

# The Effects of Anthropogenic Nutrient Enrichment on Turtle Grass (*Thalassia testudinum*) in Sarasota Bay, Florida

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**ABSTRACT:** Four meadows of turtle grass (*Thalassia testudinum* Banks ex König) in Sarasota Bay, Florida were sampled on a bimonthly basis from June 1992 to July 1993 to determine spatial and temporal variation in short shoot density, biomass, productivity, and epiphyte loads. Concurrent with the seagrass sampling, quarterly water-quality monitoring was undertaken at  $\geq 3$  sites in the vicinity of each studied seagrass meadow. Three months after termination of the seagrass sampling effort, a biweekly water-quality monitoring program was instituted at two of the seagrass sampling sites. In addition, a nitrogen loading model was calibrated for the various watersheds influencing the seagrass meadows. Substantial spatial and temporal differences in turtle grass parameters but smaller spatial variation in water quality parameters are indicated by data from both the concurrent quarterly monitoring program and the biweekly monitoring program instituted after termination of the seagrass study. Turtle grass biomass and productivity were negatively correlated with watershed nitrogen loads, while water quality parameters did not clearly reflect differences in watershed nutrient inputs. We suggest that traditional water-quality monitoring programs can fail to detect the onset or continuance of nutrient-induced declines in seagrass health. Consequently, seagrass meadows should be monitored directly as a part of any effort to determine status and/or trends in the health of estuarine environments.

## Introduction

Worldwide, the most often-cited cause of seagrass decline and disappearance is that of anthropogenic nutrient enrichment of nearshore waters (Larkum 1976; Kemp et al. 1983; Cambridge and McComb 1984; Orth and Moore 1984; Silberstein et al. 1986; Neverauskas 1987; Valiela et al. 1990; Reyes and Merino 1991; Tomasko and Lapointe 1991; Lapointe et al. 1994). Despite the well-documented relationship between increased water-column nutrient input and degradation of seagrass habitats, seagrass losses continue at unprecedented rates on a global scale.

In Tampa Bay, Florida, seagrass coverage has de-

clined approximately 80% from historic levels (Lewis et al. 1985; Zieman and Zieman 1989). Charlotte Harbor, the second largest estuary in Florida, has lost approximately 29% of its seagrasses (Duke and Kruczynski 1992). Trend analysis for seagrass distribution in Sarasota Bay is not as current as efforts in Tampa Bay and Charlotte Harbor, but seagrasses are thought to have declined approximately 30% since the late 1940s (Mangrove Systems, Inc. 1988), leaving only 26% of the bay bottom covered with seagrasses (Culter 1992).

In Sarasota Bay, losses of seagrasses are thought to be due to a combination of both direct (i.e., burial and/or removal during dredge and fill operations) and indirect (i.e., degraded water quality) impacts (Haddad 1989; Tomasko et al. 1992), as is the case in most of the estuaries in Florida. Despite successful efforts to reduce direct impacts,

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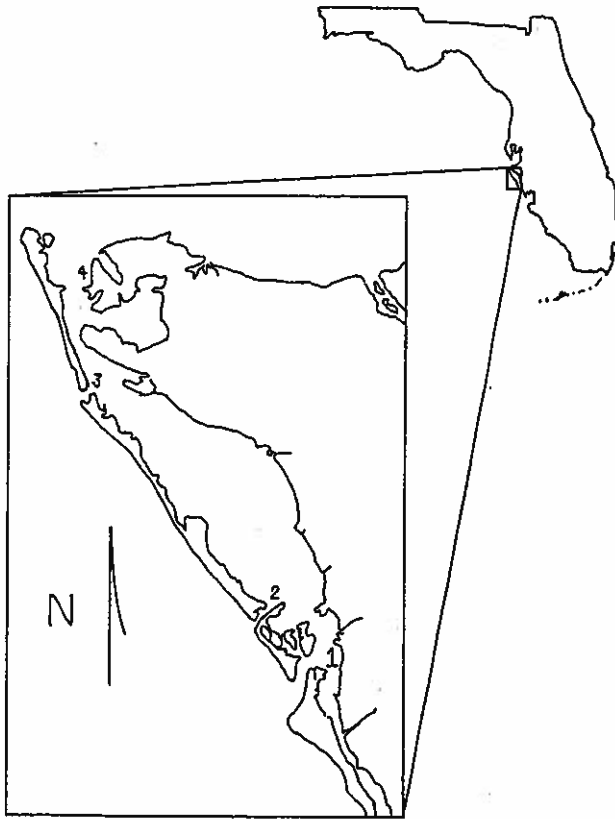


Fig. 1. Map of Sarasota Bay, showing location of study sites. 1 = Siesta Key, 2 = Raccoon Key, 3 = Leffis Key, 4 = Perico Isle.

indirect impacts continue to stress these important fishery habitats, mostly through the deleterious effects of nutrient-stimulated phytoplankton and epiphytic algal growths.

