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P. H. Doering and R. H. Chamberlain

Made in United States of America
Reprinted from Journal of the American Water Resources Association
Vol. 35, No. 4, August 1999

JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION
VOL. 35, NO. 4 AMERICAN WATER RESOURCES ASSOCIATION AUGUST 1999

WATER QUALITY AND SOURCE OF FRESHWATER DISCHARGE TO THE CALOOSAHATCHEE ESTUARY, FLORIDA[1]

P. H. Doering and R. H. Chamberlain[2]

ABSTRACT: The Caloosahatchee River has two major sources of freshwater: one from its watershed and the other via an artificial connection to Lake Okeechobee. The contribution of each source to the freshwater discharge reaching the downstream estuary varies and either may dominate. Routine monitoring data were analyzed to determine the effects of total river discharge and source of discharge (river basin, lake) on water quality in the downstream estuary. Parameters examined were: color, total suspended solids, light attenuation, chlorophyll *a*, and total and dissolved inorganic nitrogen and phosphorus. In general, the concentrations of color, and total and dissolved inorganic nitrogen increased, and total suspended solids decreased, as total discharge increased. When the river basin was the major source, the concentrations of nutrients (excepting ammonia) and color in the estuary were relatively higher than when the lake was the major source. Light attenuation was greater when the river basin dominated freshwater discharge to the estuary. The analysis indicates that water quality in the downstream estuary changes as a function of both total discharge and source of discharge. Relative to discharge from the river basin, releases from Lake Okeechobee do not detectably increase concentrations of nutrients, color, or TSS in the estuary.
(KEY TERMS: aquatic ecosystems; water quality; ocean studies; estuary; nutrients; freshwater discharge.)

INTRODUCTION

Freshwater discharge from rivers influences estuarine water quality directly through changes in quantity and composition (Nixon, 1981; Loder and Reichard, 1981; Cifuentes *et al.*, 1990). In response to rainfall events, increases in the quantity of discharge generally result in decreased salinity and a higher fraction of fresh water at a given geographical location in an estuary (Balls *et al.*, 1997). Because the concentrations of nutrients and other water quality parameters in fresh water and seawater differ, changes in the freshwater fraction result in changes in concentration. Superimposed on effects arising from the mixing of fresh and salt water, are those arising from changes in composition of the fresh water that are often themselves related to the quantity of discharge (Hirsch *et al.*, 1982; Balls *et al.*, 1997).

Freshwater discharge also indirectly influences estuarine water quality through its effect on residence time. The degree to which the concentration of material is altered by reaction within the estuary is largely a function of residence time, which depends on river discharge (Doering *et al.*, 1994; Balls, 1994; Eyre and Twigg, 1997). In combination, these factors have made it difficult to demonstrate the extent to which compositional changes in the freshwater endmember are reflected in the downstream estuary (Balls *et al.*, 1997).

Over the long term, anthropogenic activities also affect river discharge. Direct alterations such as chanelization, diversion, damming, and withdrawal cause changes in both the quantity and quality of discharge (Hopkinson and Vallino, 1995). In the contiguous United States, it is hard to find a body of fresh water that has not been directly impacted (Naiman *et al.*, 1995). In upland watersheds, conversion of land from one use to another can cause additional changes in the quantity and composition of runoff (Correll *et al.*, 1992; Balls, 1994).

In south Florida, the artificial connection of Lake Okeechobee to the Caloosahatchee River and its estuary represents a unique anthropogenic manipulation of hydrology. As a result, the Caloosahatchee River has two major sources of fresh water: one from its watershed, and the other from Lake Okeechobee. The contribution of each source to the total discharge

[1]Paper No. 98022 of the *Journal of the American Water Resources Association*. **Discussions are open until April 1, 2000.**
[2]Ecosystem Restoration Department, South Florida Water Management District, 3301 Gun Club Road, West Palm Beach, Florida 33406 (E-Mail/Doering: pdoering@sfwmd.gov).

reaching the estuary varies and either may dominate. Management questions concerning downstream effects of discharge have centered on both quantity and quality and particularly on effects of releases from Lake Okeechobee. In this report, we evaluate the effects of total river discharge and the source of discharge (river basin or lake) on the concentration of nutrients and other water quality parameters in the downstream estuary.

STUDY AREA

The Caloosahatchee River and its estuary are located on the southwest coast of Florida (Figure 1). The river was first extended eastward and connected to Lake Okeechobee in 1884. The combination lock and dam structures at the towns of Moore Haven (S-77) and Ortona (S-78) were completed in 1937 to control river flow and discharge from Lake Okeechobee (Figure 1). The river was straightened and deepened in 1937, 1941, and again in 1966 to improve navigation and provide flood control. The final structure, the Franklin Lock and Dam (S-79), was added in 1966 to further control river flow and to act as a salinity barrier (Flaig and Capece, 1998).

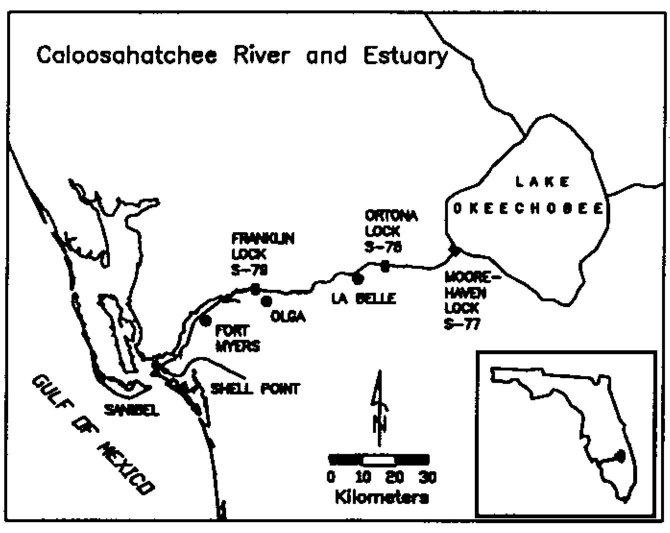

Figure 1. Map of the Caloosahatchee River and Estuary Showing the Connection to Lake Okeechobee and Location of Water Control Structures.

The Caloosahatchee River drains a watershed of 344,000 ha. Runoff and groundwater seepage from the watershed and discharge from Lake Okeechobee at S-77 are the major sources of water to the river (Flaig and Capece, 1998). Discharges from Lake Okeechobee occur for flood control and water supply. Regulatory releases are made to maintain the Lake level below a prescribed schedule. Water is also released to flush algal blooms and salt water out of the river, which contaminate potable water supplies (Flaig and Capece, 1998).

Freshwater discharge from the river to the estuary at S-79 is significant. Enough freshwater enters the estuary at S-79 to fill its entire volume over eight times per year. The volume of the estuary (Shell Point upstream to S-79, Figure 1) is approximately 105 x 10^6 m^3 (Stoker 1992), while the median annual discharge at S-79 is 870 x 10^6 m^3 (Flaig and Capece, 1998).

METHODS

Data Sets

The data used to address the effects of freshwater input and source (basin vs. lake) on water quality in the downstream estuary derive from several different sampling programs. Water quality in the Caloosahatchee estuary, San Carlos Bay, and Pine Island Sound was sampled monthly at stations 1-17 (Figure 2) from December 1985 to May 1989. A subset of these stations (0, 2, 4, 8, 10, 11, 12, 16, 17) was revisited on a monthly basis from November 1994 to February 1996. All these data were used to evaluate effects of discharge on nutrient, color, chlorophyll *a*, and total suspended solids concentrations in the estuary.

The light extinction coefficient (K) for photosynthetically active radiation (PAR: 400-700 nm) was measured only during November 1994 to February 1996. These data were combined with measurements taken as part of another project at 'HB' stations (Figure 2) from May 1996 to March 1997. Combined data were used to evaluate effects of discharge and source on K and the relationship between K and other water quality parameters.

