Appendix 4B-9: Florida Lake and River Survey

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Long-term phosphorus removal in Florida aquatic systems dominated by submerged aquatic vegetation

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Abstract

Anthropogenic phosphorus (P) loads have been implicated in eutrophication of lakes and wetlands throughout Florida. One technology that holds considerable promise for controlling these loads in a cost-effective manner is the use of treatment wetlands. Preliminary research in south Florida on the use of submerged aquatic vegetation (SAV) as the dominant vegetation in these treatment wetlands is reporting higher P removal performance than wetlands dominated by rooted, emergent plants. This research has been based to-date primarily on relatively small-scale mesocosms (5–2000 m^2) and on a larger scale treatment wetland (148 ha) that has been operated for about 7 years. Considering the magnitude of engineering decisions and project costs to implement P control in the Everglades Agricultural Area and elsewhere in Florida, it is prudent to look for additional confirmation of P removal performance from other existing SAV-dominated systems in Florida that have a longer operational period. This paper describes an analysis of existing data collected from a number of SAV-dominated, flow-through lakes and rivers in Florida with characteristics similar to the proposed SAV treatment systems. While these existing input–output data were not specifically collected for the purpose of preparing mass balances and P removal rate estimates, they can be judiciously applied to that analysis. The overall conclusion of this analysis is that SAV-dominated lakes and rivers do typically remove P from the water column. The likely long-term sink for this P is the newly accreted sediment. The long-term average P removal rate for 13 SAV-dominated lake and river systems in Florida was 1.2 g/m^2 per year. This result compares favorably with an average net sediment P accumulation rate of 1.2 g/m^2 per year reported by others for 11 SAV-dominated Florida lakes. These estimated long-term P removal rates are higher than those for full-scale wetlands dominated by emergent vegetation (Treatment Wetlands (1996); Wetlands Ecol. Mgmt. 4 (1997) 159). Average first-order P removal rate constants for SAV-dominated lakes (15 m/year) and rivers (46 m/year) are generally less than those estimated in SAV-dominated mesocosms (60–140 m/year) and similar to a large-scale SAV-dominated stormwater treatment area (STA) (40 m/year). P removals in all of these SAV-dominated systems are influenced by inlet P loading rates, with removal rates positively correlated to both P inlet concentration and hydraulic loading rate (HLR). Based on this analysis, caution is recommended when extrapolating the P removal results from relatively short-term or small-scale mesocosm studies to the design of full-scale, long-term operating SAV-dominated wetlands.

Keywords: Everglades; Phosphorus; Submerged aquatic vegetation; Treatment wetlands; Stormwater treatment; Florida lakes

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1. Introduction

Over 30 years of research have convincingly demonstrated that elevated levels of phosphorus (P) have contributed to the alteration of natural aquatic and wetland ecosystems in south and central Florida (Reddy et al., 1999a; Miao and DeBusk, 1999; McCormick and Scinto, 1999; Schelske et al., 1999). A variety of efforts are underway to reverse the historic trend of increasing releases of total P (TP) to these impaired ecosystems. These activities include source control through improved best management practices, and treatment of TP-contaminated effluents and stormwater through conventional wastewater treatment facilities and natural nutrient polishing systems such as treatment wetlands (Goforth, 2000; Reddy et al., 1999b; Kadlec, 1999; Kadlec and Knight, 1996; Guardo et al., 1995; Kadlec, 1994). Stormwater treatment areas (STAs) were constructed in south Florida to reduce TP from surface waters (Moustafa et al., 1999). These STAs incorporate a variety of dominant plant communities ranging from emergent macrophytes to periphytic algae.

In Florida, many shallow aquatic systems are dominated by submerged aquatic vegetation (SAV) (Duarte et al., 1986). These systems often show better water quality (clarity, total suspended solids, pH, Chl *a*, TP and TN) than other systems (O'Dell et al., 1995; Canfield and Hoyer, 1992; Leslie et al., 1983). Submerged macrophytes have also been used in wastewater treatment for nutrient removal (Gu et al., 2001; Brix and Schierup 1989; McNabb, 1976). There are several advantages to using SAV as opposed to cattail dominated systems as a means for removing P. First, SAV systems utilize nutrients from both the water column and sediments. Non-rooted submerged macrophytes such as *Ceratophyllum demersum* and *Chara* spp. appear to rely exclusively on dissolved nutrients from the water column (Barko and James, 1997). Moreover, the ability of SAV to occupy most of the water column allows it to remove nutrients from the water column effectively and to minimize effects of hydraulic short-circuiting. Second, the high rate of photosynthesis by submerged macrophytes raises the water column's pH, which in turn may lead to co-precipitation of soluble reactive P (SRP) with $CaCO_3$ (Murphy et al., 1983). Third, a dense SAV community provides extensive surface area for periphyton growth that in turn helps remove nutrients from the water column. Lastly, an SAV plant community may physically filter, detain and cause sedimentation of suspended solids which contain organic P or adsorbed inorganic P.

A treatment system dominated by SAV is one of several advanced treatment technologies being evaluated by the South Florida Water Management District (District) and Florida Department of Environmental Protection. As required by the United States Army Corps of Engineers (USACE) 404 permit, the District has been evaluating selected treatment technologies for their effectiveness in the removal of P (PEER Consultants P.C./Brown and Caldwell, 1996; Jorge et al., 2002). Recent findings indicate that SAV-dominated mesocosms (stocked with *C. demersum*, *Chara* spp. and *Najas guadalupensis*) are capable of reducing TP concentrations in agricultural runoff to levels as low as ~20 μg/l (Gu et al., 2001; Dierberg et al., 2002). Between May 1998 and April 1999, the mean TP concentration in Cell 4 (148 ha) of Stormwater Management Area 1 West (STA-1W), a full-scale constructed wetland dominated with *Najas* and *Ceratophyllum*, was reduced from 36 μg/l in the inflow to 14 μg/l in the outflow (Jorge et al., 2002).

SAV has also been associated with increased releases of P from the sediments of shallow aquatic systems. Two potential processes for increased P cycling in SAV-dominated ecosystems are: uptake of sediment P through the roots and eventual plant tissue decomposition and nutrient release (Barko and Smart, 1980), and the altered redox potential in the surface sediments due to high rates of SAV metabolism and resulting releases of iron-bound P under chemically reducing conditions (Frodge et al., 1991). Of interest in design of SAV-dominated treatment wetlands is the net balance between the competing processes of P uptake and release that ultimately result in the long-term sequestering of P in stable sediments.

The ultimate sink for P removal in an SAV-dominated treatment wetland is burial of P in stable accreted sediments. Direct measurement of this net burial rate is one approach to assessing the long-term P removal performance of SAV wetlands. A second approach is preparation of P mass balances based on net differences between inflow and outflow P masses. The preferable analytical method to quantify long-term P removal in aquatic and wetland systems is to conduct both types of measurements on a single system.

Research on SAV treatment systems has utilized several different experimental scales, ranging from small mesocosms to full-scale constructed wetlands (over 1000 ha; Gu et al., 2001). While data from mesocosms and constructed wetlands are convincing, these systems have been operated from only 1 to 7 years. Long-term TP accumulation may not be accurately reflected in inflow/outflow mass balance data or from sediment cores collected from these relatively young systems. Additional information on long-term system performance can be obtained from large aquatic systems with a long history of SAV dominance to verify the short-term performance for phosphorus removal in the recently constructed systems. For the purposes of this study, P removal rates in natural Florida aquatic systems are analyzed in an attempt to validate the results of SAV-dominated treatment wetlands in south Florida.

Many SAV-dominated lakes and rivers exist in Florida and some of them have been closely monitored for several key water quality parameters (FDEP, 2000; Florida LAKEWATCH, 1997; Canfield and Hoyer, 1992). This paper provides an evaluation of TP removal data from a number of these Florida systems. Two lines of evidence on long-term net P removal as a function of SAV are examined: input-output mass balances and P accumulation rates in lake sediments.