This study was designed to determine seagrass responses to anthropogenic nutrient enrichment of nearshore waters in Sarasota Bay. Specifically, the study sought to determine if *Thalassia testudinum* 1) is adversely affected by increased watershed nitrogen loads, and 2) is a more sensitive barometer of pollutant loads than traditional water-quality parameters.

## Materials and Methods

### FIELD SITES

Four study sites were chosen within Sarasota Bay (Fig. 1). The southernmost site, Siesta Key, was the most degraded location. A nutrient loading model for Sarasota Bay's airshed and watershed, discussed below, indicates the highest nitrogen loads in the bay degrade these nearshore waters. The location is characterized by sparse, heavily epiphytized meadows of *Thalassia testudinum* that are limited to depths of less than 1 m (MSL). The second site,

located north of Siesta Key, was offshore of a mangrove island called Raccoon Key. This location receives the second highest nutrient load of the four sites chosen. The third site, Leffis Key, is in a relatively pristine portion of the bay, as is the fourth site, Perico Isle. Sample sites were approximately 50 cm below MSL. All sites were visited on the same day on a bimonthly basis, from June 1992 to July 1993. A severe storm in March 1993, with waves up to 2 m high in the bay, displaced equipment in three of four sites, destroying that particular experiment.

### SEAGRASS SAMPLING

Short shoot density, productivity, biomass, turnover rates, and blade epiphyte loads were determined using standard procedures (i.e., Barber and Behrens 1985; Tomasko and Dawes 1989; Tomasko and Lapointe 1991; Tomasko et al. 1993). Density was determined by enumerating short shoots in 10 replicate 25 cm by 25 cm quadrats. The needle-marking technique was used for productivity estimates. Fifteen short shoots randomly located within 25 cm of a submerged stake were tagged and their blades marked with a 25-gauge hypodermic needle. After 7 d to 14 d (depending on season) the marked short shoots were brought back to the laboratory and all leaf material was severed from the short shoot at the appropriate reference point. Newly formed blade material was then separated from older material. Epiphytes were removed by lightly scraping with a double-edged razor blade, and the few remaining calcareous types (mainly *Fosiella* sp.) were brushed from the blades after they had dried for at least 24 h at 65°C. Epiphyte levels are expressed as a percent of blade weight.

Values for biomass per short shoot and productivity per short shoot were multiplied by average short shoot densities to determine areal biomass ( $\text{gdw m}^{-2}$ ) and areal productivity ( $\text{gdw m}^{-2} \text{d}^{-1}$ ).

### WATER QUALITY

Data were from two sources: a concurrent water quality monitoring program that was spatially-intensive and was quarterly in its timing, and a later monitoring program that incorporated two of the same sites as the seagrass sampling study and was biweekly in its timing. The later water-quality monitoring program was designed to correlate with a more intensive seagrass study, the results of which are being currently analyzed. Station density for the concurrent monitoring program (stations per square kilometer) averaged 0.39, a value 10 to 20 times higher than typical water-quality monitoring programs for estuarine systems in Florida (Tomasko unpublished data).

For the quarterly monitoring program, a mini-

mum of three surface water samples were collected within 1 km of each study site, fixed with sulfuric acid to a pH of <2, and immediately placed on ice. Values of total nitrogen (TN) were determined using an Autoanalyzer II according to methods of Slawyk and MacIssac (1972) and Technicon (1973). Chlorophyll *a* concentrations were determined after acidification and trichromatic spectrophotometry (American Public Health Association 1989). Data are summarized from Lowrey et al. (1993). The biweekly water-quality monitoring program used the same sampling and analytical techniques as the concurrent program, along with sampling for turbidity, color, and light attenuation coefficients. Light attenuation coefficients were determined for downwelling irradiance using cosine sensors (LI-COR 192SB) attached to a lowering frame with depths marked in 20-cm increments. Incident irradiance was simultaneously recorded with a cosine sensor in air (LI-COR 190SB) for later corrections in incident conditions. Two separate depth profiles were recorded at each site, with measurements taken at the surface and at depths of 20 cm, 40 cm, and 60 cm. All light measurements were made as close to solar noon as possible (i.e., between 1000 h and 1400 h, local time). Data are summarized from Dixon and Kirkpatrick (1995).

