance weighting from the topographic survey data. The bottom elevation in Cell 4 ranged from approximately 2.7 to 3.2 m (NGVD 29) (Fig. 2a). The internal hydrology and transport processes calibration period of December 16, 1999 to January 15, 2000 (31 days) was selected based on the concurrent tracer test of Dierberg et al. (2005). The validation period was January 10, 1995 through December 31, 2000 for outlet discharge, and from January 10, 1995 through December 31, 1999 for chloride concentrations. Chloride was assumed to act as a conservative tracer that provided a basis for further testing of water balance and transport processes.

Depth-dependent hydraulic resistance was simulated with a power function of the following form (SFWMD, 2005a,b):

$$n = Ad^B; \quad \text{for } d > d_t \quad (1)$$

where n = Manning's flow resistance factor [T M$^{-1/3}$]; A and B are empirical constants, usually determined through model calibration; and d_t is detention depth [L] which was assigned 0.01 m throughout the model domain. The values of A and B primarily depend on the wetland habitat (vegetation type) or vegetation density. The model area was divided into three primary habitats: SAV, cattail, and open water/sparse submerged vegetation. The channels and ditches parallel to the flow direction along the levee were represented by open water/sparse submerged vegetation (Dierberg et al., 2005). Calibration was used to determine final A values for these three zones. Previous work found B = −0.77 to be appropriate for most Everglades wetland plant communities (SFWMD, 2005a) and this value was used here for all vegetation types. The groundwater flow resistance was described by the hydraulic conductivity (k = 3.5 × 10^{-4} m s^{-1}, Harvey et al., 2000) which was assigned spatially constant throughout the model.

To simulate the tracer test of Dierberg et al. (2005), a constant flow-weighted tracer concentration (6.78 mg/L) was assigned in all five inflow culverts for 35 min on 16 December 1999. The average stage (3.48 m) measured immediately upstream of G256 culvert was used as a constant head boundary to simulate flow at the outlet structure. The initial water surface elevation was set to 3.64 m throughout the domain. In Cell 4, there was a significant difference in stage between the wetland and the adjacent seepage return canal along the western boundary; therefore, the flow in the surficial aquifer was driven towards the canal from the wetland. Nungesser and Chimney (2006), in studies of ENR Project hydrology for the period from January 13, 1995 to June 20, 1999, found that the seepage from the wetland to the canal across the western levee was about 11% of outflow discharge. However, groundwater inflow was believed to be a minor component of the water budget (Chimney et al., 2000) because the head difference between Cell 4 and adjacent treatment Cell 1 was relatively small. Therefore, seepage inflow was not modeled here, whereas the seepage outflow was modeled using the cell head boundary condition which was based on the difference in stage between the wetland and the seepage return canal. The value of the seepage coefficient (K_s) that controls the rate of flow through the western boundary wall was estimated based on the residual term of the water budget to make seepage close to the mass

Table 1
Primary model parameters/inputs, descriptions, values and sources.

Symbol	Description	Value	Sources
Initial conditions			
–	Water level, m	3.64	DBHYDRO
C_{TP}^{SW}	Water column TP concentration, $\mu g\,L^{-1}$	40	DBHYDRO
S_{TP}	Soil TP, $g\,m^{-2}$	Varies	Unpublished data, SFWMD
Parameters			
–	Outlet boundary head, m	3.48	DBHYDRO
A	Empirical constant of flow resistance, $m^{0.44}\,s$		
	SAV	0.72	Calibrated
	Cattail	1.28	Calibrated
	Open water/sparse submerged vegetation	0.065	Calibrated
B	Empirical constant of flow resistance, unitless	−0.77	SFWMD (2005a)
K_{veg}	Vegetation reference crop potential ET correction coefficient, unitless (0–1)	0.64	Calibrated
k	Hydraulic conductivity, $m\,s^{-1}$	3.5×10^{-4}	Harvey et al. (2000)
K_S	Seepage coefficient, $m^2\,s^{-1}$	0.0092	Estimated from water budget (1995–2000)
α_L	Longitudinal dispersivity, m	35	Calibrated
α_T	Transverse dispersivity, m	3	Calibrated
D_m	Molecular diffusion, $m^2\,day^{-1}$	7.3×10^{-5}	Kadlec and Wallace (2008)
k_{up}	Uptake rate constant, day^{-1}		
	SAV and cattail	0.4	Calibrated
	Open water/sparse submerged vegetation	0.2	Calibrated
k_r	Release rate constant, day^{-1}	1.97×10^{-4}	Calibrated

balance in Cell 4 for the period of January 1995 through December 2000 (Table 1). Precipitation and potential evapotranspiration were assigned spatially constant values.

The hydrologic and transport parameters such as flow resistance coefficient (A), vegetation reference crop potential ET correction coefficient (K_{veg}), longitudinal dispersion (a_L), and transverse dispersion (a_T), were estimated through model calibration. These parameters were adjusted by trial and error over reasonable ranges to minimize the Root Mean Square Error (RMSE) and Mean Absolute Error (MAE) between observed and simulated tracer concentrations. The calibrated model was subsequently validated against two independent data sets: (1) outflow daily discharge at the G256 structure for the time period of 1995–2000; and (2) chloride concentrations at two internal monitoring stations (i.e., ENR401 and ENR402) for the time period of 1995–1999.