The primary objective of this study was to calculate rates of long-term TP removal in selected Florida SAV-dominated systems by analyzing previously collected inflow and outflow water quality data. This approach has been successful in comparing P removal performance in constructed wetlands (Kadlec and Knight, 1996). Specifically, this analysis was designed to answer the following two questions:

- Have Florida aquatic systems with a history of SAV dominance effectively removed phosphorus over the long term, and at what rates?
- How do aquatic systems in Florida with a history of SAV dominance respond to changes in TP loading, SAV cover, and hydraulic variations?

In addition to these two questions, we also summarized information on related water quality changes in SAV-dominated lakes. These ancillary independent variables were used to correlate with TP removal rates to provide a preliminary look at the factors that might be important during design of SAV-dominated STAs. Where available, the estimated input–output mass balances were compared with published sediment P accumulation rates to provide an independent estimate of the accuracy of this approach.

2. Methods

Florida has over 7700 lakes that range in size from 4 to over 180 000 ha (Canfield and Hoyer, 1992). In addition, there are a number of spring-fed, SAV-dominated rivers in the state. The first task of this study was to identify SAV dominated water bodies and then to locate inflow/outflow water quality and flow data for each system.

A variety of sources were used to gather historical environmental data (physical, chemical, and biological data) on SAV dominated systems. These include:

- United States Geological Survey (USGS)—surface water quality and flow database (http://water.usgs.gov/nwis).
- STORET, database operated by the US Environmental Protection Agency (EPA) Office of Water—surface water quality data (http://www.epa.gov/storet).
- Florida Department of Environmental Protection (FDEP) Bureau of Invasive Plant Management-SAV database for the years 1983–2000.
- Florida water management districts—surface water quality and flow data.

- Florida LAKEWATCH, a volunteer citizen's lake monitoring program operated by the University of Florida's Department of Fisheries and Aquatic Sciences (http://lakewatch.ifas.ufl.edu).
- Other published information sources.

2.1. Selection of SAV-dominated Florida aquatic systems for analysis

The most complete and detailed SAV information resource was the FDEP Bureau of Invasive Plant Management SAV database for the years 1983–2000, which summarizes annual inventories of hundreds of lakes and other aquatic systems (including rivers) throughout Florida. There are 460 different water bodies included in this database and it was the only data set found that provided methodologically consistent, year-to-year SAV coverage information for a large number of Florida aquatic ecosystems.

The FDEP SAV database was used to reduce the list of water bodies for further data collection. An arbitrary cutoff was used to eliminate water bodies with low SAV coverage. The lakes and streams that were kept on the candidate list were generally those with an average SAV coverage of at least 20% during at least 1 year. Only years with full plant surveys (as opposed to partial and presence/absence surveys) were included in the quantitative SAV numbers in the exhibit. A few additional lakes that did not quite make this cut but had macrophyte coverage close to the 20% cutoff and also are known to have water quality and flow data were retained. This initial selection process resulted in a list of 95 SAV-dominated aquatic systems. All of these water bodies are potential candidates for a more intensive data collection effort and analysis of TP removal in SAV-dominated aquatic systems in Florida.

This list of SAV-dominated aquatic systems was further reduced in size by focusing attention on flow-through dominated lakes and rivers and eliminating isolated lakes and lakes whose water budgets are dominated by non-point contributions. The availability of both flow and water quality data from the sources listed above for each of these aquatic systems was used to develop the final list of SAV-dominated aquatic systems which include 11 lakes and two rivers. Fig. 1 presents a general location map for the final list of SAV-dominant aquatic systems evaluated in this paper.

Site characteristics such as surface area, depth, flow, nominal hydraulic retention time, and dominant vegetation for each of the study lakes and rivers were gathered through the sources listed above. The surface area for most of the water bodies was obtained from the DEP SAV database and the remaining areas were estimated from area maps and published literature. Mean depths were obtained from Florida LAKEWATCH, published literature, and STORET. Dominant SAV plant species for each system were identified using the DEP SAV database. Stream flow data were obtained from USGS. A majority of the study lakes and rivers only had one USGS gauging station; therefore, the missing inflow or outflow data were estimated by using the value from the one available gauge.

2.2. Data analysis

Total phosphorus removal rates were estimated from existing data sources for each of the SAV-dominated systems. Mass balances for TP were prepared from available flow and concentration data for each SAV-dominated system. These data sets are unfortunately incomplete and either surface outflows or inflows had to be estimated for six of the 13 study systems. Inflow and outflow TP data were available for all of the selected study sites.

Due to the high variability in local rainfall and lack of information associated with the database, no effort was made to estimate rainfall or non-point source inputs. The selected systems are all thought to be dominated by flow-through and our experience in calculating water balances for the STAs indicate that rainfall and evapotransportation (ET) are roughly equal, and errors from ignoring rainfall and runoff may be minor. In two cases, complete water and TP mass balances prepared by others were used (Lake Panasoffkee and Lake Tarpon).

Pollutant removal rates were summarized as a simple logarithmic decay (first-order process) by

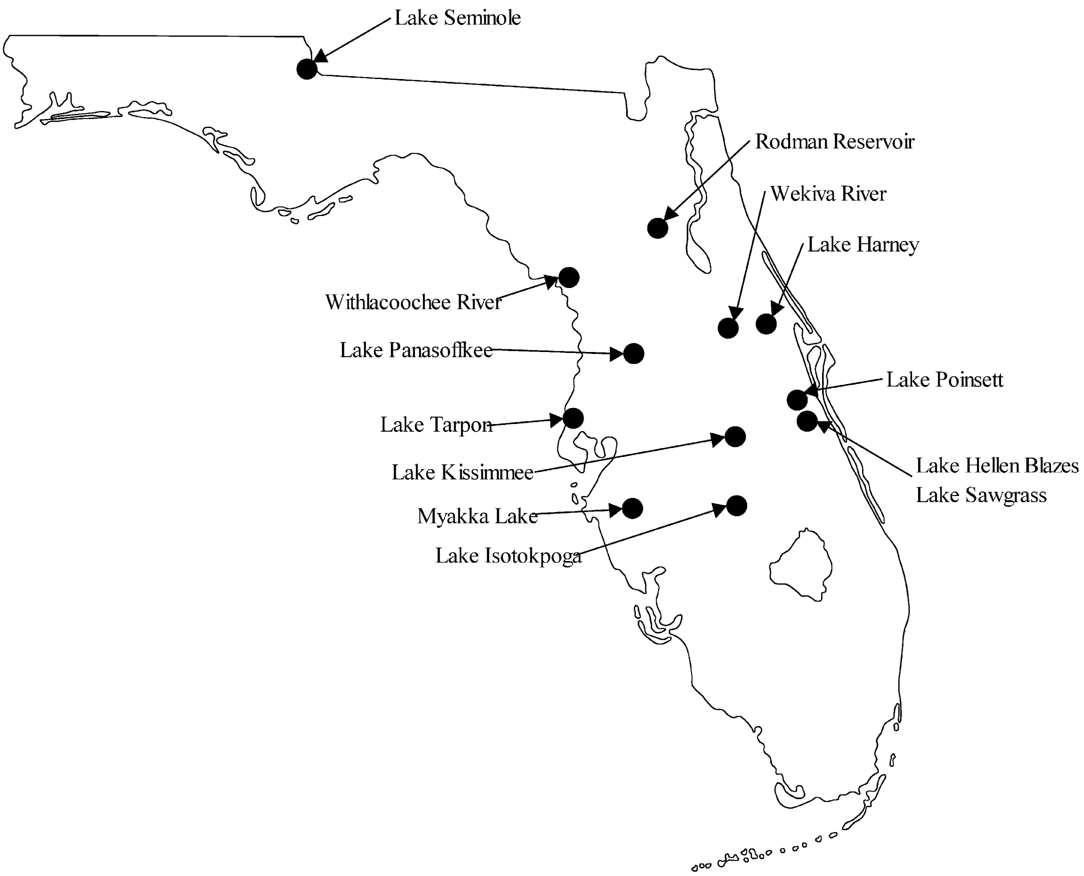

Fig. 1. General location map for final list of SAV-dominant aquatic systems.

the use of inflow/outflow concentrations and hydraulic loading data. While traditional lake eutrophication studies have focused on volume-based models (Vollenweider, 1975; Havens and James, 1997), macrophyte-dominated wetland performance is tied more closely to surface area than to water volume (Kadlec and Knight, 1996), so an area-based model is typically more appropriate than a volumetric first-order model.

