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# Seasonal and Long-term Trends in the Water Quality of Florida Bay (1989–1997)

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**ABSTRACT:** Analysis of 6 yr of monthly water quality data was performed on three distinct zones of Florida Bay: the eastern bay, central bay, and western bay. Each zone was analyzed for trends at intra-annual (seasonal), interannual (oscillation), and long-term (monotonic) scales. The variables TON, TOC, temperature, and TN : TP ratio had seasonal maxima in the summer rainy season; APA and Chl *a*, indicators of the size and activity of the microplankton tended to have maxima in the fall. In contrast, $NO_3^-$, $NO_2^-$, $NH_4^+$, turbidity, and $DO_{sat}$, were highest in the winter dry season. There were large changes in some of the water quality variables of Florida Bay over the study period. Salinity and TP concentrations declined baywide while turbidity increased dramatically. Salinity declined in the eastern, central, and western Florida Bay by 13.6‰, 11.6‰, and 5.6‰, respectively. Some of the decrease in the eastern bay could be accounted for by increased freshwater flows from the Everglades. In contrast to most other estuarine systems, increased runoff may have been partially responsible for the decrease in TP concentrations as input concentrations were 0.3–0.5 μM. Turbidity in the eastern bay increased twofold from 1991 to 1996, while in the central and western bays it increased by factors of 20 and 4, respectively. Chl *a* concentrations were particularly dynamic and spatially heterogeneous. In the eastern bay, which makes up roughly half of the surface area of Florida Bay, Chl *a* declined by 0.9 μg $l^{-1}$ (63%). The hydrographically isolated central bay zone underwent a fivefold increase in phytoplankton biomass from 1989 to 1994, then rapidly declined to previous levels by 1996. In western Florida Bay there was a significant increase in Chl *a*, yet median concentrations of Chl *a* in the water column remained modest (~2 μg $l^{-1}$) by most estuarine standards. Only in the central bay did the DIN pool increase substantially (threefold to sixfold). Notably, these changes in turbidity and phytoplankton biomass occurred after the poorly-understood seagrass die-off in 1987. It is likely the death and decomposition of large amounts of seagrass biomass can at least partially explain some of the changes in water quality of Florida Bay, but the connections are temporally disjoint and the processes indirect and not well understood.

## Introduction

One of the primary purposes for conducting long-term monitoring projects is to be able to detect trends in the measured variables over time. These programs are usually initiated as a response to public perception (and possibly some scientific data) that "the river-bay-prairie-forest-etc. is dying." In the case of Florida Bay, the major impetus was the combination of a seagrass die-off (Robblee et al. 1991), increased phytoplankton abundance (Phlips and Badylak 1996), sponge mortality (Butler et al. 1995), and a perceived decline in fisheries (Tabb and Roessler 1989; Tilmant 1989) beginning in 1987. A network of water quality monitoring stations was established in 1989 to explicate spatial patterns and temporal trends in water quality in an effort to elucidate mechanisms behind the recent ecological change.

A spatial analysis of 6 yr of data from our monitoring program resulted in the delineation of three subsets of stations within which there are robust similarities in water quality (Fig. 1; Boyer et al. 1997). We have argued that these spatially contiguous groups of stations are the result of similar loading and processing of materials, hence we call these areas zones of similar influence (ZSI). The eastern bay ZSI acts most like a conventional estuary in that it has a quasi-longitudinal salinity gradient caused by the interaction of clearly definable seawater and freshwater end-members. The central bay is a hydrographically isolated area with low and

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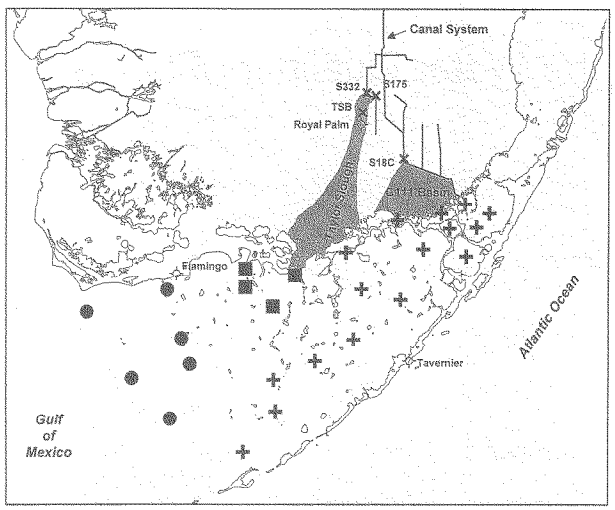


Fig. 1. Map of zones of similar influence (ZSI) as a result of cluster analysis on mean and SD of five principal component scores. Stations are labelled as eastern bay (+), central bay (■), and western bay (●).

infrequent terrestrial freshwater input, long water-residence time, and high evaporative potential. The western bay ZSI is the most tidally influenced by the Gulf of Mexico and is most isolated from direct overland freshwater sources. We present time series analyses of monthly water quality data from these three composite ZSI of Florida Bay.

## Methods

### Site Characteristics

Florida Bay is a wedge-shaped, shallow lagoonal estuary located off the southern tip of the Florida peninsula. It is bounded by the Everglades to the north and open to the Gulf of Mexico along its western margin. The main line of the Florida Keys, a Pleistocene reef, separates Florida Bay from the Atlantic Ocean. The sediments of Florida Bay are composed mostly of biogenic carbonate muds (Bosence 1989) so weathering of pre-existing substratum is not an important source of nutrients. Florida Bay is divided into relatively discrete basins by shallow mud banks, which restrict water mixing between basins and attenuate both tidal range and current. Florida Bay has a tropical savanna climate characterized by a long dry season and a wet season (Hela 1952). The dry season lasts from November to April, and generates only 18–33% of the annual total rainfall. The remainder of the annual precipitation falls during the May–October wet season (Schomer and Drew 1982). Previous analyses of Florida Bay water quality have shown that Florida Bay is a P-limited marine ecosystem that experiences periodic events of hypersalinity (Fourqurean et al. 1993).

Freshwater flow in the Everglades ecosystem and therefore into Florida Bay is strongly manipulated by human activities (Light and Dineen 1994). Streamflow into Florida Bay is largely confined to two drainage areas: Taylor Slough and the C-111 basin. Taylor Slough is a natural watershed and flow route that historically drained the extreme southeastern Everglades. Water levels in Taylor Slough are modulated by the S175 control structure and the S332 pump station (Fig. 1). Flow in Taylor Slough is gaged downstream at the Taylor Slough Bridge (TSB). The C-111 basin receives water from the controlled canal and levy system that regulates water levels in extreme south Florida. Freshwater flow into the C-111 basin is controlled at the S18C structure. Both the Taylor Slough Bridge and S18C are considerably upstream from the interface with Florida Bay, hence reported discharges should only be considered as rough approximations of freshwater inputs to Florida bay. See Fourqurean et al. (1999) for a more detailed description of the Florida Bay ecosystem.

### Sample Collection and Analysis

A total of 28 sampling sites were distributed throughout Florida Bay (Fig. 1). Stations were sampled roughly every other month from July 1989 to December 1990 and then monthly from March 1991 to the present. Two consecutive days were required to complete each sampling event.

Surface salinity and temperature were measured using a combination salinity-conductivity-temperature probe (Orion model 140). Dissolved oxygen (DO, mg $l^{-1}$) was measured 10 cm below the surface using an oxygen electrode (Orion model 840) corrected for salinity and temperature. DO was converted to percent saturation ($DO_{sat}$) using the equations of García and Gordon (1992).

For analysis of dissolved nutrients, duplicate water samples were collected 10 cm below the surface using sample-rinsed 150-ml syringes. Samples were filtered (25-mm Whatman GF/F) by hand into sample-rinsed 60-ml HDPE bottles, which were capped and immediately placed on ice in the dark for transport. The wet filters, for chlorophyll *a* (Chl *a*) analysis, were placed in 1.8-ml plastic centrifuge tubes to which 1.5 ml of 90% acetone was added (Strickland and Parsons 1972); tubes were capped and put into a dark bottle on ice for transport. Duplicate, unfiltered water samples were collected using sample-rinsed 150-ml HDPE bottles; these were kept at ambient temperature in the dark during transport.