3.5. Phosphorus cycling model simulations

Weekly composite data monitored at inlet structures G254B and G254D were used as a source boundary condition. The spatially constant water column TP concentration ($40\,\mu g\,L^{-1}$) were set as the initial condition, which was the mean observed value from monitoring stations ENR401 and ENR402 at the beginning date of the simulation. An average wet atmospheric TP deposition of $10\,\mu g\,L^{-1}$ (Ahn and James, 2001), estimated at the ENR Project site, was assigned to the entire model domain. Initial soil TP content was based on sampling conducted by SFWMD throughout the ENR Project on 20 January 1995. Four sampling stations (4-2E, 4-2W, 4-1W, and 4-1E) were within Cell 4 (Fig. 1). In the STAs and the Everglades region, a 0–10 cm upper soil layer was generally used to describe the soil TP concentrations (DeBusk et al., 2001; Pietro et al., 2009). Therefore, soil TP in the upper 10 cm soil layer for the entire ENR Project was estimated from these data using a kriging interpolation scheme, and the estimates for Cell 4 were extracted for this study. The mean soil TP was $8.56\,g\,m^{-2}$ which was consistent with the average TP in the upper 10 cm ($8.3\,g\,m^{-2}$) reported for soil collected prior to construction of the ENR Project (Reddy and Graetz, 1991). Peat accretion was monitored from mid-1995 to mid-1999 using feldspar horizon markers throughout the ENR Project (Chimney et al., 2000). Soil TP measured in samples collected at 4-1E and 4-2W on November 12, 1998 and October 20, 1999 were compared to model simulated predictions of soil TP.

Table 2
Scenarios designed to evaluate management conditions on spatial and temporal dynamics of TP in water column and soil.

Scenario	Description	Questions to be answered
S0	Evaluate current conditions using data from 1995–2000	
Internal characteristics		
S1	Identify short-circuiting locations and change vegetation pattern to improve performance	Vegetation effects of channels/ditches on TP removal effectiveness of Cell 4
External loadings from 1995 to 2000		
S2a	Original TP concentrations × 0.75	Impacts of loading on outlet TP concentrations and wetland removal performance
S2b	Original TP concentrations × 0.50	
S2c	Original TP concentrations × 0.25	
S3a	Original flow × 0.5	
S3b	Original flow × 1.5	
S3c	Original flow × 2	
Long-term simulations (1995–2018)		
S4	Repeat 6-year (1995–2000) input data and run for additional 18 years	Temporal evolution of long-term performance (TP outflow and accumulation in soil)
S5	No inflow TP concentrations applied after 2001, repeat hydrologic inputs through 2018	Release of TP from accumulated solids (i.e., internal load)

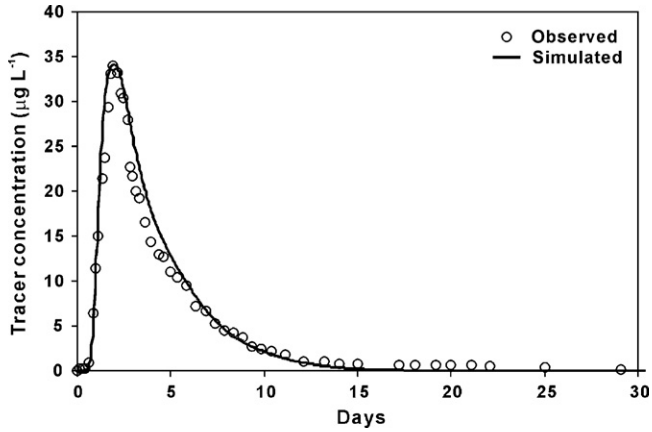

Fig. 3. Observed and simulated tracer concentrations at outflow structure (G256) of Cell 4. The Rhodamine WT dye was applied to Cell 4 on December 16, 1999 (Dierberg et al., 2005).

Spatially non-uniform values of k_u were assigned to characterize the variable uptake by different vegetation types. Since the channelized zone contained less vegetation, TP removal was set to be less effective than the remaining more dense SAV area (Dierberg et al., 2005). A much lower k_r was assigned constant over the entire Cell 4. The final values of these parameters were determined through calibration (Table 1). To ensure numerical stability, the model was run in a relatively short time step (1 h). P-cycling processes were characterized by linear kinetics in ordinary differential form, and these equations were solved by fourth-order Runge Kutta numerical integration method.

Model calibration was performed against the long-term outlet TP concentrations from Cell 4 for the period from January 10, 1995 through December 31, 1998. During calibration, the dominant parameters k_u and k_r were adjusted by trial and error to minimize RMSE and MAE. For k_u, we attempted to vary our parameters within the range of reported settling parameter from relevant past studies (Dierberg et al., 2002, 2005; Walker and Kadlec, 2005). However, the release coefficient, k_r was not available from previous studies; therefore, we tested the model for a wide range of k_r values during calibration. Validation was performed using the two-year data from 1999 and 2000 without changing calibrated parameters.

3.6. Simulation scenarios design

The following scenarios were designed to evaluate the effects of changes in management conditions on the spatial and temporal behavior of TP retention in Cell 4. The baseline scenario (S0) was designed to emulate the system operations and management applied from 1995 through 2000. The next scenario (S1) evaluated the effect of removing short-circuiting pathways on TP removal by changing the open water/sparse submerged aquatic vegetation areas to dense SAV. Changes in TP loads were studied by decreasing TP inflow concentrations by 25%, 50% and 75% as scenarios S2a, S2b, and S2c, respectively. Similarly, flow input was decreased by 50%, and increased by 50% and 100%, as simulation scenarios S3a, S3b, and S3c, respectively. These scenarios are relevant to specified load limits of TP in discharges from the EAA using agricultural Best Management Practices (BMPs) (Horn et al., 2007) that are required to achieve annual-average TP load reduction of at least 25% compared to pre-BMP period (i.e., before 1996). However, even greater reduction has been achieved than that specified in the EFA. For example, TP reductions of 73% and 59% were achieved in the EAA in 2001 and 2005, respectively (Horn et al., 2007).