#### NUTRIENT LOADING ANALYSIS

The nutrient loading analysis for Sarasota Bay modelled inputs of both phosphorus and nitrogen (Camp, Dresser and McKee, Inc. 1992). For the purposes of this discussion, only nitrogen loads are considered.

Five nutrient sources were modelled: stormwater runoff, baseflow, point sources, septic tanks, and rainfall. Below is a brief summary of techniques, with more detailed information available in Heyl (1992) and Camp, Dresser and McKee, Inc. (1992).

Stormwater loads were determined by first mapping the various land use characteristics of the entire watershed. Using data developed through the United States Environmental Protection Agency's National Urban Runoff Program (NURP), percent directly connected impervious areas and event mean concentrations of various pollutants were calculated for each land use. Rainfall and runoff relationships were determined using long-term monitoring data from local stream and rain gauges. Stormwater loads were then determined for individual watersheds.

Baseflow was modelled using known values for nutrient concentrations of uncontaminated groundwater and locally obtained rates of horizontal groundwater movement. Point sources (sewage

and industrial outfalls) were determined for the 17 wastewater plants in the Sarasota Bay watershed discharging directly or indirectly to surface waters. Flow and nutrient concentration data came from routine monitoring reports for all sewage treatment plants. No industrial outfalls exist in the watershed.

Septic tank nutrient loads were determined using an algorithm designed by Camp, Dresser and McKee, Inc. (1992) that was locally calibrated using nutrient concentrations in receiving waters. The algorithm takes into account septic tank densities per watershed, effluent nutrient concentrations, uptake and absorption rates of nutrients in groundwater, and groundwater migration rates. Loads from rainfall (direct atmospheric loading of nitrogen to surface waters) were determined using data from local rain gauges and average rainfall nitrogen concentrations from NURP data.

#### CIRCULATION DATA

Circulation data are from Sheng and Peene (1992). The data are from a three-dimensional model of circulation and salinity transport. All output was calibrated using continuous recording instrumentation at various locations throughout the bay. Model runs were used to determine percent flushing rates at the four seagrass sampling sites.

#### STATISTICAL ANALYSIS

Seagrass data were analyzed using two-way ANOVA with SAS (SAS Institute 1985) to determine the effects, if any, of location and sampling date. Differences were viewed as significant at  $p \leq 0.05$ . Prior to analysis, data were tested to ensure normality and homoscedasticity. Duncan's New Multiple Range Test was used when significant main effects were detected.

To determine the relationship between modelled nitrogen loads and seagrass biomass and productivity, correlation analysis was performed. Data from December 1992 were not used due to the much reduced biomass and productivity at all sites. Various correlations were tested for significance (Iman and Conover 1983), with the best fit for both datasets being a power equation. Generated correlation values were tested for statistical significance as in Scheffler (1979).

#### Results

Modelled nitrogen loads to the seagrass study sites varied widely (Table 1). At Siesta Key, nitrogen loads were 12 times higher than at Leffis Key, and more than twice as high as the next most impacted site, Raccoon Key. Flushing rates were similar for all sites, between 64% and 81% water exchange over a 10-d period (Table 1).

TABLE 1. Nitrogen loads, flushing rates, and water quality parameters for bay segments containing study sites. Nitrogen loads are from Camp, Dresser, and McKee, Inc. (1992), flushing rates are from Sheng and Peene (1992), and water quality data are from Lowrey et al. (1993). For total nitrogen (TN) and chlorophyll *a* (Chl *a*), values are means ( $\pm$  SD) of  $n \geq 12$ .