The simplest expression of the first-order, area-based plug flow wetland performance model, assuming no net rainfall or seepage, is:

$$\ln\left(\frac{C_1}{C_2}\right)=\frac{k_1}{q} \quad (1)$$

where C_1, average inlet concentration (mg/l); C_2, average outlet concentration (mg/l); k_1, first-order, area-based rate constant (m/year); q, average hydraulic loading rate (HLR in m/year).

This is the general form of the model and is referred to as the one-parameter or k_1 plug-flow model. When referring to TP removal in this paper, k_1 is interchangeable with $k_{1\mathrm{TP}}$.

Data from many aquatic systems indicate that internal and external loading of TP may result in non-zero, irreducible water column constituent concentrations (Kadlec and Knight, 1996). For some purposes these concentrations may be so low as to be indistinguishable from zero. In other cases, effluent discharge goals approach the lowest constituent concentrations measured in natural aquatic systems. In these situations, the plug flow model can be corrected by introducing a second parameter that represents the lowest achievable or irreducible concentration, C^*. The two-parameter first-order, area-based plug flow model, or $k-C^*$ model, is:

$$\ln\left[\frac{(C_1 - C^*)}{(C_2 - C^*)}\right] = \frac{k}{q} \tag{2}$$

where k is two-parameter model first-order, area-based removal rate constant (m/year).

Flow-weighted inlet and outlet concentration data were combined with average HLR (q), to estimate k and C^* for a given data set. Calculations are based on estimated monthly, quarterly, or annual averages with 'paired' inflow/outflow TP concentrations. The selection of the data-averaging time period was based on the nominal hydraulic residence time (HRT) of the water body. The averaging period used was always longer than the nominal HRT for each particular system.

Most wetland and aquatic systems do not behave as pure plug flow reactors (Kadlec, 2000; Kadlec and Knight, 1996). In these cases the tanks-in-series (TIS) model has been used to describe the observed deviation from plug flow. This model assumes that flow through the aquatic system is similar to a number of completely mixed stirred reactors in series. A tracer study should be performed on the wetland to determine the residence time distribution of that system and to estimate the number of reactors that would best describe the system (Levenspiel, 1999). The TIS model can be written as:

$$\frac{(C_2 - C^*)}{(C_1 - C^*)} = \left(1 + \frac{k_{\mathrm{TIS}}}{nq}\right)^{-n} \tag{3}$$

where k_{TIS} is the two-parameter TIS, area-based removal rate constant (m/year) and n is the number of TIS.

The plug flow reactor rate constant is renamed as k_{PFR} and is related to k_{TIS} by the following equations:

$$k_{\mathrm{TIS}} = nq[(e^{(-k_{\mathrm{PFR}}/q)})^{-1/n} - 1] \tag{4}$$

$$k_{\mathrm{PFR}} = nq\left[\ln\left(1 + \frac{k_{\mathrm{TIS}}}{nq}\right)\right] \tag{5}$$

Data on the number of TIS that best fits the hydraulics of each of the SAV-dominated systems were not available. For this analysis, it was assumed that $n = 1.5$ for each of the lakes and $n = 4$ for the two rivers. These assumed values reflect the expected higher hydraulic efficiency of the river systems compared with the more short-circuited lakes. A more detailed analysis might benefit from tracer testing of subject lakes and rivers to better define their actual hydraulic properties. Flow-weighted inlet and outlet TP concentration data and average HLR data were used to provide a calibration of the $k-C^*$ model. All 'paired' data (both inlet and outlet concentration data) collected from the above sources were utilized and the Excel Solver routine was employed to provide the best-fit calibration to these data sets. The two parameters k_{PFR} and k_{TIS} were corrected to k_{20} values ($T = 20$ °C) based on the modified Arrehnius relationship:

$$k = k_{20}(\theta)^{(T-20)} \tag{6}$$

where θ is the temperature correction factor (Kadlec and Knight, 1996).

3. Results

Site characteristics for each of the study lakes and rivers are presented in Table 1. The study systems ranged from a segment of the Withlacoochee River (147 ha) to larger systems like Lake Kissimmee (14 143 ha) and Lake Seminole (17 408 ha). Mean depths ranged from 1.1 (Wekiva River and Lake Poinsett) to 5.7 m (Seminole Lake). Mean inflows ranged from 0.12×10^6 (Lake Tarpon) to 50.1×10^6 m^3 per day (Lake Seminole). *Hydrilla verticillata* was the dominant SAV species observed in the majority of the study systems. Annual SAV percent cover estimates from FDEP for these selected systems are summarized in Table 2.

Table 3 summarizes the general surface water quality for the study lakes and rivers. All values are based on mean flow-weighted inflow and outflow concentration estimates over the reported period of record. Inflow and outflow water quality stations are from a number of different agencies (STORET, USGS, water management districts) and therefore, do not necessarily have 'paired'

Table 1
Site information for select SAV dominated lakes and rivers in Florida

P.Q.R. flow summary

Water body	Area (ha)	Depth (m)	Volume ($m^3 \times 10^6$}	In (m^3 per day × 10^6)		Out (m^3 per day × 10^6)		Period of record	Nominal HRT (day)	Dominant Vegetation[a]
				Average	Median	Average	Median			
Lakes										
Harney lake	2452[a]	1.7[c]	41.7	4.68	3.29	4.68	3.29	October 1981–September 1998	8.9	*Vallisneria americana*, *H. verticillata*
Hellen Blazes lake	154[a]	2.2[d]	3.4	1.47	0.79	1.47	0.79	October 1970–September 1999	2.3	*H. verticillata*
Istopoga lake	11 207[a]	1.6[c]	179	0.80	0.47	0.87	0.48	January 1988–September 1998	224	*H. verticillata*
Kissimmee lake	14 143[a]	2.1[c]	297	2.27	0.88	2.27	0.88	October 1969–September 1999	131	*H. verticillata*, *Utricularia foliosa*
Myakka Lake, Upper	413[a]	1.2[e]	5.0	0.51	0.28	0.58	0.29	April 1989–September 1991	9.7	*H. verticillata*, *C. demersum*
Panasoffkee lake[f]	1805[a]	1.3[c]	23.5	0.32	–	0.30	–	May 1992–April 1993	74.1	*V. americana*, *N. guadalupensis*
Poinsett lake	1754[a]	1.1[c]	19.3	2.44	1.50	2.44	1.50	October 1953–September 1999	7.9	*H. verticillata*, *V. americana*
Rodman reservoir	3885[a]	2.4[e]	93.2	2.62	2.21	3.32	2.79	October 1968–September 1999	35.6	*H. verticillata*, *C. demersum*
Sawgrass lake[g]	195[a]	1.5[d]	2.9	1.47	0.79	1.47	0.79	October 1970–September 1999	1.9	*H. verticillata*
Seminole lake	17 408[b]	5.7[e]	992	50.1	50.8	56.1	56.4	January 1972–September 1999	19.8	*H. verticillata*
Tarpon lake[h]	1026[a]	2.2[f]	22.6	0.12	–	0.10	0.04	January 1996–December 1988	187	*H. verticillata*
Rivers										
Wekiva River	181[b]	1.1[e]	2.0	0.71	0.64	0.71	0.64	October 1935–September 1999	2.8	*V. americana*, *Nitella* spp.
Withlacoochee river	147[b]	1.4[e]	2.1	0.81	0.35	0.95	0.43	October 1983-September 1999	2.6	*H. verticillata*, *V. americana*

Underlined data were estimated flow.

[a] FDEP SAV database (2001).

[b] Estimated.

[c] Florida LAKEWATCH.

[d] Brenner (1997).

[e] STORET.

[f] CH2M HILL (1995).

[g] Includes little lake Sawgrass.

[h] SWFWMD (2000).