Two basic hypothetical scenarios were designed to simulate the long-term response of the wetland starting from 1995: continuation of operational management adapted during the period 1995–2000 for an additional 18 years (S4), and complete elimina-

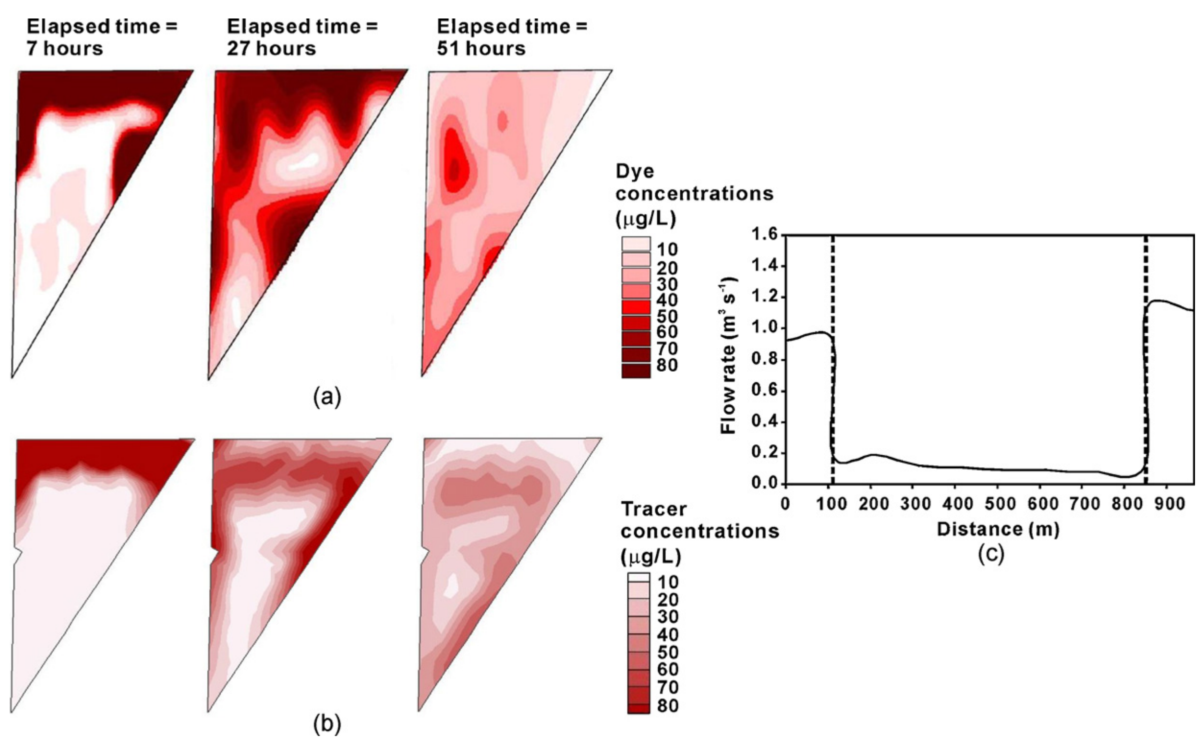

Fig. 4. Comparison between observed and simulated 2-D progression of Rhodamine WT dye plume, and simulated flow rate across the transect BB′ (Fig. 1) at Cell 4. (a) Observed dye plumes at elapsed time 7, 27, and 51 h; (b) simulated dye plume at elapsed time 7, 27, and 51 h; (c) average simulated flow rate across transect BB during the tracer test period. The observed spatial map of dye was estimated by Dierberg et al. (2005).

tion of inflow TP after 2000 (S5). In S4, the 1995–2000 input data (e.g., rainfall, ET, hydrologic and TP loads) used for the baseline were sequentially repeated for the remaining period of January 2001 through December 2018, assuming that Cell 4 would have future water and TP loadings similar to the historic one. Similarly, for S5 the 6-year historic hydrologic loading was sequentially repeated but the TP inflow concentrations were set to zero after the year 2001. It was assumed that the biological and chemical conditions within Cell 4 remained similar during the projection period to that of the period of the baseline. The scenarios listed in Table 2 were used to calculate TP dynamics in soil and water column.

4. Results and discussion

4.1. Hydrodynamics and transport

The calibrated model fit for the Rhodamine-WT tracer concentrations at the outlet structure showed an excellent agreement to the observed concentrations (Fig. 3, RMSE = 1.9 μg L^{-1}; MAE = 1.3 μg L^{-1}). The simulated spatial distribution of the tracer is compared in Fig. 4a and b to maps based on aerial photographs collected by Dierberg et al. (2005) at tracer elapsed time of 7, 27, and 51 h. The simulated and observed patterns of tracer evolution are similar, with the clearly visible short-circuiting along east and west levees of the Cell 4. Although the mesh resolution was not sufficient to capture the local effects of narrow channels seen in the photograph in Kadlec and Wallace (2008), the integrated solute transport through the system, as reflected in the tracer breakthrough curve (Fig. 3) was well matched by the model. The simulated flow rate across the transect BB′ (Fig. 1) shows that 70% of the inflow water is channelized through the zones adjacent to the levees (Fig. 4c).

The outflow discharge and chloride concentrations simulated during model validation also closely matched the observed data (Fig. 5). These results suggest that the 2-D hydrologic/transport model successfully simulated the spatial and temporal variation of internal hydrology and solute transport processes of Cell 4 during the calibration and validation periods.

The set of calibrated parameters, which provided the best-fit between simulations and observations of tracer data, are summarized in Table 1. The Manning's n values for the open water/sparse submerged vegetation and SAV-dominated areas were approximated using their respective mean water depths to be 0.056 and 1.0 s m$^{-1/3}$. These n-values were somewhat higher than those used in previous STA modeling studies. For example, DBEL (2000) applied n = 0.07 and 0.6 for open water/canal and SAV areas in Cell 4, respectively. Similarly, Sutron Corp. (2005) estimated n = 0.038 s m$^{-1/3}$ for canals, and depth-varying n values for SAV of 0.3–1.0 s m$^{-1/3}$ for corresponding water depths of 0.9–0.15 m in STA 1W Cell 5. The difference in these estimated n-values can be explained by the difference in vegetation density, bed slopes, and other hydraulic characteristics such as discharge, inlet/outlet locations, and shape of the wetland.

