Phosphorus and Nitrogen Inputs to Florida Bay: The Importance of the Everglades Watershed

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ABSTRACT: A large environmental restoration project designed to improve the hydrological conditions of the Florida Everglades and increase freshwater flow to Florida Bay is underway. Here we explore how changing freshwater inflow to the southern Everglades is likely to change the input of nutrients to Florida Bay. We calculated annual inputs of water, total phosphorus (TP), total nitrogen (TN), and dissolved inorganic nitrogen (DIN) to Everglades National Park (ENP) since the early 1980s. We also examined changes in these nutrient concentrations along transects through the wetland to Florida Bay and the Gulf of Mexico. We found that the interannual variability of the water discharge into ENP greatly exceeded the interannual variability of flow-weighted mean nutrient concentrations in this water. Nutrient inputs to ENP were largely determined by discharge volume. These inputs were high in TN and low in TP; for two ENP watersheds TN averaged 1.5 mg l^{-1} (0.11 mM) and 0.9 mg l^{-1} (0.06 mM) and TP averaged 15 µg l^{-1} (0.47 µM) and 9 µg l^{-1} (0.28 µM). Both TP and DIN that flowed into ENP wetlands were rapidly removed from the water. Over a 3-km section of Taylor Slough, TP decreased from a flow-weighted mean of 11.6 µg l^{-1} (0.37 µM) to 6.1 µg l^{-1} (0.20 µM) and DIN decreased from 240 µg l^{-1} (17 µM) to 36 µg l^{-1} (2.6 µM). In contrast, TN, which was generally 95% organic N, changed little as it passed through the wetland. This resulted in molar TN : TP ratios exceeding 400 in the wetland. Decreases in TN concentrations only occurred in areas with relatively high P availability, such as the wetlands to the north of ENP and in the mangrove streams of western ENP. Increasing freshwater flow to Florida Bay in an effort to restore the Everglades and Florida Bay ecosystems is thus not likely to increase P inputs from the freshwater Everglades but is likely to increase TN inputs. Based on a nutrient budget of Florida Bay, both N and P inputs from the Gulf of Mexico greatly exceed inputs from the Everglades, as well as inputs from the atmosphere and the Florida Keys. We estimate that the freshwater Everglades contribute < 3% of all P inputs and < 12% of all N inputs to the bay. Evaluating the effect of ecosystem restoration efforts on Florida Bay requires greater understanding of the interactions of the bay with the Gulf of Mexico and adjacent mangrove ecosystems.

Introduction

The connection between Florida Bay and the terrestrial ecosystems of south Florida is diffuse

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and poorly understood. While most estuaries are connected to upland systems with a dendritic network of channels through which water, particles, and dissolved materials flow, there is little channelized flow between Florida Bay and the Everglades. Rather, water flows in shallow sloughs
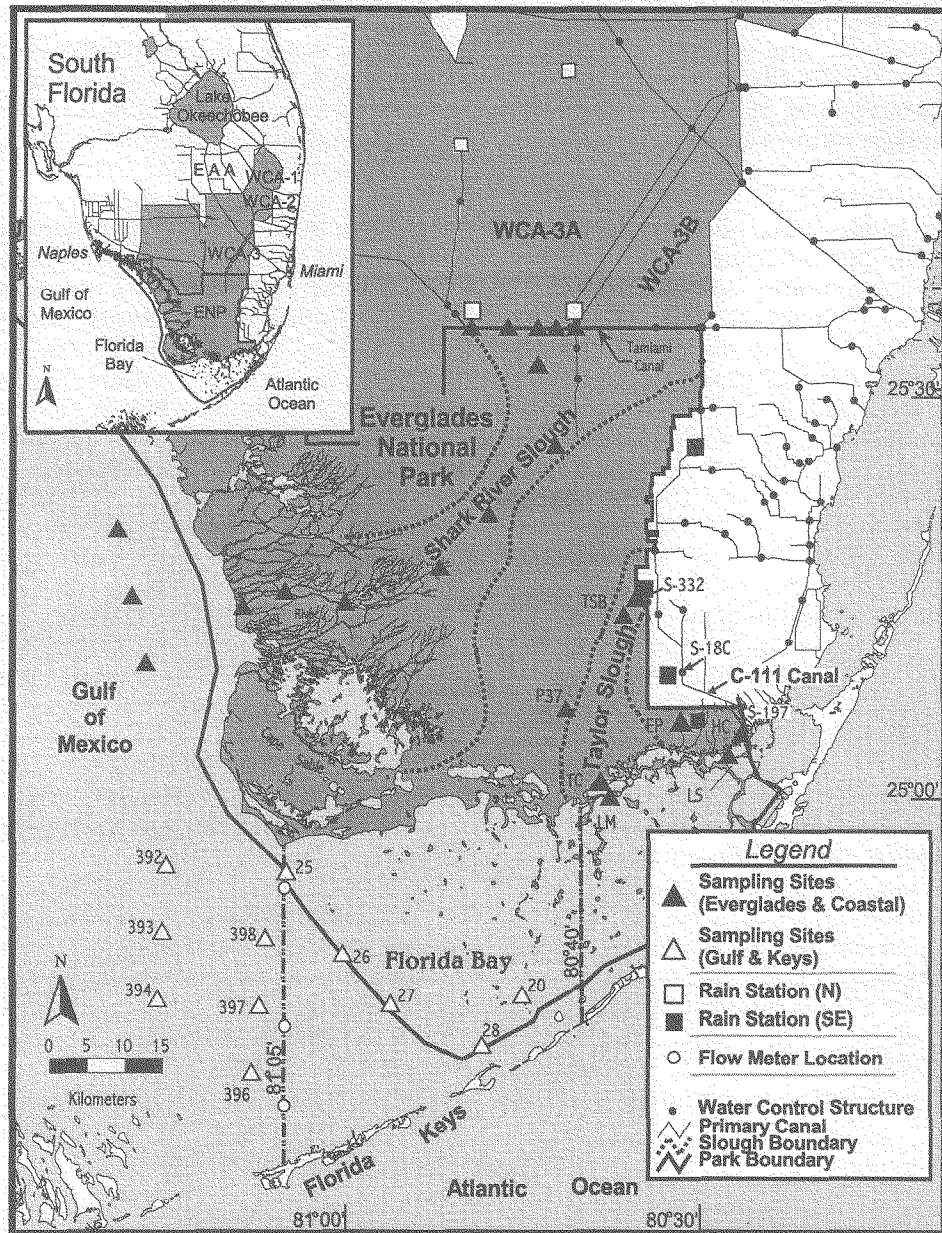
Fig. 1. Florida Bay and its watersheds. Abbreviations are as follows: Everglades Agricultural Area (EAA), Water Conservation Area (WCA), Everglades National Park (ENP), Taylor Slough Bridge (TSB), Taylor Creek (TC), Little Madeira Bay (LM), Highway Creek (HC), and Long Sound (LS). Open circles and triangles (with station identification numbers) are stations used to calculate the nutrient budget in the Discussion section.

through this large (6,200 km²), flat wetland (Fig. 1). This flow becomes channelized through broad bands of creeks only as it nears the coastline. Given this diffuse water source, as well as the complex hydrodynamics of water within Florida Bay (Wang et al. 1994), understanding of the distribution, quantity, and ecological influence of material exchanges between the bay and the Everglades watershed is quite limited (McIvor et al. 1994; Sklar and Browder 1998).

During the past century, pronounced ecological changes have occurred in the Everglades (Davis and Ogden 1994). These changes include the construction of a network of about 2,400 km of canals and levees for flood control and water supply, the fragmentation and "over-drainage" of wetlands by these canals and levees, the loss of about half of the original 12,000-km² wetland area to agriculture and residential development (Davis et al. 1994), the eutrophication of wetlands near agricultural ar-

eas (Craft and Richardson 1993; McCormick et al. 1996; Reddy et al. 1998), and invasion of these eutrophied areas by cattails (*Typha domingensis*; Urban et al. 1993; Doren et al. 1997; Wu et al. 1997).

Ecological changes within Florida Bay, including mass mortality of *Thalassia testudinum* and algal blooms, have been evident during the last two decades (Robblee et al. 1991; Butler et al. 1995; Phlips and Badylak 1996; Fourqurean and Robblee 1999), and have commonly been attributed to many of the same human activities that have changed the Everglades. The diversion of freshwater to the Atlantic coast by canals has increased the salinity of the bay (McIvor et al. 1994). In the 1980s, freshwater discharges from canals to the Atlantic Ocean were roughly four times larger than discharges to sloughs that flowed toward Florida Bay (Light and Dineen 1994). Anthropogenic nutrient inputs from the Florida Keys and the Gulf of Mexico may have also increased (Lapointe and Clark 1992). The latter inputs may include phosphorus which is transported by long-shore currents from the central Florida coast and nitrogen from the Everglades that flows into the Gulf from the Shark River Slough (Lapointe et al. 1994).

A large set of environmental engineering projects intended to restore many of the hydrological characteristics of Everglades National Park's wetlands and to decrease the salinity regime of Florida Bay is now underway. An indirect consequence of increasing freshwater flow to Florida Bay may be an increase in nutrient inputs. In this paper, we attempt to determine: 1) the quantity of nutrients transported from the Everglades to Florida Bay; 2) the relationship of these inputs to the discharge of water into the Everglades; and 3) the significance of these inputs, relative to other sources of nutrients.

Methods

STUDY AREA

This study focuses on the flow of water and nutrients into and through Everglades National Park (ENP), which is located at the southern tip of the Florida peninsula (Fig. 1). The immediate source of this water is undeveloped wetlands (Water Conservation Areas) to the north of ENP, which are used for water storage, and a network of canals that cross southeastern Florida (Light and Dineen 1994). Water flows into ENP is in three regions. First, the largest discharge is through a set of five gated spillways along the Tamiami canal, which is on the northern boundary of ENP. This surface water then flows toward the Gulf of Mexico via the Shark River Slough, a broad shallow waterway that is blanketed by a mosaic of sawgrass (*Cladium ja-*

maicense), spike rush (*Eleocharis* sp.) with thick benthic algal mats, and tree islands (Davis et al. 1994). The slough splits into a wide band of mangrove creeks and rivers within about 15 km of the Gulf.

A second site of water input to ENP is along its eastern boundary, mostly through the S-332 pump station. This water flows directly toward Florida Bay via Taylor Slough. Plant community composition in this slough is similar to that of Shark River Slough, but water is shallower and hydroperiods are shorter in this smaller slough. Carbonate soils are more common in Taylor Slough than in Shark River Slough. Taylor Slough waters flow into mangrove creeks within about 4 km of Florida Bay. A third site of water input to ENP is the southern reach of the C-111 Canal. The overflow of this canal is not routed through a natural slough, but sheet-flows in a southerly direction through a sawgrass marsh and then into mangrove creeks within about 4 km of northeastern Florida Bay.

DATA SOURCES AND ANALYSES

Most of the data that we used in this study came from monitoring programs of the South Florida Water Management District (SFWMD), ENP, the United States Geological Survey (USGS), and Florida International University (FIU). Data were made available through the DBHYDRO and Water Quality databases of the South Florida Water Management District.

Daily rainfall accumulations were collected in gauges at eight stations (Fig. 1), with half of the stations near the inflows to the northern ENP boundary and half of the stations near inflows to the eastern ENP boundary. We calculated annual rainfall for both sets of stations from the daily mean rainfall at each set's stations. If no data were available for any station of a set, we did not estimate these missing values.