Site	Annual Nitrogen Load (kg TN yr <sup>-1</sup> )	Flushing Rate (% in 10 d)	Annual Avg. TN (mg l <sup>-1</sup> )	Annual Avg. Chl <i>a</i> ( $\mu$ g l <sup>-1</sup> )
Siesta Key	219,098	76	0.73 (0.34)	8.38 (2.18)
Raccoon Key	85,713	81	0.62 (0.24)	4.86 (1.21)
Perico Isle	28,855	81	0.69 (0.28)	8.46 (3.15)
Leffis Key	17,994	64	0.60 (0.24)	4.50 (1.35)

In contrast to nitrogen load data, water column parameters from the quarterly monitoring program were somewhat similar at the four sites. Data obtained from the concurrent quarterly monitoring program showed the highest total nitrogen (TN) values, from Siesta Key, were only 21% higher than the lowest TN values, from Leffis Key (Table 1). In comparison with TN values, the variation in chlorophyll *a* concentrations (Chl *a*) was greater between sites (Table 1). However, there was no clear relationship between nitrogen loads and Chl *a* concentrations, as values from Perico Isle were nearly twice those of Raccoon Key, despite the waters off Raccoon Key being exposed to much greater nitrogen loads than the waters off Perico Isle. Turbidity values were nearly twice as high, and light extinction coefficients were 33% higher at Siesta Key compared with Leffis Key (Table 2).

Data obtained from the later biweekly water-quality monitoring program showed similar patterns, as did the concurrent quarterly monitoring program (i.e., reduced spatial variability versus nitrogen load estimates and parameters from the seagrass study). For example, annual average Chl *a* values from Siesta Key were only 26% higher than values from Leffis Key (Table 2), despite Siesta Key having a modelled nitrogen load 12 times higher than Leffis Key. Values for both turbidity and color varied less than modelled nutrient loads, although turbidity values were 26% higher and values for color were 50% higher at Siesta Key versus Leffis Key, respectively. Annual average light attenuation

TABLE 2. Comparison of water quality parameters for Siesta Key and Perico Isle from two separate monitoring programs. Quarterly data are from concurrent sampling at  $\geq 3$  locations within 1 km of seagrass study sites. Biweekly data are from seagrass study sites during a 1-yr study initiated 3 mo after termination of the seagrass study. Quarterly data are means ( $\pm$  SD) of  $n \geq 12$ . Biweekly data are means ( $\pm$  SD) of  $n = 26$ .

Site/Program	Annual Avg. Chl <i>a</i> ( $\mu$ g l <sup>-1</sup> )	Annual Avg. Turbidity (NTU)	Annual Avg. Color (Pt-Co units)	Annual Avg. K (m <sup>-1</sup> )
Siesta: quarterly	8.38 (2.18)	4.45 (1.51)	9.6 (3.2)	0.73 (0.40)
Siesta: biweekly	5.9 (3.4)	4.3 (2.0)	18 (15)	1.13 (0.40)
Leffis: quarterly	4.50 (1.35)	2.45 (0.97)	8.3 (6.6)	0.55 (0.27)
Leffis: biweekly	4.7 (2.5)	3.4 (1.1)	12 (6)	0.83 (0.19)

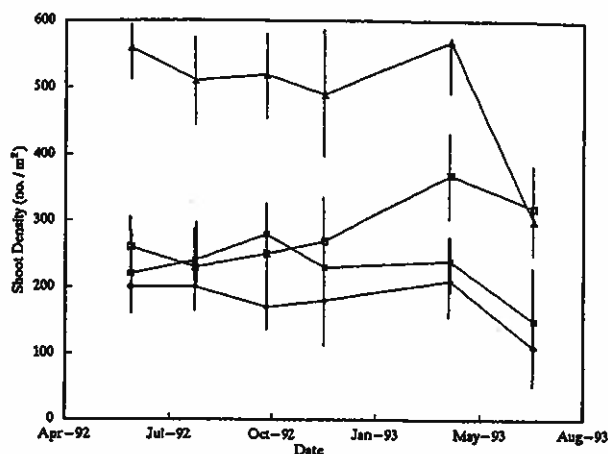


Fig. 2. Short shoot density (no. m<sup>-2</sup>) for *Thalassia testudinum* from four sites in Sarasota Bay. Values are means of  $n = 10$  ( $\pm$  SE). Siesta Key = solid squares, Raccoon Key = diamonds, Leffis Key = triangle, Perico Isle = open squares.

coefficients varied by 36% between Siesta Key and Leffis Key.

