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A model examining hierarchical wetland networks for watershed stormwater management

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ABSTRACT

There is increasing awareness that solutions to degraded quality and excessive quantity of stormwater and resulting impacts on downstream water bodies may require a watershed approach to management rather that the incremental approach that is now common. Examination of low-relief watersheds characteristic of the southeastern coastal plain reveals common hierarchical patterns of surface water convergence that may be emulated in developed watersheds to enhance the efficacy of peak-flow attenuation and pollutant removal. A dynamic systems model was developed to compare stormwater management using a hierarchical network of treatment wetlands with the standard incremental approach wherein treatment systems are designed considering only site-level effluent criteria. The model simulates watershed hydrology, suspended sediment transport and phosphorus removal and transformation. Results indicate that watershed planning of stormwater collection and treatment systems using hierarchical networks can greatly enhance overall effectiveness (annual retention improvements of 31% for flow, 36% for sediment and 27% for phosphorus) with respect to an equal area of uniformly sized wetlands. Further, network proportions can be adjusted to specific runoff characteristics. Distinct roles were observed for each wetland size class: small headwater wetlands effectively removed sediment, medium-sized mid-reach wetlands retained phosphorus, while large wetlands primarily stored and attenuated long-period hydrologic flows.

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1. Introduction

Stormwater runoff is considered the major threat to aquatic ecosystem health in the United States (Olson, 1993; USEPA/USDA, 1998). Numerous techniques have been devised for attenuating hydrologic flows and removing contaminants from urban and agricultural runoff, but the major constraint continues to be diffuse delivery, which necessitates extensive regional infrastructure, large capital investments, and intensive management. This problem will increasingly require planning at the watershed scale in order to efficiently protect aquatic resources (Loucks, 1990; Mitsch, 1993; Black, 1997; Carle et al., 2005). This paper describes use of simulation modeling to explore conceptual patterns of stormwater treatment system design at the watershed scale.

Using wetlands – both natural and man-made – for capturing stormwater runoff and pollutants, has emerged from an understanding of the role wetlands naturally play in landscapes (Ewel and Odum, 1986; Mitsch and Gosselink, 1993; Leibowitz et al., 2000). Specifically, wetland stormwater treatment areas (WSTAs) can provide the services of water storage and peak-flow attenuation (Ogawa and Male, 1986; DeLaney,

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1995), nutrient cycling and burial (Richardson, 1985; Reddy et al., 1993), metal sequestration (Thurston, 1999; Odum et al., 2000), sediment settling (Kadlec and Knight, 1996), and breakdown of organic compounds (Nix et al., 1994; Knight et al., 1999). Numerous authors have highlighted constraints, benefits, and design considerations for using wetlands to treat stormwater (Loucks, 1990; Stockdale, 1991; Rushton et al., 1997) and enhanced stormwater treatment basins exist where ecological and treatment objectives are simultaneously met (Knight, 1996; Otto et al., 2000).

Stormwater management is typically achieved incrementally (i.e., on a site-by-site basis—Emerson et al., 2005), with little attention paid to larger scale hydrologic organization that exists in all landscapes shaped by water. The common result is watersheds that lack the characteristic hierarchical hydrologic convergence found in undeveloped basins (Sullivan, 1986; Ogawa and Male, 1986; Loucks, 1990). Our hypothesis, after examining hydrologic convergence patterns found in undeveloped watersheds, is that stormwater management systems might be improved by emulating these patterns. Watershed scale planning (USEPA/USDA, 1998) warrants explicit attention to these larger scale patterns. In particular, siting and sizing stormwater treatment areas for regional management of runoff has garnered attention (Palmeri and Bendoricchio, 2002); recent work using spatial models to site treatment areas (Zhen et al., 2004; Newbold, 2005) could be used in concert with conceptual planning tools like the one we propose in this study to optimize watershed-scale runoff control system design.

Several authors have explored the role of wetland size and location on treatment. Loucks (1990) suggests small wetlands should be the focus of hydrologic restoration because they have been extensively removed from the landscape, further arguing that upstream erosion and flooding would be poorly addressed by large terminal treatment wetlands. Van der Valk and Jolly (1992) suggest that small headwater wetlands will most effectively intercept agricultural pollutants. Mitsch (1993) analyzed implementation costs of large, downstream wetlands versus small, headwater wetlands and concluded that smaller wetlands were more flexible and less expensive. In contrast, Ogawa and Male (1986) used simulation models to show that large downstream wetlands were most effective at attenuating peak basin outflow conditions, and that benefits of flow impedance were highly localized, with negligible effects observed a few miles downstream. We examine the synergistic effects of multiple size classes of WSTAs on watershed discharge.

Previous work (Tilley and Brown, 1998) focused on the role of three size classes of WSTAs separately. They suggest, based on area requirements for meeting target outflow criteria, that treatment function is scale-dependant: small wetlands sequester P, medium wetlands capture sediments and large wetlands attenuate water flows. Complementary roles suggest increased effectiveness if different sizes are used concurrently.

Wetland arrangement in a regional network can borrow from convergence characteristics observed in undisturbed basins. Sullivan (1986) analyzed low-relief watersheds in Florida and observed a hierarchical arrangement of wetlands (Fig. 1). Fig. 2A shows the distribution of wetland size in four low-relief watersheds in Florida. Fig. 2B shows mean location

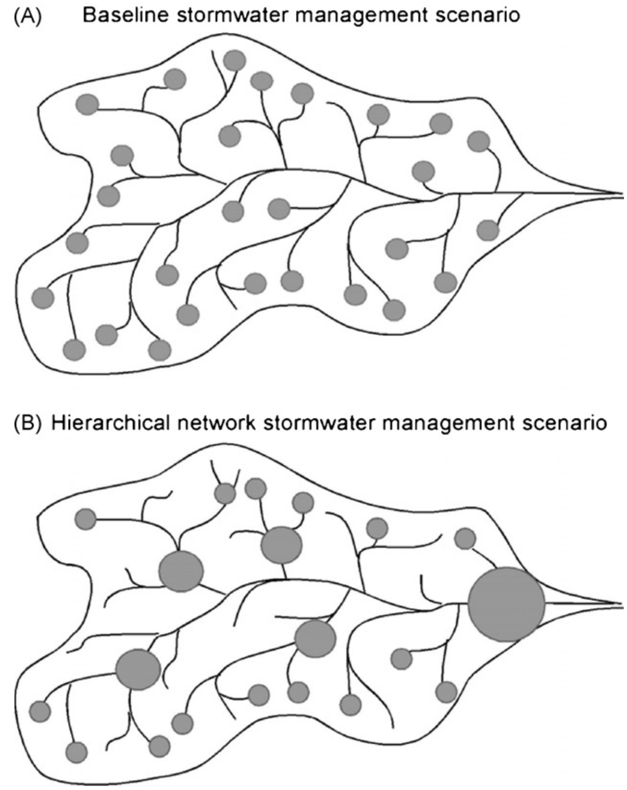

Fig. 1 – Schematic of landscapes arranged with (A) small wetlands only (baseline scenario) and (B) with hierarchical network of wetlands (network scenario).

and variance, measured as distance from watershed outlet, for each size class. Small wetlands are distributed throughout, but are the dominant size class in headwater regions. Medium wetlands (sloughs/riparian systems) were centrally located, converging flow to a few large wetlands in the lower reaches of watersheds (coastal wetlands and extensive bottomlands). Morphologically, small wetlands correspond to isolated wetlands, medium wetlands correspond to conveyance wetlands—riparian systems or sloughs, and large wetlands are regional receiving systems (coastal or bottomland).

While there is no direct evidence to suggest that wetlands are hierarchically organized in undisturbed landscapes to maximize pulse attenuation of water, nutrients or sediments, it is our hypothesis that emulating the observed spatial hierarchy for WSTAs will improve stormwater discharge properties in urbanizing watersheds in comparison with an incremental approach wherein hierarchical hydrologic convergence is ignored. We explore this hypothesis using a theoretical process-based systems simulation model.