Daily water discharge rates in DBHYDRO were calculated from daily measurements of water depth and application of a calibrated flow-rating curve for each water management structure. We calculated monthly discharge as the sum of daily values per structure and annual discharge for a region as the sum of monthly discharge of each regions' structures. For water inputs to Shark River Slough, we calculated monthly discharge from each of the five water management structures along the Tamiami Canal (S-12s and S-333). For water inputs to Taylor Slough, we calculated monthly discharge only from daily discharge at S-332. There is additional discharge (averaging 21% of Taylor Slough plus C-111 canal values) from a nearby structure (S-175), but this water is not included in our study because of the absence of associated nutrient measurements until mid 1995. We also calculated the

monthly volume of water that flowed within Taylor Slough from daily discharge values at the Taylor Slough bridge. For water inputs to the wetlands south of the C-111 Canal, we calculated monthly discharge as the sum of daily discharge through S-18C minus the sum of discharge through S-197, which when open discharges water toward Biscayne Bay. For water inputs to the Water Conservation Areas, we calculated monthly discharge at structures S-5A, S6, S7, S150, and S8 (not shown in Fig. 1), which are on the border between these areas and the Everglades Agricultural Area.

Nutrient samples were collected and analyzed on a biweekly or monthly basis at each water management structure where discharge was estimated and are reported here (since 1979 for Shark River Slough inputs and 1984 for southeastern Everglades inputs). Water samples from stations along transects through the sloughs to adjacent coastal waters (Fig. 1) were collected and analyzed on a monthly basis (since 1993 for Shark River Slough and Harney River and 1991 for southeastern Everglades), except for samples from the Gulf of Mexico, which were collected four times per year (since 1995). Freshwater samples were analyzed by the SFWMD. Estuarine samples were collected and analyzed by the Southeast Environmental Research Program of FIU. (Analyses of all estuarine samples and freshwater collected since 1987 were documented for quality assurance and control procedures. Prior to 1987, such documentation is scarce; only interlaboratory round-robin results are available [M. Loucraft-Manzano personal communication].) For freshwater samples, nitrate, nitrite, ammonium, and soluble reactive phosphorus were measured using Technicon autoanalyzers from 1979 to 1987 and Alpkem rapid flow analyzers since 1987 (Clesceri et al. 1989). Organic N was digested using the micro-kjeldahl method and organic P was digested by wet oxidation with persulfate (Clesceri et al. 1989). For estuarine samples, nitrate, nitrite, ammonium, and soluble reactive phosphorus were measured using an Alpkem autoanalyzer. Corrections for changing refractive indices were made for water with different salinity levels. Total N was analyzed using an ANTEK nitrogen analyzer (Frankovich and Jones 1998) and total P samples were digested as in Solorzano and Sharp (1980).

We calculated monthly nutrient loading from each water management structure as the product of the arithmetic monthly mean nutrient concentration and the monthly cumulative water discharge per water management structure. For these calculations, missing nutrient concentration values for a given structure were estimated by a linear interpolation between the most recent known values. Nutrient concentrations below detection were assumed to be half of the detection limit. Annual nutrient loading for a given region was the sum of monthly loads for all water management structures in a region (given above). Flow-weighted mean concentrations were calculated as the annual load divided by annual discharge.

We also collected samples on a daily basis, using an autosampler, at the mouth of Taylor Creek (Fig. 1), starting in April 1996. We analyzed these estuarine samples for total N and P as described above. Daily flow estimates in the creek were made by the United States Geological Survey, using acoustic velocity meters, for three depth layers (E. Patino personal communication). From these data, we calculated nutrient loading and flow-weighted mean concentrations at this site.

Nutrient exchange between the Gulf of Mexico and Florida Bay was estimated using nutrient concentration data and water advection rate data from three stations along the 80°05′ meridian (Fig. 1). Advection rates across the meridian at these stations were calculated by T. Pratt and N. Smith (personal communication) based on data from three SonTek upward-looking acoustic doppler current profilers (ADCP) deployed by the Waterways Experiment Station of the United States Army Corps of Engineers. ADCP data used in this paper were collected for 1 yr, starting April 1, 1996. The ADCP units measured current velocity at 15-min intervals for four to nine water layers (each layer 29 cm deep). Samples for nutrient analyses were collected at stations 25, 397, and 396 (Fig. 1) near the ADCP stations, with a monthly sampling frequency at the northern station and seasonal sampling frequency at the other two stations (five samples from March 1996 through May 1997).

Relationships between annual water discharge, annual loads, mean nutrient concentrations, and time were explored using simple regression and Kendall rank correlation analyses. We tested for the equality of means among stations or among regions by using a randomized block design ANOVA (with time as a block) after log-transforming all data. If a significant difference among means existed, the similarity of individual stations was tested using the Tukey-Kramer post-hoc comparison test. All analyses were performed using StatView and SuperANOVA programs (Abacus Concepts, Inc., Berkeley, California).

Results

NUTRIENT INPUTS TO WETLANDS OF EVERGLADES NATIONAL PARK

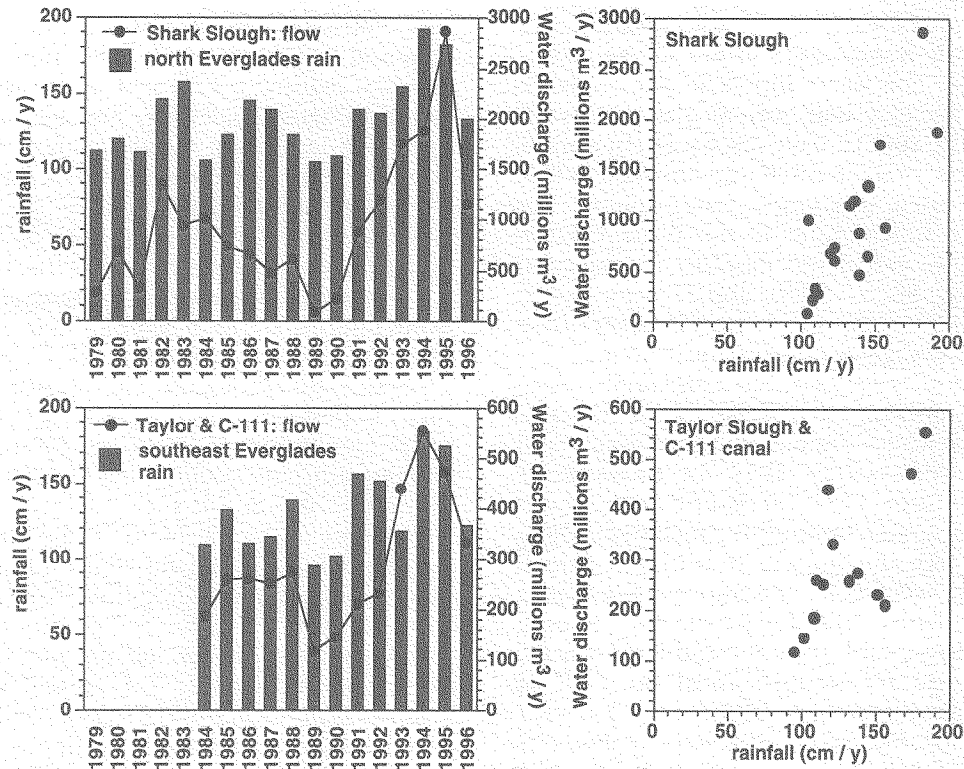The input of water from canals to the watersheds that flow toward Florida Bay was strongly influ-

Fig. 2. Annual regional rainfall and the annual discharge of water into Everglades National Park through its northern boundary into Shark River Slough and through its eastern boundary into Taylor Slough and the wetlands south of the C-111 Canal.

enced by rainfall quantity (Fig. 2). Based on linear regression, annual rainfall accounted for 68% (p < 0.01) of the variance of annual water discharge into Shark River Slough. Rainfall accounted for 48% (p < 0.01) of the variance in discharge into Taylor Slough (including the wetlands south of the C-111 Canal, hereafter Taylor Slough + C-111). The variability of annual discharge was greater than the variability of annual rainfall. For Shark River Slough, the coefficient of variation (CV) was 73% for annual discharge but only 21% for annual rainfall. For Taylor Slough + C-111, the CV was 45% for discharge and 19% for rainfall. For both regions, there was little water discharge when rainfall was about 100 cm yr^{-1}, but discharge increased as much as tenfold when there was a less than twofold increase in rainfall. With the exception of the drought of 1989 and 1990, discharge into Shark River Slough greatly exceeded discharge into Taylor Slough + C-111.

Nutrient inputs to the headwaters of Shark River Slough and Taylor Slough + C-111 were strongly influenced by water discharge. Phosphorus inputs to these wetlands were mostly in the form of organic P plus particulate inorganic P; soluble reactive P concentrations for the entire period of record were rarely above the detection limit of 4 μg l^{-1}. For Shark River Slough, total phosphorus (TP)

inputs increased with increasing discharge (Fig. 3). The increase was approximately linear for discharges up to about 10^9 m^3 yr^{-1}, but at greater discharge, TP inputs increased to a lesser extent. The nonlinearity of this relationship between load and flow occurred because TP concentrations decreased significantly with higher discharge in Shark River Slough (Fig. 4 and Table 1).

For Taylor Slough + C-111, TP concentrations were independent of discharge (Fig. 4). This resulted in a positive and generally linear relationship between TP inputs and discharge (Fig. 3). An unusually high TP input is estimated to have occurred in 1994, but 38% of this annual input occurred in September 1994. This monthly estimate is based on a single water sample taken during heavy rains. Excluding September, the 1994 discharge was 465 × 10^6 m^3 and TP input was 5.2 × 10^6 g.

Nitrogen inputs to Everglades National Park (ENP) wetlands followed the same general pattern as P inputs, increasing with increasing water discharge (Fig. 5). Total N concentrations in water flowing into Shark River Slough decreased significantly as a function of discharge (Fig. 4 and Table 1). Interannual changes in TN inputs to Shark River Slough were more dependent on changes in discharge than changes in TN concentration because
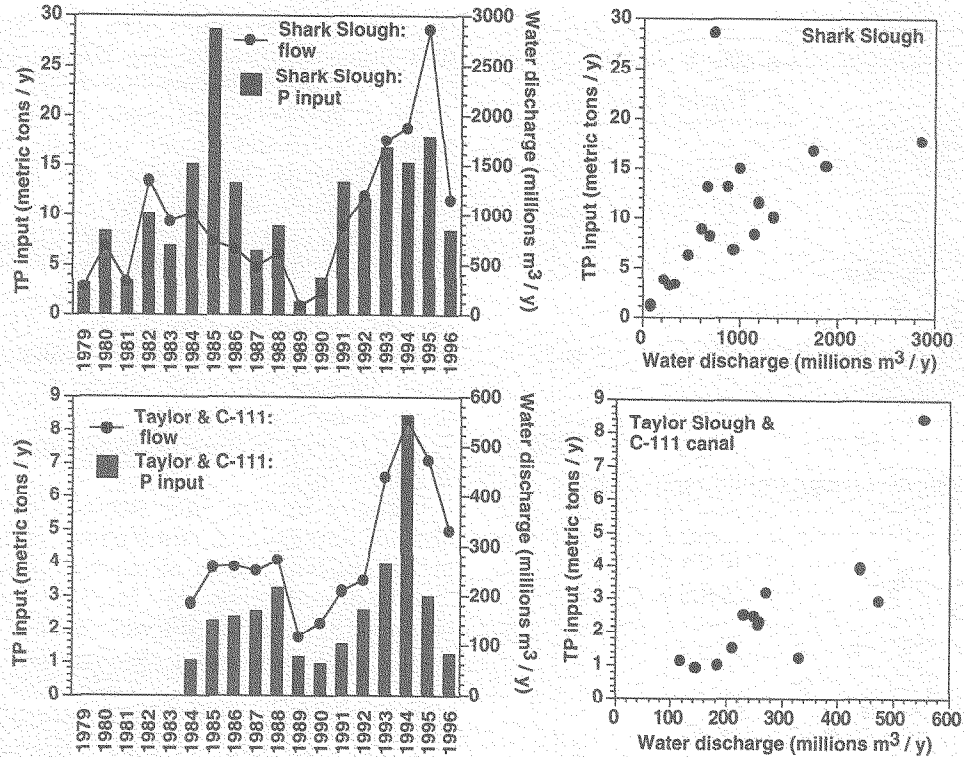
Fig. 3. Annual water discharge and annual total phosphorus (TP) input to Shark River Slough and Taylor Slough plus the wetlands south of the C-111 Canal.

of the wider range of discharge variability (thirty-fold for discharge and twofold for TN concentration). As with TP inputs, TN concentrations in Taylor Slough + C-111/water were independent of discharge (Fig. 4) and TN inputs to Taylor Slough + C-111 increased in proportion to water discharge (Fig. 5).