Unfiltered water samples were analyzed for total organic carbon (TOC), total nitrogen (TN), total phosphorus (TP), alkaline phosphatase activity (APA), and turbidity. TOC was measured by direct

injection onto hot platinum catalyst in a Shimadzu TOC-5000 after first acidifying to pH < 2 and purging with $CO_2$-free air. TN was measured using an ANTEK 7000N Nitrogen Analyzer using $O_2$ as carrier gas to promote complete recovery of the nitrogen in the water samples (Frankovich and Jones 1998). TP was determined using a dry-ashing, acid-hydrolysis technique (Solórzano and Sharp 1980). The APA assay measures the activity of alkaline phosphatase, an enzyme produced by bacteria and algae that cleaves phosphate from organic compounds (Boyer et al. 1997). Turbidity was measured using an HF Scientific model DRT-15C turbidimeter and reported in NTU.

Filtrates were analyzed for nitrate + nitrite ($NO_x^-$), nitrite ($NO_2^-$), ammonium ($NH_4^+$), and soluble reactive phosphorus (SRP) on a four-channel autoanalyzer (Alpkem model RFA 300). Filters for Chl *a* content ($\mu g\ l^{-1}$) were allowed to extract for a minimum of 3 d at −20°C before analysis. Extracts were analyzed using a Gilford Fluoro IV Spectrofluorometer (excitation = 435 nm, emission = 667 nm) with comparison to Chl *a* standards (Sigma).

Some variables were not measured directly but were calculated by difference. Nitrate ($NO_3^-$) was calculated as $NO_X^- - NO_2^-$. Total dissolved inorganic nitrogen (DIN) was calculated as $NO_X^- + NH_4^+$. Total organic nitrogen (TON) was calculated as TN − DIN. All concentrations are reported as μM unless otherwise noted. The TN:TP and DIN:SRP ratios were calculated on a molar basis.

### Hydrologic Data

Freshwater flow data ($m^3\ d^{-1}$) from 1984 to 1996 was collected at gaging stations (S175, S332, TSB, and S18C) in Everglades National Park (ENP) that are operated by the South Florida Water Management District and United States Geological Survey. Flow data were summed to get monthly and annual flow ($10^6\ m^3\ yr^{-1}$). Precipitation data ($cm\ d^{-1}$) at the Royal Palm, Flamingo, and Tavernier monitoring sites (Fig. 1) were also obtained and summed to get monthly and annual inputs in $cm\ yr^{-1}$. We did not have any volume estimates of groundwater inputs to Florida Bay at this time.

### Statistical Analysis of Water Quality Data

Prior to analysis, stations were grouped into distinct spatial ZSI by a statistical procedure outlined in Boyer et al. (1997). Principal component analysis (PCA) was used to extract composite variables (principal components) from the original data. The PCA solution was rotated (using VARIMAX) in order to facilitate the interpretation of the principal components and the factor scores saved for each data record. Mean and SD of factor scores were used in a k-means cluster analysis to aggregate stations into distinct ZSI: eastern bay—19 stations, central bay—four stations, and western bay—six stations (Fig. 1). The station located at the mouth of the C-111 Canal was not included in any of these ZSI because of its unique water quality characteristics (Boyer et al. 1997).

Four different graphical and statistical analyses were performed on the data in an effort to both visualize and quantify trends in each measured water quality variable at different temporal scales. The first analysis, at the seasonal scale, was a graphical depiction of 6 yr of data for each variable grouped by month using box-and-whisker plots. The box-and-whisker plot depicts the distribution of data around the median (as quartiles and range) as well as the 95% confidence interval of the median allowing it to be used as a graphical, nonparametric ANOVA (McGill et al. 1978). Second, the significance of these seasonal trends was also tested using the Kruskal-Wallis test (nonparametric ANOVA) on the data grouped by month. The third approach was to calculate a centered, 12-mo moving average of each variable over the period of record. For this test the water quality variable presented was a composite (median) of all the stations included in that particular ZSI. The moving average acts as a low-pass filter to disclose interannual oscillations (Chatfield 1989) and allows visual assessment of the linearity of any long-term trends. Finally, the seasonal Kendall-$\tau$ analysis was performed on each composite variable to test for monotonic trend (Hirsch et al. 1991). The seasonal Kendall-$\tau$ test is a nonparametric regression analysis which determines: direction of trend (+ or −), goodness of fit ($\tau$), statistical significance of fit (p) not accounting for serial correlation, and trend slope estimate over the period of record (TSE in units $yr^{-1}$). The seasonal Kendall-$\tau$ test requires the dataset to be contiguous, therefore our data prior to 1991 were excluded from this analysis. Reversals in trend direction, such as might be seen in the case of interannual oscillations, cannot be detected by this analysis. Statistical significance for both the Kruskal-Wallis and seasonal Kendall-$\tau$ test was set at $p < 0.10$.

## Results

Our previous work has shown that many of the water quality variables measured in Florida Bay are correlated (Fourqurean et al. 1993; Boyer et al. 1997). These groupings, as determined by PCA, were variables related to 1) the organic component (TOC, TON, and APA); 2) the dissolved inorganic nitrogen (DIN) component ($NO_2^-$, $NO_3^-$, and $NH_4^+$); 3) the phytoplankton component (turbidity, TP and Chl *a*); and 4) the component of tem-

TABLE 1. Median values of all parameters for the period of record are shown for each zone. Results of seasonal Kendall-τ test are reported as median, τ value, significance of trend (p), and trend slope estimate (TSE) in units $yr^{-1}$. Boldface indicates results significant at $p < 0.1$.

| Parameter | Eastern Bay | | | | Central Bay | | | | Western Bay | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Median | τ | p | TSE | Median | τ | p | TSE | Median | τ | p | TSE |
| TON (μM) | 46.0 | −0.21 | 0.253 | −0.79 | 80.8 | −0.24 | 0.187 | −4.21 | 30.5 | 0.12 | 0.538 | 0.39 |
| APA (μM $hr^{-1}$) | **0.40** | **−0.56** | **0.002** | **−0.05** | 1.95 | −0.18 | 0.333 | −0.13 | **0.24** | **−0.39** | **0.028** | **−0.04** |
| TOC (μM) | 695.9 | −0.21 | 0.253 | −11.52 | 1,323.0 | −0.12 | 0.538 | −32.86 | 439.9 | 0.03 | 0.930 | 4.76 |
| $NO_2^-$ (μM) | 0.23 | −0.06 | 0.790 | −0.01 | **0.15** | **0.77** | **0.000** | **0.06** | 0.11 | 0.05 | 0.860 | 0.00 |
| $NO_3^-$ (μM) | 0.71 | 0.14 | 0.480 | 0.01 | **0.26** | **0.62** | **0.000** | **0.11** | 0.12 | −0.03 | 0.929 | 0.00 |
| $NH_4^+$ (μM) | 3.41 | −0.29 | 0.112 | −0.32 | **7.27** | **0.42** | **0.018** | **1.56** | 1.05 | −0.11 | 0.596 | −0.07 |
| Turbidity (NTU) | **2.84** | **0.35** | **0.052** | **0.22** | **8.56** | **0.64** | **0.000** | **2.45** | **7.18** | **0.61** | **0.001** | **1.32** |
| TP (μM) | **0.25** | **0.56** | **0.001** | **−0.03** | **0.65** | **−0.36** | **0.042** | **−0.04** | **0.58** | **−0.33** | **0.065** | **−0.04** |
| Chl *a* (μg $l^{-1}$) | **0.85** | **−0.74** | **0.000** | **−0.14** | 2.34 | −0.03 | 0.930 | −0.01 | **1.93** | **0.33** | **0.065** | **0.12** |
| $DO_{sat}$ (%) | 92.2 | 0.03 | 0.930 | 0.05 | 86.3 | 0.12 | 0.538 | 1.00 | 88.8 | 0.18 | 0.333 | 0.57 |
| Temp. (°C) | 25.8 | 0.12 | 0.535 | 0.10 | 25.6 | 0.08 | 0.724 | 0.07 | 25.6 | −0.06 | 0.792 | −0.09 |
| Salinity (ppt) | **28.1** | **−0.56** | **0.002** | **−2.27** | **34.1** | **0.42** | **0.017** | **−1.94** | **35.2** | **−0.39** | **0.028** | **−0.93** |
| SRP (μM) | **0.03** | **−0.27** | **0.100** | **0.00** | **0.05** | **−0.52** | **0.003** | **−0.01** | **0.03** | **−0.53** | **0.002** | **−0.01** |
| TN:TP | **184.4** | **0.52** | **0.004** | **13.0** | 131.9 | 0.18 | 0.333 | 7.0 | 55.6 | 0.21 | 0.253 | 2.4 |
| DIN:SRP | 152.5 | 0.15 | 0.429 | 5.98 | **120.7** | **0.36** | **0.043** | **47.9** | 51.8 | 0.23 | 0.165 | 6.4 |

perature and $DO_{sat}$. Salinity and SRP are discussed individually, as they were not correlated with any of the other measured variables. We present results for water quality variables as grouped by previous PCA; the variables are given in descending order