2. Study site

Improving water quality entering Biscayne Bay from the low-relief watersheds on the Atlantic Coastal Ridge in Dade County, Florida (Fig. 3) was the focus of this work. Water management in the region is complex due to the network of canals and control structures, and the absence of topography

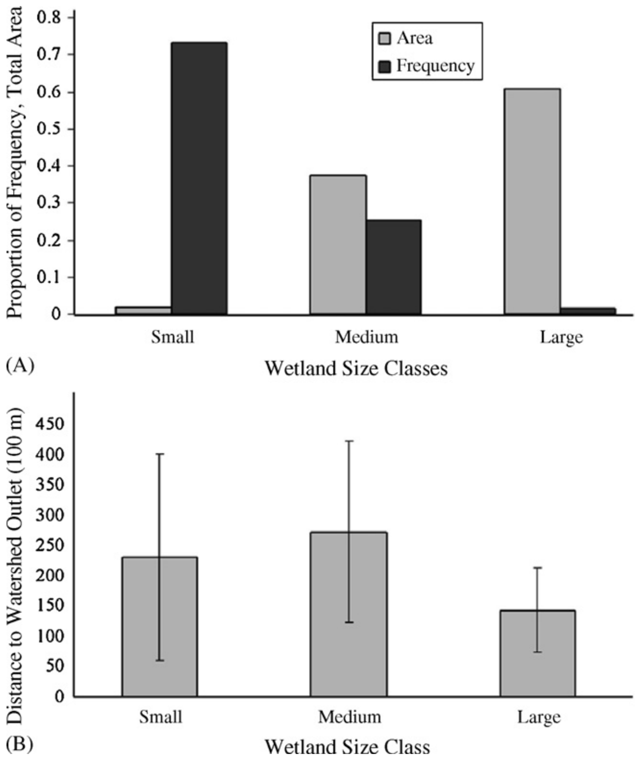

Fig. 2 – Wetland network configurations showing (A) mean proportion of wetlands in three size classes based on spatial extent and frequency of occurrence for four Florida basins, and (B) wetland spatial location measured as mean and standard deviation of distance from watershed outlet for each size class in the Wacassassa Basin, North Florida.

to delimit hydrologic flow directions. Water control infrastructure coupled with increased impervious surface resulting from ongoing development has dramatically changed regional hydrology. Most important has been the shift in water delivery from sub-surface flows, to pulsed surface flows. Concern for Biscayne Bay has arisen because fresh water delivery was historically dominated by sub-surface sources that are both strongly buffered against large pulses, and scoured of dissolved phosphorus by the sub-surface limestone. Data from the C-102 canal (Princeton Canal) basin (Fig. 3) were used to explore effects of a regional stormwater treatment system on attenuating elevated runoff and pollutant delivery. This basin (A = 8500 ha) drains eastward to northern Biscayne Bay. This and other basins are of particular regional interest because they are undergoing rapid development from predominantly agricultural areas to more developed land-uses as urbanization expands southward from Miami.

3. Methods

3.1. *Wetland modeling*

Wetland sizes were distributed based on Fig. 2; small, medium and larger wetlands represent 2%, 37% and 61% of total wetland area, respectively. In the C-102 canal basin, we assumed wetland sizes of 0.2 ha (small; $n = 410$ basin-wide), 9.5 ha (medium; $n = 33$) and 250 ha (large; $n = 2$); frequencies of each class are based on a total wetland area of 10% of the basin (850 ha). Total area and size-class distribution are model variables.

We used a systems dynamics approach to model stormwater treatment wetland processes within each wetland (Kendall, 1997; Shelley and Mudgett, 1999; Wang and Mitsch, 2000). For modeling wetland hydrology we used a daily water balance approach driven by observed rainfall data (http://www.sfwmd.gov/org/ema/dbhydro/). A standard plug-flow model, based on linear hydraulic settling rates, exists for constituent removal within wetland systems (Kadlec and Knight, 1996). This model, applied widely to predict nutrient and sediment retention in wastewater treatment wetlands, relies on average long-term behavior, with inflow volumes and constituent concentrations maintained at relatively constant levels. Several authors (e.g., Wong and Geiger, 1997; Werner and Kadlec, 2000) have suggested that this plug-flow model may be inappropriate for stormwater systems, particularly for phosphorus dynamics, because of the stochastic nature of water and pollutant delivery. Others (Carleton et al., 2001) observe that first order kinetics adequately represent stormwater treatment dynamics. In this model we assume concentration-based settling (first-order kinetics) for sediment removal, but we model internal phosphorus processing explicitly (*sensu* Wang and Mitsch, 2000) to avoid making assumptions of linear settling. We back calculate an effective linear settling rates for P to ensure that realistic results given the observed range of removal efficiencies (Kadlec and Knight, 1996). We assume that the wetland treatment areas are completely mixed systems. Further refinement of this approach might include use of partial mixing models (Werner and Kadlec, 2000).

The energy systems language (Odum, 1994) was used to devise model structure. The goal was to produce a model of sufficient process detail to capture water storage and constituent processing dynamics without excessive parameter estimation and complexity. The energy systems language facilitates aggregation to mathematically simple models. Furthermore, each symbol encodes a specific mathematical formulation that allows rapid extraction of difference equations to compile the simulation.

The model was constructed and evaluated in Microsoft Excel using Euler integration of the difference equations inferred from energy systems diagrams. Spreadsheet modeling was chosen because it allows a user to view and modify calculations, change parameter estimates, and immediately evaluate results within a framework typically used for preparation and analysis of input/output data. The transparency of spreadsheet operations readily permits user modifications (e.g., accounting for additional water quality parameters) and examination of model calculations.

The model was run on a daily time step, driven by study area rainfall observations (Sculley, 1986). It is a lumped parameter model in that the land-uses surrounding the wetlands are given single parameters to describe infiltration capacity, runoff coefficients, event mean concentrations, etc. However, because model formulation includes a hierarchy of WSTAs, it has an implicit spatial component, as water and materials converge through the network.