Short shoot densities of *Thalassia testudinum* had a significant effect of location ( $p < 0.001$ ; Fig. 2). Densities were significantly higher at Leffis Key and significantly lower at Raccoon Key compared with all other sites, but densities were not significantly different between Siesta Key and Perico Isle. Densities did not vary with date. In this and subsequent graphs, sampling dates for individual sites are slightly offset to allow viewing of variance estimates.

Blade epiphyte values (percent of blade weight) showed significant effects of sampling date ( $p < 0.001$ ), location ( $p < 0.001$ ), and the interaction of date by location ( $p < 0.001$ ). Epiphyte levels did not vary consistently between sites or dates (Fig. 3). For example, in December 1992, epiphyte loads were much higher at Siesta Key than any other site, yet on other dates there was little difference among sites.

Turnover rates of blade tissue showed significant effects due to sampling date ( $p < 0.001$ ) but not location. For all locations, blade turnover was lowest in December 1992 (Fig. 4). Blade turnover

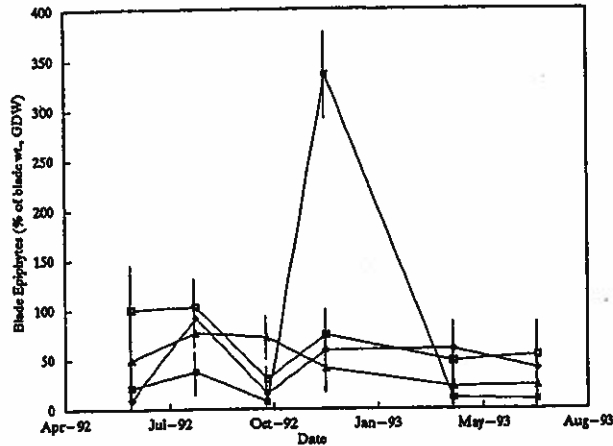


Fig. 3. Blade epiphyte values (% of blade weight, gdw) for *Thalassia testudinum* from four sites in Sarasota Bay. Values are means of  $n = 10$  ( $\pm$ SE). Siesta Key = solid squares, Raccoon Key = diamonds, Leffis Key = triangle, Perico Isle = open squares.

rates in August 1992 and October 1992 were not significantly different from each other but were significantly lower than in June 1992, May 1993, and July 1993.

Blade biomass (Fig. 5), showed significant effects due to date ( $p < 0.001$ ) and the interaction of date by location ( $p < 0.05$ ) but no significant effect of location alone.

Blade productivity (Fig. 6) showed significant effects of date ( $p < 0.001$ ), location ( $p < 0.006$ ), and the interaction of date by location ( $p < 0.001$ ). At all locations, productivity was lowest in December 1992, paralleling blade turnover rates (Fig. 4). Blade productivity for Perico Isle and Raccoon Key

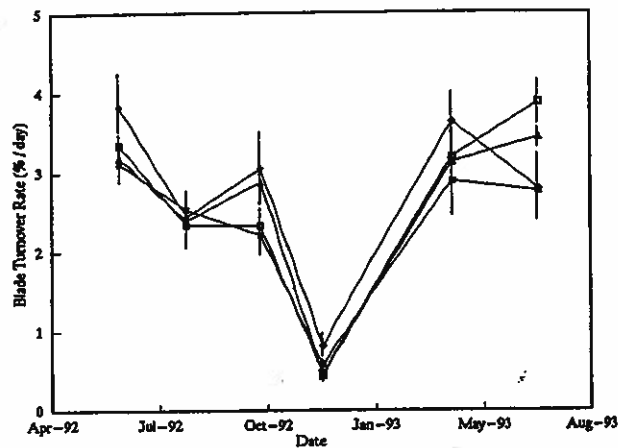


Fig. 4. Blade turnover rates (%  $d^{-1}$ ) for *Thalassia testudinum* from four sites in Sarasota Bay. Values are means of  $n = 10$  ( $\pm$ SE). Siesta Key = solid squares, Raccoon Key = diamonds, Leffis Key = triangles, Perico Isle = open squares.