The temporal pattern of flow-weighted mean nutrient concentrations (Fig. 6) shows that the mean parcel of water discharged into Shark River Slough steadily and significantly (Table 1) decreased in P and N concentrations since the mid-1980s and generally decreased in N concentration since 1979. Since 1987 (conservatively avoiding the peak near 1985) TP concentrations decreased (regression slope of -1.0 μg l^{-1} yr^{-1}). However, a longer-term decrease in TP did not occur; TP concentrations since 1992 were similar to TP concentrations of the early 1980s. Total N concentrations in water flowing into Shark River Slough since 1987 decreased (regression slope of -37 μg l^{-1} yr^{-1}). During this same time period, annual Shark River Slough discharge also significantly increased (Table 1). For water discharged into Taylor Slough + C-111, there was a significant (p < 0.05) decrease in TN concentration since 1984 (regression slope of -38 μg l^{-1} yr^{-1}). No significant change occurred in TN

concentration since 1987 or in TP concentration since 1984 or 1987.

Another notable difference between the watersheds of the Everglades is evident from these flow-weighted mean concentration data. Concentrations of N and P inputs to Shark River Slough were consistently higher than inputs to Taylor Slough + C-111 (Table 2 and Fig. 6). Both TP and TN flow-weighted mean concentrations were about two times higher in Shark River Slough than Taylor Slough + C-111. However, despite its lower total N content, the flow-weighted mean concentration of dissolved inorganic N (DIN, the sum of nitrate, nitrite, and ammonium) in water flowing into Taylor Slough + C-111 was about two times higher than in water flowing into Shark River Slough. Flow-weighted mean DIN concentrations did not change significantly (and no r^2 values were > 0.1) in Shark River Slough or Taylor Slough + C-111 as a function of year or discharge (data not shown).

NUTRIENT CHANGES WITHIN WETLANDS

Changes in the nutrient content of waters flowing through the Everglades are evident from comparisons of concentration gradients along flow pathways. Nutrient changes within the wetlands of
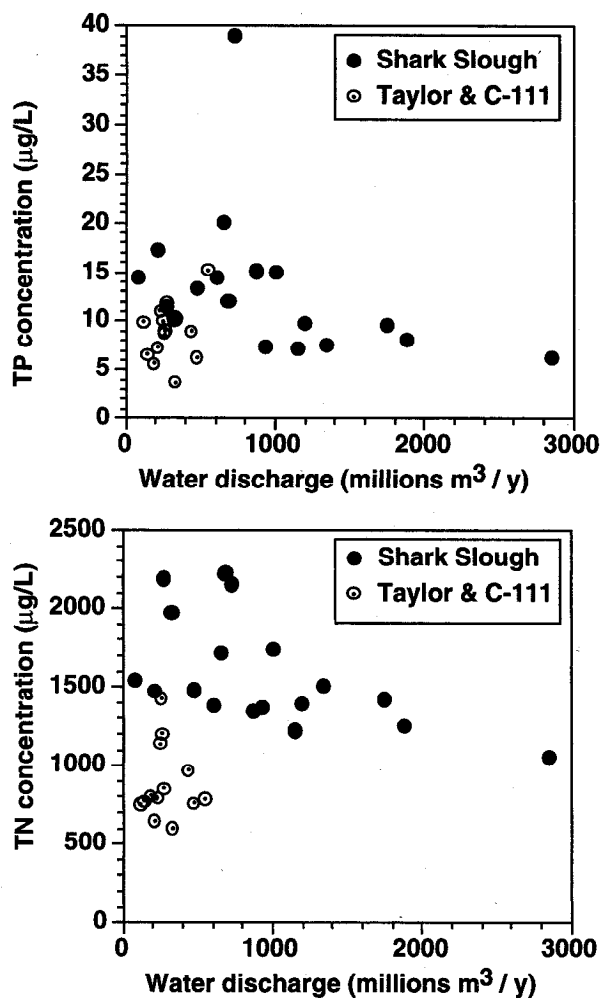
Fig. 4. Annual flow-weighted mean concentrations of total phosphorus (TP) and total nitrogen (TN) as a function of annual water flow volume into Shark River Slough and Taylor Slough plus the wetlands south of the C-111 Canal.

the Water Conservation Areas (WCAs) and the canals that circumscribe the WCAs, which are the sources of water flowing into ENP, are evident in Table 2. Flow-weighted mean N and P concentrations in water discharged into the WCAs and associated canals (Fig. 1) from the Everglades Agricultural Area were much higher than N and P concentrations in water that was discharged into ENP. More than 90% of P and 66% of N (based on concentration change, Table 2) was removed within the WCAs and associated canals. The greater removal of P yielded a higher molar N:P ratio in water flowing into ENP (mean = 260 for Shark River Slough and mean = 242 for Taylor Slough + C-111) than in water flowing into the WCAs (N:P ratio = 62).

Changes in nutrients within ENP wetlands are also evident from a comparison of flow-weighted mean concentrations along the flow pathway of Taylor Slough (Fig. 1). Most of the water discharged from S332 did not reach Taylor Slough Bridge and most water flowing past this line did not reach the mouth of Taylor Creek (Table 3). With decreasing flow volume, nutrient loads also decreased along this gradient. The mean nutrient content of this water also decreased over the 3-km pathway between S332 and Taylor Slough Bridge. Total P concentrations decreased from a mean of 11.6 μg l^{-1} to 6.1 μg l^{-1}. While TN concentrations remained nearly unchanged, DIN concentrations decreased nearly 90%. While DIN constituted 26% of the TN flowing into Taylor Slough, almost all of the N flowing past Taylor Slough Bridge was in an organic form. In 1996 the TP and TN content of water about 25 km further down-slough, in Taylor Creek, was higher than in water at either site at the north end of Taylor Slough. A net flux of TP (0.34 $\times$ 10^6 g yr^{-1}) and TN (26.8 $\times$ 10^6 g yr^{-1}) from

TABLE 1. Linear regression best fit equations and Kendall rank correlation Tau values for annual water discharge (D, in million m^3 y^{-1}) into Shark River Slough and annual flow-weighted mean N and P concentrations (in μg l^{-1}) in this water as a function of year (Y) and annual water discharge. Slopes that are significantly different from zero are given with $p < 0.05$ (*) and $p < 0.01$ (**) levels.

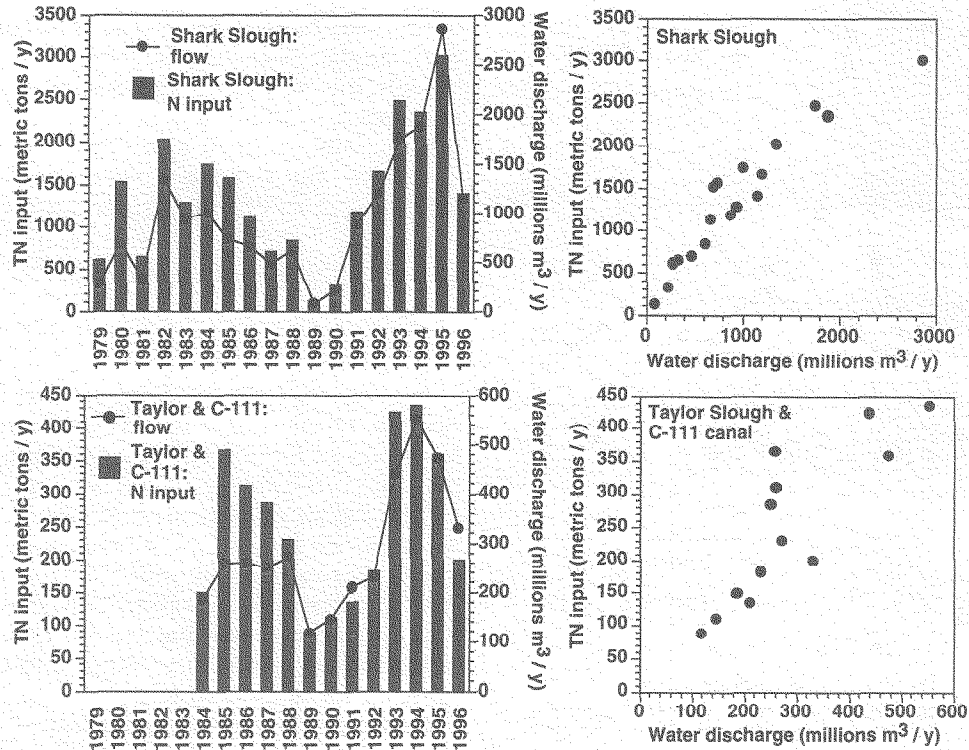|	Year		Discharge	
1987–1996				
Discharge	D = −80.5 + 216.5Y	(r^2 = 0.584**)		
	Tau = 0.644**			
N concentration	N = 1561 − 37.10Y	(r^2 = 0.598**)	N = 1517 − 0.144D	(r^2 = 0.723**)
	Tau = −0.600*		Tau = −0.600*	
P concentration	P = 17.26 − 1.04Y	(r^2 = 0.688**)	P = 15.71 − 0.00376D	(r^2 = 0.702**)
	Tau = −0.556*		Tau = −0.644**	
1979–1996				
Discharge	D = −268 + 71.6Y	(r^2 = 0.308*)		
	Tau = 0.320			
N concentration	N = 2077 − 52.34Y	(r^2 = 0.651**)	N = 1851 − 0.286D	(r^2 = 0.324*)
	Tau = −0.673**		Tau = −0.386*	
P concentration	P = 15.83 − 0.275Y	(r^2 = 0.038)	P = 17.203 − 0.00420D	(r^2 = 0.149)
	Tau = −0.203		Tau = −0.412*	

Fig. 5. Annual water discharge and annual total nitrogen (TN) input to Shark River Slough and Taylor Slough plus the wetlands south of the C-111 Canal.

Taylor Creek into Florida Bay was measured in 1996.

Nutrient changes in a more extensive area of the ENP wetlands can also be assessed using results from sites lacking flow measurements. Figure 7 presents these results for three watersheds, the Shark River Slough, Taylor Slough, and the southeastern Everglades from the C-111 Canal to Long Sound. In Shark River Slough, nutrients dramatically changed along the flow pathway from the Tamiami Canal to the Gulf of Mexico. Wet season (May 1–October 31) mean TP concentrations decreased in the first 5 km of Shark River Slough but increased more than twofold at the most upstream mangrove river site and fourfold at the river mouth. Total P concentrations in the Gulf of Mexico (mean of three stations about 10 km offshore from the river mouth) were higher than in the freshwater wetland but much lower than in the mangrove zone, with only about 12.5 µg l⁻¹.