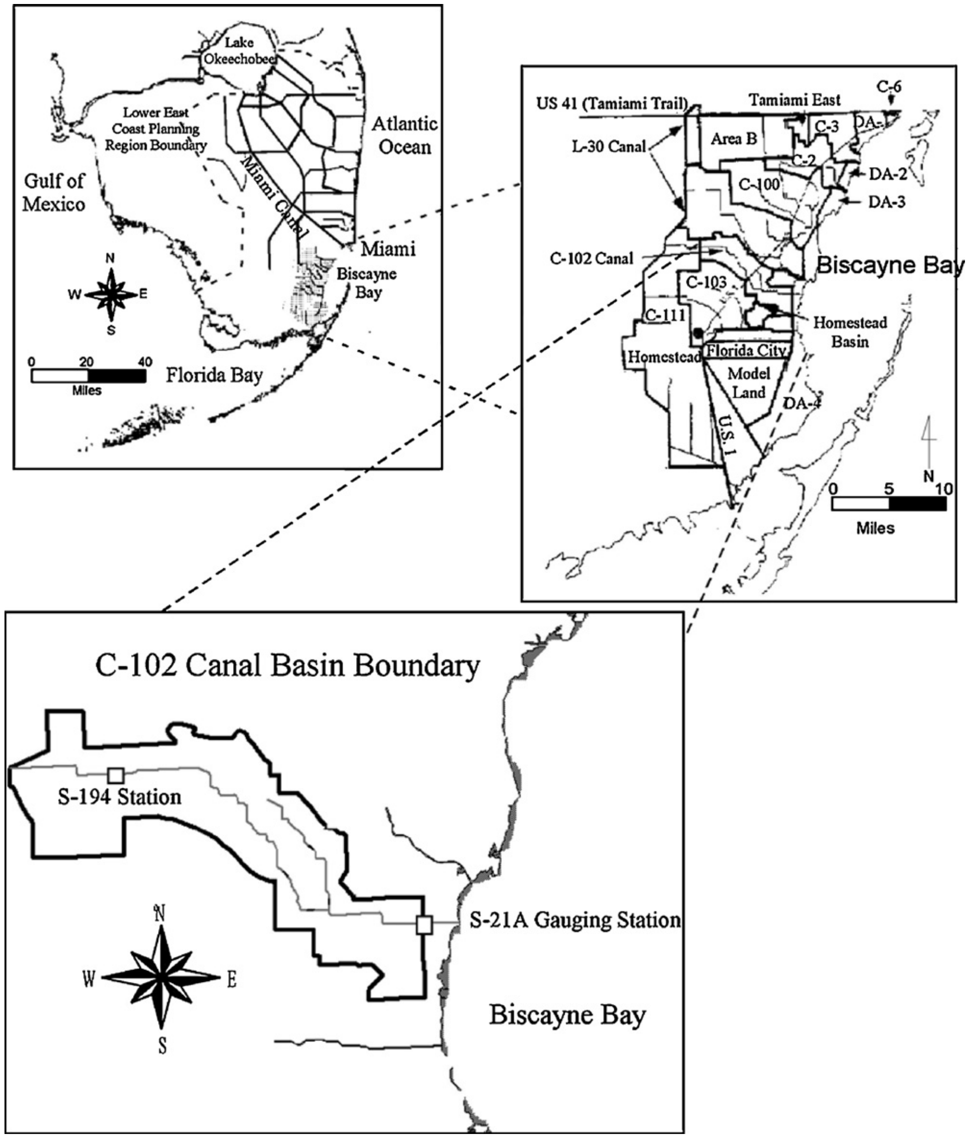

Fig. 3 – Study area showing southern peninsular Florida, South Miami/Dade County, and the C-102 Canal basin. Outflows at the S-21A gaging station were used for calibration; inflows at the S-194 gaging station were used as boundary conditions.

Most model rate coefficients were obtained directly from literature sources. Where rate coefficients were unavailable, we used standard calibration techniques to arrive at reasonable values. Model calibration for the hydrology was performed using data from an intensively studied stormwater system in central Florida (Rushton, 1996); for constituent processing, data on treatment efficiency aggregated as a function of residence time from several stormwater wetlands were used (Kehoe et al., 1994; Carr and Rushton, 1995; Rushton et al., 1996). The effects of land-use on surface runoff were generalized from the runoff coefficient and event mean concentration data presented in Harper (1994) for Central and South Florida. The calibration criterion was minimization of bias between predicted and observed retention efficiencies and flow volumes annually, and by season. For P dynamics, calibration also included matching observed effects of turnover time.

Hydrologic validation was done using data from the terminal S-21A gaging station on the C-102 canal (Fig. 3). Inflow data from the upstream regional canal system discharging flow into the western end of the basin were included as a boundary condition (S-194 station). Mean constituent concentrations in canal water were used to ensure sediment and nutrient portions of the model were approximating flows accurately, but no daily water quality data were available for validating model dynamics. The current land-use condition, where wetlands represent 6% of the basin area, was used for validation.

A sensitivity analysis was done to determine which model parameters contributed most to model behavior. Each parameter was increased and decreased by 25% and the resulting output was compared to the calibration condition.

3.2. *Hydrologic module*

The central components of a water balance (rain, runin, evapotranspiration, seepage and outflow) were included, but concerted effort was made to simplify processes to allow rapid

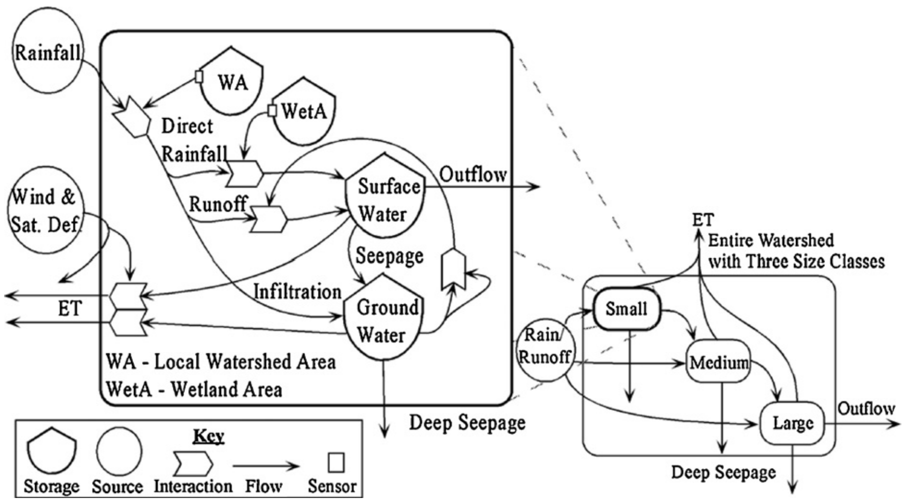

Fig. 4 – Energy systems representation of model hydrologic component, with details shown for one size class. Difference equations for model simulation are shown in Table 1.

model execution and tractable parameterization from literature sources.

An overview diagram of the hydrology for a single wetland is given in Fig. 4. The difference equations extracted from that diagram are given in Table 1. The schematic shown in Fig. 1 demonstrates how hydrologic modules are linked in the simulated basin: water runs off the landscape into one of three size classes based on the source area for that wetland, and then converges to progressively larger wetlands until terminal discharge from large wetlands. Rate coefficients controlling unit-area hydrologic behavior (infiltration, evapotranspiration) were identical for each wetland size class. Wetlands were assumed cylindrical (i.e., they maintain constant area with depth).

Rainfall data from a local climatological station (S21A_R—DBHYDRO database, http://www.sfwmd.gov/org/ema/dbhydro) for wet (1666 mm in 1995), average (1239 mm in 1999) and dry years (988 mm in 1996) were used. Another average year (1299 mm in 2001) for which gauging station flow data were available was selected for model validation. Constituent flows, which were calibrated at the individual wetland scale, were validated using annual mass flow data and mean discharge concentration observations from 1992 (1526 mm of rainfall).

Uplands draining to a wetland directly are termed the catchment. We assumed each wetland had a catchment of equal radius; as a result, small frequently occurring wetlands capture proportionally more direct runoff, and larger less numerous wetlands receive more of their flow in channels from upstream retention areas. Runoff was computed as a function of catchment landuse, based on a nominal infiltration capacity (Harper, 1994) modified by modeled antecedent soil moisture conditions. The variable source area (VSA) concept, used to describe hydrology for isolated wetlands (Sun, 1995), was employed in this study. The VSA model suggests that the capacity of the land to absorb rainfall is inversely proportional to the square of soil moisture (in a 3 m profile), with saturated conditions resulting in maximum runoff generation. As such, the portion of the catchment area that is contributing surface runoff to a wetland changes dynamically (Table 1).

Monthly mean evapotranspiration rates were computed using the Blaney–Criddle method (SCS, 1967) based on published monthly averaged values from a local weather station (Homestead Research and Extension Center;

Table 1 – Model parameters and difference equations for hydrologic module

Sources	
R	Rain
ET	Evapotranspiration; based on monthly average values for temperature, humidity, radiation and wind speed
CI	Channelized inflow (outflow from upstream wetland)
Storages	
WA	Watershed area (upland)—constant
WetA	Wetland area—constant
SW	Surface water stage in wetland
WH	Weir height (above grade)
GW	Groundwater (surficial soil storage in top 3 m of soil beneath source area)
Flow coefficients	
KLanduse	Infiltration coefficient based on aggregated land use (Harper, 1994)
KSeepage	Rate of seepage from wetlands to groundwater based on hydraulic conductivity of peat and average peat depth
KDeepSeepage	Rate of deep seepage from GW (residence time in upper 3 m of soil = 180 days)
Equations	
dSW/dt	$R \times WetA + CI + \{R \times WA - [(R \times WA \times KLanduse/GW^2) + WA \times ET]\} - KSeepage \times (SW/WetA) - ET \times WetA - ((SW - WH)/WetA)^{1.5}$
dGW/dt	$R \times WA \times KLanduse/GW^2 + (SW/WetA) \times Kseepage - KdeepSeepage \times GW - WA \times ET$
dt	1 day

Note: Equations and rate parameters are the same for each size class.

accessed through the Florida Automated Weather Network—http://www.fawn.ifas.ufl.edu) of temperature, relative humidity, wind speed and solar radiation. Resulting monthly ET estimates (in mm day^{-1}) are: {1.18: January; 1.85: February; 2.85: March; 4.10: April; 5.27: May; 6.31: June; 6.07: July; 5.65: August; 4.29: September; 3.60: October; 2.78: November; 1.59: December}. These monthly estimates are used for both the wetland and catchment area in multiple locations in the water budget equations (Table 1).