The effects of source on the concentration of nutrients, color, and TSS in the fresh water discharged at the Franklin Lock and Dam were evaluated using routine monitoring data collected bimonthly from 1986-1996 by the South Florida Water Management District. Average daily freshwater discharge (cfs) from Lake Okeechobee at Moore Haven (S-77) and to the estuary at Olga (S-79) were obtained from records maintained by the South Florida Water Management District.

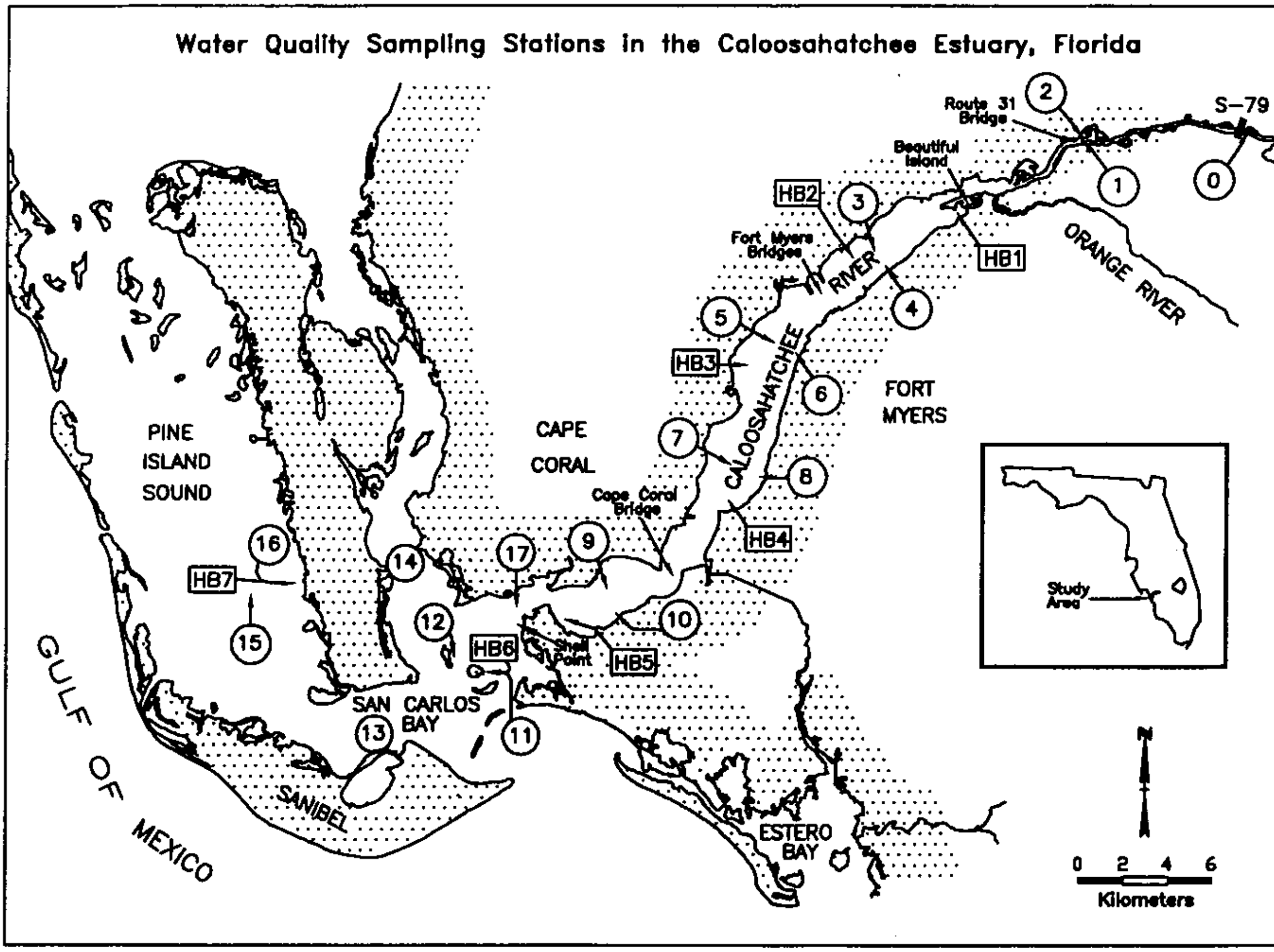

Figure 2. Location of Water Quality Sampling Stations.

Water Sampling and Analysis

Water sampling and analytical techniques were consistent across all data sets. Water samples were obtained with a van Dorn bottle from a depth of 0.5 meters. Conductivity was measured at a depth of 0.5 m with a sonde unit: Hydrolab Surveyor or YSI 600XL. In the field, samples for dissolved inorganic nitrogen (NH_4, NO_2 + NO_3 = NO_X) and phosphorus (DIP) and color were passed manually through 0.4 um filters using a syringe. Whole water samples were retained for total kjeldahl nitrogen (TKN), total phosphorus (TP), total suspended solids (TSS), and chlorophyll *a*. Samples for NH_4, NO_X, TKN, and TP were acidified to pH = 2 with 50 percent H_2SO_4. All samples were stored on ice until their return to the laboratory. Chlorophyll samples were filtered and analyzed in the laboratory within 24 hours of collection (SFWMD, 1994).

The light extinction coefficient (K) was calculated from vertical profiles of PAR taken with a LI-193SA spherical quantum sensor. Measurements were taken at 0.5 m intervals except near the surface where they were taken at 0.1, 0.25, and 0.5 m. Concurrent measurements of incident radiation in air were taken with a deck cell (LI-190SA quantum sensor). K was calculated from the equation:

$$\ln(I_Z/I_0) = -KZ \quad (1)$$

where I_Z is PAR at depth (Z), and I_0 is the deck cell PAR reading.

In the laboratory, all water quality parameters were analyzed by standard methods (APHA *et al.*, 1985) and as described in the South Florida Water Management District QA Plan (SFWMD, 1994). In brief, dissolved inorganic nutrients were analyzed colorometrically on an Alpkem Rapid Flow Analyzer (DIP within 24 hrs of collection, NH_4 and NO_X within 28 days). Color was quantified spectrophotometrically at 465 nm against a platinum cobalt standard within 48 hrs of collection. Chlorophyll *a* was analyzed spectrophotometrically. TP was analyzed by the molybdate-ascorbic acid method after persulphate digestion, within 28 days of collection.

Calculations and Statistics

In river dominated estuaries, water quality varies as a function of the magnitude of freshwater discharge and the geographic distance from the discharge. In addition to evaluating the effects of freshwater source, the analysis also accounted for these factors. To account for spatial variation, the estuary was divided into five regions based on geography and the distribution of stations (Table 1). For each sampling date, water quality data were averaged across stations within a region. This averaging procedure employed all available information for a particular date within a region.

TABLE 1. Estuarine Regions. Distances are kilometers downstream of Structure-79.

Region	Kilometers from S-79	Stations
Head	0-4	0,1,2,3,4, HB1
Upper Estuary	14-28	5,6,7,8, HB2, HB3
Lower Estuary	28-41	9,10,17, HB4, HB5
San Carlos Bay	41-49	11,12,13,14, HB6
Pine Island Sound	59	15,16, HB7

Total discharge was estimated as the average daily discharge at S-79 during the 30 days prior to each sampling event. The 30-day time period was chosen first because numerical modeling of salinity in the estuary indicates that at average discharges (~1000 cfs), the hydraulic residence time in the estuary is about one month (Bierman, 1993). Secondly, as compared to 14- and 21-day averages, the 30-day average yielded higher correlations (Spearman Rank) with salinity on the day of sampling in each of the five estuarine regions.

For each sampling date, the predominant source of water was classified as coming either from the river basin or the lake. The predominant source was determined by difference between average daily discharge (mean of 30 days) at S-79 and S-77. This difference represents an estimate of the 30-day mean discharge from the Caloosahatchee River basin. If the ratio of calculated basin discharge:discharge at S-79 was greater than or equal to 0.5 for a particular sampling date, then the source was classified as 'Basin.' If the ratio was less than 0.5, then the lake was considered the predominant source.