Table 2
SAV percent cover from FDEP SAV database for select lakes and rivers in Florida

Lake/river	FDEP 'Full' SAV survey % cover									
	Annual							Statistics		
	1983	1984	1986	1988	1990	1992	1995	Average	Maximum	Minimum
Lakes										
Harney lake	0.7	0.9	27	28	28	25	26	19	28	0.7
Hellen Blazes lake	–	–	38	24	88	93	33	55	93	24
Istopoga lake	1.9	2.1	3.8	47	32	12	69	24	69	1.9
Kissimmee lake	6.8	8.5	12	23	33	40	62	26	62	6.8
Myakka lake, Upper	41	98	29	25	57	36	11	42	98	11
Panasoffkee lake	94	69	69	87	63	58	71	73	94	58
Poinsett lake	1.3	4.4	25	32	33	19	30	21	33	1.3
Rodman Reservoir	27	48	44	40	36	44	18	37	48	18
Sawgrass lake	11	85	35	26	23	79	93	50	93	11
Sawgrass, little lake	–	–	–	34	86	82	96	75	96	34
Seminole lake	4.5	7.1	5.4	29	64	63	8.7	26	64	4.5
Tarpon lake	8.5	6.1	2.3	3.5	2.5	20	5.5	6.9	20	2.3
Rivers										
Wekiva River	–	–	17	9.3	6.7	93	47	35	93	6.7
Withlacoochee river	75	30	66	57	42	24	7.2	43	75	7.2

monthly data (include both inflow and outflow for a particular month).

Mean inflow TP concentrations in study lakes and rivers ranged from 54 to 371 μg/l, while mean outflow TP concentrations ranged from 33 to 386 μg/l. Mean inflow total nitrogen (TN) ranged from 0.90 to 2.01 mg/l and mean outflow TN ranged from 0.60 to 1.98 mg/l. Mean inflow color ranged from 46 to 195 pcu and mean outflow color ranged from 22 to 191 pcu. Mean inflow Secchi depth ranged from 0.52 to 1.88 m, which was similar to the mean outflow Secchi depth range of 0.66–1.89 m. Mean alkalinity ranged from 15 to 137, and 25 to 109 mg/l at inflow and outflow, respectively. Calcium concentration ranged from 8.7 to 62.3 mg/l at the inflow while only from 12.0 to 54.8 mg/l at the outflow. All of the constituent concentration averages are summarized in Table 3.

Table 4 includes a summary of estimated TP removal performance for the study systems. The mean TP inflow concentration was 120 μg/l for the lakes and 138 μg/l for the river systems. The mean TP outflow was 97 and 120 μg/l for the lakes and rivers, respectively. Estimated mean TP mass removals ranged from −1.6 (Upper Myakka Lake) to 7.1 g/m^2 per year (Lake Hellen Blazes). The average TP mass removal rates for lakes and rivers were 1.2 and 3.7 g/m^2 per year, respectively. Estimated k_1 values ranged from −7.5 to 67 m/year. The highest k_1 value was estimated for Lake Hellen Blazes and the lowest was estimated for Upper Myakka Lake. The estimated mean k_1 value is 15 m/year for lakes and 46 m/year for the rivers.

These values can be compared with a global estimate of k_1 of about 12 m/year for treatment wetlands dominated by emergent macrophytes (Kadlec and Knight, 1996), an estimated value of 18.4 m/year in the entire Everglades Nutrient Removal Project (ENRP) in south Florida (Chimney et al., 2000), estimated values ranging from 62 to 142 m/year for SAV-dominated mesocosm systems, and 28–55 m/year based on annual averages of ENRP Cell 4 (SAV-dominated), researched as part of the advanced technology demonstration project conducted by the SFWMD (DB Environmental Laboratories, 2000, DBEL). It is well known that k_1 is highly correlated with

Table 3
Summary of surface water quality of select SAV dominated lakes and rivers based on flow weighted inflow and outflow concentration estimates over the reported period of record

Lake/river	Wtr temp (°C)		pH (SU)		Color (pcu)		Secchi depth (m)		Alkalinity (mg/l)		Calcium (mg/l)		TSS (mg/l)		Period of record
	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out	
Lakes															
Harney Lake	23.7	24.1	7.08	7.56	167	112	0.86	0.93	51	55	45.2	52.3	–	–	October 1981–September
Hellen Blazes Lake	23.3	23.1	6.96	7.01	195	191	0.72	0.86	–	–	36.1	33.8	10.2	4.4	December 1979–September
Istopoga Lake	24.4	25.0	6.47	7.43	95	69	0.52	1.12	15	25	8.7[c]	12.0[c]	4.8	–	1966–2000
Kissimmee Lake	24.7	23.9	7.03	7.18	123	69	0.67	0.66	29	42	12.7	12.2	–	12.8	May 1970–July
Myakka Lake, Upper	23.6	24.2	6.77	6.68	168	149	1.04	–	29	23	38.1	43.5	–	7.8	October 1962–November
Panasoffkee Lake	23.4	24.1	7.37	8.05	71	22	–	1.28	115	—	51.5[c]	38.5[c]	– –	–	May1966–September
Poinsett Lake	23.7	24.0	7.03	7.31	164	132	1.15	0.97	63	51	55.3	52.8	–	6.9	October 1953–December
Rodman Reservoir	21.7	23.1	7.56	7.59	46	65	1.88	1.89	137	109	62.3	54.8	–	4.8	May 1970–December
Sawgrass Lake[a]	23.1	23.3	7.08	7.01	165	180	0.86	0.98	–	–	33.2	29.5	4.6	3.2	December 1979–September
Seminole Lake	22.5	20.8	7.13	7.51	47	48	1.06	0.87	34	33	9.5[C]	12.4[C]	–	18.5	1972–1999
Tarpon Lake	–	24.5	–	7.19	–	70	–	1.18	–	42	–	44.2	–	2.6	October 1977–September
Rivers															
Wekiva River	22.1	22.7	7.13	7.43	20	46	1.08	1.07	108	95	–	36.0	–	–	Apr 1965–November
Vvtthlacoochee River	23.3	23.3	7.08	7.20	114	115	–	–	101	121	40.3[C]	39.5[C]	–	–	May 1956–September

Lake/river	Turbidity (NTU)		NO_x–N (mg/l)		NH_4–N (mg/l)		TKN (mg/l)		TN (mg/l)		*Ortho*-P (mg/l)		Total P (mg/l)		Period of record
	In	Out	In	Out	In	Out	In	Out	In	out	in	Out	In	Out	
Lakes															
Harney Lake	2.2	2.2	0.130	0.080	0.095	0.175	1.34	1.23	1.47	1.38	0.096	–	0.079	0.080	October 1981–September
Hellen Blazes Lake	3.3	2.6	0.053	0.027	0.082	0.084	1.91	1.95	1.96	1.98	–	–	0.149	0.134	December 1979–September
Istopoga Lake	5.1	4.1	0.230	0.010	0.067	0.040	1.04	0.95	1.27	0.92	0.029	0.021	0.054	0.046	1966–2000
Kissimmee Lake	25.3	21.0	0.046	0.043	0.148	0.108	1.50	1.42	1.55	1.46	0.112	0.020	0.104	0.052	May 1970–July
Myakka Lake, Upper	2.1	2.1	0.037	0.034	0.063	0.075	2.82	1.30	1.19	1.36	0.232	0.309	0.371	0.386	October 1962–November
Panasoffkee Lake	2.8	2.4	0.572	0.012	0.021	0.035	0.69	0.75	1.28	0.77	0.065	0.015	0.080	0.033	May1966–September
Poinsett Lake	20.8	16.3	0.365	0.088	0.197	0.112	1.55	1.47	2.01	1.56	0.071	0.041	0.115	0.075	October 1953–December
Rodman Reservoir	8.4	14.6	0.530	0.055	0.056	0.058	0.54	0.62	1.04	0.60	0.038	0.020	0.057	0.047	May 1970–December
Sawgrass Lake[a]	1.9	1.3	0.034	0.023	0.068	0.071	1.66	1.72	1.70	1.74	–	–	0.128	0.103	December 1979–September
Seminole Lake	–	10.2	0.509	0.284	0.085	0.042	0.46	0.41	0.90	0.66	0.024	0.019	0.054	0.039	1972–1999
Tarpon Lake	–	2.36	–	0.023	–	0.048	–	0.89	1.16[b]	0.92	–	0.027	0.121[b]	0.042[b]	October 1977–September
Rivers															
Wekiva River	1.4	2.3	0.829	0.619	0.030	0.063	0.49	0.66	1.39	1.21	–	0.119	0.193	0.162	April 1965–November
WjthJacoochee River	–	–	0.057	0.034	0.036	0.040	0.85	0.90	0.91	0.93	0.056	0.038	0.091	0.061	May1956–September

[a] Includes little lake Sawgrass.