Seepage from the wetland was based on Darcy's law, using average conductivity estimates (0.144 cm h^{-1}) for peat from Wise et al. (2000) and a nominal peat thickness of 25 cm. Groundwater in the uplands is assumed to fill a 3 m vadose zone; seepage from surficial groundwater into deeper aquifers is a linear function of groundwater stage.

Surface outflow was assumed to be controlled by a rectangular weir, where outflow is proportional to the 1.5th power of the gravity head above discharge height, which was set at 0.2 m for small and medium wetlands, and 0.0 m for large wetlands.

3.3. Constituent module

Phosphorus is the primary water quality constituent considered because of its role in driving eutrophication in Biscayne Bay. Modeling P dynamics required inclusion of a suspended sediment component to account for P sorption. This module (Fig. 5) overlays the hydrologic module; parameter values were constant for each size class (Table 2).

3.3.1. Sediment loading and settling

Sediment inflows were simulated as a function of flow volume (Turner et al., 1975), and delivery concentrations were determined using land use information from Harper (1994). Sediment settling was modeled using first-order kinetics (Kadlec and Knight, 1996) resulting in sediment concentration varying as a function of inflowing load and hydraulic residence time. Hydraulic residence times ($HRT_t = storage_t/outflow_t$; where t is time) are computed in the hydrologic module. This approach is similar to models for wastewater wetland design (Kadlec and Knight, 1996), but concentrations and flows are computed on a daily basis to accommodate dynamic hydrology. Sediment outflows were proportional to hydrologic outflows, so effluent concentrations equal water column concentrations.

3.3.2. Phosphorus loading

P processes in wetlands (Reddy et al., 1999) and on the landscape were aggregated to track three pools: sediment bound P, dissolved inorganic P and dissolved organic P. P loading from the source area was computed for each species separately. Loading of P attached to sediments (~40% of total annual load—Harper, 1994) was estimated in proportion to sediment mass delivered. Dissolved fraction loading (60% of total annual load) was proportional to the interval between runoff events. Carr and Rushton (1995) showed inter-event time to be an effective predictor of P loads. This implies daily P deposition for each land use (DAR in Table 2), computed from annual mass loading data presented in Harper (1994). The resulting storage (POL—Table 2), of which 80% is inorganic

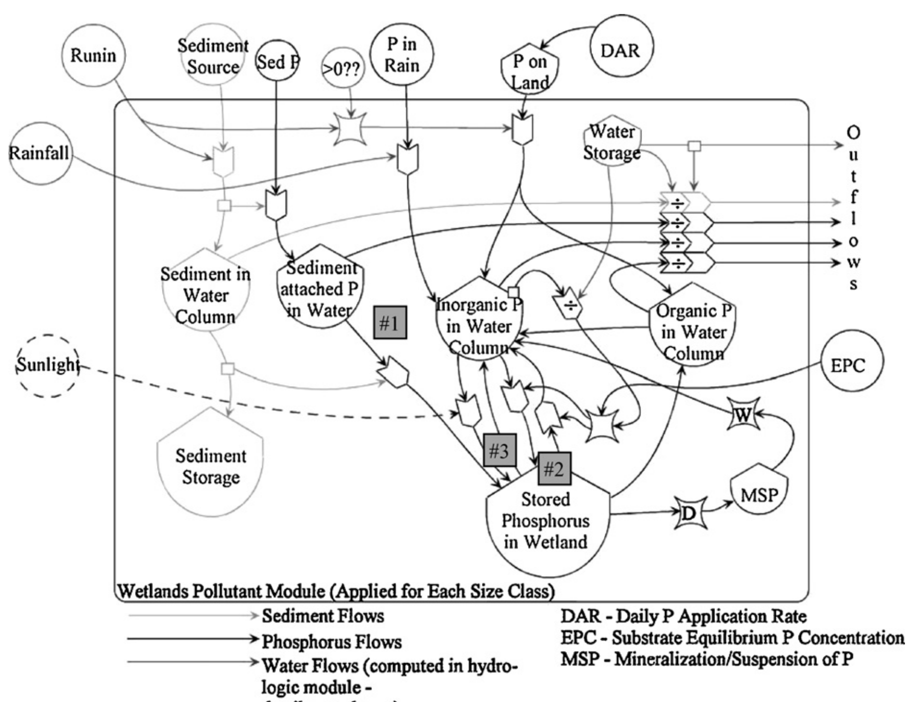

Fig. 5 – Constituent dynamics for a single wetland in network. Shown are flows of sediments, P (sediment-bound, dissolved inorganic and dissolved organic) and driving flows of water from Fig. 4. Sediment bound P settles in proportion to sediment removal (#1). Soil sorption processes are represented as pathway #2, with both adsorption and desorption paths, both driven by the equilibrium P concentration (EPC). Annual biomass production (driven by sunlight) controls one pathway of inorganic uptake (#3).

Table 2 – Model parameters and difference equations for constituent module

Sources	
R	Rainfall
RI	Run-in from source area (from hydrologic module)
Outfl	Water outflow from wetland (from hydrologic module)
SW	Water storage in wetland (from hydrologic module)
Sed	Sediments
Prain	P in rainfall
DAR	Daily accumulation rate of P on source area
POL	P stored on land
EPC	Equilibrium P concentration (EPC)
Sun	Solar insolation
Storages	
SedW	Sediment in the water column
SedS	Sediment stored in the wetland
PsedW	Phosphorus attached to sediment in the water column
PinorgW	Inorganic P fraction dissolved in water column
PorgW	Organic P fraction dissolved in water column
Pstored	P in the sediments (labile and refractory)
MSP	Mineralized surface P (accumulates when wetland is dry)
Flow coefficients	
Ksed	Constant for sediment load in runoff (sediment event mean concentration)
KsedSettle	Sediment linear settling rate
KPSed	P attached to sediment
KinorganicP	Proportion of P from surface runoff that is inorganic
KorganicP	Proportion of P from surface runoff that is organic
KinorgPRel	Rate of release of inorganic P from sediments
Kdesorb	Rate of P desorption
KOtoIn	Rate of conversion from organic to inorganic P in water column
Kbiomass	Rate of biomass uptake of P
Kadsorb	Rate of P adsorption
KorgPRel	Rate of release of organic P from sediments
Equations	
dSedW/dt	Ksed × RI − KsedSettle × SedW − (SedW/SW) × Outfl
dPOL/dt	DAR × X − Z1 × POL
X	Number of days between rainfall events causing runoff
Z1	1 if surface runoff occurs in that day, 0 otherwise
dPSedW/dt	KPSed × Ksed × RI − KPSed × KsedSettle × SedW − KPSed × (SedW/SW) × Outfl
dPstored/dt	KPsed × KsedSettle × SedW + Z5 × KAdsorb × [(PInorgW/SW) − EPC] − Z4 × KDesorb × [EPC − (PInorgW/SW)] − KInorgPRel × PStored
dPInorgW/dt	Prain × rain + Z2 × POL × KInorganicP + Z3 × MSP + KInorgPRel × PStored + Z4 × KDesorb × [EPC − (PInorgW/SW)] + KOtoIn × POrgW − Sun × Kbiomass − Z5 × KAdsorb × [(PInorgW/SW) − EPC] − (PInorgW/SW) × Outfl
Z2	1 if surface runoff occurs in that day; 0 otherwise
Z3	1 if wetland is wet; 0 if wetland in dry
Z4	1 if concentration of inorganic P is less than soil EPC; 0 otherwise
Z5	1 if concentration of inorganic P is greater than soil EPC; 0 otherwise
dPOrgW/dt	Z6 × POL × KOrganicP + KOrgPRel × PStored − KOtoIn × POrgW − (POrgW/SW) × Outfl
Z6	1 if surface runoff occurs in that day; 0 otherwise

Notes: The phosphorus attached to sediments is considered non-labile for the purposes of this model. Coefficients on adsorption/desorption are different to due to hysteresis effects (Reddy et al., 1999). Adsorption is modeled as a linear isotherm. Rates are identical for each wetland size class.