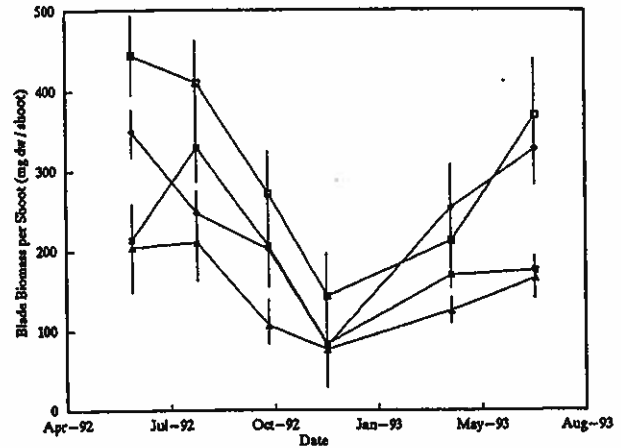


Fig. 5. Blade biomass values (gdw short shoot $^{-1}$ ) for *Thalassia testudinum* from four sites in Sarasota Bay. Values are means of  $n = 10$  ( $\pm$ SE). Siesta Key = solid squares, Raccoon Key = diamonds, Leffis Key = triangles, Perico Isle = open squares.

was generally higher than Leffis Key, which was itself not significantly different from Siesta Key.

Areal blade biomass (Fig. 7) is the product of average short shoot density ( $n = 10$ ) and average blade biomass per short shoot ( $n = 10$ ). In times other than December, areal biomass at Perico Isle and Leffis Key was higher than at Raccoon Key and Siesta Key.

Similar to areal blade biomass, areal blade productivity showed a strong seasonal influence (Fig. 8). Productivity was high for most of the year, with much lower rates in December. For months other than December, areal production at Perico Isle and Leffis Key was higher than at Raccoon Key and Siesta Key.

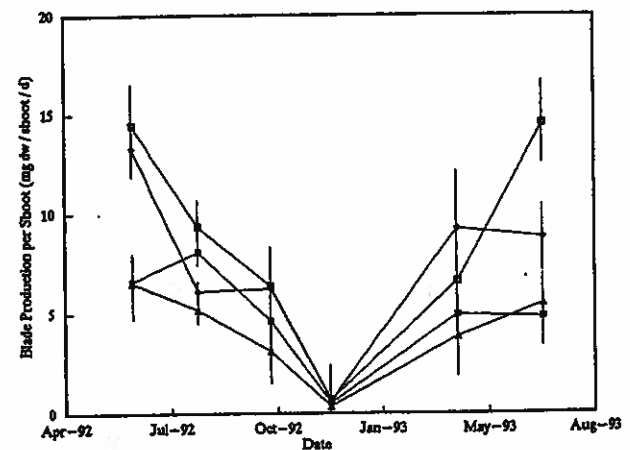


Fig. 6. Blade productivity values (gdw short shoot $^{-1} d^{-1}$ ) for *Thalassia testudinum* from four sites in Sarasota Bay. Values are means of  $n = 10$  ( $\pm$ SE). Siesta Key = solid squares, Raccoon Key = diamonds, Leffis Key = triangles, Perico Isle = open squares.

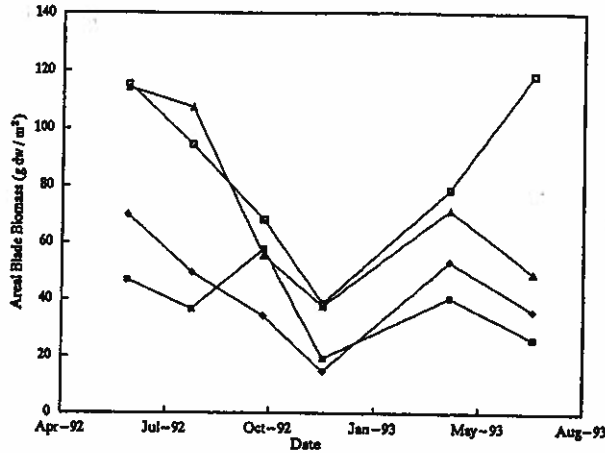


Fig. 7. Areal blade biomass values (gdw m<sup>-2</sup>) for *Thalassia testudinum* from four sites in Sarasota Bay. Values are calculated from short shoot biomass means (n = 10) multiplied by short shoot density means (n = 10). Siesta Key = solid squares, Raccoon Key = diamonds, Leffis Key = triangles, Perico Isle = open squares.