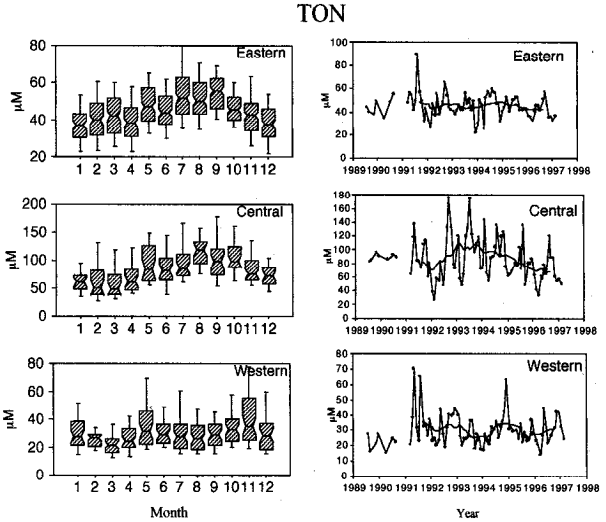


Fig. 2. Box-and-whisker and time series plots of TON (μM) by zones of similar influence (ZSI). Box-and-whisker plots include all monthly data for 19 stations in the eastern bay, four stations in the central bay, and six stations in the western bay for 1991–1997. The center horizontal line in the box is the median of the data, the top and bottom of the box are the 25th and 75th percentiles (quartiles), and the ends of the whiskers are the 5th and 95th percentiles. The notch in the box is the 95% confidence interval of the median. Outliers were omitted from all graphs to reduce compression. For time series plots, the open circles are medians of all stations in each ZSI for each month sampled. The heavy smoothed line is the centered 12-mo moving average of the data. The closed circles are discontinuous data collected prior to March 1991, These data were not included in the analyses; they are reported only for comparative purposes.

the amount of variance explained in the original dataset.

## Organic Component: TON, APA, and TOC

The median concentration of TON in the central bay (80.8 μM) was double that of the eastern and western bays (Table 1). There were significant differences in TON concentrations when the data were grouped by month in both the eastern bay ($p < 0.001$) and central bay ($p = 0.001$), with maximum TON concentrations occurring in the summer (wet season) months (Fig. 2). The western bay did not display a significant seasonal cycle in TON ($p = 0.21$). The moving average for TON showed no evidence of interannual fluctuation in the eastern bay (Fig. 2), but TON in the central bay showed an increase during 1993–1994 followed by a decline. The moving average for the western bay showed some evidence of interannual oscillation on a 2-yr period. The Kendall-τ test showed no significant trend in TON for any of the ZSI (Table 1).

Median APA in the central bay was five times greater than the eastern bay and eight times greater than the western bay (Table 1). APA showed seasonal variation in all three zones. The APA in the eastern bay showed a modest increase in the fall months ($p = 0.013$, Fig. 3) while the central bay was highly variable with lowest values in March–May ($p = 0.003$). In the western bay, the seasonal pattern in APA showed a dramatic increase from October to January ($p = 0.0001$). No interannual oscillation in APA was evident in the eastern bay (Fig. 3). In the central bay the moving average increased linearly until 1994 when it began to decline. The Kendall-τ of Eastern Bay was negative and indicated a significant decreasing trend

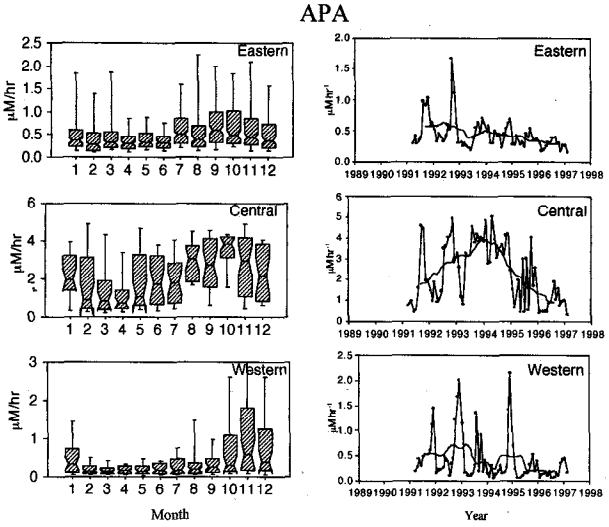


Fig. 3. Box-and-whisker and time series plots of APA (μM $h^{-1}$) by zones of similar influence (ZSI).

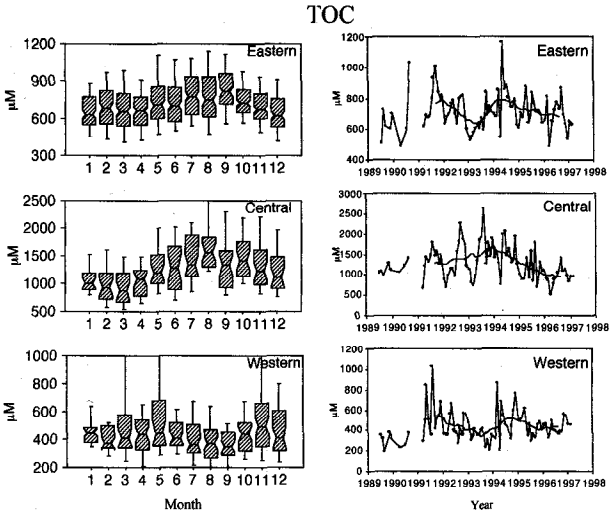


Fig. 4. Box-and-whisker and time series plots of TOC (μM) by zones of similar influence (ZSI).

in APA of 0.05 μM $h^{-1}$ $yr^{-1}$ (Table 1). No monotonic long-term trend was detected in the central bay because APA rose through 1991–1993 and then declined in 1993–1997. Some interannual oscillation in APA was seen in the western bay (Fig. 3), but overall activity declined by 0.04 μM $h^{-1}$ $yr^{-1}$ (Table 1).

Median TOC in the central bay (1323 μM) was more than twice as high as that of the eastern and western bays (Table 1). There were no significant seasonal trends in TOC concentration in the eastern and western bays (Fig. 4), but TOC in the central bay was higher during May–October ($p = 0.02$). The moving average of TOC in the eastern and western bays showed a dip in 1993, with recovery by 1994 (Fig. 4). TOC in the central bay increased over the period of record until mid 1994 when it also began to decline (Fig. 4). No significant trends in TOC for Florida Bay were evident (Table 1).

## DIN Component: $NO_2^-$, $NO_3^-$, and $NH_4^+$

Median $NO_2$ concentration in the eastern bay (0.23 μM) was almost twice as large as the central and western bays (Table 1). The seasonal pattern for $NO_2^-$ for all ZSI was similar; $NO_2^-$ concentration was highest during the cooler, dry season and lowest during the wet, hot season ($p < 0.0001$ for all, Fig. 5). The moving average for the eastern bay showed some evidence of a 2 yr oscillation (Fig. 5) with no significant monotonic long-term trend (Table 1). The moving average for the central bay showed strong evidence of increasing $NO_2^-$ concentration as well as increasing range (Fig. 5); the Kendall-$\tau$ confirmed an increase of 0.06 μM $yr^{-1}$. There was no long-term trend in $NO_2^-$ concentration in the western bay.