(KinorganicP = 0.8), is transported during the next runoff event (Z1—Table 2).

3.3.3. *Phosphorus removal*

Sediment bound P was removed in proportion to sediment settling rates (pathway #1—Fig. 5; KsedSettle in Table 2). No mechanism was built into the model to allow P sorbed to sediments to desorb while suspended in the water column.

Dissolved inorganic P removal took place along two pathways: (1) adsorption to anion exchange sites on the substrate and complexation with humic materials and cations (primarily calcium and aluminum under low redox conditions) in the soils (pathway #2 in Fig. 5; Kadsorb in Table 2), and (2) biomass uptake and subsequent deposition of recalcitrant organic material (pathway #3 in Fig. 5; Kbiomass in Table 2).

The first removal mechanism – adsorption – is reversible, continuing as long as the water column concentration remained above the equilibrium phosphorus concentration (EPC—Table 2) of the soil (Reddy et al., 1993). The EPC was set at 0.07 mg/l to reflect the high adsorptive capacity of the mineral substrate in South Florida, though even this value is likely conservative. The adsorption process was simulated as a linear isotherm, based on the concentration gradient between water and soil and the residence time of water in the wetland. The maximum exchange rate was 2.4×10^{-4} g P g sediment^{-1} day^{-1} (Kayek and Yousef, 1993), and the process was characterized by hysteresis, whereby adsorption occurs more rapidly than desorption (Kadsorb > Kdesorb in Table 2; after Reddy et al., 1999).

The second removal mechanism, peat deposition, is driven by wetland plant productivity, which is in turn varies seasonally (Sun in Table 2). While initial vegetative uptake may represent a net sink, the system eventually reaches equilibrium, where only a fraction of the phosphorus is permanently stored. Reddy et al. (1999) summarize the literature and report that approximately 25% of total P uptake is eventually deposited in refractory forms that can be considered removed from internal cycling. The remainder is returned to the water column as dissolved inorganic or organic P. We assume that concentration gradients between the pore water and water column that result from plant uptake are equilibrated instantly; as a result, P in the water column is available to plants.

The pathway of dissolved organic P removal is represented as a bacterial cycling loop. The turnover time of dissolved organic P conversion to inorganic P (controlled by KOtoIn—Table 2) in the water column was estimated as 4 days (and assumed independent of season or water level), greatly aggregating the complex array of degradation rates for dissolved organic materials with varying degrees of recalcitrance.

When a wetland dries, any P in the water column is deposited to the MSP tank (Fig. 5) which is re-released when the wetland is re-flooded (Z3—Table 2). We do not account for accelerated decomposition of organic sediments and resulting inorganic P release.

Export was proportional to the water column concentration and the volume of water that leaves the wetland during each time step (e.g., Outfl × PInorg/SW—Table 2).

3.4. *Evaluation of model output and scenario development*

The model predicts outflow volumes and constituent concentrations on a daily basis. Multiple criteria were used to determine the optimal configuration and distribution of WSTAs in the landscape. The primary goal was to maximize reduction of peak volumetric discharges and large pulses of reactive phosphorus. Overall retention efficiencies were considered to be of secondary concern, though also important. The optimal configuration was the one that maximized both of these criteria. An ancillary goal was to maximize the area of wetlands with a hydroperiod and nutrient-loading signature characteristic of natural wetlands under the assumption that such a signature would allow treatment wetlands to function as wildlife habitat.

The effectiveness of hierarchical networks of wetland stormwater treatment areas – hereafter referred to as the network scenario – was made in comparison with a configuration – hereafter referred to as the baseline scenario – which contains only one size class. In this baseline condition, an equal total area of WSTAs are distributed throughout the basin without hierarchical hydrologic convergence, simulating the condition that results from incremental, site-by-site management of stormwater. It should be noted that this model was designed only to explore the effects of emulating natural landscape organization conceptually; it was not developed to aid in engineering design of specific STAs.

4. Results

4.1. *Model calibration and validation*

A tabular summary of calibrated model performance for a single wetland (Table 3) shows an annual error rate in hydrologic predictions less than 15%. Basin-scale model hydrologic dynamics were calibrated to high, low and medium rainfall years and validated using rainfall and observed flows from 2001 (rainfall = 129.9 cm). Monthly and annual calibration errors were {29%, 18%}, {15%, 8.8%} and {20%, 13%} for low (1996; 98.8 cm), medium (1998; 127.2 cm), and high (1995; 166.6 cm) rainfall years, respectively. Validation mean absolute error averaged 33% monthly but was only 4.3% for the annual flow volume. Fig. 6 shows the rainfall pattern and observed flows at the S-21a gauging station, and model flow predictions for the validation year (2001). While the cumulative annual flow error is small, the delivery pattern highlights some substantial differences between predicted and observed. In particular, the model underestimates major peak flows (mean error = −54%) and overestimates dry-season (March–June) base flows. However, general correspondence in timing and volume were considered adequate for this conceptual model; improved prediction accuracy would be needed for planning purposes (Fig. 6).

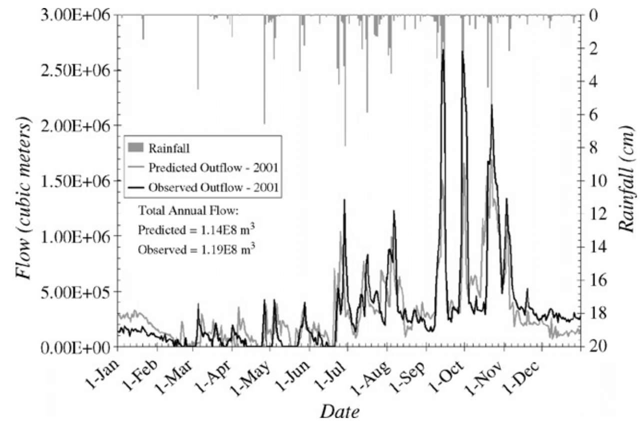

Fig. 6 – Model hydrology validation comparing observed and predicted total annual and daily flows from the C-102 basin for 2001. Rainfall delivery is also shown; total annual precipitation was 130 cm. The predicted condition is for 6% wetland coverage, configured without hierarchical convergence.

As the residence time of the wetland is manipulated, observed P retention varies considerably (Table 3B). The calibration performs well for medium turnover times, and varies in the manner that would be expected (i.e., reduced turnover time decreases retention efficiency). Finally, seasonal performance (Table 3C) indicates that the model error is within 5% for both dry season and wet season monthly loads.

The sensitivity of the model to changes in select parameters (Table 4) demonstrates that the hydrologic module is most sensitive to changes in evapotranspiration rate, hydraulic residence time, weir characteristics and seepage rates. The sediment/P module is most sensitive to changes in sediment settling rate, EPC and OM release rates. In general, the model appears relatively insensitive to moderate uncertainty (25%) in rate parameters.