Estuarine water quality data were analyzed by a 3-factor ANOVA with interaction. The factors were source, with two levels (river basin, lake), total discharge at S-79, with three levels (< 1000 cfs, 1000-4000 cfs, > 4000 cfs), and region, with four levels (head, upper estuary, lower estuary, San Carlos Bay). Statistically significant differences between main effect and interactions means were evaluated using the Student-Newman-Keuls procedure (Winer, 1971). This analysis treats total discharge, a continuous variable, as a stratified categorical variable. The number of observations in each cell of the analysis depends on how the levels of discharge are stratified. The levels of discharge chosen yielded at least three observations per cell. The Pine Island Sound region was not included in the analysis because stations in this region were visited less frequently than the others and we had no observations at the highest discharge level, when the lake was the major source. Because the light extinction data set was relatively small, the two highest levels of discharge were combined to ensure three observations per cell.

Since ANOVA treated discharge as a categorical variable, the effects of total discharge also were investigated using the non-parametric Spearman's Rank Correlation Coefficient (SAS Institute, 1989). Correlations between total discharge and water quality parameters were calculated separately for all five regions. Correlations were judged significant at $p < 0.05$ and no correction for multiple testing was applied.

Water quality data collected at S-79 were analyzed by a two factor ANOVA with interaction. The factors were source, with two levels as above, and total discharge, with three levels as above. Statistical analyses were performed using SAS Version 6 (SAS Institute, 1989).

Before final selection of ANOVA models, tests for normality, homogeneity of variance and serial correlation were conducted (Table 2). For the estuarine water quality data, tests were conducted for each region as sampling occurred over time in each region. While ANOVA is generally considered to be robust with respect to deviation from normality (Keppel, 1973), the normality of ANOVA residuals was assessed using the Shapiro-Wilk Statistic (SAS Institute, 1989). For the raw data, 66 percent of the tests (total = 47) revealed significant ($p < 0.05$) deviation from normality. Application of a log (value + 1) transformation to all water quality parameters except K, the light extinction coefficient, and salinity reduced this proportion to 43 percent. Transformation of K and salinity did not reduce instances of deviation from normality. Homogeneity of variance was examined using the F-Max test (Sokal and Rholf, 1981) and 21 percent of the tests were significant ($p < 0.05$, Table 2). ANOVA is also robust to violations of this assumption and even sizable differences among variances do not seriously affect significance tests (Keppel, 1973).

TABLE 2. Testing Assumptions of ANOVA. All data except K and Salinity were Log (value +1) transformed. N = Shapiro-Wilk test for normality, H = F-max test for homogeneity of variances and S = test for serial correlation. Tests conducted for each of four regions in the estuary and for freshwater data obtained at Structure S-79. *Statistically significant at $p < 0.05$, ns = not significant.

Parameter	Test	Head	Upper	Lower	Bay	S-79
Color	N	ns	ns	*	ns	ns
	H	ns	ns	ns	ns	ns
	S	ns	ns	ns	ns	ns
TSS	N	ns	ns	ns	ns	*
	H	ns	ns	ns	ns	ns
	S	ns	ns	ns	*	ns
TN	N	ns	ns	*	*	*
	H	ns	ns	ns	ns	*
	S	ns	ns	ns	ns	ns
NO_X	N	*	*	*	*	*
	H	ns	ns	ns	ns	*
	S	ns	ns	ns	ns	ns
NH_4	N	*	*	*	*	*
	H	ns	*	ns	*	*
	S	ns	ns	ns	*	ns
TP	N	*	ns	*	ns	*
	H	ns	ns	*	ns	*
	S	*	*	*	ns	ns
DIP	N	ns	ns	ns	*	ns
	H	ns	ns	ns	ns	*
	S	*	*	*	ns	ns
Chlorophyll *a*	N	ns	ns	ns	ns	–
	H	ns	ns	ns	ns	–
	S	ns	ns	ns	ns	–
K	N	*	ns	ns	ns	–
	H	ns	ns	ns	ns	–
	S	ns	ns	ns	ns	–
Salinity	N	*	ns	ns	ns	–
	H	*	*	ns	ns	–
	S	ns	ns	ns	ns	–

ANOVA is most sensitive to serial correlation (Keppel, 1973). Positive serial correlation causes F-tests to be liberal (Snedecor and Cochran, 1967). Serial correlation was assessed by ordering residuals in time and calculating the correlation between the residuals at times t and t-1 (Chatfield, 1989). There were eight instances of serial correlation, mostly (six) confined to the phosphorus parameters (Table 2). As these correlations were positive, significance tests reported here for TP and DIP are liberal and should be viewed with caution.

Light Extinction

While the light extinction coefficient (K) was measured in the estuary, it was not measured in the freshwater discharged at S-79. To better understand variability in K and how freshwater discharge might affect K, the relationship between light extinction and selected water quality parameters in the estuary was investigated by stepwise multiple regression. The independent water quality variables examined were TSS, color, and chlorophyll *a*. All are known to affect light extinction in estuarine waters (Gallegos and Kenworthy, 1996; Chamberlain and Haywood, 1996). Multiple regressions were calculated for each of the five regions and a log (value + 1) transformation was applied to the TSS, color and chlorophyll *a* data to better meet the normality assumption of regression (Sokal and Rholf, 1981).

RESULTS

Discharge

Average monthly discharge from Lake Okeechobee at S-77 and to the estuary at S-79 is shown for the period, January 1985 to September 1996 in Figure 3. The difference between the two lines represents an estimate of the freshwater discharge at S-79 that is due to runoff from the Caloosahatchee River basin. Runoff from the basin comprised more than half the total discharge in 62 percent of the 141 months, while the lake dominated in 38 percent. Discharge from the lake can dominate at both low and high total discharges.

Water quality in the estuary was sampled 64 times and though events were evenly distributed between sources, some months of the year were left unrepresented. Runoff from the basin dominated discharge at S-79 in 34 samples and at least one time during each month of the year. Discharge from Lake Okeechobee