The α_L and α_T values of 35 and 3 m compare favorably with the values reported from tracer experiments conducted in the ridge and slough area of the Everglades marsh (Ho et al., 2009). They reported longitudinal dispersion coefficients of 0.037 and 0.26 m^2 s^{-1} for corresponding mean velocities of 1.5×10^{-3} and 8.0×10^{-4} m s^{-1}, for the first and last three days of experiments, respectively. The equivalent α_L would be within the range of 25–325 m. Similarly, the transverse dispersion coefficient of 0.012 m^2 s^{-1} reported for the entire experiment period suggested α_T = 11 m for the mean velocity of 1.1×10^{-3} m s^{-1}. However, a relatively low a_L was reported based on the particle tracer experiments in surface-water field flumes located in the Everglades (Saiers et al., 2003; Huang et al., 2008). The spatial scale of the experiment is one of the primary reasons for the difference in dispersion rate (Ho et al., 2009) because of the wide range of local velocities due to heterogeneities in landscape and vegetation pattern (Variano et al., 2009) in a large-scale experiment like Cell 4.

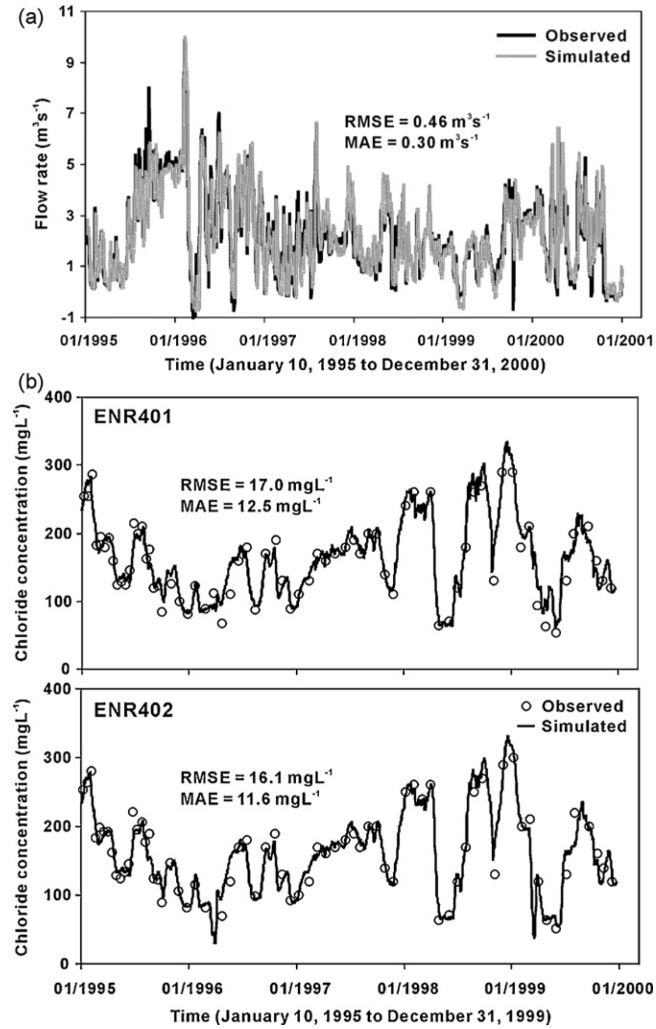

Fig. 5. Comparison between observed data and model predictions (a) outflow discharge at outlet structure (G256), (b) chloride concentrations at internal monitoring stations (ENR401 and ENR402).

4.2. Calibration and validation of total phosphorus

The overall agreement between simulated and observed outlet TP concentrations during the calibration period was moderate, and a similar prediction accuracy was observed during the validation period (Fig. 6). During periods of relatively steady conditions, such as May 1997 through September 1999, the model simulations closely tracked the observed data. However, during periods of dynamic conditions, such as January 1995 through April 1997, the model substantially under-predicted Cell 4 outlet TP concentrations (Fig. 6). In contrast, the model over-predicted the outlet TP concentrations from January 1999 through August 2000 of the validation period. During the calibration period (1995–1998), simulated cumulative TP removed from the water column showed excellent agreement with the observed data; however the model slightly under-predicted during the validation period (Fig. 7). As indicated above, limited data were available to compare soil TP

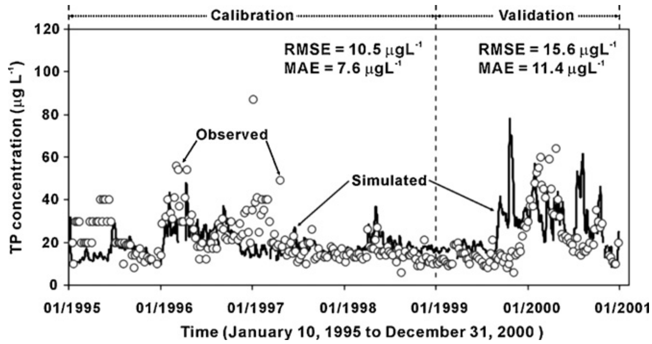

Fig. 6. Observed and simulated TP concentrations at the outflow structure (G256) of Cell 4 for the model calibration period (January 10, 1995 to December 31, 1998) and validation period (January 1, 1999 to December 31, 2000).

changes in Cell 4. In location 4-1E, the model was generally able to capture the overall trend of the observed soil TP, but somewhat underestimated the data at station 4-2W (Fig. 8). Overall, the model was able to reproduce the spatio-temporal variations of soil TP content over a 5-year period; and predicted the integral effect of TP accumulation in the soil.