The data available to validate the sediment and nutrient component of this model were limited to estimates of total annual delivery to Biscayne Bay (Alleman, 1995) and nominal discharge concentrations for the same period. Constituent flow validation results (Table 5) indicate that the model is over-estimating both the magnitude of annual P flows and sediments; over-estimation is more pronounced for P. Given the large basin area and the complexity of regional hydrology, these results are encouraging; model temporal dynamics cannot be confirmed from the available data, nor can the

Table 3 – Aggregated calibration results to literature wetland storage and treatment data

Hydrologic flow	Hydrologic budget	
	Measured values[a]	Calibration values
(A) Shows the hydrologic calibration		
Rainfall (m)	1.44	1.44
Runin (m)	6.41	6.44
ET (m)	1.03	1.02
Seepage (m)	1.24	1.17
Outflow (m)	5.32	5.81
Annual retention (%)	32.2	26.3

Turnover time (days)	Annual phosphorus retention	
	Measured values[b]	Calibration values
(B) Shows the phosphorus retention changes with varying hydraulic residence time		
2	62	32.1
5	57	57
14	90	77.5

	Constituent	Measured values[c]	Calibration values
(C) Shows the seasonal retention efficiencies for water, sediments and phosphorus			
Wet season	Water	46.5	45.1
	Sediments	82.0	81.7
	Phosphorus	60.0	55.1
Dry season	Water	94.3	97.1
	Sediments	97.0	90.9
	Phosphorus	95.0	96.1
Overall	Water	58.8	55.6
	Sediments	89.0	84.0
	Phosphorus	80.0	79.0

[a] Data from Rushton (1996).
[b] Data from Rushton et al. (1996).
[c] Data from Carr and Rushton (1995).

effects of changing the network organization. Because this model addresses primarily conceptual issues, correspondence between annual predicted and observed flows was deemed sufficient (Table 5).

4.2. *Minimum area determination*

The calibrated model was first used to determine the area requirements to prevent flooding during an average rainfall year under current land use conditions. A flooding event is defined simply as any day during which the storage capacity of the regional stormwater collection systems is exceeded, releasing large pulses of water to Biscayne Bay.

To prevent overflow during an average rainfall year, a minimum basin coverage of 8.2% was required for the network scenario, compared with approximately 7.4% for the baseline scenario. For a large rainfall year, a coverage of 10% resulted in only one overflow event (a 6″ storm event) for both scenarios. Dade County land-use projections for 2015 (SFWMD, 1994) were used to plan for future development. The minimum WSTA area to prevent overflow for the medium rainfall year rose slightly to 9.0 and 8.3% for the network and baseline scenarios, respectively. A WSTA land area of 10% was used for the remainder of the study to ensure that the model results represent conservative estimates.

4.3. *Outflow comparisons*

4.3.1. *Flow attenuation*

Stormwater treatment systems are designed primarily to buffer downstream ecosystems from large pulses of water. Fig. 7 presents a comparison of the baseline and network scenarios for daily outflow for a medium rainfall year. For maximum flow events, the network scenario reduced peak flow between 30 and 70% with a mean of 48%. It is clear that the basin hydrograph recession is slower under the network scenario.

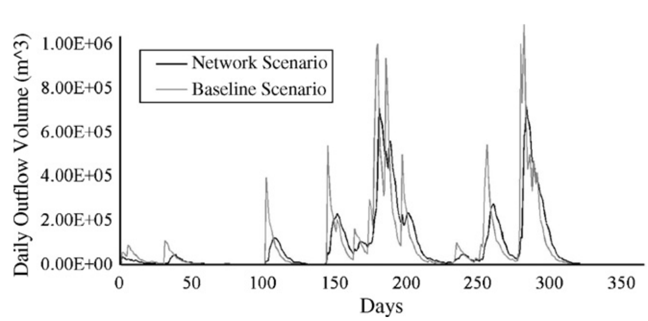

Fig. 7 – Comparison of daily outflow volumes (cm^3) for baseline and network scenarios for a medium rainfall year.

Table 4 – Model sensitivity analysis (increase and decrease model parameters by 25%)

Parameter	Base value	Units	Increase parameter 25%				Decrease parameter 25%			
			Peak flow	Water	SS	P	Peak flow	Water	SS	P
Watershed and hydraulic parameters										
Calibration water depth	0.2	m	−4.42	2.31	1.42	0.20	13.11	−3.67	−3.47	−2.39
Depth of upland soil	3	m	−8.01	5.72	0.31	−0.03	9.92	−7.06	−0.59	−2.58
Peat depth	0.25	m	−0.03	−3.41	−0.01	−0.39	−0.22	11.91	0.04	2.94
Calibrated HRT	10	day	−8.55	4.60	2.57	2.81	34.65	−6.26	−7.58	−6.63
Hydrologic module parameters (Table 1)										
KSeepage	4.00E−06	$cm\,s^{-1}$	−0.11	5.96	0.02	1.47	−0.06	−6.81	−0.02	−0.77
Weir height (WH)	0.25	m	1.99	4.85	−0.13	1.60	−1.30	−3.61	0.10	−1.83
Evapotranspiration (ET)	Varies[a]	$m\,day^{-1}$	−14.31	23.82	0.89	1.05	9.48	−8.73	−0.89	0.23
KDeepSeepage	5.10E−03	$m\,day^{-1}$	−6.51	7.40	0.12	−4.28	7.99	−6.70	−0.73	−0.63
Sediment/phosphorus module parameters (Table 2)										
KsedSettle	0.6	$\%\,day^{-1}$	0.00	0.00	3.01	0.81	0.00	0.00	−7.74	−2.09
KPSed	40	%	0.00	0.00	0.00	0.69	0.00	0.00	0.00	−0.96
Kbiomass	6	$g\,m^{-2}\,year^{-1}$	0.00	0.00	0.00	−0.56	0.00	0.00	0.00	−2.11
Kadsorb	0.75	$\%\,day^{-1}$	0.00	0.00	0.00	−1.24	0.00	0.00	0.00	2.37
Kdesorb	0.3	$\%\,day^{-1}$	0.00	0.00	0.00	−1.56	0.00	0.00	0.00	3.32
EPC	0.07	mg/l	0.00	0.00	0.00	−4.97	0.00	0.00	0.00	4.20
KOtoIn	0.15	$\%\,day^{-1}$	0.00	0.00	0.00	1.98	0.00	0.00	0.00	−1.50
KOrgPrelease	0.173	$\%\,day^{-1}$	0.00	0.00	0.00	−2.42	0.00	0.00	0.00	4.64

Given are the percent change observed in peak flow reduction, and annual water, sediment and phosphorus retention for a standardized change in each model parameter. Sensitivity was evaluated for the baseline scenario.

[a] To test the sensitivity of the model to changes in ET, we modified each monthly ET estimate by 25%. This magnitude of ET uncertainty is reasonable for the Blaney–Criddle method vis-à-vis the Penman–Monteith method (Shih et al., 1981).

Table 5 – Constituent flow validation results

Constituent	Observed annual load (g)	Predicted annual load (g)	Simulated-to-estimated	Observed mean discharge concentration (mg/l)	Predicted mean discharge concentration (mg/l)	Simulated-to-estimated
Total phosphorus	2.73E+06	4.42E+06	1.62	0.024	0.043	1.79
Total suspended solids	3.63E+08	4.56E+08	1.26	3.19	4.38	1.37

Model predictions for 1992 (medium rainfall year) were compared with observed annual discharge mass (g) and mean discharge concentration (mg/l). Estimates of actual loads to Biscayne Bay from the C-102 (Princeton Canal) basin are from Alleman (1995). Annual hydrologic flows were 1.14E8 m^3 (observed) vs. 1.04E8 m^3 (predicted).

4.3.2. *Pollutant removal*

Mass removal of sediments and phosphorus are compared between scenarios (Fig. 8) with strong evidence for improved retention of both under the network scenario. Maximum sediment flows (Fig. 8a) are reduced an average of 82% using a network approach, with a range from 56 to 96%. For phosphorus (Fig. 8b), peak flows are reduced 30–90% with a mean of 68%. The same general trend exists for outflow concentrations.