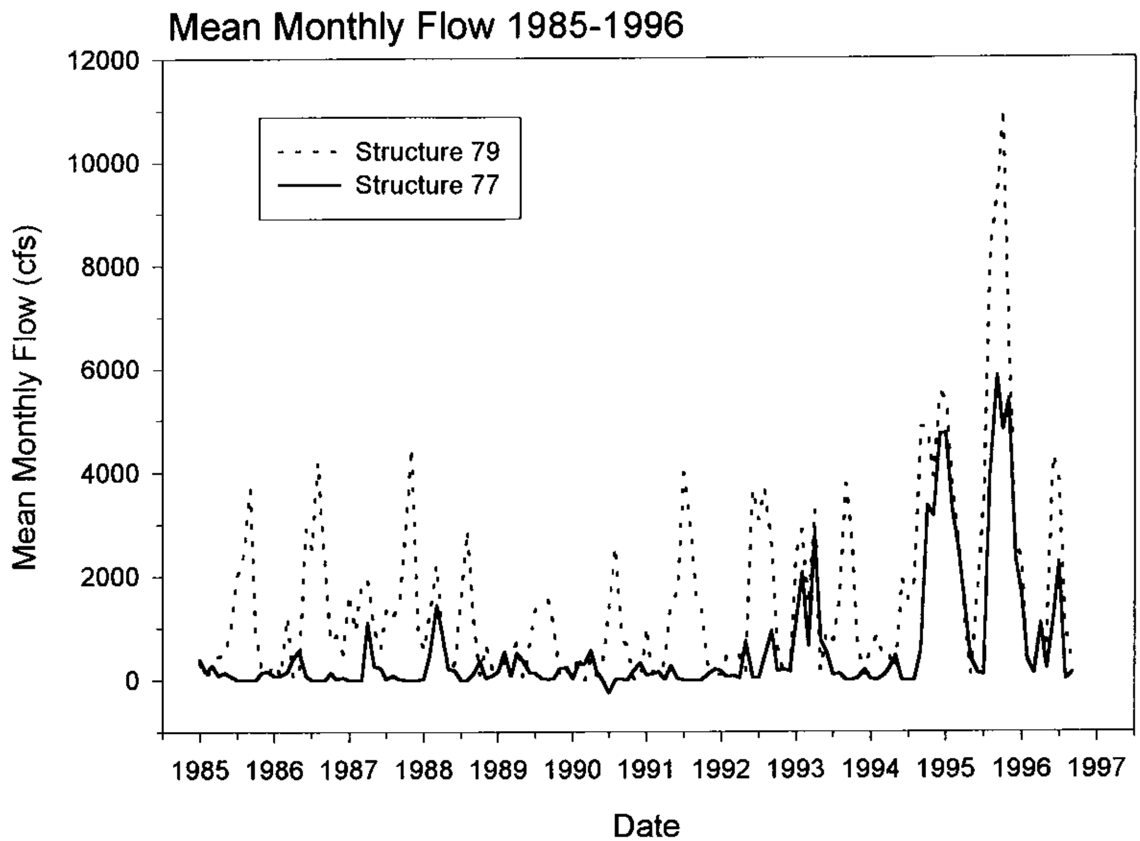

Figure 3. Average Monthly Discharge at the Franklin Lock and Dam (S-79) and at Moore Haven (S-77) from 1985 to 1996.

dominated in 30 samples. All months were represented in these samples except July, August, and September. For the basin-dominated cases, discharge at S-79 ranged from 187 cfs to 11,024 cfs (mean ± SD = 2345 ± 2460, median = 1510). When the Lake comprised the primary source, discharge ranged from 62 cfs to 9,300 cfs (mean ± SD = 1739 ± 2221, median = 571).

Water Quality at the Franklin Lock and Dam (S-79)

Between 1986 and 1996 water quality upstream of S-79 was sampled 59 times. Basin discharge comprised the major portion of total discharge in 33 samples, while discharge from the Lake dominated in 26 samples. In these samples, basin discharge dominated at least once in all months of the year except February, May, June, and December. Lake discharge dominated at least once in all months except April, July, and August.

Correlation analysis (Spearman's Rank) showed that most water quality parameters at S-79 did not vary as a function of total discharge. Only the concentrations of color and NH4 increased with increasing discharge ($p < 0.05$).

Results of the two-way ANOVA showed that the magnitude of discharge at S-79 did not affect the concentrations of TN, NO_X, or TSS (discharge effect $p > 0.05$; interaction with source $p > 0.05$). In agreement with the correlation analysis, the concentration of color increased significantly with each successive increase in discharge category (SNK test, $p < 0.05$). The concentration of NH_4 was higher at medium (1000-4000 cfs) than at lowest (< 1000 cfs) or highest (> 4000 cfs) discharges. No source x discharge interactions were detected for color and NH_4.

For both TP and DIP, the source x discharge interaction was significant. Results of the SNK test indicated that when total discharge was dominated by the basin, concentrations of TP and DIP were similar at all discharges ($p > 0.05$). When the lake dominated discharge, TP and DIP concentrations were higher at low discharge (< 1000 cfs) than at the two higher discharges (1000-4000; > 4000 cfs; Table 3).

Significant source effects also were detected. Concentrations of TN and Color were higher by 26 and 64 percent, respectively, when the river basin dominated the discharge at S-79 (Table 3). Analysis of the significant source x discharge interaction for TP and DIP showed that at the lowest discharge, concentrations were similar in the two sources. At the two higher discharges, concentrations were greater when the basin comprised most of the discharge at S-79 (Table 3). This pattern may have arisen from the previously described contrasting response of TP and DIP concentrations to discharge for the two sources.

TABLE 3. Water Quality as a Function of Source (River Basin or Lake Okeechobee) at the Franklin Lock and Dam (S-79). Arithmetic means (+ std) are shown. Statistical analyses conducted on transformed (log +1) data. Means for discharge classes shown where discharge x source interaction was significant. n = 32-33 for River Basin, n = 26 for Lake Okeechobee. *Indicates statistical difference between sources ($p < 0.05$). Letters indicate statistical difference between discharge classes when the lake was the major source.

Parameter	Discharge (cfs)	Basin	Lake
Color (cu)*	–	105 (48)	64 (20)
TN (mg/l)*	–	1.80 (0.75)	1.42 (0.40)
NO_X (mg/l)	–	0.31 (0.18)	0.25 (0.20)
NH_4 (mg/l)	–	0.04 (0.04)	0.04 (0.03)
TSS (mg/l)	–	3.6 (2.4)	3.8 (2.3)
TP (mg/l)	< 1000	0.15 (0.07)	0.12 (0.01)a
	1000-4000*	0.16 (0.05)	0.07 (0.01)b
	> 4000*	0.22 (0.12)	0.06 (0.01)b
DIP (mg/l)	< 1000	0.09 (0.04)	0.08 (0.03)a
	1000-4000*	0.11 (0.04)	0.04 (0.01)b
	> 4000*	0.12 (0.05)	0.03 (0.01)b

Water Quality in the Estuary

Correlation analysis (Table 4) indicated that every water quality parameter examined was correlated with freshwater discharge in some region of the estuary. Salinity showed significant inverse correlations in all regions (r = -0.8 to -0.9) verifying that as discharge at S-79 increases, the percentage of freshwater in any given region of the estuary also increases. Several general patterns emerged. Some parameters (color, TN, NO_X, NH_4, K) were positively correlated with discharge when the correlation was significant. By contrast TSS was negatively correlated with discharge. TP and chlorophyll *a* were negatively correlated with discharge at the head of the estuary but positively correlated farther downstream. Like TP, DIP was negatively correlated with discharge at the head of the estuary. Lastly, the effects of discharge on some parameters (color, salinity) could be detected statistically up to 59 km from S-79 in Pine Island Sound. For others (NH_4, TSS) the influence of discharge could be detected only in the upstream regions.

The three-way ANOVA yielded the following results. In no case was the three-way interaction (region x source x discharge) or the two-way, region x source, interaction statistically significant. Depending on the parameter, there were significant main effects of region, total discharge, or source and significant interactions for discharge x source and discharge x region. Further description of ANOVA results will

TABLE 4. Spearman's Rank Correlation (r) Between Mean Daily Discharge (cfs) at the Franklin Lock and Dam (S-79), Calculated for the 30 Days Prior to Sampling and Water Quality in Five Regions of the Caloosahatchee Estuary. Kilometers are distance downstream from S-79. Correlation coefficients calculated using transformed (log + 1) data. All r statistically significant ($p < 0.05$) except where noted by ns = not statistically significant. Number of observations (n) for k was 21-36.

	Head (0-14 km)	Upper Estuary (14-28 km)	Lower Estuary (28-41 km)	San Carlos Bay (41-49 km)	Pine Island Sound (59 km)
Salinity	-0.939	-0.968	-0.889	-0.901	-0.832
Color	0.844	0.902	0.880	0.776	0.449
TN	0.133 ns	0.439	0.532	0.286	-0.035 ns
NO_X	0.506	0.724	0.434	0.355	0.196 ns
NH_4	0.487	0.282	0.231 ns	0.208 ns	0.118 ns
TP	-0.538	-0.050 ns	0.251 ns	0.297	0.002 ns
DIP	-0.328	-0.061 ns	0.084 ns	0.183 ns	0.131 ns
TSS	-0.748	-0.660	-0.244	-0.111 ns	-0.047 ns
Chlorophyll *a*	-0.525	0.041 ns	0.470	0.442	0.136 ns
K	0.071 ns	0.668	0.743	0.820	0.423 ns
n	55-62	56-62	56-62	57-60	37-40

first consider regional differences (main effect of region) and effects of discharge on these differences (discharge x region interaction). Secondly, effects of source on concentrations throughout the estuarine system (main effect of source) and how source effects vary with the magnitude of discharge (discharge x source interaction) will be considered. The results for the light extinction coefficient are presented along with results for other parameters. It should be noted that the ANOVA model for K had two, rather than three, levels of total discharge.