Median $NO_3^-$ concentration in the eastern bay was threefold and sixfold higher than the central bay and western bay, respectively (Table 1). Seasonal patterns for $NO_3^-$ among ZSI were similar to those for $NO_2^-$ ($p < 0.0001$ for all, Fig. 6), with declines during the period from May to September. The moving average for the eastern bay showed some evidence of an oscillation (Fig. 6), but no long-term trend was noted (Table 1). The moving average in the central bay showed evidence similar to $NO_2^-$ of increasing $NO_3^-$ concentration and range (Fig. 6). The Kendall-$\tau$ confirmed an increase of 0.11 μM $yr^{-1}$ (Table 1). There was no

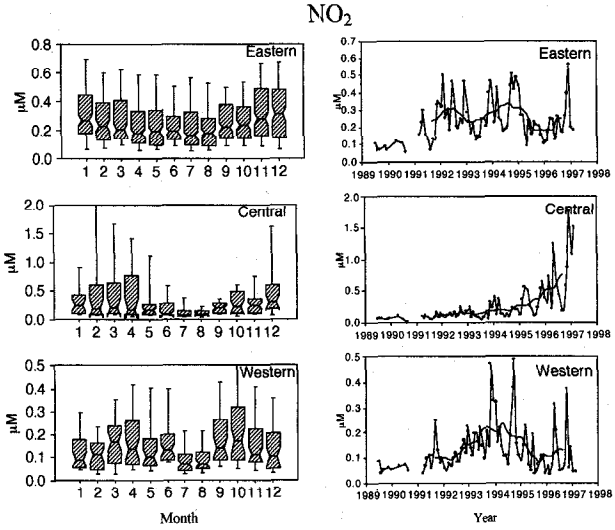


Fig. 5. Box-and-whisker and time series plots of $NO_2^-$ (μM) by zones of similar influence (ZSI).

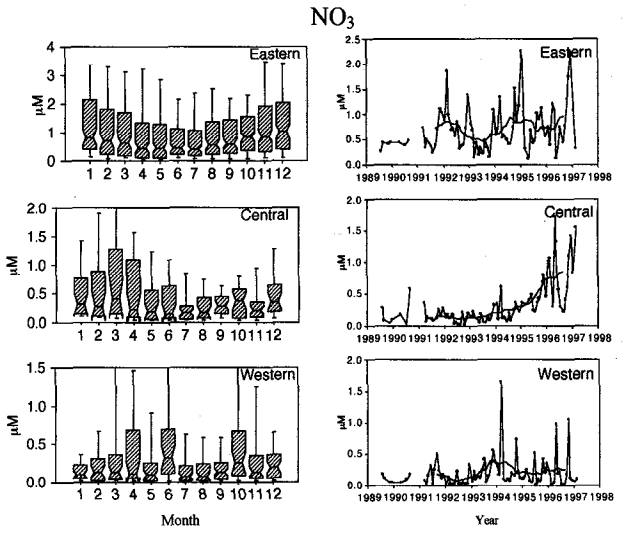


Fig. 6. Box-and-whisker and time series plots of $NO_3^-$ (μM) by zones of similar influence (ZSI).

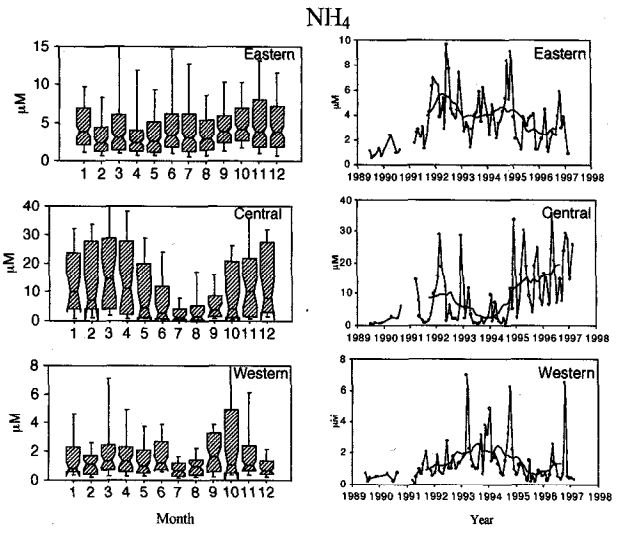


Fig. 7. Box-and-whisker and time series plots of $NH_4^+$ (μM) by zones of similar influence (ZSI).

long-term trend in $NO_3^-$ concentration in the western bay.

The central bay had very high peaks in $NH_4^+$ concentration, up to 120 μM. The median was 7.3 μM, which was twofold that of the eastern bay and seven times higher than the western bay (Table 1). There were no seasonal patterns for $NH_4^+$ in the eastern and western bays, but the central bay showed a marked decline in concentration during May–September (Fig. 7). The moving average for eastern bay showed some evidence of a 2-yr oscillating decline (Fig. 7), but the long-term downward trend was not significant ($p = 0.29$). Concentration of $NH_4^+$ in the central bay rose by 1.6 μM $yr^{-1}$ (Fig. 7 and Table 1). The western bay showed an increase in concentrations during 1993–1994, and later declined to earlier levels. For this reason, there was no significant long-term trend.

## Phytoplankton Component: Turbidity, TP, and Chl *a*

Highest median turbidity was found in both the central and western bays (8.6 NTU and 7.2 NTU, respectively) and lowest in the eastern bay (2.8 NTU). Turbidity was strongly seasonal across all of Florida Bay ($p < 0.0001$), with maximum turbidities observed during the winter-spring when winds speeds are highest and generally from the northwest (Fig. 8). Turbidity increased in all zones during the period of record (Fig. 8), with the greatest increase in the central bay (2.4 NTU $yr^{-1}$) followed by the western bay (1.3 NTU $yr^{-1}$) and eastern bay (0.2 NTU $yr^{-1}$; Table 1).

Median TP concentration was lower in the eastern bay (0.25 μM) than the central and western zones (Table 1). All zones of Florida Bay showed no seasonal signal in TP concentration and little interannual fluctuation (Fig. 9). TP declined significantly in the eastern, central, and western bays at rates of 0.03–0.04 μM $yr^{-1}$ (Table 1).

Chl *a* concentrations in the eastern bay were remarkably consistent (and low) over the annual cycle of wet and dry periods (Fig. 10), with the median being 0.96 μg $l^{-1}$. Chl *a* dropped precipitously in June 1994 and remained low for the remainder of the sampling period (Fig. 10). This type of discontinuity influenced the Kendall-τ test by overestimating the TSE (Table 1); however, it is clear that Chl *a* concentrations have declined over this time period. Concentrations of Chl *a* in the central bay

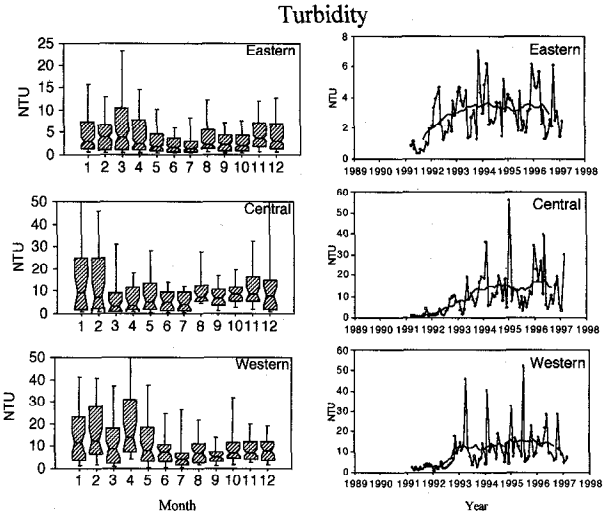


Fig. 8. Box-and-whisker and time series plots of turbidity (NTU) by zones of similar influence (ZSI).