4.3.3. *Overall retention*

Basin annual retention rates (Fig. 9) shows overall flow reduction, indicating that water spends longer on the landscape in the network scenario. On an annualized basis, the network

Table 6 – Comparison of hydrologic effects of anticipated future development with various network configurations, showing the impact of increasing development (current vs. future) and the potential for mitigating those effects by augmenting portions of the network by 2% of the total basin area

Scenario	Annual retention	Annual retention decrease (%)	Mean peak flow	Peak flow increase (%)
Current	2.97E+07	–	6.34E+05	–
Future	3.53E+07	18.67	7.02E+05	10.63
+2% all	3.37E+07	13.37	6.92E+05	9.07
+2% large	3.35E+07	12.81	6.84E+05	7.75
+2% medium	3.28E+07	10.27	5.90E+05	−7.01
+2% small	3.30E+07	11.18	6.54E+05	3.07

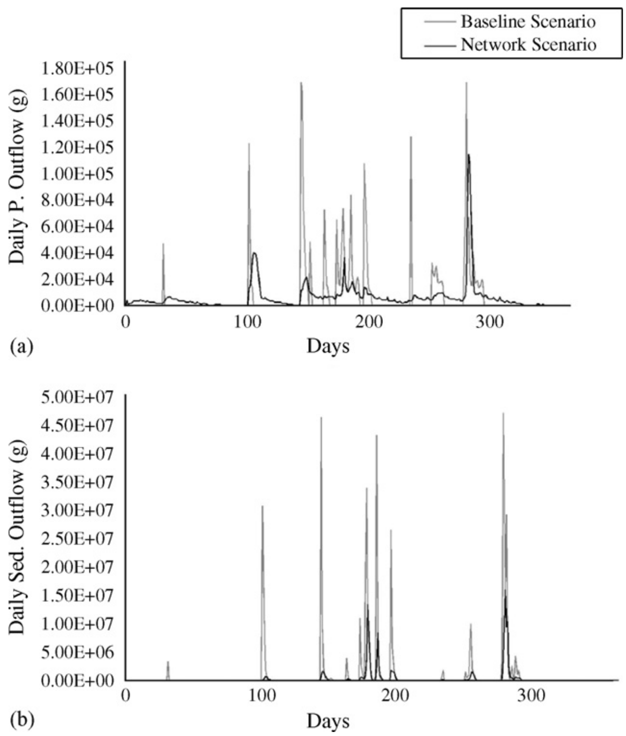

Fig. 8 – (a) Total phosphorus daily export (g) and (b) total sediment daily export (g) under baseline and network scenarios.

approach retains water longer than the baseline scenario, resulting in reduced outflows (31% less flow), and concomitant reductions in sediments (36%) and phosphorus (27%). It should be noted that longer hydraulic residence times make the stormwater system in the network scenario more susceptible to overflow during large rainfall events. This tradeoff needs to be weighed in a more detailed design-oriented model.

4.4. *Treatment roles at each scale*

Fig. 10 suggests that different wetland sizes provide different treatment roles. For water, large wetlands retained the most (53%), while small wetlands with flashy hydro-graphs retained

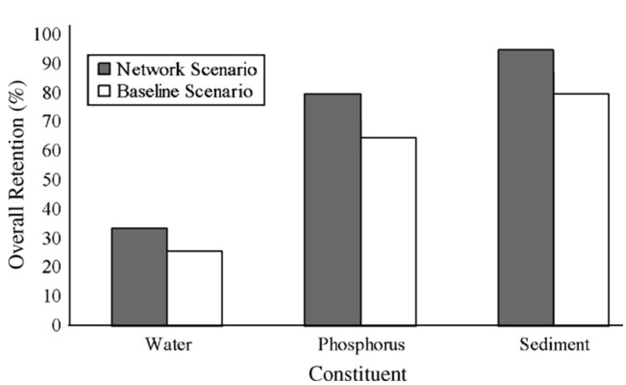

Fig. 9 – Comparison of overall annual retention for water (m^3), total P (g) and sediments (g) between baseline and network scenarios.

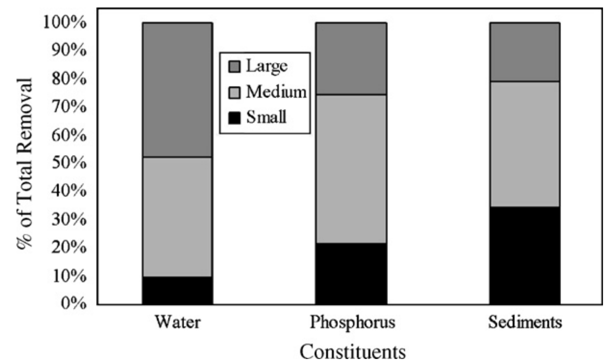

Fig. 10 – Removal roles for each wetland size class in a hierarchical network. Roles are shown as a percentage of overall removal achieved by each size class.

the least (5%). For phosphorus, medium wetlands retained the most (65%) while for small wetlands retained 30% of the sediment load despite their small total area.

4.5. *Optimizing network configuration for developed landscapes*

Results reported to this point were for a network configured to emulate spatial pattern and size class distributions found in reference basins (hereafter called the original network). However, in a developed basin, it is possible that an alternative configuration could be more effective. Fig. 11 shows results of network optimization using annual retention as the objective function. As the relative contribution of small wetlands in the network increases from the original formulation (2% of total wetland area), annual retention increases, though slightly (18, 7 and 2% for water, total phosphorus and sediment, respectively). When small wetlands represent about 60% of the total wetland area, maximum annual retention is observed; beyond this level, increasing the small wetland fraction is detrimental.

However, using annual retention as the objective does not necessarily ensure that maximum peak flow attenuation is

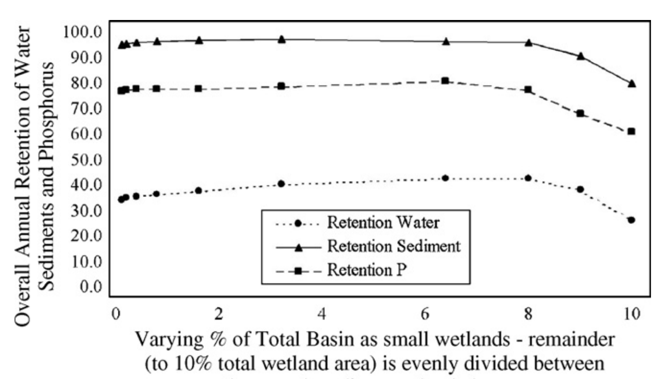

Fig. 11 – Illustrates effects of altering the network configuration using overall annual retention as sole optimization criteria. Total wetland area is constant (10% of the watershed) but allocation to small wetlands is varied, with remaining area allocated equally to medium and large wetlands.

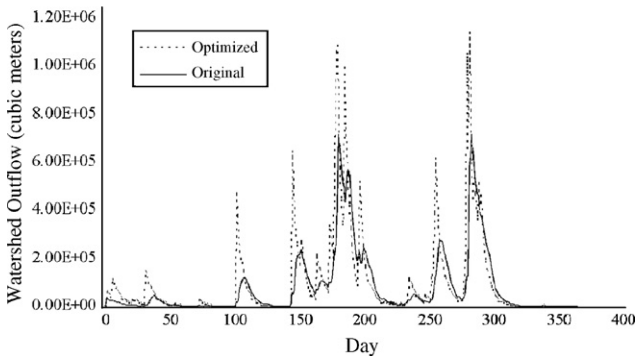

Fig. 12 – Comparison of daily flow delivery from the original network, prior to optimization, with the network optimized for annual retention.