Given the complexity of internal hydrology and P biogeochemical processes, the simple P cycling model calibration and validation results are encouraging. Although outflow concentration simulations show moderate RMSE and MAE values, the model generally captures the trends and temporal variations of the outlet TP concentrations. The discrepancies are the most noticeable for the first six months of the calibration period (January 1995 to June 1995), where the model fails to capture the observed TP concentrations (Fig. 6). We suspect that these variances may be caused by the biological changes that occurred during the adaptation (i.e., start-up) period of the wetland, which were primarily dictated by antecedent soil conditions (Kadlec and Wallace, 2008). The lack of establishment of complete coverage of vegetation would have contributed less removal of P through vegetation during the first few months of 1995, as the flow-through operation of the ENR Project was started in August 1994 (Chimney and Goforth, 2006). During the validation period, the model over-predicted the observed outflow TP concentrations, particularly on two instances (Fig. 6). This may have been due to the wide disparity between inflow and outflow TP concentrations during the calibration and validation periods, with higher inflow TP concentrations during the validation period. Also, the calibration parameters might have been impacted by the start-up period of the wetland. Other factors such as model spatial resolution, uncertainties in observed data, and model simplification could also affect the predicted results. In treatment wetlands, many P cycling processes involve feedbacks or processes that are also not discernible in linear representations.

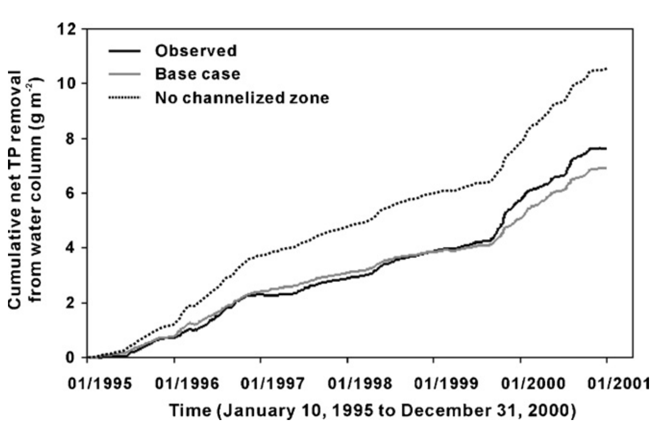

Fig. 7. Predicted cumulative TP removal from water column based on the mass balance calculations from measured data and model predictive scenarios (S0 and S1).

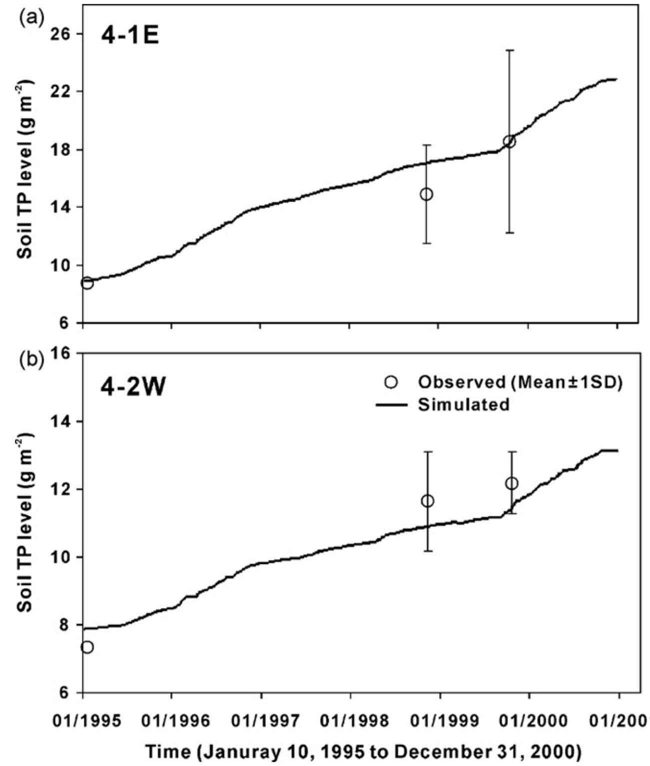

Fig. 8. Observed and predicted soil TP content at two locations within Cell 4 for the upper 10 cm of the soil profile (a) upstream area, 4-1E and (b) downstream area, 4-2W.

4.3. Phosphorus cycling parameters

The best-fit volumetric TP uptake rate coefficient for the SAV-dominated interior marsh ($k_u = 0.4\,\text{day}^{-1}$) was double that for the channelized zone ($k_u = 0.2\,\text{day}^{-1}$). These values are consistent with those of Dierberg et al. (2005), who estimated $k_u = 0.5\,\text{day}^{-1}$ for SAV-occupied areas and $0.24\,\text{day}^{-1}$ for short-circuiting channels. Because it is also common in the treatment wetlands literature to report first-order uptake coefficient on an areal basis, the volumetric uptake rate constants were multiplied by the mean water depth and water column porosity to obtain areal values (Kadlec and Wallace, 2008): $90\,\text{m}\,\text{yr}^{-1}$ for SAV/cattail and $45\,\text{m}\,\text{yr}^{-1}$ for channelized zones. A cell-average areal $k_u = 59\,\text{m}\,\text{yr}^{-1}$ was estimated by flow-weighting these values, based on 70% flow through the channelized zone (Fig. 4c). This value is consistent with the range for net TP settling rate coefficient of $43–64\,\text{m}\,\text{yr}^{-1}$ estimated by Walker and Kadlec (2005) using DMSTA for SAV treatment cells of STAs. Also, for SAV-dominated mesocosms, the areal uptake rate constant was estimated in the range of $63–132\,\text{m}\,\text{yr}^{-1}$ (Dierberg et al., 2002). Cell 4 had a lower k_u value than the mesocosms because the mesocosms received higher concentrations of labile P (i.e., SRP) in its inflows compared to Cell 4 inflows (Dierberg et al., 2002). In Cell 4 inflows, the labile P species were partially treated by upstream cells in the ENR Project. Thus, higher concentrations of more recalcitrant DOP and PP forms (less labile P species) accounted, in part, for the lower k_u value for Cell 4 than the mesocosms. Walker (1995) estimated the 90% confidence interval for settling rate coefficients in WCA-2A in the range of 8.9–11.6 m yr^{-1} based on soils data and 11.3–14.8 m yr^{-1} based on water column data during the period of continuous inundation. The higher uptake rates for Cell 4 are attributed to the SAV community, which is highly effective in P removal from wetlands. The TP release rate constant from soil to overlying water (k_r) was found to be 1.97×10^{-4} day^{-1}. While net P uptake rate constants are widely reported in the literature, no published release rates were available for comparison.