A comparison of mean annual TP concentrations along the salinity gradient from lower Shark River Slough to the Gulf of Mexico (Fig. 8) shows that TP concentrations at the three mangrove zone sites were much higher than would be expected based on conservative mixing of slough and Gulf waters. The convex distribution of values along this

gradient indicates a net internal P source within the mangrove zone.

Unlike phosphorus, TN concentrations increased slightly in the wetland after leaving the Tamiami Canal but decreased at the mangrove creek sites (Fig. 7). As inorganic N remained almost unchanged through the entire slough transect, the decrease in TN was caused by a decrease in organic N in the mangrove zone. Total N changes in this zone nearly followed the conservative mixing line of Shark River Slough and Gulf of Mexico end-members (Fig. 8). The distribution of values from two stations fell slightly below this mixing line, indicating some net removal of TN.

The resultant TN:TP molar ratio changed dramatically through the Shark River Slough system. This ratio initially increased from about 310 in water flowing into the slough to 730 at a site 5 km down-slough. The ratio then decreased down the entire transect to a minimum of 40 at the mouth of the Harney River.

The trends of nutrient change in the Taylor Slough and C-111 Canal watersheds were similar but more subtle than in Shark River Slough (Fig. 7). In both of these eastern watersheds, TP concentrations were lower in the freshwater wetlands than in the source canal water. However, concen-
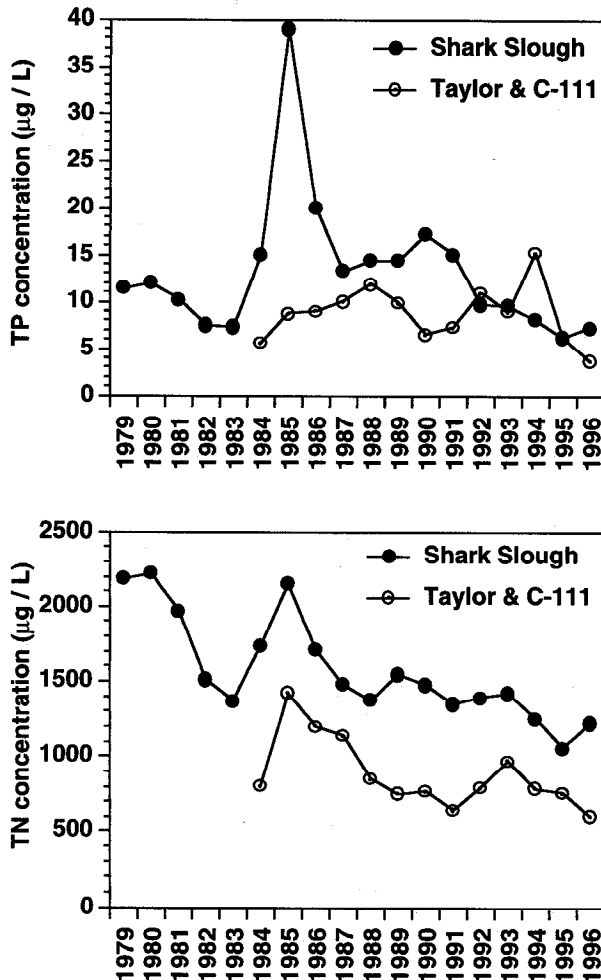
Fig. 6. Annual flow-weighted mean concentrations of total phosphorus (TP) and total nitrogen (TN) in water flowing into Everglades National Park.

trations were higher in the saline mangrove creek (wet season mean salinity = 6‰) of the C-111 Canal watershed and in coastal Florida Bay waters (wet season mean salinity = 12‰ in Long Sound and 23‰ outside of Little Madeira Bay) than in

the upstream wetland. Mean TN concentrations were similar in each watershed. As in Shark River Slough, the N:P ratios were highest in the wetland and lowest in saline waters. Unlike in Shark River Slough, DIN concentrations in source canal waters were high and decreased dramatically in concentration in the wetland. This trend reversed at the mangrove creek and bay sites, where DIN increased to magnitudes exceeding 100 μg l⁻¹.

EXCHANGE BETWEEN FLORIDA BAY AND THE GULF OF MEXICO

A summary from measurements of water advection and nutrient concentration fields at the boundary of Florida Bay and the Gulf of Mexico is presented in Table 4. Water advection across the 81°05′ meridian was eastward (from the Gulf toward Florida Bay) at the northern and middle stations and westward (toward the Gulf from Florida Bay) at the southern station. Advection rates were highest in the north and lowest in the south and yielded a net transport of water into the bay from the Gulf on an annual basis.

Nutrient concentrations for both P and N were higher north of the 81°05′ meridian than south of the meridian (Table 4). Concentration gradients in an east-west direction were less steep than these north-south gradients. Total P concentrations were nearly uniform from east to west, but both TN and DIN were higher in western Florida Bay than in waters about 14 km west of the 81°05′ meridian.

Discussion

The results presented here address an important environmental management issue—whether restoration of the Everglades' hydrology could entail an increase in nutrient loading of the wetland and Florida Bay. The results show that, without a decrease in the upstream nutrient supply, increasing freshwater flow into ENP will increase nutrient loads. It is also clear from these results that the nutrients that enter the wetland are not conserva-

TABLE 2. Annual means and interannual variations of discharge and associated nutrients loads and concentrations in water entering the Water Conservation Areas (WCA), Shark River Slough, and Taylor Slough plus C-111 basin wetlands. WCA values are from 1984–1995 and other values are from 1984–1996. Across these regions, mean concentrations and N/P ratios that share the same superscript letter are not significantly different (during 1984–1995), with p > 0.05. Values not calculated are given as "nc."

	WCA		Shark Slough		Taylor/C-111	
	Mean	SE	Mean	SE	Mean	SE
Annual discharge (millions m³)	1300	153	1040	210	286	36
Annual TP load (metric tons)	196	21	12.3	2.0	2.63	0.55
Annual TN load (metric tons)	5390	576	1430	235	252	33
Flow weighted mean TP (μg l⁻¹)	155[a]	11	14.6[b]	2.3	8.8[c]	0.8
Flow weighted mean TN (μg l⁻¹)	4250[a]	225	1480[b]	77	884[c]	66
Flow weighted mean DIN (μg l⁻¹)	nc	nc	86.9[a]	13.1	209[b]	17
Molar TN/TP	62.4[a]	3.6	260[b]	22	242[b]	23

TABLE 3. Annual means and interannual variations of discharge and nutrients loads and concentrations through Talyor Slough watershed. Data from the drought year of 1990 were excluded because of the absence of water at the Taylor Slough bridge site. Significant differences between the mean concentrations and N/P ratios from the S332 pump station and Taylor bridge sites are given for $p < 0.05$ (*) and $p < 0.01$ (**) levels. The N values from a Taylor Bridge sample of June 1986 were replaced by interpolated values because of suspected nitrate + nitrite contamination. Values not calculated are given as "nc."

	S332			Taylor Slough Bridge			Taylor Creek Mouth
	1986–1996 Mean	SE	1996 Mean	1986–1996 Mean	SE	1996 Mean	1996 Mean
Annual discharge (millions m³)	116	27	160	48.2	11.1	64.0	22.8
Annual TP load (metric tons)	1.06	0.23	0.56	0.249	0.042	0.35	0.34
Annual TN load (metric tons)	106	25.6	96.6	38.4	9.4	55.4	26.8
Flow weighted mean TP (μg l⁻¹)	11.6	2.5	3.5	*6.11	1.02	5.50	14.9
Flow weighted mean TN (μg l⁻¹)	942	57	605	795	59	865	1180
Flow weighted mean DIN (μg l⁻¹)	242	18	nc	**36.3	8.6	nc	nc
Molar TN/TP	234	30	384	*339	46	348	174

tive; long-term P and N outputs to the coastal zone are likely to be less than P and N inputs to the wetland. The question of whether this output is a significant portion of Florida Bay's nutrient budget is addressed in the third and fourth parts of this discussion.

NUTRIENTS INPUTS TO EVERGLADES NATIONAL PARK

Results from this study of the freshwater flow and nutrient inputs to ENP establish that nutrient inputs increase as a function of increasing flow (Figs. 3 and 5). This result is the consequence of relatively small interannual changes in the concentration of nutrients of upstream waters compared with large interannual changes in water discharge; nutrient concentrations were not diluted in direct proportion to increasing discharge (Fig. 4). Nutrient concentrations in these waters are dictated by several dynamics, including inflow of nutrients from Lake Okeechobee, runoff of nutrients from agricultural and residential areas, the processing of these and ambient nutrients in the wetlands of the WCAs, nutrient processing within canals, the relative contribution of water from WCA wetlands versus canals, the exchange of groundwater and surface water, rainfall and evapotranspiration, and atmospheric nutrient deposition. For inputs to the Shark River Slough, the balance of these upstream sources and sinks yielded negative slopes for both N and P concentrations as a function of water discharge (Table 2), with -144 μg l⁻¹ N per 10^9 m³ yr⁻¹ discharge and -3.8 μg l⁻¹ P per 10^9 m³ yr⁻¹ discharge since 1987. The negative slope for P since 1979 was significant if the anomalous years of 1984–1986 were excluded (Fig. 6).

Examination of those anomalous years reveals the sensitivity of nutrient loading to water management. During this time, two major changes occurred in water management. First, water delivery from June 1983 through May 1985 was unusual, with an experimental flow-through plan that al-

lowed uncontrolled inflow to Shark River Slough from the north (Light and Dineen 1994). Prior to this time, water was delivered to ENP following a static minimum delivery schedule, and since this time, water has been delivered following a statistical relationship with rainfall. Second, by late 1985, back-pumping of agricultural runoff into Lake Okeechobee had almost stopped (Walker 1999). Instead, this runoff was directed toward the WCAs. These two management changes, combined with the dry conditions of 1984 and 1985, appear to have greatly increased nutrient loading to ENP in the mid 1980s.

Our results show a significant temporal trend of decreasing flow-weighted mean TP and TN concentration after 1986 in water flowing into Shark River Slough (Fig. 6 and Table 1). This finding is consistent with Walker's (1991) analysis of TN inputs to Shark River Slough from 1977 to 1989. However, while Walker (1991) documented a trend of increasing P concentrations in water flowing into Shark River Slough during this time (also apparent in Fig. 6), we have documented a reversal of this trend after 1986. This trend of decreasing P concentrations was caused at least partially by dilution from increased rainfall and water discharge since 1989 (Figs. 3 and 4). Decreased inputs from agricultural sources associated with improved agricultural practices also have contributed to decreasing P concentrations (South Florida Water Management District 1997).