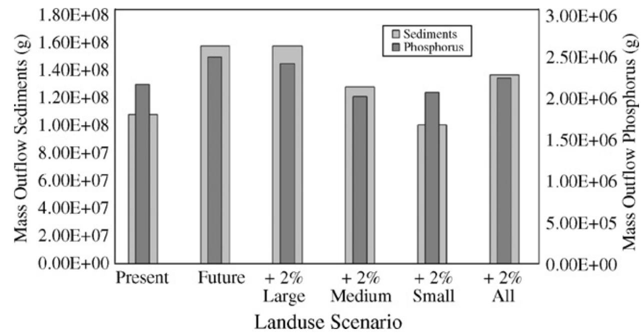

Fig. 13 – Response of various network manipulations to increased development. To mitigate effects of increased runoff and pollutant load, an additional 2% of the basin was allocated to stormwater retention, with varying effects depending on how that additional area was allocated.

achieved. To verify this, we compared the original network (emulating the natural size class distribution) with the network that maximizes annual retention. The results for the daily outflow pattern are shown in Fig. 12. If the objective is to reduce peak flows, the original network outperforms the network selected based on optimized annual retention. One quantitative measure of this effect is the mean percentage difference between the major peak flow events. For water flows, the original network outperforms the annual optimized network by almost 50%. The same results are observed for P (average peak flow reduction = 36%) and TSS outflows (average peak reduction = 39%). Increased treatment efficiency for the annual optimized network arises from reduced base-flows rather than attenuated peak flows.

4.6. *Modifications of the network to manage increased development*

The model was used to predict effects of increased development on watershed outflow characteristics. Using original hierarchical network with 10% of the total catchment area devoted to WSTAs as a baseline, we explored various strategies for mitigating effects of increased development. The first simply maintains the relative spatial distribution of wetlands; that is, increase the area of small, medium and large wetlands proportionally. Another approach is to increase the area of just one size class. We compared flows of water and constituents under anticipated future development (2015 Dade County projected land-use coverage—SFWMD, 1994) using the baseline network with the flows when an additional 2% of the basin was allocated, in various ways discussed above, to wetland treatment areas. Fig. 13 and Table 6 summarize the results.

The hydrologic flows were most effectively mitigated by adding the additional area to medium wetlands, both for total flow and for peak flow attenuation (Table 6). For the constituent mass flows (Fig. 13), the additional area was optimally allocated to small wetlands for reducing sediments, and to medium wetlands for reducing phosphorus flows. In all cases, adding the additional area to the large terminal wetlands did not provide improved treatment. Likewise, simply increasing the network area without changing the configuration did poorly in comparison with modifications targeted at specific size classes.

5. Discussion

This study explored the role of hierarchy in the design and management of stormwater collection and treatment systems in human dominated, low-relief watersheds. Analogous hierarchical patterns can be found for all landscapes dominated by flowing water (Pelletier, 1999), in the spatial distribution of cities (Brown, 1980), in the national transportation system, and in the spatial patterns of mineral deposits and mountain ranges (Odum, 2000). General systems theory suggests that hierarchy emerges in self-organizing systems to converge and concentrate energy from diffuse sources. While the hierarchy proposed herein is clearly not self-organizing, explicitly emulating the observed natural pattern in a human-dominated landscape has intuitive appeal, and this research demonstrates considerable promise for improving the characteristics of basin outflows.

The model constructed to explore the potential for a hierarchical network to improve stormwater treatment is a considerable simplification of watershed hydrology and pollutant transport. The hydrologic component has been validated with moderate accuracy, sufficient to assume that simplifications are appropriate for macroscopic analysis. The pollutant component was verified for annual flows, but temporal dynamics remain unverified. While the model is driven by rate coefficients available in the literature, a necessary next step in evaluating these results for anything more than conceptual inference is acquisition of adequate (i.e., daily) water quality validation data at the watershed scale.

Several other considerations might also improve the model. First, interactions between surface and groundwater are passive. Upward groundwater flow to surface water is neglected, resulting in the tacit assumption that WSTAs are perched above the water table. Given the porous nature of the regional geology, this assumption is limiting. Second, the dramatic absence of relief in the study area slow flow conveyance, resulting in backflows or lengthening considerably the flow time between wetlands (assumed equal to 1 day in the model). Only a highly detailed distributed parameter model would be capable of addressing these issues, introducing substantial complexity. We sought to avoid such model complexity for this conceptual effort to understand the role of spatial hierarchy, but perceive a need for such models for further research.

The conventional approach to sizing wetlands for wastewater processing involves application of statistically determined linear settling rates (Kadlec and Knight, 1996). As presented, stormwater wetland systems likely violate steady-state linearity assumptions, particularly with regard to stochastic variation in water column concentration gradients. We chose to model phosphorus dynamics in more detail. However, to ensure that the model was behaving in a manner reasonable given what is known about wetland processing potential, effective linear settling rates for total P removal for each wetland size class were computed. The back-calculated effective linear settling rates for total phosphorus were 9.33, 7.14 and 4.25 m/year for small, medium and large wetlands, respectively. Kadlec and Knight (1996) report settling rates for dissolved phosphate of between 3.6 and 21.6 m/year, which compares favorably to our model values. Decreasing settling rates with increasing size class arises due to reduced concentration gradients in downstream wetlands.

The spatial extent of treatment wetlands was set at 10% to be conservative. Tilley and Brown (1998), also using conservative estimates, found that, for regional runoff characteristics, approximately 10% of basin area should be devoted to stormwater retention. It is important to note that this value was used for both scenarios (baseline and network) in the model, and that coefficients describing wetland process dynamics were held constant.For all proposed success criteria outlined, the network outperforms the baseline. Improvement in annual retention, while certainly encouraging, is considered less noteworthy than dramatic attenuation of peak water and constituent flows. The constituent results presented were on a mass basis; mean concentrations of P were 0.04 mg/l versus 0.07 mg/l for network and baseline scenarios, respectively. This difference is significant considering the low concentrations in Biscayne Bay (7–10 ppb—SFWMD, 1994). This nominal outflow concentration is most sensitive to the equilibrium P concentration, the estimate of which we consider conservative.

The observed roles of each wetland size class loosely confirm earlier results presented in Tilley and Brown (1998), and Rushton (1997). Rushton (1997) showed that, in a small-scale treatment train, sediments were removed first, phosphorus was removed in the primary retention basin, and flows of water were attenuated in the large secondary wetland at the terminal collection point. While the scale of assessment is quite different, the general finding of partitioned network roles reinforces the conclusion of this model.

The potential to modify the network configuration to mitigate the increased flows associated with development indicates a regional planning strategy. Because the result of augmenting the area of large wetlands to mitigate effects of increased development was relatively small, network alterations in response to increasing development would be for small and medium wetlands only. Implementation of the regional network could proceed, therefore, first with large wetlands, increasing the spatial extent of other classes as needed. An ancillary benefit of the network approach is that the loading signature (flow regime and nutrient concentrations) to large wetlands is similar to that observed in natural wetlands, implying that these areas may function as viable habitat.

This model was designed to test a concept; as with other models, intent informs design, and numerous extensions and refinements are needed. In particular, the hydrologic accuracy of this model is of limited utility for basin design purposes, and would need to be transcribed to more sophisticated finite-element models (e.g., the regional simulation model under development at the South Florida Water Management District) in order to predict fluxes and storages with sufficient precision for planning and siting. The same can be said of the spatial and temporal resolution, which were selected in this model for the purpose of parsimony. However, the numerous models that have been developed for predicting the hydrologic and biogeochemical behavior of wetland systems (Wong and Geiger, 1997; Wang and Mitsch, 2000; Raghunanthan et al., 2001; Musacchio and Grant, 2002; Zhang and Mitsch, 2005) generally adopt similar levels of process specificity with similar system-scale accuracy levels, and could be adapted for use as basin-scale planning tools with integration of network flow convergence. Algorithms for internal processes may be easily amended to accommodate emerging data and theory; the primary conclusion of this work, however, is that large scale organization of treatment systems makes the most substantial difference in predicted water quality.

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