[b] Calculated from SWFWMD Lake Tarpon Surface Water Improvement and Management (SWIM) Plan, August 2000 Draft.

[c] Dissolved.

inlet loading of both TP and water (Kadlec, 2000). This also appears to be the case with the analyzed data sets.

Table 4 also summarizes estimated two-parameter model values for these Florida lakes and rivers. The mean value estimated for k_{PFR} for lakes was 29 and 117 m/year for the rivers. The mean value of k_{TIS} for lakes was 49 and 133 m/year for rivers. In all cases, k_{TIS} equals or exceeds k_{PFR}. This result reflects the mathematical relationship of these parameters described in Eqs. (4) and (5). This relationship is based on the fact that plug flow hydraulics are more efficient than more completely mixed hydraulics and a lower k value in the plug flow model describes the same mass removal as a higher k value in the TIS model.

It is important to note that since this is a two-parameter model, values for k_{PFR} and k_{TIS} cannot be compared between wetlands except among those with similar C^* estimates. A high C^* results in a higher value for the settling rate constant (k) for a given amount of TP removal. Some of the individual lake and river data sets were not robust enough to allow simultaneous calibration of k and C^*, so in some cases where Solver could not find a solution it was assumed that C^* was approximately equal to the lowest monthly average for a given data set. The mean C^* estimate for lakes was 36 μg/l with a range from 6 to 90 μg/l. The mean C^* estimated for the rivers was 56 μg/l with a range from 54 to 58 μg/l.

Statistics for the estimated monthly k_1 values are presented graphically in Fig. 2. Average and plus/minus one standard error ranges are shown for each of the 12 systems where these data are available. Lake Hellen Blazes, Lake Seminole, Lake Sawgrass, and Lake Kissimmee had the highest average k_{1TP} of the study lakes. The Withlacoochee River exhibited the highest average k_{1TP} of the two rivers studied. Typically, the water bodies with the highest average values of k_{1TP} also have the largest standard errors.

Fig. 3 illustrates the observed relationship between average annual TP mass loading and TP outflow concentrations. A linear regression equation was fitted to these data that allows the estimation of average TP outflow concentrations based on TP mass loading:

$$C_2 = 0.051\text{LR}^{0.271} \quad R_2 = 0.261 \tag{7}$$

where: LR, average TP mass loading rate (g/m^2 per year); C_2, average outlet concentration (mg/l); q, average HLR (m/year).

This equation explains only 26% of the variability of the data, so its use for estimating average TP outflow concentration should be made with caution. Fig. 4 provides the same type of summary of TP loading and k_{1TP}. The linear regression equation for these data is:

$$k_{1TP} = 0.5.14\text{LR}^{0.662} \quad R^2 = 0.49 \tag{8}$$

Although the fit of this line is better than that of Eq. (7), with the equation describing 49% of the variability in the data, it must be noted that k_{1TP} is not totally independent of the loading rate, as indicated in Eq. (1).

Additional correlations for select parameters are summarized in Table 5. The correlation between percent SAV based on full FDEP surveys and a variety of possible dependent and independent variables was tested. An inverse relationship ($R^2 = 0.41$) was observed between ammonium nitrogen inflow concentration and average SAV cover for these systems. Average annual alkalinity concentrations, both in ($R^2 = 0.29$) and out ($R^2 = 0.37$), were positively correlated with percent of SAV cover, as was inflow Ca concentration ($R^2 = 0.30$). SAV cover was weakly correlated with TP mass removed ($R^2 = 0.12$) but not with k_1.

4. Discussion

4.1. Comparisons of phosphorus removal rates in SAV-dominated Florida natural aquatic systems and constructed treatment wetlands

The results summarized in this analysis of selected SAV-dominated Florida systems extend over various periods-of-record (PORs), with a mean POR of 30 years and ranging from 18 years for Lake Harney to about 45 years for Lake Poinsett. These data indicate that these systems typically removed TP over the long-term. Ten of the 13 systems examined had positive average mass removals, with a mean TP percent mass retention

Table 4
Summary of estimated TP removal performance for select SAV dominated lakes and rivers in Florida

Lake/river	Data average period	HLR (cm/day)		Nominal HRT (day)	Average TP (μg/l)		Mass (g/m^2 per year) Calc k_1 (m/year)		Calc k_1 (m/year)	k_{20PFR} (m/year)	k_{TIS} (m/year)	C^* (mg/l)	Count	Period of Record
		Average	Median		In	Out	Loading	Removed						
Lakes														
Harney Lake	M	21.6	14.4	7.9	72	74	5.68	−0.15	−4.92	95.06	145.91	0.077	122	January 1982–December 1998
Hellen Blazes Lake	M	89.3	39.1	2.5	148	134	48.1	7.06	67.21	24.71	25.35	0.083	84	December 1979–September 1999
Istopoga Lake	A	0.73	0.68	218	51	50	0.14	−0.01	0.02	0.17	0.17	***0.018***	12	November 1972–September 1999
Kissimmee Lake	A	7.3	7.1	29	112	61	2.96	1.69	23.44	25.41	35.57	***0.012***	14	January 1973–July 1998
Myskka Lake, Upper	M	16.4	6.2	7.3	393	411	23.5	−1.57	−7.50	−0.01	−0.01	***0.090***	52	December 1972-September 1998
Panasoffkee Lake[a]	Q	1.8	2.0	72.6	64	17	0.42	0.20	8.66	25.71	125.27	0.016	5	May 1992–April 1993
Poinselt Lake	M	11.6	7.3	9.4	114	82	4.86	0.55	2.33	11.17	12.21	***0.020***	138	October 1973-May 1998
Rodman Reservoir	Q	8.2	7.7	29.3	57	48	1.72	0.17	9.43	7.27	7.89	0.006	64	September 1970–December 1998
Sawgrass Lake[b]	M	73.8	36.1	2.0	132	113	35.4	3.92	30.88	23.42	24.11	0.040	64	December 1979-September 1999
Semirole Lake	A	28.9	29.2	19.7	54	39	5.66	1.57	32.09	105.72	150.40	***0.025***	24	January 1972 - September 1998
Tarpon Lake[c]	3 years	1.2	–	187	121	42	0.52	0.34	4.54	5.34	8.32	***0.010***	1	January 1996–December 1998
Average	–	23.7	15.2	53.2	120	97	11.7	1.24	15.11	29.45	48.65	0.036	52.7	
Rivers														
Wekiva River	M	39.6	36.1	2.8	193	175	27.9	3.64	28.65	12.68	12.82	***0.058***	73	January 1980–February1997
Withlacoochee River	M	56.6	30.8	2.5	83	65	17.1	3.78	63.24	221.04	253.41	0.054	35	June 1994–June 1999
Average	–	48.1	33.4	2.6	138	120	24.2	3.71	45.94	116.86	133.12	0.056	54.0	

[a] Calculated from CH2M HILL (1995) Lake Panasoffkee Water and Nutrient Budget Study final report.

[b] Includes little lake Sawgrass.

[c] Calculated from SWFWMD Lake Tarpon SWIM Plan. August 2000 Draft results are based on estimated averages for the noted analysis period with paired inflow/outflow P concentrations number of TIS fixed at 1.5 for lakes and 4.0 for rivers in for k_{20TIS} model bold italicized values were fixed in the model calibration.