NUTRIENTS TRANSPORT THROUGH EVERGLADES NATIONAL PARK

The rapid removal and retention of phosphorus in Everglades wetlands is evident in Taylor Slough from the decrease in flow-weighted mean TP concentrations between the entry to the slough (S332) and the Taylor River Bridge (a decrease of 11.6 μg l⁻¹ to 6.1 μg l⁻¹; Table 3). The decrease in the TP flux between S332 and the bridge was even greater
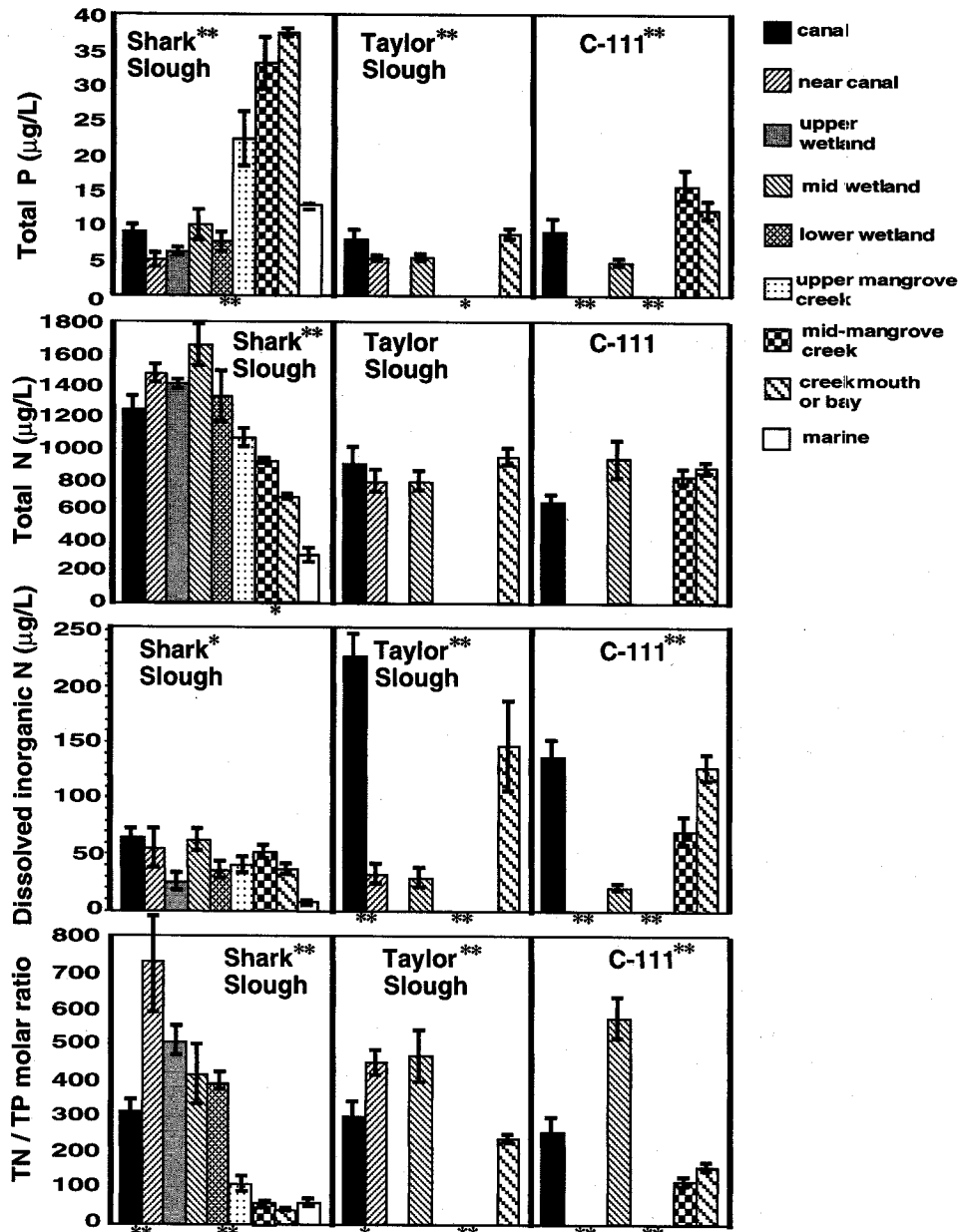
Fig. 7. Means and standard errors of wet season nutrient concentrations and N:P molar ratios in water flowing through three watersheds in ENP. Each panel shows a transect through a watershed, from a canal water source, through freshwater wetlands, and to the coastal zone. Each bar corresponds to a sampling site (filled triangle) in Fig. 1 (Taylor Slough transect site sequence = S332, TSB, P37, LM; C-111 transect site sequence = S18C, EP, HC, LS). All values are arithmetic means of May through October concentrations. Periods of record are 1993–1996 for Shark Slough (except Gulf of Mexico station, 1995–1996), and 1991–1996 for Taylor Slough and C-111 watersheds. Significant differences among the means of all sites along each transect are given adjacent to a watershed's label, p < 0.05 (*) and p < 0.01 (**). Significant differences between means of adjacent sites are given along the base of each panel. Gulf of Mexico data were not included in the statistical comparison because of the shorter period of record.

(1.06 × 10⁶ g yr⁻¹ to 0.25 × 10⁶ g yr⁻¹; Table 3), but this also reflects a loss of water through this 3-km segment of the slough. We infer that the loss of P through this segment is largely attributable to biogeochemical uptake in the marsh rather than input or output of waters with different P concentrations. A net loss of water through the segment

occurred because evaporation and seepage outputs exceeded local rainfall and seepage inputs. Net evaporation would increase P concentrations. Seepage from the slough into groundwater, toward the southeast, is known to be an important route for waters of Taylor Slough (Fennema et al. 1994) and probably exceeded seepage inputs. Seepage
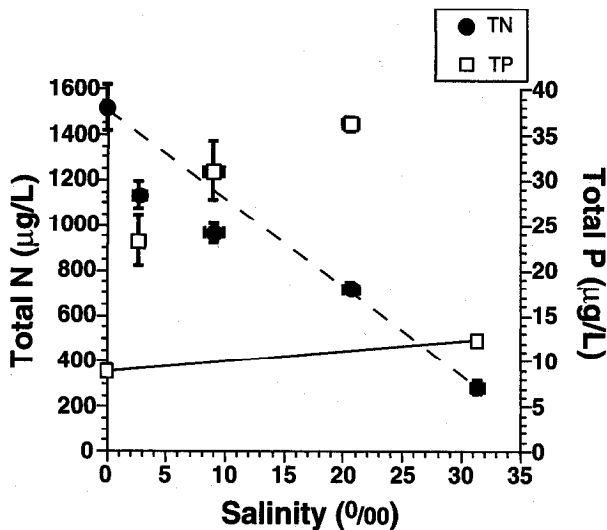
Fig. 8. Means and standard errors of annual arithmetic mean TN and TP concentrations (1993–996) and salinity at four stations along a transect from southern Shark River Slough to the Gulf of Mexico. The conservative mixing line of Gulf and slough waters is given.

outputs would not affect P concentrations in the slough.

The same trend of decreasing TP concentrations, and apparent P removal and retention, is evident from the comparison of wet season TP concentrations in source waters with concentrations in the first downstream wetland stations in Shark Slough, Taylor Slough, and the southeastern Everglades (Fig. 7). The inference that P is retained within the freshwater wetlands of ENP is consistent with previous studies that demonstrated rapid removal and strong retention in the Everglades system (Amador et al. 1992; Jones and Amador 1992; Koch and Reddy 1992, Reddy et al. 1998). An important mechanism involved in this removal and retention is the precipitation of carbonate within

periphyton (algal) mats (Gleason and Spackman 1974), which can result in the co-precipitation of phosphate (Diaz et al. 1994). Partly because of the carbonate-rich nature of the Everglades, primary productivity is P-limited, as demonstrated by P-enrichment experiments and tissue stoichiometry patterns (Steward and Ornes 1983; Koch and Reddy 1992; McCormick and O'Dell 1996; Newman et al. 1996; Koch and Snedaker 1998). This strong autotrophic demand for P, combined with biotic and abiotic P immobilization within both carbonate and peat soils (Jones and Amador 1992; Reddy et al. 1998) results in sharp gradients of P concentration in water, plants, and soils removed from P sources (Koch and Reddy 1992; DeBusk et al. 1994; McCormick et al. 1996; Doren et al. 1997; Reddy et al. 1998). Based on such evidence, we expect that little of the P that enters ENP wetlands from canal waters is transported through the wetlands to the estuaries of the watershed.

Despite this expectation, we found that the export of P from southern Taylor Slough into Florida Bay in 1996 $(0.34 \times 10^6 \text{ g yr}^{-1})$ was similar in magnitude to the P flux past the Taylor Slough Bridge (Table 3). This occurred even though water discharge from Taylor Creek was only about one third of water discharge past Taylor Slough Bridge. The resultant flow-weighted mean P concentration was much higher $(14.9 \ \mu\text{g l}^{-1})$ in Taylor Creek than in water at the bridge or at S332. It is probable that water discharge was lower through Taylor Creek than past Taylor Bridge because the creek is not the only pathway for water flowing from the slough; other creeks exist and significant seepage losses may also occur. If these other water outputs have P concentrations similar to Taylor Creek, then P export from the slough would exceed P loads to the northern slough.

Given the low P concentrations in the slough and our understanding of P biogeochemistry in

TABLE 4. Summary of water advection and nutrient concentrations (total P, total N, and dissolved inorganic N) near the Florida Bay–Gulf of Mexico boundary from April 1996 through March 1997. Values represent annual means, with temporal standard errors in parentheses. Advection values are east-west transport across a 1 m long cross-section (per station) of the 81°05' meridian (monthly cummulative values from 15 min. interval measures; negative value for transport toward the west). Nutrients were sampled monthly at the "north" station and the "east" stations, but nutrients were sampled seasonally (n = 4) at all other stations. The location of all stations is shown in Fig. 1.

	Water Advection (m³ month⁻¹)	TP (µg l⁻¹)	TN (µg l⁻¹)	DIN (µg l⁻¹)
Annual means at 81°05' meridian flow meter stations and nearby water quality stations				
North (station 25)	179,600 (26,400)	17.2 (3.0)	401 (26)	24.9 (9.4)
Middle (station 397)	82,500 (16,900)	10.8 (2.3)	275 (27)	9.2 (2.7)
South (station 396)	−55,000 (14,700)	7.3 (1.3)	252 (30)	8.4 (1.6)
Annual means at stations along east-west gradient				
West (stations 392, 393, 394)		14.3 (3.6)	254 (30)	7.3 (1.9)
81°05' meridian (stations 25, 398, 397, 396)		13.4 (3.5)	270 (24)	14.5 (7.0)
East (stations 26, 27)		13.7 (1.6)	349 (29)	15.4 (5.0)

the slough, it seems unlikely that relatively high P concentrations and P fluxes in Taylor Creek are derived from P transported through the freshwater marsh. Rather, we hypothesize the origin of much of the P that was exported to bay was the bay itself from a time prior to 1996. During the dry season, extended periods occur when bay waters flow northward through Taylor Creek (E. Patino personal communication) and intrude into the mangrove zone. Even with the unusually high rainfall of 1994 and 1995 (Fig. 2), this zone became saline in the dry season; mean daily salinity at a site about 3 km inland from Florida Bay was 10‰ in May 1995 and during the drought of 1989 and 1990, salinity levels reached 59‰ (D. Smith personal communication). As bay water has higher TP concentrations than Taylor Slough water (Fig. 7), the mangrove zone may remove and store this estuarine P. Much of the P transported to the bay in 1996 may have come from the bay during previous, drier years.

A cycle of marine P import, storage, and time-lagged export also appears to occur in the mangrove zone of the Shark River Slough. From 1993 through 1996, a steep P gradient with very high TP concentrations persisted within the Harney River (Fig. 7), despite tidal flushing with Gulf of Mexico water and the net advection of freshwater from Shark River Slough. Based on the mixing diagram for the Harney River (Fig. 8), one of several rivers and creeks flowing from the Shark River Slough, it is likely that the mangrove zone was a net P source to coastal waters from 1993 through 1996. If the source of this P were truly internal to the mangrove zone, then P export from this zone must be temporary. As we hypothesized for Taylor Slough, internal P releases during the exceptionally wet years of 1993–1996 (Fig. 1) may have followed a period of P accumulation from Gulf waters during previous, drier years. Perhaps more importantly, P release during 1993–1996 may be from litter produced by Hurricane Andrew, which devastated this region in August 1992 (Smith et al. 1994). Alternatively, P releases during 1993–1996 may not be pulses derived from drought or catastrophic mangrove mortality, but rather only indicate that we are not adequately measuring external P sources in our Shark River Slough and Gulf of Mexico monitoring programs. Two possible external sources are sediment from the Gulf and groundwater. Sediment transported near the sediment-water interface is unlikely to be collected in our surface water sampling. Such particles could be trapped in the river system and associated P mobilized under reducing conditions in the river's sediment (at the middle river site, both surface and bottom water have low dissolved oxygen, averaging < 3 mg l^{-1}).