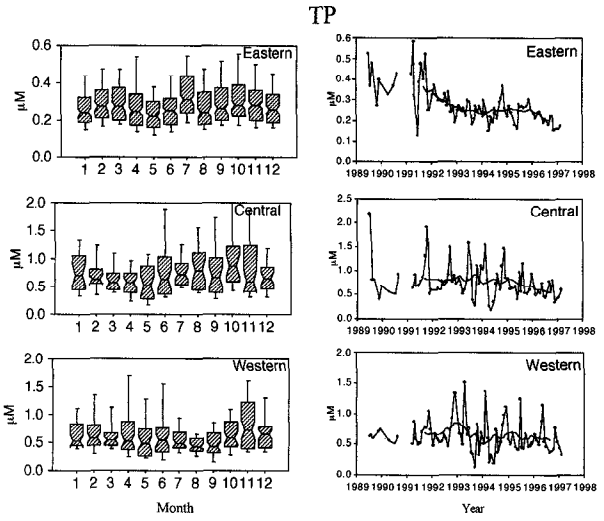


Fig. 9. Box-and-whisker and time series plots of TP (μM) by zones of similar influence (ZSI).

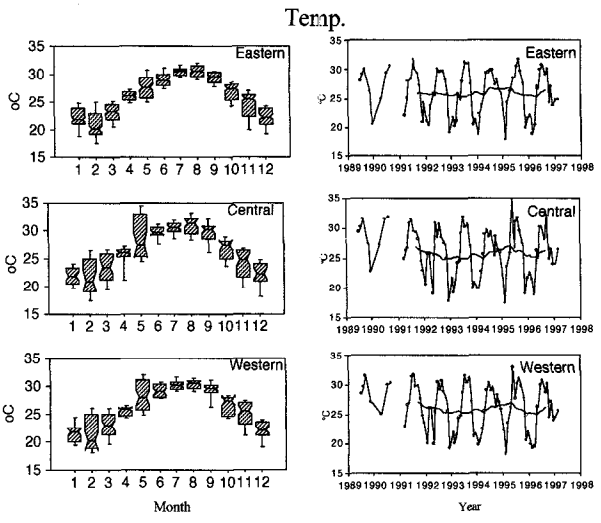


Fig. 11. Box-and-whisker and time series plots of temperature (°C) by zones of similar influence (ZSI).

were variable, with the lowest amounts found in early spring ($p = 0.002$, Fig. 10). Chl *a* increased greatly from 1991 to 1994 at an approximate rate of 1.0 μg $l^{-1}$ $yr^{-1}$ (Fig. 10). It then began to decline at an equal rate until the 1997, leading us to conclude that there was no long-term trend in Chl *a* concentration in the central bay (Table 1). Concentrations of Chl *a* in western bay are usually highest during October–February, coinciding with the cooler and windier dry season ($p < 0.0001$, Fig. 10). Chl *a* increased in this zone at a rate of 0.12 μg $l^{-1}$ $yr^{-1}$ (Fig. 10); a small but significant ($p = 0.065$, Table 1) amount relative to the median (1.93 μg $l^{-1}$).

## Temperature and $DO_{sat}$ Component

Water temperature in the eastern, central, and western bays showed pronounced seasonal signals, as expected, but did not display any significant interannual trends (Fig. 11). Median temperature for all areas was ca. 26°C. It is clear that the moving average is effective at filtering out the seasonal signal in temperature. No significant long-term trends in temperature were found for any of the zones of Florida Bay (Table 1).

Intra-annual cycles of $DO_{sat}$ were observed in all regions of Florida Bay (Fig. 12). The median $DO_{sat}$ was 92.2%, 86.3%, and 88.8% for the eastern, central, and western bays respectively. The eastern bay

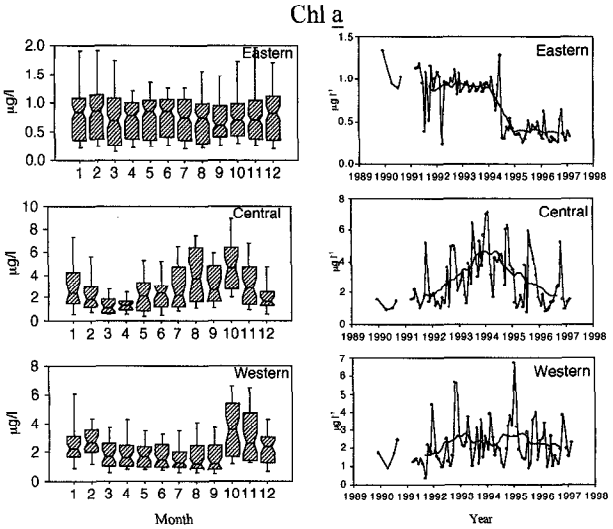


Fig. 10. Box-and-whisker and time series plots of Chl *a* (μg $l^{-1}$) by zones of similar influence (ZSI).

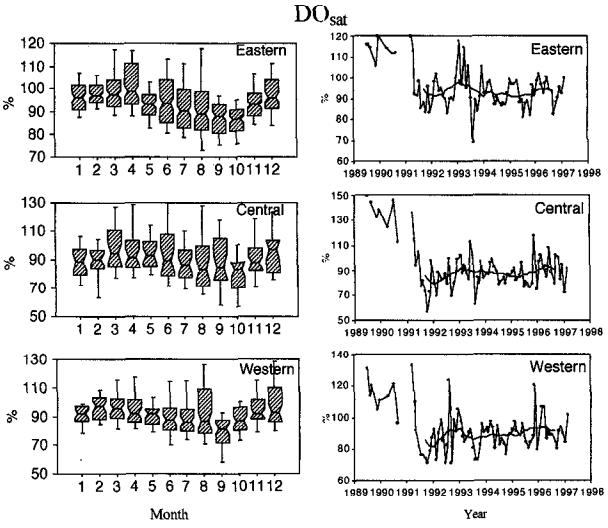


Fig. 12. Box-and-whisker and time series plots of $DO_{sat}$ (%) by zones of similar influence (ZSI).

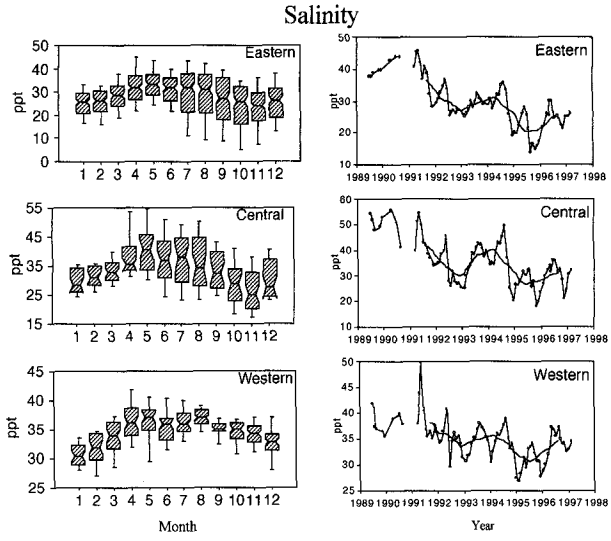


Fig. 13. Box-and-whisker and time series plots of salinity (‰) by zones of similar influence (ZSI).

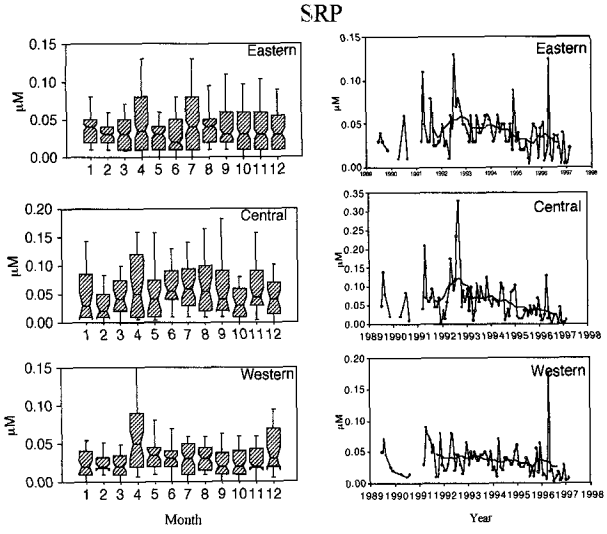


Fig. 14. Box-and-whisker and time series plots of SRP (μM) by zones of similar influence (ZSI).

displayed a decreasing trend from May to October, with rapid recovery in November. This trend was mirrored by the central and western bays but was more variable. The moving average for all showed little interannual fluctuation in $DO_{sat}$ in any of the ZSI (Fig. 12). There was no significant trend in $DO_{sat}$ over the period of record for any of the zones of Florida Bay (Table 1). Notably, across Florida Bay the oxygen concentration in the water column was supersaturated (with respect to air) during the sampling events prior to 1991. These data imply there was a large decline in $DO_{sat}$ during 1989–1991 that remains to be explained.