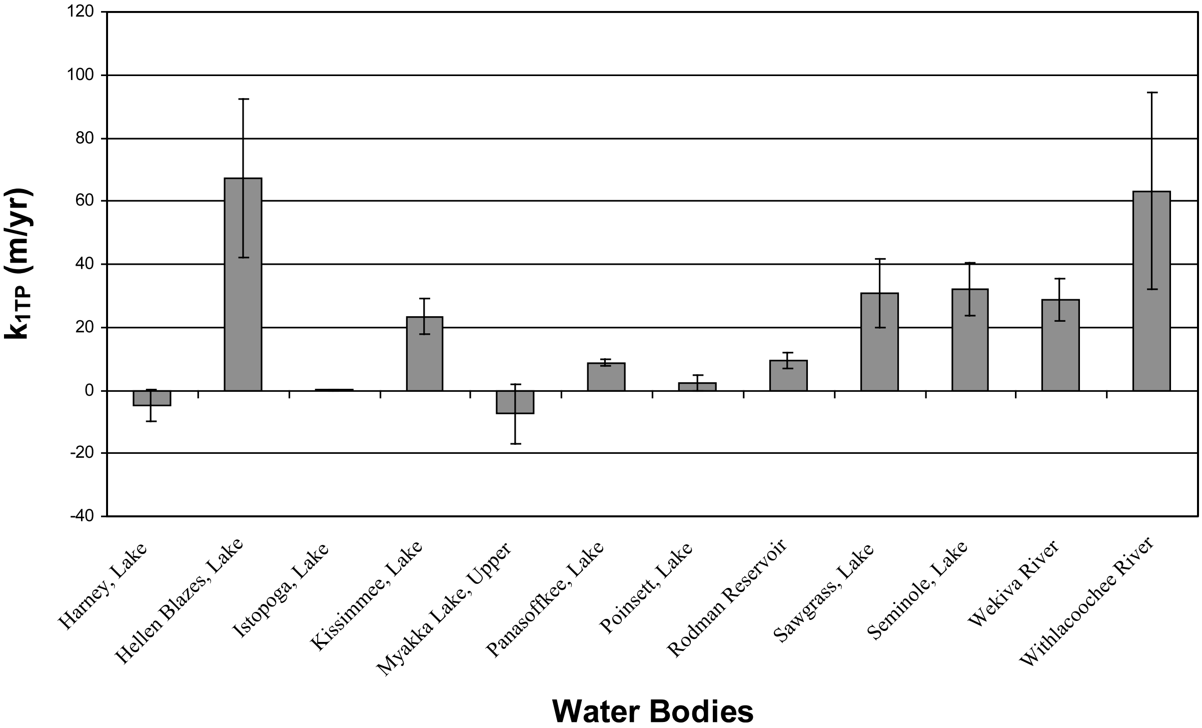

Fig. 2. Average k_{1TP} and standard errors for select SAV dominated lakes and rivers in Florida.

efficiency of about 12% over all the systems. This retention efficiency is low compared with the 56% mean TP mass retention efficiency reported for the SAV-dominated Cell 4 of STA-1W after 5 years of operation (1994–1999) (Nungesser et al., 2001). However, the mean TP mass removal for the Florida aquatic systems (average TP mass removal = 1.2 g/m^2 per year for lakes and 3.7 g/m^2 per year for rivers) is similar to STA-1W Cell 4 after 7 years of operation (1.3 g/m^2 per year) and in smaller test systems with comparable HRTs.

Although the mean TP mass removed is similar between the natural aquatic systems and the SAV treatment wetlands, the average TP mass loading rate (lakes, 11.7 g/m^2 per year; rivers, 24.2 g/m^2 per year) in the Florida aquatic systems was five to ten-fold of that (2.34 g/m^2 per year) for the STA-1W Cell 4 (Nungesser et al., 2001). Mean HLRs for the study lake and river systems (lakes, 24 cm/day; rivers, 48 cm/day) were also higher than in the SAV research systems (15 cm/day). Inflow TP concentrations of the study lakes were also higher, with a mean of 120 μg/l for the study lake systems, 138 μg/l for the rivers, and 40 μg/l for STA1W Cell 4 (from 1994 to 1999).

Nearly half of the long-term TP mass removal rates estimated in lakes and rivers in our study (−1.57 to 7.06 g/m^2 per year) are higher than the 1 g/m^2 per year maximum hypothesized by Richardson et al. (1997) for emergent wetlands. The work of Binford and Brenner (1986) and Brenner (1997) provides similar evidence that there is no limit to P removal in the range of 1 g/m^2 per year in SAV-dominated lakes and rivers. Our results illustrated in Fig. 3 are very similar to those published by Kadlec (1999). Outflow TP concentration from emergent wetlands and SAV-dominated lakes and rivers are apparently subject to the same types of constraints as wetlands, including hydraulics and hydrology, source strength, detection limits, and internal removal processes.

Our average k_1 value for SAV-dominated lakes and the rivers was 15 and 46 m/year, respectively. The k_1 value for the STA-1W Cell 4 was estimated

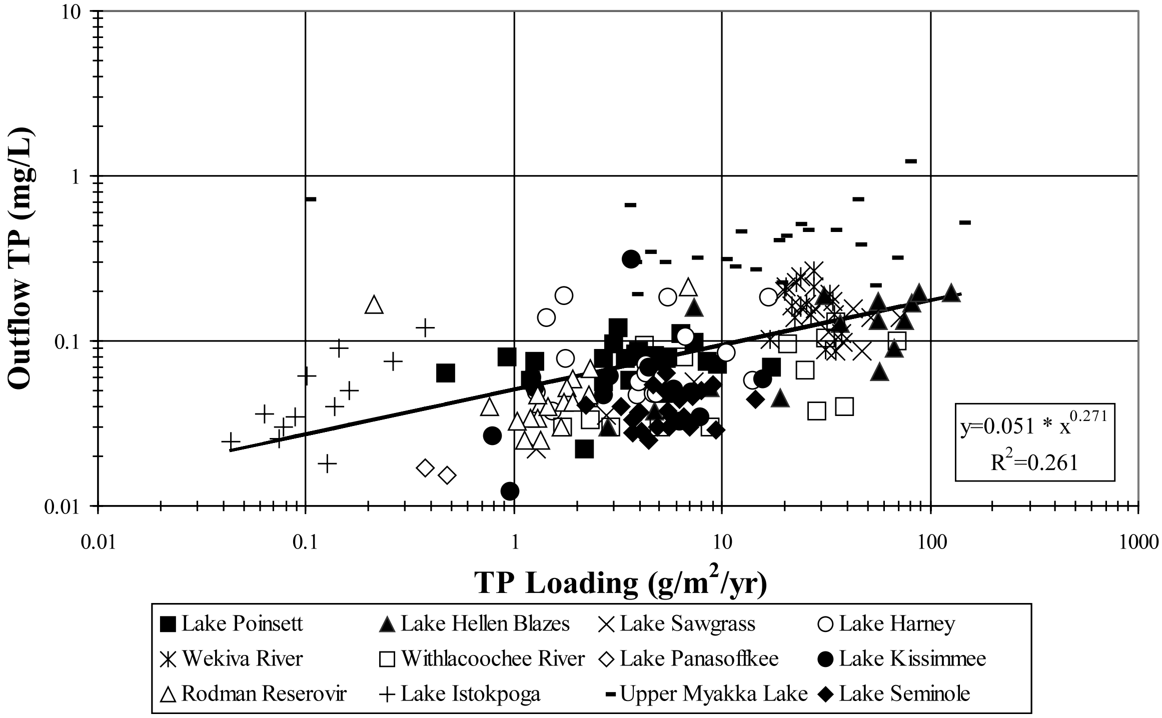

Fig. 3. Relationship between annual average TP loading and annual average TP outflow concentrations for select SAV dominated lakes and rivers in Florida.

as 40 m/year. Higher values for k_1 have been estimated for the SAV mesocosms (Kadlec, 2000; DB Environmental Laboratories, 2000).