The values of k_u and k_r primarily depend on P equilibrium relationships between soil and the water column. Factors that affect this relationship include the amount of labile P stored in the soil, inflow TP concentration, new soil accretion rate and P concentration therein, and rate and extent of organic matter mineralization. The variability in these factors may affect uptake and release parameters, and increase uncertainty of predicting future TP dynamics in soil and the water column. To reduce uncertainty in future predictions, as with all other models, continued refinement of these parameters is suggested as process understanding and data become more available.

4.4. Sensitivity analysis

The sensitivity of outflow discharge and outlet TP concentration to variations in selected input parameters and initial/boundary conditions was evaluated to help identify the inputs that contribute most to output uncertainty. The sensitivity of a state variable (x) to the changes in parameter (p) was evaluated using the sensitivity coefficient,

$$S_p = \frac{\Delta x / x}{\Delta p / p}. \tag{7}$$

A positive sensitivity coefficient of p indicates an increase in x when p is increased. For example, a sensitivity coefficient of 0.3 indicates that a 10% increase in p would increase x by 3%. The sensitivity test results are summarized in Table 3 to changes in initial/boundary conditions and process parameters within a reasonable range from the baseline data of Cell 4 (Table 3). Absolute changes in water level and outlet boundary head were evaluated (±20 and ±10 cm respectively), and percent change (±30%) was used for the other parameters. Outflow discharge was primarily influenced by the seepage coefficient (K_s), and outlet boundary head. The TP outflow was relatively insensitive to uncertainty (30%) in transport parameters α_L and α_T, but was most sensitive to TP cycling parameters (k_u and k_r) and initial soil TP content. An increase of 30% in soil TP content could increase the annual average outlet TP concentrations by about 7%. Such a high sensitivity of initial soil TP emphasizes the importance of spatial soil TP data. The high sensitivity of TP outflow to the uptake and release parameters indicates the importance of accurately calibrating these parameters.

Table 3
Model sensitivity analysis for Cell 4.

Input parameters and variables	Change	$S_{P,V}$	$S_{P,C}$
Initial/boundary conditions			
Water level	+20 cm	−0.013	0.019
	−20 cm	−0.013	−0.012
Outlet boundary head	+10 cm	−0.149	−1.616
	−10 cm	−0.076	−1.180
C_{TP}^{SW}	+30%	0.000	0.002
	−30%	0.000	0.002
S_{TP}	+30%	0.000	0.244
	−30%	0.000	0.244
Parameters			
A	+30%	−0.012	−0.141
	−30%	−0.014	−0.178
K_{veg}*	+30%	−0.012	0.010
	−30%	−0.013	0.004
k	+30%	0.000	0.001
	−30%	0.001	−0.001
K_S	+30%	−0.090	−0.007
	−30%	−0.091	−0.015
α_L	+30%	0.000	0.006
	−30%	0.000	0.005
α_T	+30%	0.000	−0.019
	−30%	0.000	−0.023
k_{up}	+30%	0.000	−0.588
	−30%	0.000	−0.852
k_r	+30%	0.000	0.325
	−30%	0.000	0.341

The parameters are changed from the baseline data of Cell 4 (Table 1). The selected response variables are the total volume of water discharge from outlet structure (V), and mean annual TP outflow concentrations (C) for the period of simulation (1995–2000).
* To test the sensitivity of the model to changes in ET, we modified the vegetation reference crop potential ET correction coefficient (K_{veg}).

4.5. Effects of management alternatives on treatment effectiveness

The simulated effect of removal of preferential flow paths (S1) was to increase cumulative TP removal by 52% (Fig. 7), and average annual TP removal by 25% (Table 4), suggesting that removing preferential flow paths through vegetation manipulation within the treatment cell could be a good management alternative to improve the P removal performance. These results also highlight how assumptions of uniform flow might be a poor approximation to flow through the wetland, and could substantially limit the ability to accurately model the P transport and cycling processes, especially to those wetlands with highly heterogeneous topographical

Table 4
Simulated average annual TP retention from the water column, and predicted outlet TP concentration for Cell 4.

Simulations	Flow (m yr^{-1})	TP-inflow (g m^{-2} yr^{-1})	TP-inflow conc. (μg L^{-1})	Predicted TP outlet conc. (μg L^{-1})	Total TP-input (g m^{-2} yr^{-1})*	Total TP-output (g m^{-2} yr^{-1})+	TP-removal (g m^{-2} yr^{-1})	TP-removal (%)++
S0	51.7	2.41	48.1	20.3 (0.17)	2.42	1.27 (0.14)	1.15 (0.16)	48 (0.15)
S1	51.7	2.41	48.1	10.9 (0.29)	2.42	0.66 (0.28)	1.76 (0.11)	73 (0.11)
S2a	51.7	1.81	36.1	16.5 (0.17)	1.82	1.00 (0.14)	0.82 (0.18)	45 (0.18)
S2b	51.7	1.20	24.0	12.7 (0.17)	1.21	0.74 (0.15)	0.47 (0.24)	39 (0.24)
S2c	51.7	0.60	12.0	8.9 (0.19)	0.61	0.48 (0.16)	0.13 (0.62)	21 (0.60)
S3a	25.7	1.19	48.1	15.9 (0.26)	1.20	0.55 (0.21)	0.65 (0.19)	54 (0.19)
S3b	77.7	3.62	48.1	22.9 (0.14)	3.63	2.08 (0.12)	1.55 (0.16)	42 (0.17)
S3c	103.8	4.84	48.1	25.0 (0.13)	4.85	2.96 (0.11)	1.89 (0.17)	39 (0.18)

Simulated values are based on calibrated parameters (k_u and k_r). Shown in parentheses are the coefficient of variation (standard deviation divided by the mean) of four simulations using calibrated k_u and $k_r \pm 30\%$.
* Total TP-input includes wet TP deposition (10 μg L^{-1}).
+ Total TP-output includes seepage loss.
++ Percent TP removed calculated relative to the amount of TP that entered Cell 4.