Region and Discharge Effects

Significant discharge x region interactions were detected for all parameters except color and NH_4. For these, the main effects of region and total discharge were significant ($p < 0.05$). Color decreased steadily with distance from S-79 (head > upper estuary > lower estuary > San Carlos Bay; SNK test, $p < 0.05$) and increased with each successive increase in total discharge category (SNK test, $p < 0.05$). The concentration of NH_4 in the system was greatest at the highest discharge (> 4000 cfs) and lowest at the two lower discharge categories, which were statistically equivalent. Results of the SNK test indicated that highest concentrations occurred in San Carlos Bay and lowest in the upper estuary, with concentrations at the head and lower estuary being intermediate.

An interaction between region and total discharge may indicate that the effect of discharge on concentration is different in different regions. This possible outcome has been addressed by the correlation analysis for the cases in which concentrations increase or decrease with increasing total discharge. Alternatively, the interaction may indicate that differences between regions vary as a function of discharge. In other words, the spatial gradient of concentration varies as a function of discharge.

The effect of discharge on concentration gradients in the estuary is most clearly exemplified by salinity (Figure 4). At low discharge (< 1000 cfs) there was a distinct and progressive increase in salinity with increasing distance from S-79. As discharge increased and the salinity of the system decreased, the gradient in the head and upper estuary gradually disappeared and the salinity gradient was pushed downstream.

Most parameters behaved much like salinity. TN provides a good example (Figure 4). At low discharge, there were distinct spatial gradients of decreasing concentration with increasing distance from S-79. As discharge increased, the concentration gradient in the head and upper estuary disappeared and significant differences occurred further downstream. At the highest discharges, gradients became less distinct and even statistically non-detectable. Although inversely related to distance from S-79, TSS behaved similarly to salinity and TN (Figure 4).

By contrast, chlorophyll *a* behaved differently. At low discharge, there was a progressive decrease in concentration with distance from S-79. At intermediate discharge, chlorophyll *a* peaked in the upper estuary and decreased farther downstream. At highest discharges, concentrations were lower at the head of the estuary and higher farther downstream (Figure 4).

Source Effects

For color, TN, NO_X, NH_4, and K, ANOVA detected no significant source x discharge interaction ($p > 0.05$). Excepting NH_4, all exhibited a significant main effect of source ($p < 0.05$). When the basin dominated discharge at S-79, concentrations across the four regions were higher by 18 to 125 percent depending on the parameter (Table 5). For another group of parameters (TP, DIP, TSS, chlorophyll *a*), the source x discharge interaction was significant. For all parameters except chlorophyll *a*, the response of concentration in the estuary to increases in discharge varied according to source (Table 6). These differing responses to increases in discharge may account for differences in concentration between the two sources at specific levels of discharge. Although the results for TP and DIP must be viewed with caution, both showed source effects at the two higher discharge categories (1000-4000, > 4000). Again concentrations were higher when the major source of discharge was the basin. At intermediate discharges (1000-4000 cfs) concentrations of TSS and chlorophyll *a* were higher when the basin dominated discharge (Table 6).

Light Extinction in the Estuary

The variability in the extinction coefficient within each region was examined through stepwise multiple regression (Table 7). Throughout most of the system, color explained a significant proportion of variation in K. However, at either end of the system, other variables were more important: chlorophyll *a* in Pine Island Sound and TSS at the head of the estuary. Elsewhere, color explained over half the variation in K (Table 4).

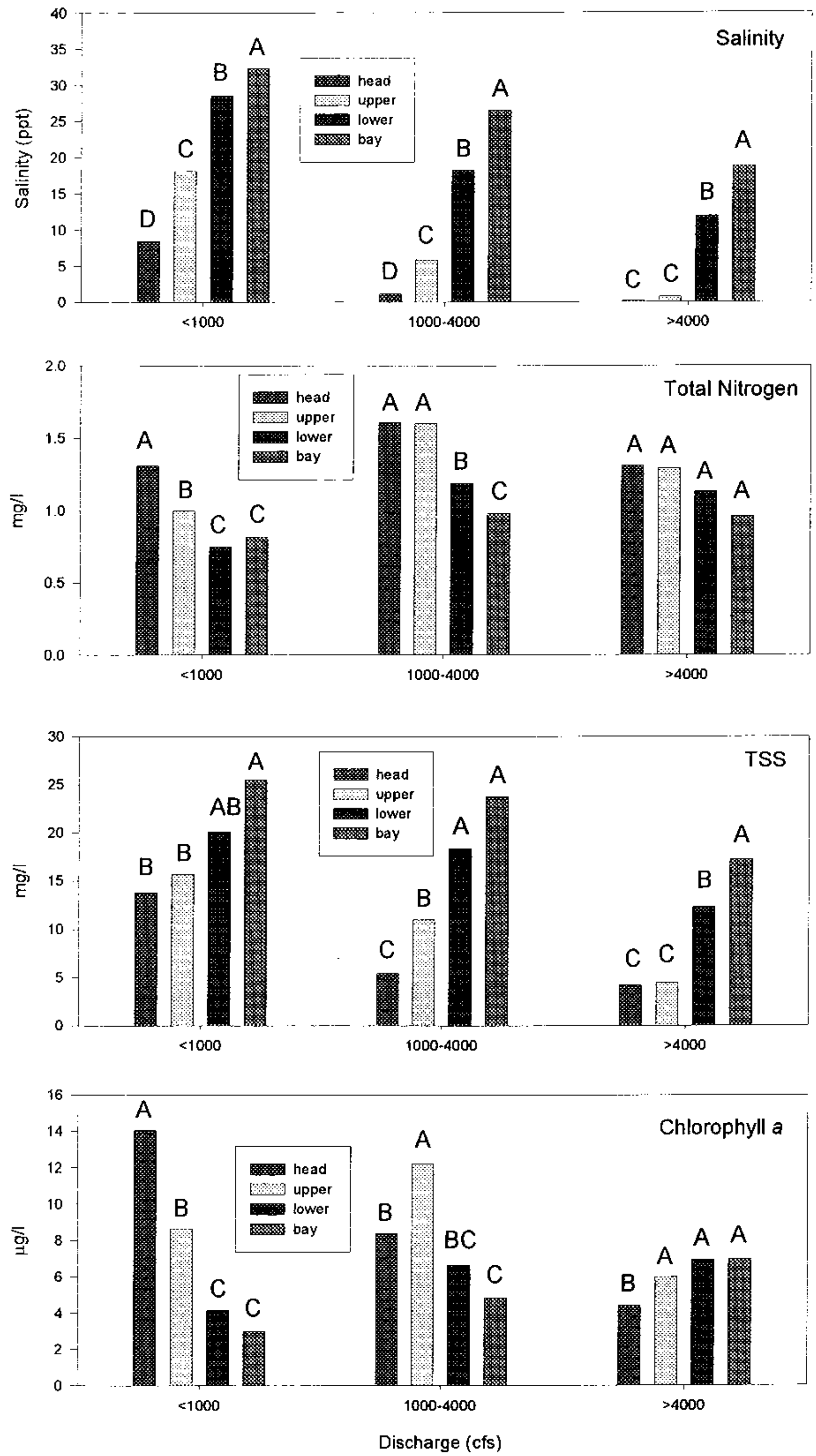

Figure 4. Effect of Freshwater Discharge at Structure S-79 on Spatial Gradients of Water Quality in the Downstream Estuary. Letters summarize results of the SNK-Test examining potential differences between regions at each level of discharge. Bars with different letters are significantly different ($p < 0.05$).

TABLE 5. Effects of Source of Freshwater (River Basin or Lake Okeechobee) Discharged at S-79 on Water Quality in the Caloosahatchee Estuary. Arithmetic means (std) are shown. Statistical analysis conducted on log (value + 1) transformed data. Parameters are those for which ANOVA showed a significant source effect and no interactions between the source effect and other main effects. *Significant difference between sources at $p < 0.05$, n = 122-130 for basin and n = 98-102 for lake.