We estimated an average C^* for TP in SAV-dominated lakes of 36 μg/l with individual values as low as 6 μg/l, while the average TP concentration exiting the study lakes was 97 μg/l. The lowest annual average TP concentration exiting STA-1W Cell 4 was 13 μg/l and the 5-year average outflow concentration was 20 μg/l (Gu et al., 2001). In our study, k_{PFR} was calculated using the C^* value and averaged 29 m/year for the SAV study lakes, which is almost double the mean k_1 of 15 m/year calculated without the C^* value.

4.2. Long-term P accretion in SAV-dominated lake sediments

A certain amount of evidence that can be used to test the mass balances presented above does exist from Florida lakes. Sediment cores from lakes throughout Florida have been collected and analyzed for ^{210}Pb and TP in a continuing effort to document recent and historical TP loads and removals (Binford and Brenner, 1986; Brenner, 1997). These authors reported average recent (upper 4 cm) TP burial rates of 0.45 and 0.30 g/m^2 per year for Lakes Hellen Blazes and Sawgrass, respectively (Brenner, 1997). The net TP removal rates estimated from input/output mass balances for these same lakes were much higher: 7.1 and 3.9 g/m^2 per year, respectively. The reason for the discrepancy between these two estimating procedures is not clear but may be due to the use of estimated inflow in some of our mass balances and spatial variability in sediment P burial rates.

Binford and Brenner (1986) reported TP accumulation rates in recent sediments for 34 other Florida lakes based on the same method of ^{210}Pb dating. TP accumulation rates ranged from 0.26 to 5.9 g/m^2 per year. Their data for 11 SAV-dominated lakes are reproduced in Table 6 along with the POR average SAV cover estimates provided by FDEP (2001). TP accumulation rates

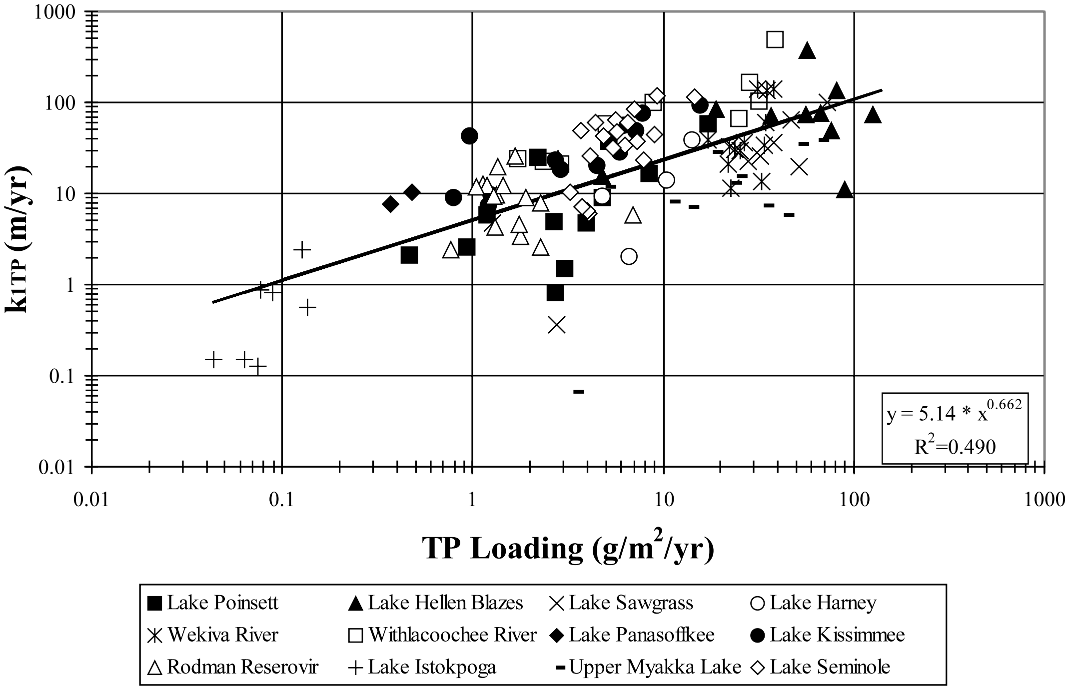

Fig. 4. Relationship between annual average TP loading and annual average $k_{1\mathrm{TP}}$ for select SAV dominated lakes and rivers in Florida.

estimated for these lakes ranged from 0.26 to 3.2 g/m^2 per year. There was a weak correlation found between SAV cover and sediment TP accumulation in these lakes ($R^2 = 0.17$). One of these lakes also overlaps with our study. They estimated a recent TP mass accumulation rate for Lake Kissimmee of 2.1 g/m^2 per year while our mass balance estimated 1.6 g/m^2 per year. With each of these comparisons it should be recognized that both methods are not precise and that both temporal and spatial variability exist in estimated TP loads and sediment TP concentrations.

4.3. *Environmental effects on phosphorus removal from natural SAV-dominated systems*

We found that TP removal in SAV-dominated aquatic systems is positively correlated with the TP inflow mass loading rate and HLR. This relationship has also been observed in the SAV-dominated STAs and research mesocosms (Kadlec, 2000). Fig. 5 compares the relationship between HLR and k_1 for the data assembled in this study and for the quarterly data for STA-1W Cell 4 and the south Florida SAV mesocosms (Kadlec (2000). The trend line for the lake and river data is clearly below STA-1W Cell 4 and the SAV mesocosms. The lower values of k_1 estimated in this study compared with values from the STA-1W Cell 4 and the SAV mesocosms may indicate that the non-optimized, SAV-dominated lakes are not as efficient for TP removal as engineered treatment wetlands.. However, there are a number of possible reasons why the long-term TP removal performance from the natural aquatic systems may be less than optimal. These include the following:

- The potential for greater hydraulic short circuiting in lakes and rivers with variable depths, irregular shorelines, incomplete plant cover, and inlet and outlet points with close proximity as compared with carefully engineered mesocosms and constructed treatment wetlands.
- The effects of plant management practices on SAV cover in the natural lakes and rivers and resulting lower TP removal rates compared with

Table 5
Correlation analyses for SAV and associated parameters in select Florida lakes

Independent variable (x)	Dependent variable (y)	R^2	Slope (m)	Intercept (b)	N
NH_4–N in (mg/l)	SAV cover (%)	0.410	−144	50.4	12
SAV cover (%)	Alkalinity out (mg/l)	0.367	1.67	17.8	9
Ca in (mg/l)	SAV cover (%)	0.296	0.485	20.2	10
Alkalinity in (mg/l)	SAV cover (%}	0.290	0.222	20.1	8
SAV cover (%)	TP mass rem. (g/m^2 per year)	0.122	0.066	−0.329	12
SAV cover (%)	NH_4–N out (mg/l)	0.115	−0.001	0.095	11
Depth (m)	TP out (mg/l)	0.105	−0.026	0.157	12
Alkalinity in (mg/l)	TP out (mg/l)	0.078	−0.001	0.171	8
SAV cover (%)	TN out (mg/l)	0.059	0.005	1.05	12
SAV cover (%)	k_{1TP} (m/year)	0.042	0.439	7.20	12
NO_3–N in (mg/l)	SAV cover (%)	0.033	−11.3	39.1	8
TP in (mg/l)	SAV cover (%)	0.031	27.4	34.2	12
Depth (m)	k_{1TP} (m/year)	0.027	4.48	15.0	11
SAV cover (%)	TP out (mg/l)	0.022	0.001	0.078	13
Alkalinity in (mg/l)	k_{1TP} (m/year)	0.014	−0.060	15.3	8
Color in (pcu)	SAV cover (%)	0.012	−0.030	43.1	10
SAV cover (%)	Color out (pcu)	0.010	0.311	98.8	12
TN in (mg/l)	SAV cover (%)	0.008	4.82	31.1	12
SAV cover (%)	NO_3–N out (mg/l)	0.005	0.002	0.194	9
Ca in (mg/l)	k_{1TP} (m/year)	0.001	−0.072	25.0	10
Ca in (mg/l)	TP out (mg/l)	0.000	0.000	0.100	10
SAV cover (%)	Ca out (mg/l)	0.000	0.004	31.0	12

Linear model: $y = mx + b$.

those in engineered and highly managed constructed wetlands with higher SAV cover and density.