The fate of N in the Everglades has received far less attention than the fate of P. Based on our nutrient loading estimates (Table 2), about 70% of the N in agricultural runoff was removed within the WCAs and associated canals in transit to ENP and about half of this removed N entered the WCAs in the form of organic N (South Florida Water Management District 1992). Rapid ammonium removal in a P-enriched area of the WCA wetlands has previously been shown (McCormick et al. 1996). Recent studies in an artificial wetland, designed to remove P from agricultural runoff, have also found that about half of the N input is retained within the wetland (M. Chimney personal communication). It appears that with relatively high P availability, there is net removal of both inorganic and organic N from surface waters.

Within ENP, rapid removal of inorganic N occurred in Taylor Slough and the southeastern Everglades (Table 3 and Fig. 7). Furthermore, a recent study in the sawgrass marsh immediately downstream of the C-111 Canal found rapid uptake of both P and DIN derived from the canal (D. Childers unpublished data). We found no evidence of TN removal in Taylor Slough, the C-111 Canal wetland, or the mangrove zone of Taylor Creek (Table 3 and Fig. 7). Only in the saline portion of Shark Slough was a TN decrease observed (Fig. 7), and most of this decrease was attributable simply to mixing with less N rich Gulf of Mexico water. Based on a mixing diagram (Fig. 8), some N removal apparently occurred in the Harney River. This apparent removal occurred in the same region where high P concentrations were found. This finding is consistent with a hypothesis that N cycling, and particularly organic N mineralization, is P limited through much of the Everglades. While little information from the Everglades exists to support this hypothesis, one study of anaerobic activity in Everglades soils found that acetate and methane production was stimulated by P enrichment (Drake et al. 1996). However, other studies of methane production and denitrification in these soils found no P-enrichment effect (Gordon et al. 1986; Bachoon and Jones 1992). The finding of molar N:P ratios that typically exceeded 400 in ENP wetlands (Fig. 7) is consistent with this hypothesis.

A NUTRIENT BUDGET FOR FLORIDA BAY: CALCULATIONS

In this section, we estimate the major inputs and outputs of nutrients at Florida Bay's boundaries (Table 5). We define the western boundary of the bay as the 81°05' meridian, which yields a total area of 2220 km^2 for the bay (Fig. 1). We also calculate

TABLE 5. Estimated N and P inputs to and outputs from Florida Bay, as total phosphorus (TP), total nitrogen (TN), and dissolved inorganic nitrogen (DIN). All values are in metric tons per year. Values in parentheses are not included in totals and question marks are given for potentially important terms for which no estimate was made. Everglades values are inputs *to* the wetland and are given as maximum values (with "<") when evidence for net uptake in the wetland was found. Maximum values ("<") are included in total values. The basis for each estimate is explained in the text.

	Entire Florida Bay			Eastern Florida Bay		
	TP	TN	DIN	TP	TN	DIN
INPUTS (metric tons per year)						
Everglades (from Taylor Slough & C-111 canal)	<2.6	250.	<60.	<2.6	250.	<60.
Everglades (from Shark Slough)	(<12.)	(1,400.)	(78.)			
Atmosphere	38.	710.	490.	10.	190.	130.
Florida Keys' wastewater	42.	170.	<170.	23.	94.	<94.
Gulf of Mexico boundary advection (north + middle)	500.	13,000.	660.			
Nitrogen fixation mainland groundwater	?	?	?	?	?	?
Total	580.	14,000.	1,400.	36.	530.	280.
OUTPUTS (metric tons per year)						
Gulf of Mexico boundary advection (south)	87.	3.500.	110.			
Gulf of Mexico boundary dispersion	7.	1,500.	150.			
Florida Keys' passes	180.	12,000.	780.	?	?	?
Denitrification, groundwater	?	?	?	?	?	?
Total	270.	17,000.	1,000.			

nutrient inputs to eastern Florida Bay, which we define as the area east of the 80°40' meridian, with an area about 600 km². This area lies to the east of a set of shallow banks that restrict water exchange with the rest of the bay (Fourqurean and Robblee 1999). Given the uncertainty that exists for all calculations involved in this exercise, we will only use these estimates as a basis for assessing the relative importance of the Everglades as a source of nutrients for Florida Bay.

From the results we have presented, we know that our estimate of the quantity of nutrients flowing into Florida Bay from the Everglades based on inputs to ENP wetlands is erroneous. Phosphorus inputs to the wetlands from the canal system greatly exceed P outputs from the freshwater wetlands. Over a decadal time scale, net P outputs from the mangrove zone to coastal waters probably are equivalent to inputs from the freshwater marshes. Nonsteady state conditions may exist over a time scale of several years, as indicated in Fig. 8. Compared to P transport, total N transport from canals, through the marshes, and to the coast is more conservative. An estimate of N loading to the coast based on N outputs from canals is probably only a slight overestimate. A similar estimate of inorganic N loading entails much greater error because of the reactive nature of inorganic nutrients, unknown rates of nitrogen fixation and denitrification, and the known capacity of Everglades marshes to sequester these nutrients (Table 3).

In Table 5, we use the calculated N and P inputs to Taylor Slough + C-111 as an estimate of inputs to Florida Bay. We expect that this overestimates the P and inorganic N actually reaching the bay. Estimates of nutrient inputs to the Gulf of Mexico

from Shark River Slough are likewise based on inputs to Shark River Slough from canal water, despite potential errors from wetland sources, sinks, and nonsteady state conditions. The estimate of nutrient output from Shark River Slough does not directly affect the Florida Bay nutrient budget. This slough flows to the Gulf of Mexico, not Florida Bay. Currently, the proportion of this slough water that flows south and east, entering Florida Bay across the 81°05' meridian, is unknown. The actual contribution of nutrients from Shark River Slough to Florida Bay must be less than the total Shark River Slough nutrient output. The total nutrient input from the Gulf of Mexico is calculated below and this calculation is independent of the Shark River Slough estimate.

Nutrient inputs from the groundwaters of the Florida mainland have not been quantified and are not estimated here. Given the high tranmissivity of limestone in the region, inputs to Florida Bay proper and the coastal zone of the Gulf of Mexico are possible and are the subject of ongoing investigations (L. Brand personal communication). It seems more likely that a subsurface wedge of saline water, which extends up to 10 km inland (Fitterman and Deszcz-Pan in press), causes most subsurface freshwater to surface within the Everglades.

Nutrient inputs from the atmosphere have been estimated regionally in Florida and locally at a central site in ENP and in the Florida Keys. Hendry et al. (1981) reported bulk N and P deposition on Bahia Honda Key, near Key West, in biweekly samples in 1978 and 1979. They reported TN deposition of 0.32 g N m^{-2} yr^{-1} and TP deposition of 0.017 g P m^{-2} yr^{-1}. In Florida statewide, they reported inorganic N was 69% of TN in this depo-

sition. These values are used in Table 5. This may underestimate N inputs to Florida Bay. Hendry et al. (1981) concluded that N deposition had increased about 3-fold from the mid1950s through 1979, and rates may have increased further since that time. Prospero et al. (1996) reported that inorganic N deposition in ENP was 0.305 g N m^{-2} yr^{-1} in 1990. This value may be unusually low because of low rainfall in 1990. We expect that values from Hendry et al. (1981) are reasonable approximations for Florida Bay.

Atmospheric P deposition has rarely been measured and, because it is mainly in the form of dry deposition and lower in concentration than N, is more prone to both sampling and analytical errors (Prospero et al. 1996). In the Everglades, attempts to measure P deposition have been prone to contamination by insects and birds. A likely minimum deposition rate is 0.006 g P m^{-2} yr^{-1}, a rate estimated for Bermuda (Graham and Duce 1982). A likely maximum deposition rate is 0.030 g P m^{-2} yr^{-1}, a rate estimated for the Everglades wetlands (Redfield 1998). Thus, we cautiously use values from Hendry et al. (1981) to estimate P deposition on Florida Bay waters (Table 5).

Our estimate of nutrient inputs to Florida Bay from the Florida Keys is based on a United States Environmental Protection Agency report (United States Environmental Protection Agency 1993). In this report, wastewater nutrient production from each populated island of the Florida Keys was estimated for wastewater treatment plants, domestic waste fields (including cesspits), and live-aboard boats. Nutrient production estimates were based on measured water use for treatment plants, estimated domestic water use, and nutrient concentrations that are considered typical of wastewater in Florida and the United States. The most important potential errors associated with using these estimates as a nutrient input to Florida Bay is the accuracy of the nutrient production rate, reactivity of nutrients between the waste site and coastal waters, and the direction of transport. We did not assess the accuracy of the United States Environmental Protection Agency estimates and expect that these are reasonable estimates. Based on studies of groundwater nutrients in the Keys, which are porous limestone islands with high transmissivity, it is likely that most N and a lesser proportion of P entering the waste-stream are mobile and quickly flow to coastal water (Lapointe et al. 1990). We assume that half of this flow is toward Florida Bay and half is toward the Atlantic Ocean. Nutrient inputs from stormwater runoff are not included in our estimate.

Finally, we estimate the flux of nutrients across the boundary between Florida Bay and the Gulf of Mexico. This flux includes advective transport and dispersion of nutrients. We calculated the advective nutrient flux for a 1-yr period (starting April 1, 1997) from the results of the ADCP current meters and the analysis of nutrient samples at adjacent stations (Table 4) along the 80°05' meridian (Fig. 1). This flux was calculated as the sum of the monthly products of the net east-west water flux and the nutrient concentration at each of the three sites along the meridian. For the southern and middle sites, nutrients were sampled seasonally (n = 5 from March 1996 to May 1997), so monthly concentrations were estimated by linear interpolation. Fluxes from each of the three sites were then extrapolated over the length of the entire bay-Gulf boundary (44.3 km) by dividing this boundary into three segments, with current meters at the midpoint of each segment, as in Smith and Pitts (1995). The estimated integrated water inflow to Florida Bay was 2.24 × 10^{10} m^3 yr^{-1} through the northern segment and 1.36 × 10^{10} m^3 yr^{-1} through the central segment. Estimated outflow from Florida Bay through the southern segment is 1.32 × 10^{10} m^3 yr^{-1}. Nutrient flux into the bay from the north and central segments are reported in the Input section of Table 5 and the flux from the bay through the southern segment is reported in the Output section. Because the nutrient flux estimates are directly dependent upon the water flux estimates and these estimates are based on only three current meters along the long and open bay-Gulf boundary, this nutrient flux estimate probably entails more uncertainty than other calculations in Table 5.

The finding of a net transport of water from the Gulf of Mexico into Florida Bay along the bay's western boundary is consistent with previous studies of Florida Bay circulation (Smith 1994; Wang et al. 1994). The main region of outflow from the bay is the southern portion of the Gulf boundary and the passes between the Florida Keys. We calculate nutrient loss from the bay through these passes as the product of annual water flux (here estimated as the difference between inflow and outflow on the western boundary) and mean 1991–1996 TN and TP at two water quality monitoring stations near the largest passes in the Keys.