Parameter	Basin	Lake
Salinity (ppt)	14.78 (10.91)	17.55 (11.78)
Color (cu)*	58 (45)	36 (30)
TN (mg/l)*	1.24 (0.39)	1.05 (0.46
NO_X (mg/l)*	0.09 (0.13)	0.04 (0.07)
NH_4 (mg/l)	0.03 (0.03)	0.03 (0.04)
K (/meter)*	2.78 (1.88)	1.70 (1.18)

TABLE 6. Effects of Source of Freshwater (River Basin or Lake Okeechobee) Discharged at S-79 on Water Quality in the Caloosahatchee Estuary. Arithmetic means (std) are shown. Statistical analysis conducted on log (value + 1) transformed data. Parameters are those for which ANOVA detected source effects only at specific levels of discharge. *Significant difference between sources at $p < 0.05$, letters indicate differences between discharge classes at specific levels of source, means with different letters are significantly different at $p < 0.05$. n = 16-64.

Parameter	Discharge (cfs)	Basin	Lake
TP (mg/l)	< 1000	0.10 (0.05)	0.09 (0.05)a
	1000-4000*	0.10 (0.05)	0.07 (0.03)b
	> 4000*	0.10 (0.03)	0.06 (0.02)b
DIP (mg/l)	< 1000	0.05 (0.04)b	0.05 (0.03)a
	1000-4000*	0.05 (0.04)b	0.03 (0.02)b
	> 4000*	0.07 (0.03)a	0.03 (0.01)b
TSS (mg/l)	< 1000	17.4 (8.3)a	20.1 (13.8)a
	1000-4000*	16.0 (9.3)a	12.0 (8.4)b
	> 4000	9.4 (6.9)b	9.7 (10.2)c
Chlorophyll a (μg/l)	< 1000	7.1 (5.3)	7.9 (7.9)
	1000-4000*	9.4 (7.4)	5.7 (3.4)
	> 4000	6.8 (4.2)	5.2 (2.5)

TABLE 7. Variability in the Light Extinction Coefficient. Percentage of variation explained by certain water quality parameters in stepwise multiple regressions. Independent variables were transformed (log + 1). $p < 0.05$ in all cases except where noted as ns = not statistically significant.

Region	Color	Chlorophyll *a*	TSS
Head	0.15	ns	0.45
Upper Estuary	0.65	0.07	ns
Lower Estuary	0.55	ns	ns
San Carlos Bay	0.87	0.11	ns
Pine Island Sound	ns	0.39	ns

DISCUSSION

The relationships that we observed between total freshwater discharge and water quality in the estuary likely arose (1) directly from the magnitude of freshwater discharge and its subsequent mixing with sea water in the estuary (Nixon, 1981); (2) indirectly from the effects of freshwater discharge on residence time in the estuary (Balls, 1994; Eyre and Twigg, 1997); and (3) directly from differences in source of fresh water (Balls *et al.*, 1997).

The magnitude of freshwater discharge at S-79 influenced the concentration of water quality parameters within each of five regions (correlation analysis) and the location and strength of spatial gradients within the system (ANOVA). This latter pattern most likely results from a progressive 'freshening' of the system as discharge increases. As the spatial extent of freshwater influence increases, spatial differences begin to disappear as concentrations approach those in the freshwater source.

Within regions of the system, concentrations of most parameters always increased (color, nitrogen nutrients) or always decreased (TSS) with increasing discharge, when correlations were statistically significant. These patterns also most likely arose directly from the mixing of fresh and salt water. The direction of change (positive or negative) in concentration with increasing discharge in the estuary depends on the relative concentrations in fresh and salt water. In the Caloosahatchee system nutrients and color are more concentrated in fresh water, while TSS is more concentrated in salt water (McPherson and Miller, 1990; Doering and Chamberlain, 1998). Correlations were positive when the constituent was more concentrated in fresh water (e.g., color) and negative when more concentrated in seawater (e.g., TSS).

Discharge correlations for chlorophyll *a* and TP were negative at the head of the estuary and positive further downstream. Such relationships may be explained by an internal source that accumulates at the heads of the estuary. For chlorophyll *a* the source may be in situ growth. At low flow a phytoplankton biomass maximum accumulates at the head of the estuary. As flow increases the biomass maximum is pushed further downstream (Welsh *et al.*, 1972; Doering *et al.*, 1994). Thus at the head of the estuary, chlorophyll *a* decreases with increasing discharge, while it increases with discharge further downstream. Our data are consistent with this hypothesis. The maximum chlorophyll *a* concentration encountered during a sampling event occurs farther downstream of S-79 as discharge increases (Figure 5).

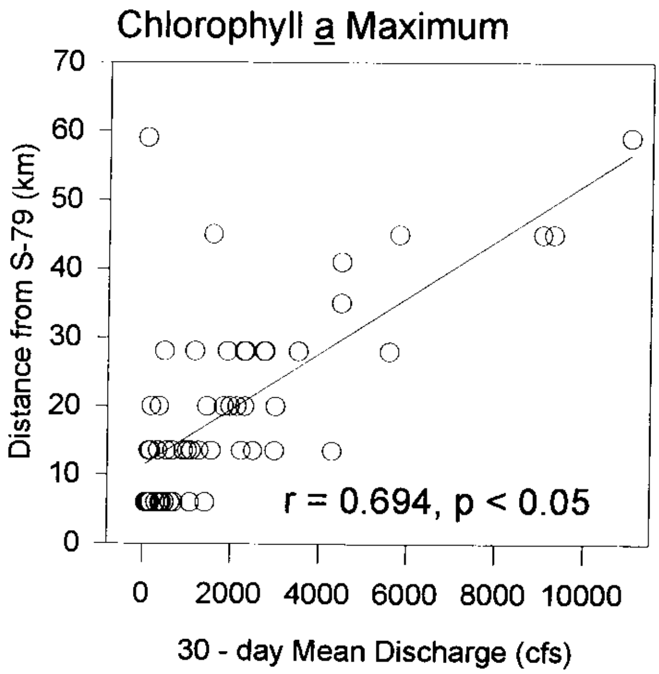

Figure 5. Location of the Chlorophyll *a* Maximum, for Each Sampling Date, as a Function of Discharge at the Franklin Lock and Dam (S-79). Distance is kilometers downstream from S-79.

The negative correlation between TP and discharge at the head of the estuary may arise from three different processes. There is a sewage treatment plant (STP) discharge between station 4 and the Ft. Myers bridges (Figure 2). STP discharge could accumulate under low flow conditions. At low discharge the concentration of TSS increases, probably by advection of seawater to the head of the estuary. This may increase the concentration of particle bound P (Froelich, 1988; Jitts, 1959). Lastly, the concentration of P may increase at low discharge owing to enhanced release of DIP from sediments, brought on by increased salinity (Caraco *et al.*, 1990; Pomeroy *et al.*, 1965). Farther downstream, the positive correlation between discharge and TP may reflect the mixing of endmembers with different concentrations. Evaluating these alternatives is beyond the scope of this investigation. The behavior of TP in the Caloosahatchee requires further examination.

Theoretical considerations have demonstrated that changes in both the magnitude and composition of freshwater discharge should be reflected in the downstream estuary (Loder and Reichard, 1981; Cifuentes *et al.*, 1990). Because the composition of the freshwater endmember itself varies as a function of discharge (Hirsch *et al.*, 1982), it has been difficult to experimentally demonstrate the extent to which compositional changes in the freshwater endmember are reflected in the downstream estuary (Balls *et al.*, 1997).