Areal blade biomass was negatively correlated with watershed nitrogen loads (R<sup>2</sup> = 0.584, p < 0.001; Fig. 9). With areal blade biomass represented by y and daily watershed nitrogen loads represented by x, the relationship between these two factors is explained by

$$y = 336.15(x)^{-0.342}$$

Areal blade productivity was also negatively correlated with watershed nitrogen loads (R<sup>2</sup> = 0.536, p < 0.001; Fig. 10). With areal blade productivity represented by y and daily watershed nitrogen loads represented by x, the relationship between these two factors is explained by

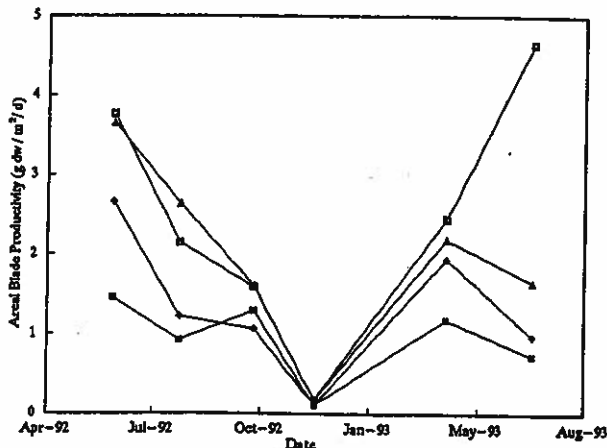


Fig. 8. Areal blade productivity values (gdw m<sup>-2</sup> d<sup>-1</sup>) for *Thalassia testudinum* from four sites in Sarasota Bay. Values are calculated from short shoot productivity means (n = 10) multiplied by short shoot density means (n = 10). Siesta Key = solid squares, Raccoon Key = diamonds, Leffis Key = triangles, Perico Isle = open squares.

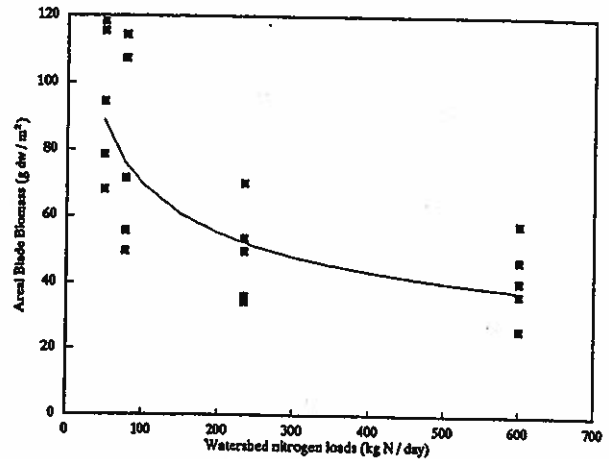


Fig. 9. Areal blade biomass plotted against watershed nitrogen loads for *Thalassia testudinum* from four sites in Sarasota Bay. Line is best-fit relationship (see text). Data from December 1992 are not included.

loads represented by x, the relationship between these two factors is explained by

$$y = 11.35(x)^{-0.371}$$

### Discussion

#### SEASONAL AND SPATIAL VARIATION IN SEAGRASS BIOMASS AND PRODUCTIVITY

The *Thalassia testudinum* meadows at all four sites exhibited strong seasonality in biomass per short shoot and productivity per short shoot but little seasonality in short shoot density. Consequently, it appears that the observed seasonality in areal biomass and areal productivity reflects changes occurring on the level of individual short

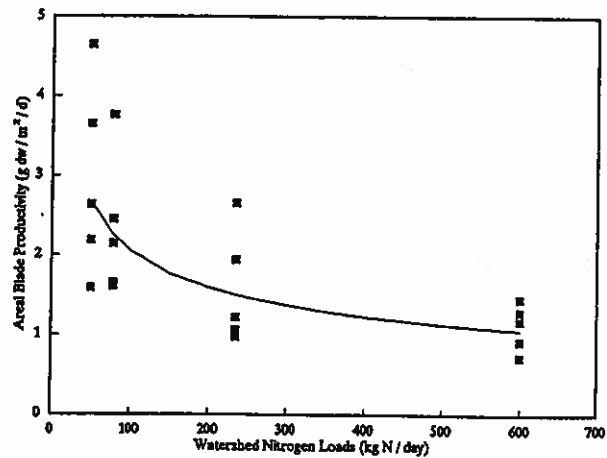


Fig. 10. Areal blade productivity plotted against watershed nitrogen loads for *Thalassia testudinum* from four sites in Sarasota Bay. Line is best-fit relationship (see text). Data from December 1992 are not included.

shoots, as opposed to changes in the number of short shoots.