## Salinity Component

Salinity in the eastern, central, and western bay showed evidence of distinct seasonal patterns ($p < 0.0001$), with highest values occurring from April to September, during the wet season (Fig. 13). The increase in salinity during the summer is caused by high summer evaporation rates and long water-residence times in Florida Bay (Fourqurean et al. 1993). This seasonal pattern was more pronounced in the central bay where advective mixing is most restricted. In the western bay the seasonal signal in salinity was damped by open contact with the Gulf of Mexico.

The median salinity in the eastern bay for the 6-yr period of record was 28.1‰, with the moving average showing evidence of an interannual oscillation of ~2-yr periodicity (Fig. 13). In addition, there was a significant long-term decline of $-2.3$‰ $yr^{-1}$ or almost 14‰ over the period of record (Table 1). Median salinity in the central bay was 34.1‰, and despite large interannual oscillations, declined significantly by almost 12‰ over the period of record (TSE = $-1.9$‰ $yr^{-1}$, Table 1). In the western bay, the moving average showed less evidence of interannual oscillation due to the influence of the Gulf of Mexico (Fig. 13), and the Kendall-τ analysis showed a more modest decline in salinity of $-0.9$‰ $yr^{-1}$ (Table 1) or 5.5‰ over the period of record.

## SRP

SRP was not included in any of the original principal components (Boyer et al. 1997), meaning it did not explain a significant amount of variance in the data. SRP concentrations are generally an order of magnitude lower than existing TP concentrations Florida Bay (Table 1) and are maintained at these levels by biological uptake and chemical scavenging by the carbonate sediments. SRP in the eastern, central, and western bays showed no coherent seasonal change in concentration (Fig. 14). Little interannual oscillation was evident; there were small but significant declines in SRP for all areas of Florida Bay (Table 1). The concentrations are so low (0.03–0.05 μM) as to be at or below the kinetic threshold of many organisms.

## N:P Ratios

N:P ratios have been used as an indicator of relative N or P limitation of algae for many years (Redfield 1967). The TN:TP ratio in the eastern, central, and western bays showed some evidence of a seasonal pattern, with higher values occurring in summer months and significant peaks in May (data not shown). The median TN:TP in the eastern bay was 184 but showed a sustained increase of 13 $yr^{-1}$ ending at ~235. The median TN:TP in

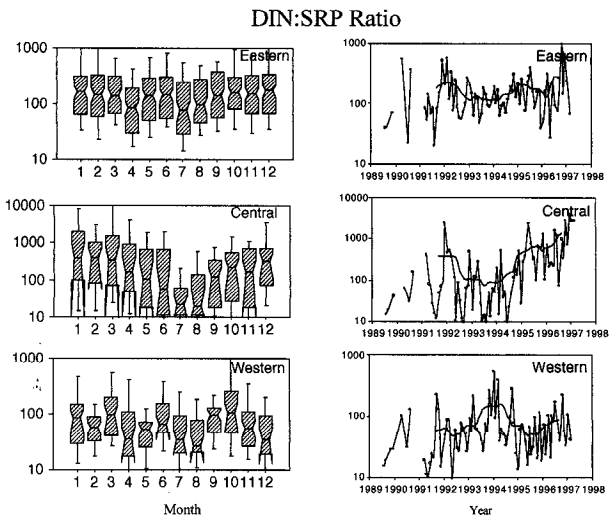


Fig. 15. Box-and-whisker and time series plots of DIN : SRP by zones of similar influence (ZSI).

the central and western bays was 132 and 55.6, respectively, with little interannual oscillation and no significant trends evident.

Since most of the TN in Florida Bay is organic, the DIN : SRP ratio may be a better indicator of nutrient limitation. The DIN : SRP ratio in the eastern bay showed very little seasonal cycling and was consistently high (median 152; Table 1), which indicated the potential for continuous P limitation (Fig. 15). Both the central and western bay showed evidence of fluctuation between P limitation and N limitation (Fig. 15). Lowest DIN : SRP ratios were observed during July in the central bay, but the western bay showed no obvious seasonal pattern. For the period of record, the lowest DIN : SRP ratios in the central bay occurred during 1992–1994.

### Hydrologic Data

The S18C canal accounted for the largest proportion of potential overland freshwater flow, ranging from $70 \times 10^6$ m$^3$ yr$^{-1}$ to $412 \times 10^6$ m$^3$ yr$^{-1}$ (Fig. 16). Losses and gains from groundwater between this structure and Florida bay are unquantified, so the actual amount of water entering Florida Bay from this system is not known. Due to its proximity to distribution, the ratio of actual freshwater input to S18C gage flow is probably the closest to unity than any of the other structures. S18C flow for the period 1990–1995 (post-1989 drought) increased significantly, by $38 \times 10^6$ m$^3$ yr$^{-2}$ ($\tau = 0.69$, $p = 0.02$). Flows through the S175 fluctuated from lows of $8.3 \times 10^6$ m$^3$ yr$^{-1}$ and $3.6 \times 10^6$ m$^3$ yr$^{-1}$ in 1989–1991 to a high of $152 \times 10^6$ m$^3$ yr$^{-1}$ in 1993. No trend in flow through the S175 was evident ($\tau = 0.09$, $p = 0.72$). The S332 flows prior to 1992 were $\sim 40 \times 10^6$ m$^3$ yr$^{-1}$; from 1992 onward the flows

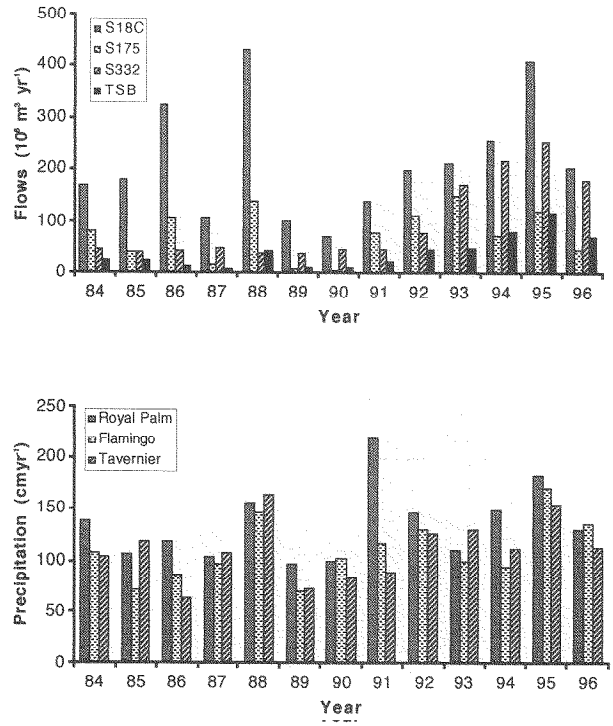


Fig. 16. Annual freshwater flows ($10^6$ m$^{-3}$ yr$^{-1}$) at individual water control structures in the Everglades for 1989–1996. Annual precipitation records (cm yr$^{-1}$) for rain gages at Royal Palm, Flamingo, and Tavernier.

increased dramatically, reaching $256 \times 10^6$ m$^3$ yr$^{-1}$ in 1995 (Fig. 16). In addition there was a change in the S332 flow relative to the S18C flow. Prior to 1992 the S332 made up 19.7% of the combined flow; from 1992 onward it contributed 33.2% of the total. This increase was also seen downstream of S332 as a concurrent increase in flows at the Taylor Slough Bridge (Fig. 16). Annual flows at the Taylor Slough Bridge increased from an average of $20 \times 10^6$ m$^3$ yr$^{-1}$ prior to 1992 to $73 \times 10^6$ m$^3$ yr$^{-1}$ afterward.