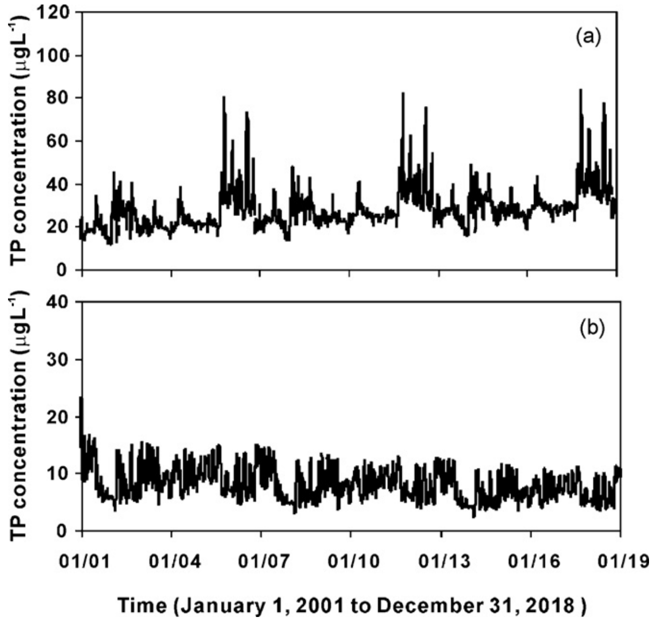

Fig. 9. Predicted outlet TP concentrations for (a) S4 and (b) S5.

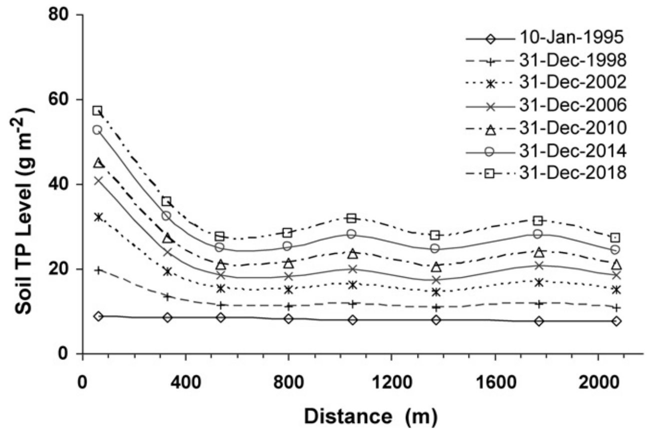

Fig. 10. Predicted soil TP content along the transect AA′ for the upper 10 cm of the soil profile (from inflow to outflow culverts) under S4 from January 10, 1995 to December 31, 2018.

features or vegetative communities (Wang and Jawitz, 2006; Min and Wise, 2009).

Table 4 summarizes the simulation results of changes in inflow TP concentrations and hydraulic loading rates (S2–S3) for Cell 4. A 75% reduction in inflow TP concentration resulted in predicted six-year (1995–2000) average outlet TP concentration of 8.9 μg L^{-1}, which is less than the Everglades Forever Act long-term goal of the target TP concentration of 10 μg L^{-1} in STA discharges. Percent removal, or removal effectiveness, is directly related to inflow TP concentration. Thus, reduction/increase of TP concentrations in inflow water results in a decrease/increase in removal effectiveness (Table 4). Low-flow conditions correspond to higher retention time, which leads to higher TP removal effectiveness, consistent with observations in south Florida Boney Marsh (Moustafa et al., 1998). In addition, sensitivity of simulation results to ±30% uncertainty in k_u and k_r values was estimated (Table 4). Sensitivity of TP removal to k_u and k_r was inversely related to the inflow TP concentrations, but did not vary significantly with flow rate (Table 4). Therefore, extension of the calibrated uptake and release parameters to low-concentration input conditions should be conducted with caution.

4.6. Long-term simulations of current and reduced load

Under conditions of no significant long-term management actions or load changes (S4), predicted outflow TP concentration gradually increased to 41.0 μg L^{-1} after 18 years of operation