Table 6
Comparison of average SAV dominance and recent sediment accumulation rates for select Florida lakes

Lake	SAV average cover (%)	P accumulation rate (g/m^2 per year)
Butler	56	1.06
Fairview	32	3.25
Jassamine	36	3.02
Kissimmee	26	2.08
Minnehaha	25	0.32
Okahumpka	73	0.24
Pierce	47	1.18
Townsend pond	39	0.26
Yale	26	0.83
Helen Blazes	55	0.45
Sawgrass	50	0.30
Average		1.18

Sources: SAV cover. FDEP (2001); P accumulation. Binford and Brenner (1986) and Brenner (1997).

- The highly variable hydraulic loading experienced by these lakes compared with more controlled SAV test systems.
- The potential for nitrogen limitation in some of the study lakes (as indicated by N:P ratios in six of the 13 study systems less than 14:1) or effects of other potential limiting factors (e.g. iron).
- Higher calcium content in south Florida stormwaters (typically 50–70 mg/l in agricultural drainage waters) compared with the subject lakes and rivers (8.7–62 mg/l) and the assumed relationship between calcium and co-precipitation of P.
- The greater mean water depth of the lake and river systems described in this paper (1.5–5 m) than the SAV-dominated treatment wetlands (0.2–0.8 m).
- The predominance of *Hydrilla* in the study lakes as opposed to *Najas*, *Chara*, and *Ceratophyllum* in the engineered SAV treatment wetlands.
- All of the study lakes were located north of Lake Okeechobee and are subjected to a slightly

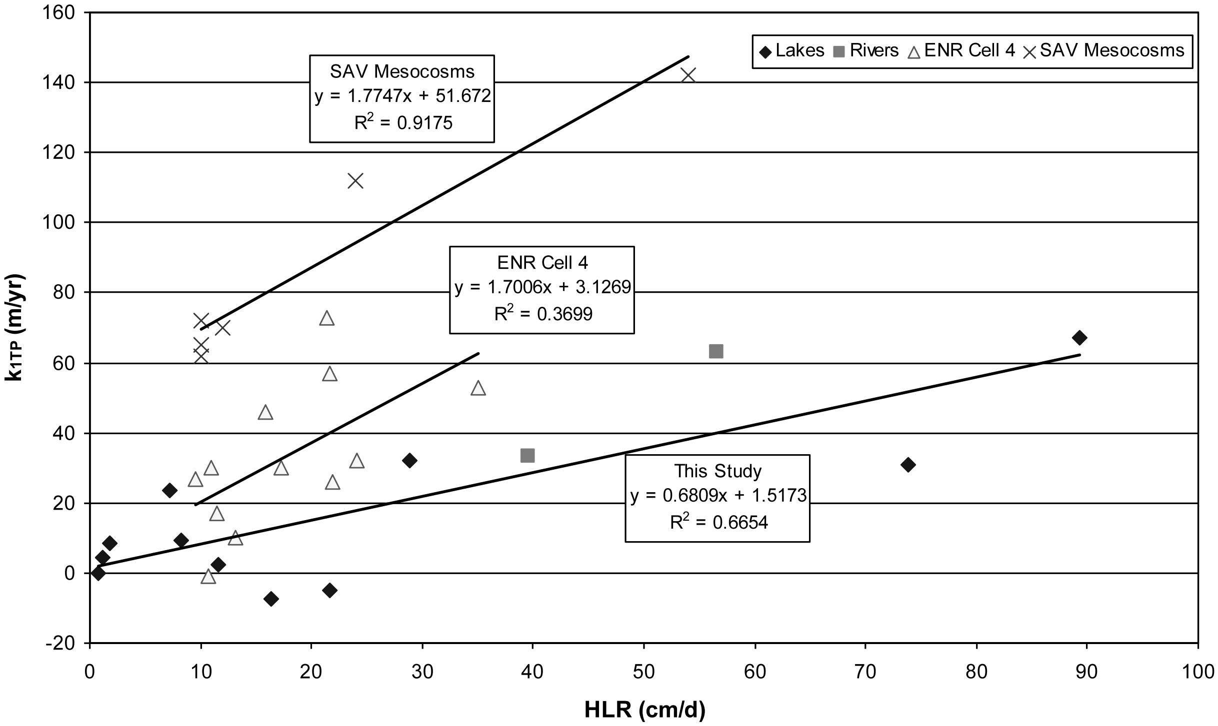

Fig. 5. Relationship between HLR and k_{1TP} for SAV dominated lakes and rivers in this study, STA-1W Cell 4, and SAV mesocosms.

different climate and have different sediments than the controlled SAV test systems.

It is also possible that relatively short-term k_1 estimates from constructed SAV treatment wetlands such as STA-1W Cell 4 and small-scale mesocosms may reflect smaller scale and short-term or start-up conditions. Small mesocosms typically have higher ratios of surface area to volume than full-scale systems and biogeochemical processes that are affected by surface area are often enhanced in these smaller systems. The ‘clothesline’ effect on wetland mesocosm temperature and affected processes is a case-in-point (Kadlec and Knight, 1996). It is also well recognized that startup phenomena can take many years to disappear, resulting in either higher or lower performance, depending upon the process considered, and that ‘...phosphorus removal rates (*k* values) should be determined from post-startup operation’ (Kadlec, 2002). Additional study of the TP removal performance of SAV-dominated lakes will be needed to better quantify the importance of these potential treatment variables. In the interim, it is prudent to recognize that mesocosm based TP removal estimates from SAV-dominated treatment wetlands may over predict long-term performance.

Other findings from this research may shed additional light on design of SAV-dominated STAs. In the group of lakes and rivers examined in detail in this study, SAV cover was not found to be highly correlated with either TP or TN inflow or outflow concentrations. Other researchers have also observed poor correlations between these parameters as well as SAV cover and chlorophyll *a*, a known correlate of TP and TN (Huber et al., 1982; Canfield and Hoyer, 1992). This observation may indicate that some SAV plant communities are most affected by sediment nutrient concentrations as determined by Barko and Smart (1980) rather than water column nutrient concentrations. The lack of a significant correlation between SAV cover and outflow TP concentration may be due to the relatively stronger dependence of outflow TP on the inflow TP concentration in relatively low loaded SAV systems.

SAV estimates summarized in this paper were also poorly correlated with other water quality parameters, except for alkalinity, calcium, and ammonium nitrogen. The positive correlation between alkalinity and SAV cover may indicate a possible hard-water habitat preference for some of the Florida species of SAV. Calcium has been linked in TP removal through co-precipitation, especially in light of elevated pH conditions in highly productive SAV lakes and wetlands (Gu et al., 2001; DB Environmental Laboratories, 2000; Murphy et al., 1983). Regressions between inflow calcium and alkalinity concentrations and TP outflow concentration and $k_{1\mathrm{TP}}$ were not found to be significant in this study. Once again, if there is an effect of calcium on TP burial, it may be masked by a stronger relationship between inlet TP load and these indices of TP removal performance. There was also no effect observed between calcium and alkalinity and C^*_{TP} in this study.

4.4. Implications for full-scale wetland designs

The most important result of this study is the finding that SAV-dominated lakes and rivers may provide a wealth of information to designers of 'green' technologies and that after longer operational periods than STA-1W, they are averaging a mean TP mass removal of 1.2 g/m^2 per year. As it is probable that these ecosystems operate by the same biogeochemical principles as constructed treatment systems and that they have the added benefit of their large scale and their long history of TP removal, we feel that long-term k and k_1 values are meaningful and justify comparison to treatment systems with a shorter operational history. Therefore, we suggest that extrapolations of TP settling rates (k and k_1) from short-term operational systems to full-scale designs be made with care. Additionally, in light of limited economic resources for solving eutrophication problems in Florida and elsewhere, it is suggested that further resources be consigned to tighten the TP mass balances for some of these lake systems. This work should also be coordinated with confirmatory sediment analyses as described above.

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