Precipitation at the Royal Palm, Flamingo, and Tavernier stations were used as a proxy of rainfall input to Florida Bay. Annual precipitation during 1989–1996 ranged from 97 cm yr$^{-1}$ to 220 cm yr$^{-1}$ (mean 136 cm yr$^{-1}$) for Royal Palm, 70 cm yr$^{-1}$ to 172 cm yr$^{-1}$ (mean 110 cm yr$^{-1}$) for Flamingo, and 74 cm yr$^{-1}$ to 155 cm yr$^{-1}$ (mean 111 cm yr$^{-1}$) for Tavernier (Fig. 16). No significant trend was found at any of these sites for this period of record.

## Discussion

Classification of 28 sampling stations into three ZSI allowed us to characterize trends in water quality variables for broad regions of Florida Bay. This spatial scale is larger than that of the individual station but smaller than the whole ecosystem and is the result of statistical analysis and not arbitrary

TABLE 2. Seasonal highs in water quality parameters from Kruskal-Wallis test ($p < 0.1$) for period of record 1991–1997.

| Parameter | Eastern Bay | Central Bay | Western Bay |
|---|---|---|---|
| TON | Summer | Summer | |
| APA | Fall | Fall | Winter |
| TOC | | Summer | |
| $NO_2^-$ | Winter | Winter | Winter |
| $NO_3^-$ | Winter | Winter | Winter |
| $NH_4^+$ | | Winter | |
| Turbidity | Winter/Spring | Winter/Spring | Winter/Spring |
| TP | | | |
| Chl *a* | | Fall | Fall/Winter |
| Temperature | Summer | Summer | Summer |
| $DO_{sat}$ | Winter/Spring | Winter/Spring | Winter/Spring |
| Salinity | Summer | Summer | Summer |
| SRP | | | |
| TN:TP | Summer | Summer | Summer |
| DIN:SRP | | Fall/Win./Spr. | |

grouping. Our approach is justified by the heterogeneous nature of the system and the different water-residence times therein (Boyer et al. 1997), and greatly simplifies the analysis of the data. In the temporal domain, we have analyzed for three scales of variability: intra-annual (seasonal), inter-annual (oscillations), and long-term monotonic trends.

The water quality of Florida Bay, as described by the variables measured in this monitoring network, is highly variable in both space and time. Many of the measured variables had seasonal maxima in the summer rainy season (Table 2). Obviously, water temperature followed a predictable pattern, with warmer temperatures in summer months, but concentrations of TON and TOC also tended to be greatest in the summer. This may be a result of enhanced decomposition of particulate organic matter in the warmer months or simply the concentration of solute by evaporation (salinity was also higher in the summer). Because of the lack of seasonality of TP and the fact that TON is the major component of TN, TN:TP ratios were also highest in the summer. Indicators of biomass and activity of the phytoplankton, Chl *a* and APA, tended to have maxima in the fall.

Some variables were highest in the winter dry season, such as the DIN pool, turbidity, and $DO_{sat}$. What drives this pattern in DIN is unclear, but it is not due to increased allochthonous loading from streamflow as this is the dry season. Possibly the lower summer-fall concentrations are due to uptake by phytoplankton and seagrass. Winds during the winter are higher than the rest of the year and generally come from the northeast (Schomer and Drew 1992). Higher winds and the long fetch from this direction may result in more resuspension of the fine muds from the bottom of Florida Bay (increased turbidity). The $DO_{sat}$ maxima in winter and spring may be due to increased diffusion of oxygen across the air-water interface in the more turbulent and windy seasons, as well as there being less respiratory consumption of oxygen in the cooler months.

After removing the seasonal signal from the time series of water quality data, many aspects of water quality displayed directional change over the period of record of this dataset (Table 3). Three var-

TABLE 3. Significant trends ($p < 0.10$) in water quality parameters from Kendall-$\tau$ test for period of record 1991–1997.

| | Eastern Bay | Central Bay | Western Bay |
|---|---|---|---|
| Parameter | Change | Change | Change |
| Salinity (ppt) | ↓ 13.6 (−385) | ↓ 11.6 (−28%) | ↓ 5.6 (−15%) |
| TP (μM) | ↓ 0.18 (−51%) | ↓ 0.21 (−28%) | ↓ 0.25 (−34%) |
| Chl *a* (μg $l^{-1}$) | ↓ 0.86 (−63%) | | ↑ 0.71 (1.5×) |
| Turbidity (NTU) | ↑ 1.3 (1.6×) | ↑ 14.7 (20×) | ↑ 7.9 (4×) |
| SRP (μM) | ↓ 0.01 (−22%) | ↓ 0.05 (−66%) | ↓ 0.03 (−63%) |
| APA (μM $hr^{-1}$) | ↓ 0.30 (−52%) | | ↓ 0.21 (−58%) |
| $NO_2^-$ (μM) | | ↑ 0.33 (3×) | |
| $NO_3^-$ (μM) | | ↑ 0.65 (2.8×) | |
| $NH_4^+$ (μM) | | ↑ 9.4 (6×) | |
| TN:TP | ↑ 78 (1.5×) | | |
| DIN:SRP | | ↑ 241 (5×) | |

iables displayed downward trends for salinity, TP, and SRP in all three zones. Salinity declined in all sections of Florida Bay as a function of variations in climate: the beginning of our data collection coincided with the ending of a regional drought, and the years 1994–1996 have all been wetter than normal, but no statistically significant trend was evident. Fresh water enters Florida Bay as rainfall, surface water runoff, and groundwater; more rainfall has led to a general increase in the surface water inputs since 1987 (Fig. 16). Regression of detrended monthly precipitation and combined monthly flows (S18C + TSB) was significant ($p < 0.01$) but explained < 2% of the variance, supplying very little predictive power for any type of modeling effort. Full-scale prediction of salinities in Florida Bay will require a more detailed assessment of inputs and mixing across Florida Bay

Fresh water entering Florida Bay from overland sources has a lower TP concentration than seawater entering Florida Bay from the Gulf of Mexico (Boyer and Jones 1999; Rudnick et al. 1999) and from precipitation (0.15–0.30 μM; R. Jones unpublished data). This dilution effect may help to explain some of the decline in P concentrations in the bay, but without process rate measurements we can only speculate as to actual cause.

Other water quality variables had different long-term trends in the different zones (e.g., Chl *a*, Table 3). These regional differences are the result of the varying importance of freshwater runoff, oceanic exchange, and internal processing of nutrients across Florida Bay and are discussed below.

### Eastern Bay

The eastern bay is most influenced by freshwater inputs from land, specifically Taylor Slough and the C-111 basin. Salinity in eastern Florida Bay showed a 38% decline over the period of record (Table 3). This was most probably due to increased freshwater flows and rainfall as a result of both climatic factors (alleviation of drought) and management efforts (increased flows to Taylor Slough). Boyer and Jones (1999) reported that the annual median salinity of the eastern bay was correlated to freshwater inputs ($r^2 = 0.85$, $p = 0.001$).

Concurrent with increased freshwater inputs was a 51% decrease in TP and a 22% decrease in SRP concentrations (Table 3). This type of trend is contrary to observations from most estuarine ecosystems, where increased runoff causes greater loading and therefore, higher nutrient concentrations. While TP loading did increase with flow, the TP concentration of the incoming water (0.30 μM) did not change significantly (Boyer and Jones 1999). This means that the trend of decreasing TP concentrations in eastern Florida Bay may have been due simply to dilution by increased freshwater inputs. SRP concentrations are an order of magnitude lower than TP, so a significant decline over the period of record was unexpected. Greater sediment absorption with decreased salinity could possibly account for this decline but has not been quantified. The increased turbidity caused by resuspension of bottom sediments over the period 1989–1997 did not result in increased concentrations of TP or SRP in the water column. Another response to the decrease in TP was an increase in the TN:TP ratio of the system (Table 1). We emphasize that the high TN:TP ratio of eastern Florida Bay is due to the very low concentrations of TP in the system and not to increased or high concentrations of N.