We calculate the dispersive flux of nutrients across the bay-Gulf boundary from the east-west gradient of nutrient concentrations (Table 4), as determined from three pairs of sampling stations (stations 392, 25; 393, 26; 394, 27 in Fig. 1). Each pair of stations lies approximately on the same line of latitude and crosses the bay-Gulf boundary. For the western sites, which were sampled seasonally, monthly concentrations were estimated by linear interpolation. Dispersion for a 1-yr period (starting

April 1, 1997) was calculated as the sum of monthly dispersion across three segments of the bay-Gulf boundary (with the east-west gradient lines crossing a segment midpoint), using a dispersion coefficient of 100 m² s⁻¹ (Fischer et al. 1979) and water depth of 3 m.

A NUTRIENT BUDGET FOR FLORIDA BAY: INFERENCES AND MANAGEMENT IMPLICATIONS

The nutrient budget shows that the Gulf of Mexico is the dominant source of both N and P inputs to Florida Bay (Table 5). These inputs are attributable to the net advection of Gulf water into the bay, particularly through the northern portion of the bay-Gulf boundary, where the fastest water transport and the highest nutrient concentrations were found (Table 4). This advective input of nutrients to the bay was countered by a much smaller dispersive nutrient output from the bay. The resultant net import of nutrients from the Gulf was very large—at least 12 times larger than TN or TP inputs from any other source. This finding is consistent with inferences from previous studies of the distribution of nutrients in the Florida Bay's water and seagrass, which found that N : P and C : P ratios decreased toward the Gulf along an east-west gradient, thus pointing toward the Gulf as a dominant P source for the bay (Fourqurean et al. 1992; Fourqurean et al. 1993; Boyer et al. 1997). Given the large magnitude of the P and N inputs from the Gulf that are estimated in Table 5, the main question is not whether they are important to Florida Bay, but where they are important. Because of the bay's large, shallow banks, which are to the east ENP's western boundary in the bay (Fig. 1), it is likely that most of the nutrients transported across the bay-Gulf boundary only skirt the western banks and flow south toward the Atlantic. The flux of nutrients to the east of the banks and into the interior of Florida Bay is unknown.

The estimated magnitude of N and P inputs to Florida Bay from terrestrial and atmospheric sources is very low compared to inputs to the Chesapeake Bay system and other well-studied estuaries (Boynton et al. 1995; Boyer and Jones 1999). On an areal basis, TP input to Florida Bay (37 mg m⁻² yr⁻¹) is less than 10% of P inputs to the 20 estuaries reported in Boynton et al. (1995) and TN input to Florida Bay (510 mg m⁻² yr⁻¹) is less than 50% of N inputs to these estuaries. The resultant molar N : P ratio of terrestrial and atmospheric inputs to Florida Bay (N : P = 30) is only slightly higher than that of many other estuaries, and similar to that of Chesapeake Bay, because of the relatively minor contribution of inputs from the Everglades in the Florida Bay nutrient budget.

Because of the isolation of eastern Florida Bay

from the Gulf of Mexico, we can use a budget of this area (Table 5) to evaluate the relative influences of the Everglades, the atmosphere, and the Keys apart from the overwhelming influence of the Gulf. For P in eastern Florida Bay, inputs from the Keys and the atmosphere are of roughly the same magnitude and larger than the input from the Everglades. For N, the Everglades is the largest source, but inputs from the Keys and the atmosphere are of similar magnitude. The resultant molar N : P ratio for eastern Florida Bay inputs (N : P = 33) is far lower than the N : P ratio of ambient bay water in this region (median = 180; Boyer et al. 1997). The difference between the N : P ratios of nutrient inputs and ambient water reflects the long residence time of water in this region of the bay and the importance of internal nutrient processing. It is likely that the N : P ratio of this water is elevated by a combination of binding and burial of P in carbonate sediments (Orem et al. in press) and nitrogen fixation in seagrass beds and benthic algal mats (Powell et al. 1989). As in Shark Bay, a hypersaline lagoon with carbonate sediments in Australia (Smith and Atkinson 1984; Atkinson 1987), both of these processes drive eastern Florida Bay toward a state of P limitation. The high N : P ratio of inputs from the Everglades reinforce this P limitation.

Based on the balance of total inputs and total outputs for the bay as a whole (Table 5), it appears that about half of the P that entered the bay was retained within the bay. It also appears that N exports may exceed N imports. This result is consistent with the expectation that the bay sediments are a P sink and that nitrogen fixation is an important N source within the bay. However, without measurements of the rates of these internal processes (plus denitrification) and the seaward exchange of nutrients through the Keys' passes, the actual magnitude of the net P and N exchange with the Gulf of Mexico and the Atlantic (through the Keys' passes) remains quite uncertain. We conclude only that the importance of seaward exchange in Florida Bay differs considerably from its importance in much more eutrophied and river-dominated estuaries, where nutrient imports are dominated by terrestrial and atmospheric sources and seaward exchange results in the export of these nutrients from the estuary (Kelly 1997).

Given the results in Table 5, we can infer the relative importance of nutrient inputs from the Everglades, including the Shark River Slough, and the potential impact of increasing water flow through these wetlands. Even assuming that all P flowing into Everglades National Park, including Shark River Slough, is delivered into Florida Bay (i.e., all Shark River Slough waters flowing into the

Gulf of Mexico then flow into the bay), this source is minor for the bay as a whole; the freshwater Everglades (Taylor Slough + C-111 watershed plus Shark River Slough) apparently contribute less than 3% of all P inputs to the bay. The freshwater Everglades are more important as an N source but still apparently contribute less than 12% of all N inputs to the bay. Despite the large magnitude of estimated N input to the Gulf of Mexico from Shark River Slough, the Gulf itself is a much more important N source. Even if all N that enters Shark Slough were directly injected into western Florida Bay, this N would only represent about 11% of the calculated Gulf input. This is, of course, only a tentative conclusion because of the relatively short duration (1 yr) of flow measurements on the Gulf boundary and the coarse spatial resolution of flow and nutrient data at the boundary. If future measurements and models confirm the magnitude of this net water and nutrient flux, then any increase in freshwater flow with associated N into the Gulf of Mexico is unlikely to impact Florida Bay.

For eastern and central Florida Bay, changes in water flow and associated N inputs could yield a significant increase in the total N pool. However, the water that flows from the wetland is likely to be depleted in inorganic N (Table 3), while atmospheric N inputs are mostly inorganic N (Hendry et al. 1981). The ecological influence of the readily available inorganic N probably exceeds the influence of the more refractory organic N pool. Given the high concentrations of inorganic N already in eastern Florida Bay (Fourqurean et al. 1993; Boyer et al. 1997), additional N inputs to eastern Florida Bay are unlikely to have a local effect. This expectation is supported by the statistical independence of chlorophyll a concentrations and estimated N loading (as well as P loading) from Taylor Slough and the C-111 Canal to eastern Florida Bay (Boyer and Jones 1999). The effect of N loading to eastern Florida Bay is more likely to be remote. As in the Everglades, N from this P-limited area may circulate to a boundary where P is relatively more available and may then stimulate productivity and decomposition.

We conclude that the ongoing effort to hydrologically restore the Everglades and Florida Bay is unlikely to have a pronounced effect on the bay directly via increased P and N loading from the Everglades. Based on the above calculations, P input to the bay from the freshwater Everglades is a very small portion of the bay's P budget. Even ignoring the potentially large N input from the Gulf, N input to the bay from the Everglades is similar in magnitude to N input from the atmosphere. The ecological effect of N from the Everglades is

probably far less than N from the atmosphere because Everglades N is more refractory, organic N.

Understanding the relative importance of the Everglades in Florida Bay's nutrient budget requires much greater understanding of the interactions of the bay and the Gulf of Mexico. The Gulf appears to have an overwhelming influence on nutrient transport to and, potentially, through the bay. Given all of the shortcomings of our nutrient budget calculations, we conclude that a much more rigorous effort is needed in order to quantify the importance of the Gulf. This includes the development of a hydrodynamic model that can estimate the fate of waters after they flow into the bay from the Gulf.

Predicting the effect of increasing freshwater flow on nutrient inputs to Florida Bay requires considerably more information on the dynamics of nutrients within the freshwater Everglades and the mangrove zone. From our results, it appears that the direct effect of increasing freshwater flow on the bay's nutrient budget cannot be considered in a simple, proportional manner. Even if freshwater flow to the bay is doubled, this will not result in a doubling of nutrient inputs to the freshwater marshes of ENP or through these marshes. Phosphorus inputs are unlikely to change with increasing flow because of the strong P retention within Everglades wetlands. Nitrogen inputs will increase, but operation of large constructed wetlands designed to decrease P loading of the WCAs and ENP (Walker 1995) are also likely to decrease N loading to these wetlands and Florida Bay. Finally, an important unknown that affects the fate of nutrients from the Everglades and coastal waters is the processing of nutrients within the mangrove zone and how this processing is affected by changing freshwater flow and salinity. While hydrological restoration of the Everglades may not greatly alter direct nutrient loads from the Everglades, it probably will decrease the extent of salinity intrusion into the mangrove zone and decrease the extent to which the mangrove zone acts as a sink for estuarine nutrients.

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LITERATURE CITED

AMADOR, J. A., G. H. RICHANY, AND R. D. JONES. 1992. Factors affecting phosphate uptake by peat soils of the Florida Everglades. *Soil Science* 153:463–470.

ATKINSON, M. J. 1987. Low phosphorus sediments in a hypersaline marine bay. *Estuarine, Coastal and Shelf Science* 24:335–347.

BACHOON, D. AND R. D. JONES. 1992. Potential rates of methanogenesis in sawgrass marshes with peat and marl soils in the Everglades. *Soil Biology and Biochemistry* 24:21–27.

BOYER, J. N., J. W. FOURQUREAN, AND R. D. JONES. 1997. Spatial characterization of water quality in Florida Bay and Whitewater Bay by multivariate analysis: Zones of similar influence (ZSI). *Estuaries* 20:743–758.

BOYER, J. N. AND R. D. JONES. 1999. Effects of freshwater inputs and loading of phosphorus and nitrogen on the water quality of eastern Florida Bay, p. 545–561. *In* K. R. Reddy, G. A. O'Connor, and C. L. Schelske (eds.), Phosphorus Biogeochemistry of Subtropical Ecosystems: Florida as a Case Example. CRC/Lewis Publishers, Boca Raton, Florida.

BOYNTON, W. R., J. H. GARBER, R. SUMMERS, AND W. M. KEMP. 1995. Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries* 18:285–314.

BUTLER, IV, M. J., J. H. HUNT, W. F. HERRNKIND, M. J. CHILDRESS, R. BERTELSON, W. SHARP, T. MATTHEWS, J. M. FIELD, AND H. G. MARSHALL. 1995. Cascading disturbances in Florida Bay, USA: Cyanobacteria blooms, sponge mortality, and implications for juvenile spiny lobsters *Pamulirus argus*. *Marine Ecology Progress Series* 129:119–125.

CLESCERI, L. S., A. E. GREENBERG, AND R. R. TRUSSEL (EDS.). 1989. Standard Methods for the Examination of Water and Wastewater, 17th edition. American Public Health Association, Washington, D. C.

CRAFT, C. B. AND C. J. RICHARDSON. 1993. Peat accretion and phosphorus accumulation along a eutrophication gradient in the northern Everglades. *Biogeochemistry* 22:133–156.

DAVIS, S. M., L. H. GUNDERSON, W. A. PARK, J. R. RICHARDSON, AND J. E. MATTSON. 1994. Landscape dimension, composition, and function in a changing Everglades ecosystem, p. 419–444. *In* S. M. Davis and J. C. Ogden (eds.), Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, Florida.

DAVIS, S. M AND J. C. OGDEN (EDS.). 1994. Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, Florida.