In our case, source of discharge accounted for changes in composition of fresh water at S-79 and for some parameters, differences were detected downstream. On average, fresh water from the basin was darker (high color) and had higher concentrations of TN than water from Lake Okeechobee. When discharge from the river basin dominated, both color and TN concentrations were higher throughout the estuary than when discharge was from the lake. At low discharges (< 1000 cfs), the concentrations of TP and DIP did not differ between sources at S-79 or downstream in the estuary. At higher discharges, TP and DIP were more concentrated both in the fresh water at S-79 and in the downstream estuary, when the major source of freshwater discharge was the basin. For NH_4, source had no effect on freshwater concentrations and no source effects were detected in the downstream estuary.

By contrast NO_X and TSS did not behave as expected. No source effects were detected for NO_X at S-79, but the downstream estuarine concentrations were higher when discharge was from the river basin. Despite the lack of a statistical difference in the fresh water discharged at S-79, mean NO_X concentrations varied in the expected direction being higher when the basin was the major source. TSS concentrations did not vary as a function of source at S-79 but at intermediate discharge, concentrations were higher in the estuary when discharge was from the river basin. No explanation for this behavior was readily apparent.

The results of the statistical analysis presented here clearly demonstrate that both the magnitude of freshwater discharge and the source of discharge have an important impact on water quality in the downstream estuary. In a relative sense, discharge can cause more substantial changes in estuarine water quality than can a change in source. Figure 6 illustrates these relative effects using color as an example. At any one discharge, changes in source resulted in a 30-50 percent change in the concentration of color. Depending on the source, changes in discharge, from low to intermediate or intermediate to high, resulted in concentration increases of 70 percent to over 100 percent.

Light Extinction

Although other factors could be important, color explained the bulk of variation in light extinction throughout most of the system. Water from both the lake and river is brown in color, a condition probably caused by dissolved humic material. The finding that color is an important determinant of light extinction is not unusual for coastal systems fed by highly colored rivers (Doering *et al.*, 1994; Gallegos and Kenworthy, 1996) and in part explains the dependence of K on freshwater discharge.

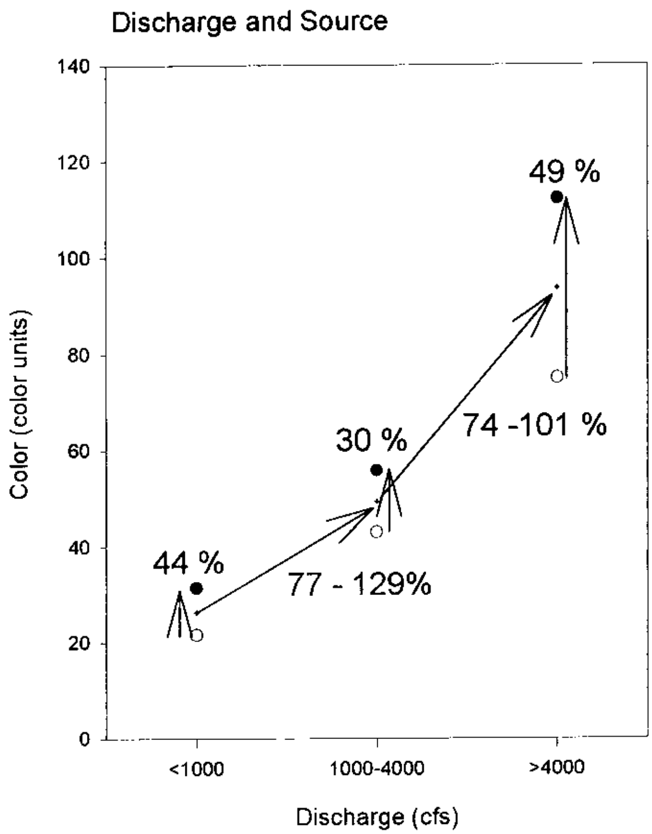

Figure 6. Relative Effects of Total Discharge and Source of Discharge on the Concentration of Color in the Caloosahatchee Estuarine System. Solid circles = basin. Open circles = lake. Vertical arrows indicate effect of changing source at specific levels of discharge. Other arrows indicate effect of changing discharge category, while source is held constant.

In shallow systems like the Caloosahatchee, light extinction can exert an important control on the growth and distribution of submerged aquatic vegetation (Carter and Rybicki, 1990; Onuf, 1996; Lee and Dunton, 1997). Beds of wild celery, *Vallisneria americana*, are often prominent in the head and upper estuarine regions, while mixed beds of turtle grass, *Thalassia testudinum*, and shoal grass, *Halodule wrightii*, occur in San Carlos Bay and Pine Island Sound. Historically, there has been a decline in seagrass coverage in Pine Island Sound and San Carlos Bay (Harris *et al.*, 1983). The loss has been primarily in deeper portions of the system (> 1.0 m) and may have resulted from reduced water clarity (Harris *et al.*, 1983).

In general, seagrass beds extend to depths at which 10-30 percent of surface irradiance penetrates (Kenworthy and Haunert, 1991). Taking the values in Table 5, this ecological compensation depth (Gallegos and Kenworthy, 1996) averages from 0.4 to 0.8 m when the river basin is the major source of freshwater. When the lake dominates discharge, the ecological compensation depth averages from 0.7 to 1.3 m. Although the increases in compensation depth associated with lake discharge are small (0.3 to 0.5 m), in a shallow system like the Caloosahatchee, small changes in depth may have disproportionately large ecological impacts.

Management Implications

Management questions concerning downstream effects of discharge have centered on both quantity and quality and particularly the effects of releases from Lake Okeechobee on these. Water quality in the downstream estuary changed as a function of total discharge, regardless of source. Nevertheless, relative to discharge from the river basin, releases from Lake Okeechobee did not detectably increase concentrations of nutrients, color, or TSS in the estuary. When effects were detected, concentrations were relatively lower and the water was relatively clearer when discharge at S-79 was dominated by water from Lake Okeechobeee. When the two sources are compared, the lower concentrations associated with discharges from Lake Okeechobee suggest that the effects of discharges from the lake are less severe on estuarine water quality than discharges from the basin. The lower concentrations originating from Lake Okeechobee may be attributable to the unique morphometry of this system. This large, shallow lake is enclosed within a dike for flood control purposes. The western edge of the lake is comprised of a large (40,000 ha), wetland marsh, which effectively filters nutrients before water leaves the lake at S-77 (cf., Steinman *et al.*, in press; Hwang *et al.*, in press). As a consequence, improving the quality of water discharged from the basin would have the greatest effect on downstream conditions in the estuary.

The water and material loads associated with controlled freshwater releases from Lake Okeechobee represent artificial, anthropogenic additions to the discharge and loads to the estuary from the Caloosahatchee River. This study has addressed effects of releases from Lake Okeechobee on water quality in the estuary. The effects of additional loading from the lake on processes (e.g., primary productivity, nutrient cycling) occurring in the downstream estuary remain to be elucidated.

ACKNOWLEDGMENTS

We thank Dan Crean, Kathy Haunert, and the staff of the Water Quality Monitoring Division of the South Florida Water Management District for their help in collection and analysis of the many samples taken over the years. We also thank Keith Donohue, George Kraemer, and Dennis Hanisak of the Harbor Branch Oceanographic Institution for their help in sample collection and scientific insight. The comments of Alan Steinman and Becky Robbins greatly improved the manuscript.

LITERATURE CITED

American Public Health Association, American Water Works Association, and Water Pollution Control Federation (APHA, AWWA, WCPA), 1985. Standard Methods for Examination of Waste Water (16th Edition). American Public Health Association, Washington, D.C.

Balls, P. W., A. Macdonald, K. B. Pugh, and A. C. Edwards, 1997. Rainfall Events and Their Influence on Nutrient Distributions in the Ythan Estuary (Scotland). Estuarine, Coastal and Shelf Science 44 (Supplement A):73-81.

Balls, P. W., 1994. Nutrient Inputs to Estuaries from Nine Scottish East Coast Rivers; Influence of Estuarine Processes on Inputs to the North Sea. Estuarine, Coastal and Shelf Science 39:329-352.

Bierman, V., 1993. Performance Report for the Caloosahatchee Estuary Salinity Modeling. South Florida Water Management District contract deliverable from Limno-Tech Inc.