Concurrent with the decrease in salinity, TP, and SRP in eastern Florida Bay was a loss of density of the seagrass *Halodule wrightii* (Hall et al. 1999). Light penetration of the water column, which often controls seagrass density (e.g., Dennison 1987), decreased slightly in eastern Florida Bay over this interval (turbidity increased by 1.3 NTU over the POR, Table 3). It is unlikely that this increase in turbidity caused the observed decline in *H. wrightii* density because there was no concurrent decline in the dominant seagrass of the region, *Thalassia testudinum* (Hall et al. 1999). There is a possible connection between decreased P in the water column and the loss of *H. wrightii;* this species has a higher demand for nutrients than *T. testudinum* (Fourqurean et al. 1992, 1995) and is therefore more restricted to areas of higher nutrient availability. Powell et al. (1989) speculated that historic (pre-1984) declines in the prevalence of *H. wrightii* in eastern Florida Bay (documented in Schmidt 1979 and Zieman 1982) were caused by a decrease in P loading from the Everglades watershed, in turn caused by a decrease in the amount of freshwater input via streamflow. This work suggests the opposite: as freshwater inflow increased over the period 1989–1997, concentrations of TP and SRP in the water column decreased. Perhaps this decrease in the availability of water-column P led to the decline in *H. wrightii* abundance over the same period.

Chl *a* concentrations decreased in eastern Florida Bay by 63% (Table 3). We believe the decline in phytoplankton biomass during this period is further evidence of the P-limited nature of this zone (Fourqurean et al. 1993; Phlips and Badylak 1996). With the decline in TP concentrations we expected to see an increase in APA as the bacteria and algae produced more exoenzyme to alleviate P limitation, but the reverse was true—APA declined by 52% (Table 3). We do not have data on bacterial numbers and cannot determine whether the APA

decline was due to a decrease in the bacterial population, a shift in community structure, or a decrease in cell-specific enzyme production. Since some algae and most cyanobacteria produce alkaline phosphatase, the decline in APA may be due to the observed general decline in phytoplankton biomass.

### Central Bay

Salinity, TP, and SRP in the central bay all decreased over the period of record (Table 3). Salinity declined by 28%, likely due to the regional climatic factors that influenced rainfall and runoff into Florida Bay. TP declined by 0.21 μM (Table 3), a drop of 28%; and SRP declined by 0.05 μM, a 66% decline. There was no longterm monotonic trend in Chl *a* for the period of record, but this is misleading. During 1991–1994, median Chl *a* increased by a factor of four but then declined to previous levels by 1996. Concurrent with the phytoplankton blooms was a depression in the DIN: SRP ratio (Fig. 15).

The central bay was the zone most strongly affected by the seagrass die-off (Robblee et al. 1991); over 4,000 ha were lost between 1987 and 1988, yet the increase in phytoplankton biomass and decrease in light penetration of the water column did not begin until 1991 (Fig. 10). We can only speculate as to why it took 3 yr for increases in phytoplankton biomass to occur. Hypersaline conditions were unlikely to have kept phytoplankton growth in check because a significant component of the bloom community was *Synechococcus,* which is known for its great salinity tolerance (Phlips and Badylak 1996). We also observed a decline in $NH_4^+$ during 1992–1994 that coincided with the large increase in Chl *a* (Figs. 7 and 10). Quite possibly the 1992–1994 drawdown was a result of phytoplankton N demand, but it is unclear why there was no such response in the TP or SRP pools (Figs. 9 and 14). The DIN:SRP ratio during this period showed evidence of potential N limitation (Fig. 15), which may help to explain the preponderance of $N_2$ fixing cyanobacteria during the bloom event. Although no P or Chl *a* data exist for Florida Bay during the period of active seagrass die-off, the waters overlying the dying seagrass beds were remarkably clear (10 m visibility, J. W. Fourqurean personal observations).

In contrast to the rest of Florida Bay, the concentration of DIN in the central bay showed significantly increasing trends (Table 3). Both $NO_2^-$ and $NO_3^-$ concentrations tripled while $NH_4^+$ increased sixfold, with the largest increases occurring after 1994. These increases may be due to remineralized N from decaying belowground biomass of seagrass. There are five possible sources of N to Florida Bay: advective mixing, $N_2$-fixation, terrestrial runoff, groundwater input, and precipitation. In central Florida Bay, import by advective mixing is restricted by the western banks resulting in long water-residence times. $N_2$-fixation has been shown to be a significant source of N to seagrass in Biscayne Bay (Capone and Taylor 1980); porewater and seagrass measurements led Fourqurean et al. (1992) to speculate that $N_2$-fixation may be an important N source for the central bay. The importance of $N_2$-fixation may also have increased due to the recent (post-1991) presence of cyanobacterial blooms in this region (Phlips and Badylak 1996). Nitrogen loading from land runoff in this area is low and probably not significant. Groundwater input is unknown. Precipitation in this region is high in the summer months but probably does not get more rainfall than the rest of Florida Bay.

There are only three significant sinks for N in this ecosystem: advection, burial, and denitrification. As previously mentioned, advective mixing is low. Burial of N is a significant sink but total N concentrations in deeper sediments are not excessive (P. Carlson personal communication). Denitrification has been shown to be a significant sink of N in most other estuaries; with relatively high and increasing $NO_3^-$ and TOC concentrations in the central bay we would not expect denitrification to be N-limited or C-limited. Apparently, over the period 1989–1996, the sources of N to the water column of the central bay increased relative to sinks.

### Western Bay

The trends in the western bay were similar to those of the eastern bay, with the exception of Chl *a*. Salinity declined by 5‰ over the period of record, and TP, SRP, and APA declined by 34%, 63%, and 58% overall (Table 3). The salinity decline may have been due partially to increased freshwater flows in the eastern bay. More likely it was the result of a freshening of the waters of the southwest Florida Shelf from Shark Slough drainage, and the influence of increased rainfall. Evidence is incresing that inshore shelf waters become entrained with Gulf of Mexico waters flowing south through Western Florida Bay and out to the Atlantic Ocean through passes in the Keys (Smith 1994; T. Lee personal communication). This mixing of low P waters from Shark Slough with Gulf of Mexico might also explain the reduction in P concentrations as well.

Turbidity in the western bay ZSI increased fourfold between 1989 and 1991 (Table 3), probably as a result of resuspension of the bottom sediments that had become destabilized by the seagrass die-

off which began in 1987 (Robblee et al. 1991; Hall et al. 1999; Zieman et al. 1999). As in the eastern bay, increased benthic resuspension of carbonate sediments did not lead to an increase of P concentrations in the water column; in fact, concentrations of both TP and SRP declined over the study period. Concurrent with the decline in water-column P, Chl *a* concentrations increased in this zone by 0.7 $\mu g\ l^{-1}$. The long-term trend in the DIN:SRP ratio shows periodic drops, suggesting the phytoplankton in the western bay experience frequent N limitation (Fig. 15). Interestingly, during 1992–1994 the central bay showed a dip in the DIN : SRP ratio while in the western bay it was higher than the long-term median (Fig. 15).

## Conclusions

Rapid environmental changes have precipitated a general concern for the state of the Florida Bay ecosystem. Clearly there have been major changes in some water quality variables of Florida Bay over the study period. Salinity and P concentrations have declined baywide while turbidity has increased dramatically. In contrast to most other estuarine systems, increased runoff from the watershed may have actually decreased the concentration of P, the limiting nutrient, in the water column. Chl *a* concentrations were particularly dynamic and spatially heterogeneous in Florida Bay. In the eastern bay, which makes up roughly half of the surface area of Florida Bay, phytoplankton abundance has significantly declined. The hydrographically isolated central bay zone underwent a fivefold increase in phytoplankton abundance from 1989 to 1994, but then rapidly declined to 1989 levels. In western Florida Bay, there was a significant rise in phytoplankton abundance, yet median concentrations of Chl *a* in the water column remain modest (~2 $\mu g\ l^{-1}$) by most estuarine standards.

The major factor accounting for the anecdotal evidence of decreased water clarity in Florida Bay since 1991 is apparently fueled by the increase in sediment-derived turbidity. Turbidity in the western bay increased fourfold from 1991 to 1996; while turbidity in the central bay increased by a factor of 20. It important to note that these changes in turbidity and phytoplankton abundance happened years after the poorly understood seagrass die-off of 1987. The death and decomposition of large amounts of seagrass biomass probably can partially explain some of the changes in water quality of Florida, but the connections are temporally disjointed and the processes indirect and not well understood. It is important to remember that 6 yr of data collection represents a narrow window of time relative to natural climatic fluctuations of the ecosystem.

## Acknowledgments

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