DEBUSK, W. F., K. R. REDDY, M. S. KOCH, AND Y. YANG. 1994. Spatial distribution of soil nutrients in a northern Everglades marsh: Water Conservation Area 2A. *Journal of the Soil Science Society of America* 58:543–552.

DIAZ, O. A., K. R. REDDY, AND P. A. MOORE, JR. 1994. Solubility of inorganic phosphorus in stream water as influenced by pH and calcium concentration. *Water Research* 28:1755–1763.

DOREN, R. F., T. V. ARMENTANO, L. D. WHITEAKER, AND R. D. JONES. 1997. Marsh vegetation patterns and soil phosphorus gradients in the Everglades ecosystem. *Aquatic Botany* 56:145–163.

DRAKE, H. L., N. G. AUMEN, C. KUHNER, C. WAGNER, A. GRIESSHAMMER, AND M. SCHMITTROTH. 1996. Anaerobic microflora of Everglades sediments: Effects of nutrients on population profiles and activities. *Applied and Environmental Microbiology* 62:486–493.

FENNEMA, R. J., C. J. NEIDRAUER, R. A. JOHNSON, T. K. MACVICAR, AND W. A. PERKINS. 1994. A computer model to simulate natural Everglades hydrology, p. 249–289. *In* S. M. Davis and J. C. Ogden (eds.), Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, Florida.

FISCHER, H. B., E. J. LIST, R. C. Y. KOH, J. IMBERGER, AND N. H. BROOKS. 1979. Mixing in Inland and Coastal Waters. Academic Press, New York.

FITTERMAN, D. V. AND M. DESZCZ-PAN. In press. Helicopter EM mapping of saltwater intrusion in Everglades National Park, Florida. *Geophysical Exploration*.

FOURQUREAN, J. W., R. D. JONES, AND J. C. ZIEMAN. 1993. Processes influencing water column nutrient characteristics and phosphorus limitation of phytoplankton biomass in Florida Bay, FL, USA: Inferences from spatial distributions. *Estuarine and Coastal Shelf Science* 36:295–314.

FOURQUREAN, J. W. AND M. B. ROBBLEE. 1999. Florida Bay: A history of recent ecological changes. *Estuaries* 22:345–357.

FOURQUREAN, J. W., J. C. ZIEMAN, AND G. V. N. POWELL. 1992. Phosphorus limitation of primary production in Florida Bay: Evidence from the C:N:P ratios of the dominant seagrass *Thalassia testudinum. Limnology and Oceanography* 37:162–171.

FRANKOVICH, T. A. AND R. D. JONES. 1998. A rapid, precise and sensitive method for the determination of total nitrogen in natural waters. *Marine Chemistry* 60:227–234.

GLEASON, P. J. AND W. SPACKMAN. 1974. Calcareous periphyton and water chemistry in the Everglades, p. 225–248. *In* P. J. Gleason (ed.), Environments of South Florida: Past and Present. Miami Geological Society, Coral Gables, Florida.

GORDON, A. S., W. J. COOPER, AND D. J. SCHEIDT. 1986. Denitrification in marl and peat sediments in the Florida Everglades. *Applied and Environmental Microbiology* 52:987–991.

GRAHAM, W. F. AND R. A. DUCE. 1982. The atmospheric transport of phosphorus to the western north Atlantic. *Atmospheric Environment* 16:1089–1097.

HENDRY, C. D., P. L. BREZONIK, AND E. S. EDGERTON. 1981. Atmospheric deposition of nitrogen and phosphorus in Florida, p. 199–215. *In* S. J. Eisenreich (ed.), Atmospheric Pollutants in Natural Waters. Ann Arbor Science, Ann Arbor, Michigan.

JONES, R. D. AND J. A. AMADOR. 1992. Removal of total phosphorus and phosphate by peat soils of the Florida Everglades. *Canadian Journal of Fisheries and Aquatic Sciences* 49:577–583.

KELLY, J. R. 1997. Nitrogen flow and the interactions of Boston Harbor with Massachusetts Bay. *Estuaries* 20:365–380.

KOCH, M. S. AND K. R. REDDY. 1992. Distribution of soil and plant nutrients along a trophic gradient in the Florida Everglades. *Soil Science Society of America Journal* 56:1492–1499.

KOCH, M. S. AND S. C. SNEDAKER. 1997. Factors influencing *Rhizophora mangle* L. seedling development in Everglades carbonate soils. *Aquatic Botany* 59:87–98.

LAPOINTE, B. E. AND M. W. CLARK. 1992. Nutrient inputs from the watershed and coastal eutrophication in the Florida Keys. *Estuaries* 15:465–476.

LAPOINTE, B. E., J. D. O'CONNELL, AND G. S. GARRETT. 1990. Nutrient couplings between on-site sewage disposal systems, groundwaters, and nearshore surface waters of the Florida Keys. *Biogeochemistry* 10:289–307.

LAPOINTE, B. E., D. A. TOMASKO, AND W. R. MATZIE. 1994. Eutrophication and tropic state classification of seagrass communities in the Florida Bay Keys. *Bulletin of Marine Science* 54:696–717.

LIGHT, S. S. AND J. W. DINEEN. 1994. Water control in the Everglades: A historical perspective, p. 47–84. *In* S. M. Davis and J. C. Ogden (eds.), Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, Florida.

McCORMICK, P. V. AND M. B. O'DELL. 1996. Quantifying periphyton responses to phosphorus in the Florida Everglades: A synoptic-experimental approach. *Journal of the North American Benthological Society* 15:450–468.

McCormick, P. V., P. S. Rawlik, K. Lurding, E. P. Smith, and F. H. Sklar. 1996. Periphyton-water quality relationships along a nutrient gradient in the northern Florida Everglades. *Journal of the North American Benthological Society* 15:433–449.

McIvor, C. C., J. A. Ley, and R. D. Bjork. 1994. Changes in freshwater inflow from the Everglades to Florida Bay including effects on biota and biotic processes: A review, p. 117–146. *In* S. M. Davis and J. C. Ogden (eds.), Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, Florida.

Newman, S., J. B. Grace, and J. W. Koebel. 1996. Effects of nutrients and hydroperiod on *Typha, Cladium,* and *Eleocharis*: Implications for Everglades restoration. *Ecological Applications* 6:774–783.

Orem, W. H., C. W. Holmes, C. Kendall, H. E. Lerch, A. L. Bates, S. R. Silva, A. Boylan, M. Corum, M. Marot, and C. Hedgman. In press. Geochemistry of Florida Bay sediments: I. Nutrient history at five sites in eastern and central Florida Bay. *Journal of Coastal Research.*

Phlips, E. J. and S. Badylak. 1996. Spatial variability in phytoplankton standing crop and composition in a shallow inner-shelf lagoon, Florida Bay, Florida. *Bulletin of Marine Science* 58: 203–216.

Powell, G. V. N., W. J. Kenworthy, and J. W. Fourqurean. 1989. Experimental evidence for nutrient limitation of seagrass growth in a tropical estuary with restricted circulation. *Bulletin of Marine Science* 44:324–340.

Prospero, J. M., K. Barrett, T. Church, F. Dentener, R. A. Duce, J. N. Galloway, H. Levy, II, J. Moody, and P. Quinn. 1996. Atmospheric deposition of nutrients to the north Atlantic basin. *Biogeochemistry* 35:27–73.

Reddy, K. R., Y. Wang, W. F. DeBusk, M. M. Fisher, and S. Newman. 1998. Forms of soil phosphorus in selected hydrological units of the Florida Everglades. *Journal of the Soil Science Society of America* 62:1134–1147.

Redfield, G. W. 1998. Quantifying atmospheric deposition of phosphorus: A conceptual model and literature review for environmental management. Technical Publication WRE #360, South Florida Water Management District, West Palm Beach, Florida.

Robblee, M. B., T. R. Barber, P. R. Carlson, Jr., M. J. Durako, J. W. Fourqurean, L. K. Muehlstein, D. Porter, L. A. Yarbro, R. T. Zieman, and J. C. Zieman. 1991. Mass mortality of the tropical seagrass *Thalassia testudinum* in Florida Bay (USA). *Marine Ecology Progress Series* 71:297–299.

Sklar, F. H. and J. A. Browder. 1998. Coastal environmental impacts brought about by alterations to freshwater flow in the Gulf of Mexico. *Environmental Management* 22:547–562.

Smith, N. P. 1994. Long-term Gulf-to-Atlantic transport through tidal channels in the Florida Keys. *Bulletin of Marine Science* 54: 602–609.

Smith, N. P. and P. A. Pitts. 1995. Long-term transport patterns in Florida Bay. Final report, Florida Department of Environmental Protection, Florida Marine Research Institute, St. Petersburg, Florida.

Smith, S. V. and M. J. Atkinson. 1984. Phosphorus limitation of net production in a confined aquatic ecosystem. *Nature* 307:626–627.

Smith, III, T. J., M. B. Robblee, H. R. Wanless, and T. W. Doyle. 1994. Mangroves, hurricanes and lightning strikes. *Bioscience* 44:256–262.

Solorzano, L. and J. H. Sharp. 1980. Determination of total dissolved phosphorus and particulate phosphorus in natural waters. *Limnology and Oceanography* 25:754–758.

South Florida Water Management District. 1992. Surface water improvement and management plan for the Everglades. South Florida Water Management District, West Palm Beach, Florida.

South Florida Water Management District. 1997. Everglades best management practices program. South Florida Water Management District, West Palm Beach, Florida.

Steward, K. K. and W. H. Ornes. 1983. Mineral nutrition of sawgrass (*Cladium jamaicense* Crantz) in relation to nutrient supply. *Aquatic Botany* 16:349–359.

United States Environmental Protection Agency. 1993. Water-quality protection program for the Florida Keys National Marine Sanctuary. Phase II report. United States Environmental Protection Agency, Washington, D. C.

Urban, N. H., S. M. Davis, and N. G. Aumen. 1993. Fluctuations in sawgrass and cattail densities in Everglades Water Conservation Area 2A under varying nutrient, hydrologic and fire regimes. *Aquatic Botany* 46:203–223.

Walker, W. W. 1991. Water quality trends at inflows to Everglades National Park. *Water Resources Bulletin* 27:59–72.

Walker, W. W. 1995. Design basis for Everglades storm treatment areas. *Water Resources Bulletin* 31:671–685.

Walker, W. W. 1999. Long-term water quality trends in the Everglades, p. 447–466. *In* K. R. Reddy, G. A. O'Connor, and C. L. Schelske (eds.), Phosphorus Biogeochemistry of Subtropical Ecosystems: Florida as a Case Example. CRC/Lewis Publishers, Boca Raton, Florida.

Wang, J. D., J. van de Kreeke, N. Krishnan, and D. Smith. 1994. Wind and tide response in Florida Bay. *Bulletin of Marine Science* 54:579–601.

Wu, Y., F. H. Sklar, and K. Rutchey. 1997. Analysis and simulations of fragmentation patterns in the Everglades. *Ecological Applications* 7:268–276.

Sources of Unpublished Materials

Brand, L. University of Miami, Rosenstiel School of Marine and Atmospheric Science, 4600 Rickenbacker Causeway, Miami, Florida.

Pratt, T. Unpublished data from acoustic doppler current profilers. Waterways Experiment Station, United States Army Corps of Engineers, 3909 Hill Ferry Road, Vicksburg, Mississippi.

Patino, E. United States Geological Survey, 9100 N. W. 36th Street #107, Miami, Florida.

Smith, D. Everglades National Park, 40001 State Road 9336, Homestead, Florida.

Smith, N. Unpublished data from acoustic doppler current profilers. Harbor Branch Oceanographic Institution, 5600 North United States Highway 1, Fort Pierce, Florida.