Caraco, N., J. Cole, and G. E. Likens, 1990. A Comparison of Phosphorus Immobilization in Sediments of Freshwater and Coastal Marine Systems. Biogeochemistry 9:277-290.

Carter, V. and N. B. Rybicki, 1990. Light Attenuation and Submersed Macrophyte Distribution in the Tidal Potomac River and Estuary. Estuaries 13:441-452.

Chamberlain, R. and D. Hayward, 1996. Evaluation of Water Quality and Monitoring in the St. Lucie Estuary, Florida. Water Resources Bulletin 32:681-696.

Chatfield, C., 1989. The Analysis of Time Series: An Introduction (4th Edition). Chapman and Hall, New York, New York, 241 pp.

Cifuentes, L. A., L. E. Schemel, and J. H. Sharp, 1990. Qualitative and Numerical Analyses of the Effects of River Inflow Variations on Mixing Diagrams in Estuaries. Estuarine, Coastal and Shelf Science 30:411-427.

Correll, D. L., T. E. Jordan, and D. E. Weller, 1992. Nutrient Flux in a Landscape: Effects of Coastal Land Use and Terrestrial Community Mosaic on Nutrient Transport to Coastal Waters. Estuaries 15:431-442.

Doering, P. H. and R. H. Chamberlain, 1998. Water Quality in the Caloosahatchee Estuary, San Carlos Bay and Pine Island Sound. *In:* Proceedings of the Charlotte Harbor Public Conference and Technical Symposium, March 15-16, 1997, Punta Gorda, Florida, pp. 229-240. Charlotte Harbor National Estuary Program Technical Report No. 98-02, West Palm Beach, Florida, South Florida Water Management District, 274 pp.

Doering, P. H., C. A. Oviatt, J. H. McKenna, and L. W. Reed, 1994. Mixing Behavior of Dissolved Organic Carbon and Its Potential Biological Significance in the Pawcatuck River Estuary. Estuaries 17(3):521-536.

Eyre, B. and C. Twigg, 1997. Nutrient Behaviour During Post-Flood Recovery of the Richmond River Estuary Northern NSW, Australia. Estuarine, Coastal and Shelf Science 44:311-326.

Flaig, E. G. and J. Capece, 1998. Water Use and Runoff in the Caloosahatchee Watershed. *In:* Proceedings of the Charlotte Harbor Public Conference and Technical Symposium; March 15-16, 1997, Punta Gorda, Florida, pp. 73-80. Charlotte Harbor National Estuary Program Technical Report No. 98-02, West Palm Beach, Florida, South Florida Water Management District, 274 pp.

Froelich, P. N., 1988. Kinetic Control of Dissolved Phosphate in Natural Rivers and Estuaries: A Primer on the Phosphate Buffer Mechanism. Limnol. and Oceanogr. 33:649-668.

Gallegos, C. L. and W. J. Kenworthy, 1996. Seagrass Depth Limits in the Indian River Lagoon (Florida, U.S.A.): Application of an Optical Water Quality Model. Estuarine, Coastal and Shelf Science 42:267-288.

Harris, B. A., K. D. Haddad, K. A. Steidinger, and J. A. Huff, 1983. Assessment of Fisheries Habitat: Charlotte Harbor and Lake Worth, Florida. Final Report, Florida Dept. of Natural Resources, Bureau of Marine Research, Marine Research Laboratories, St. Petersburg, Florida, 211 pp.

Hirsch, R. M., J. R. Slack, and R. A. Smith, 1982. Techniques of Trend Analysis for Monthly Water Quality Data. Water Resources Research 18:107-121.

Hopkinson, C. S., Jr. and J. J. Vallino, 1995. The Relationships Among Man's Activities in Watersheds and Estuaries: A Model of Runoff Effects on Patterns of Estuarine Community Metabolism. Estuaries 18:598-621.

Hwang, S.-J., K. E. Havens, and A. D. Steinman (in press). Phosphorus Kinetics of Planktonic and Benthis Assemblages in a Shallow Subtropical Lake. Freshwater Biology.

Jitts, H. R., 1959. The Adsorption of Phosphate by Estuarine Bottom Deposits. Aust. J. Mar. Freshwater Res. 10:7-21.

Kenworthy, W. J. and D. E. Haunert, 1991. The Light Requirements of Seagrasses. (Proceedings of a workshop convened to examine the capability of water quality criteria, standards and monitoring programs to protect seagrasses.) NOAA Technical Memorandum NMFS-SEFC-287.

Keppel, G., 1973. Design and Analysis: A Researcher's Handbook. Prentice-Hall, Inc. Englewood Cliffs, New Jersey, 658 pp.

Lee, K-S. and K. H. Dunton, 1997. Effects of In-Situ Light Reduction on the Maintenance, Growth and Partitioning of Carbon Resources in Thalassia Testudinum Banks Ex Konig. Journal of Experimental Marine Biology and Ecology 210:53-73

Loder, T. C. and R. P. Reichard, 1981. The Dynamics of Conservative Mixing in Estuaries. Estuaries 4:64-69.

McPherson, B. F. and R. L. Miller, 1990. Nutrient Distribution and Variability in the Charlotte Harbor Estaurine System, Florida. Water Resources Bulletin 26:67-80.

Naiman, R. J., J. J. Magnuson, D. M. Mcknight, J. A. Stanford, and J. R. Karr, 1995. Freshwater Ecosystems and Their Management: A National Initiative. Science 270:584-585.

Nixon, S. W., 1981. Freshwater Inputs and Estuarine Productivity. *In:* Proceedings of the National Symposium on Freshwater Inflow to Estuaries. U.S. Fish and Wildlife Service, Biological Services Program, FWS/OBS-81-04, Vol. 1, pp. 31-57.

Onuf, C. P., 1996. Seagrass Responses to Long-Term Light Reduction by Brown Tide in Upper Laguna Madre, Texas: Distribution and Biomass Patterns. Marine Ecology Progress Series 138:219-231.

Pomeroy, L. R., E. E. Smith, and C. M.Grant, 1965. The Exchange of Phosphate Between Estuarine Water and Sediments. Limnol. and Oceanogr. 10:167-172.

SAS Institute Inc., 1989. SAS/STAT User's Guide, Version 6, Vol. I and Vol. II. SAS Institute Inc., Cary, North Carolina, 1686 pp.

Snedecor, G. W. and W. G. Cochran, 1967. Statistical Methods. Iowa State University Press, Ames, Iowa, 593 pp.

Sokal, R. R. and F. J. Rohlf, 1981. Biometry: The Principles and Practice of Statistics in Biological Research. W. H. Freeman and Co., San Francisco, California, 859 pp.

South Florida Water Management District, 1994. SFWMD Comprehensive Quality Assurance Plan. DRE Inventory Control No. 318.

Steinman, A. D., K. E. Havens, N. G. Aumen, R. T. James, K. R. Jin, J. Zhang, and B. Rosen (in press). Phosphorus in Lake Okeechobee: Sources, Sinks, and Strategies. *In:* Phosphorus Biogeochemistry of Subtropical Ecosystems: Florida as a Case Example, K. R. Reddy, G. A. O'Connor, and, C. L. Schelske (Editors). CRC/Lewis Publishers, New York, New York.

Stoker, Y. E., 1992. Salinity Distribution and Variation with Freshwater Inflow and Tide, and Potential Changes in Salinity Due to Altered Freshwater Inflow in the Charlotte Harbor Estuarine System, Florida. U.S. Geological Survey, Water-Resources Investigations Report 92-4062, 30 pp.

Welsh, E. B., R. M. Emery, R. I. Matsuda, and W. A. Dawson, 1972. The Relation of Periphytic and Planktonic Algal Growth in an Estuary to Hydrographic Factors. Limnol. Oceanogr. 17:731-737.

Winer, B. J., 1971. Statistical Principles in Experimental Design. McGraw-Hill, New York, New York, 907 pp.