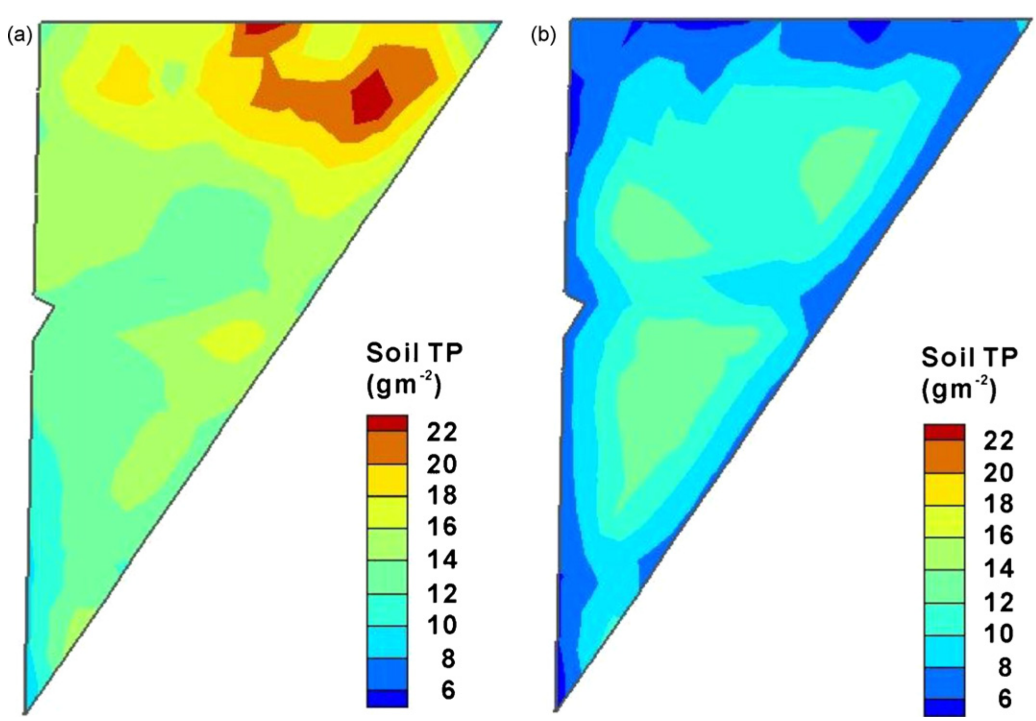

Fig. 11. Snapshots of soil TP pattern predicted under S5 for the upper 10 cm of the soil profile on (a) December 31, 2000 and (b) December 31, 2018.

(Fig. 9a). The gradual increase of TP concentrations over time results from release from TP accumulated in the soil. It should be noted that a saturation level of TP uptake was not considered in the model. Temporal changes in soil TP along longitudinal transect AA′ (Fig. 1) are shown in Fig. 10 under the assumption (S4) of continuation of operational management adapted during the period 1995–2000. The largest decrease in soil TP was in the upstream portion of Cell 4 (within 500 m of the inlet structures), with relatively stable soil TP content beyond 500 m downstream. The sharp change of soil TP content after 500 m is likely due to the transition in the topography and initial soil TP content.

Under reduced external P loading conditions, wetland soils may act as a source, releasing accumulated P to the overlying water column, contributing to eutrophication even after the external load has been eliminated (Reddy, 1991; Fisher and Reddy, 2001; Pant and Reddy, 2003). Under S5, the predicted mean annual outlet TP concentration was still 6.7 μg L^{-1} 18 years after elimination of external loading that occurred during the 6-year period 1995–2000 (Fig. 9b). The simulated soil TP distribution under S5 at the end of the year 2000 and 2018 is shown in Fig. 11a and b, respectively. Results indicated that the accumulated soil P slowly releases from Cell 4 and would require many years to approach natural Everglades background concentrations of $\leq$500 mg/kg (DeBusk et al., 2001) under the calibrated parameters and assumptions imposed in our model.

4.7. Model limitations

Although the model reasonably reproduced measured outflow TP concentrations and successfully simulated the integrated effects of soil TP accumulation, it has a number of limitations. Therefore, scenario simulation results have to be interpreted with caution when used in the management context. A major limitation is that the P uptake and release coefficients represent the combined effect of all biogeochemical processes between water column and soil. Therefore, the model cannot segregate the effects of individual processes that affect TP dynamics in Cell 4. For example, this model cannot independently specify individual processes such as diffusion due to concentration gradient between water column and soil, resuspension of sediment TP, chemical precipitation, or particulate matter deposition. Another limitation is that spatially explicit models are computationally intensive and thus constrain the use of formalized parameter estimation and uncertainty analysis techniques. A further limitation is that field data were not available to calibrate and verify the model for a wide range of conditions, for example, relaxation of the wetland after significant reduction in inflow TP concentrations, re-flooding after drying, high and low inflow/outflow TP concentrations, and time-series soil TP content under these varied conditions.

5. Conclusions

This study described the development of a flow-coupled, spatially distributed TP transport and cycling model, and investigated the effects of various management alternatives on spatial and temporal dynamics of TP in a stormwater treatment wetland. The model reasonably predicted the response of the wetland to alternate management conditions, and provided consistent information of spatial TP evolution over time in both soil and the water column. Considerations that could improve the model predictions include higher resolution spatial data to better describe the complexity of the wetland, finer resolution mesh elements that could better capture narrow channels and ditches, temporally varying (e.g., seasonal) uptake and release parameters, and a more complex representation of wetland P cycling. However, we simplified our spatial and process complexity based on data availability, computation time, reasonable calibration and ease of use. This approach demonstrated a good match of overall trends including hydrodynamics, multi-year cumulative TP removal, and multi-year changes in soil TP.

The outlet TP concentrations were highly sensitive to TP cycling parameters, and moderately sensitive to initial soil TP content and overland flow resistance parameter (A). It was found that removing preferential flow paths would remarkably improve TP removal effectiveness of the wetland (i.e., annual average increased by 25%); suggesting the utility of accurately modeling short-circuiting flow for predicting treatment wetland performance. Furthermore, if inflow TP loads were eliminated after several years of historic loadings, substantial water column TP concentrations were maintained from release of accumulated soil TP. The TP removal performance was also highly sensitive to both hydraulic and TP loads. These simulations indicate that outlet TP concentrations could be reduced by either eliminating short-circuiting pathways within the wetland (e.g., replacing channels, open water with dense SAV vegetation which can improve internal flow distribution and biological uptake) or controlling TP concentrations at inlet sources.

The modeling approach presented here is flexible, in which algorithms can be easily modified or added to accommodate additional internal processes. Ultimately, the model can be used as a management tool to assess the impacts of different management conditions. Such estimates are important in designing and developing operational strategies to maximize the P removal performance of STAs.

Acknowledgements

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