Overall, biomass and productivity exhibited unimodal growth, with values highest in spring through fall, and lowest values in winter. Similar patterns were also found by Barber and Behrens (1985) and Tomasko and Dawes (1990) for *Thalassia testudinum* at other sites along the Florida Gulf Coast.

Despite having similar areal biomass and areal productivity, the *T. testudinum* meadows from Leffis Key and Perico Isle were quite different. The Leffis Key meadow had high short shoot densities but relatively little biomass per short shoot. In contrast, the Perico Isle site had lower densities than Leffis Key but greater biomass per short shoot. These two meadows are both characterized by high biomass and productivity compared with the other two sites, but they achieve this condition by different means.

#### RESPONSE OF TURTLE GRASS TO ANTHROPOGENIC NITROGEN-ENRICHMENT

Sarasota Bay is strongly nitrogen-limited, as molar N to P ratios of seawater average less than 5, and are often less than unity (Tomasko unpublished data). Thus, Sarasota Bay is similar to Tampa Bay, to the north, and Charlotte Harbor, to the south, which are also nitrogen-limited systems (Johannsson 1991; Montgomery et al. 1991). Further, results from the nutrient loading model suggest nitrogen loads to Sarasota Bay are approximately threefold higher than those expected from a pristine, undeveloped watershed (Camp, Dresser, and McKee, Inc. 1992).

However important nutrient enrichment might be in degrading seagrass habitats, water-column nutrient concentrations themselves tend to be poor indicators of nutrient status. As shown by Suttle and Harrison (1988) and Suttle et al. (1990), water column nutrients (both phosphorus and nitrogen) can have turnover rates of less than 10 min. Because of this phenomenon, Smith et al. (1981) and Valiela et al. (1990) found that water-column concentrations of the limiting nutrient were poor indicators of trophic state in marine and estuarine systems.

It appears that in Sarasota Bay, water-column nutrient concentrations do not reflect differences in modelled watershed nitrogen loads (Tables 1 and 2). Despite having a nitrogen load 12 times higher than Leffis Key, the waters off Siesta Key had annual average TN values only 21% higher than Leffis Key. Chlorophyll *a* concentrations were slightly higher in the waters off Perico Isle compared with Siesta Key, despite having watershed nitrogen loads only 13% that of waters off Siesta Key (Table 1).

Even with the greater temporal intensity of bi-

weekly sampling, Chl *a* values off Siesta Key were only 26% higher than Leffis Key, despite a 12-fold higher value for modelled nitrogen loads (Table 2).

Based on quarterly sampling, annual average light attenuation coefficients ( $n \geq 12$ ) varied by only 33% between Siesta Key and Leffis Key. From biweekly sampling, annual average light attenuation coefficients ( $n = 26$ ) varied by only 36% between Siesta Key and Leffis Key. Clearly, traditional water-quality parameters do not sufficiently reflect the magnitude of differences in modelled nitrogen loads in nearshore waters of Sarasota Bay.

In contrast, the modelled correlations between watershed nitrogen loads and seagrass biomass and productivity more strongly suggest the negative relationships between these variables. The sparsest and least productive seagrass meadow, off of Siesta Key, was in waters that received the greatest nitrogen input. The two locations with the lowest watershed nitrogen loads, Leffis Key and Perico Isle, had the lushest and most productive seagrass meadows.

Based on biweekly data, the Leffis Key site received approximately 38% more surface light (at the sample depth of 50 cm) than the Siesta Key site. As values for seagrass biomass and productivity at Leffis Key were typically 150% higher than at Siesta Key, factors other than water clarity alone seem to be associated with the reduced biomass and productivity of the seagrass meadows off Siesta Key. The same pattern is evident for other water-quality parameters as well—differences between sites are of lesser magnitude than differences in productivity and biomass of the seagrass meadows themselves.

The specific processes by which elevated nitrogen loads adversely affect *Thalassia testudinum* meadows in Sarasota Bay are not known, but they are probably similar to the multiple impacts thought to affect *Zostera marina* in temperate estuaries (e.g., Valiela et al. 1990; Dennison et al. 1993; Short et al. 1993). While the highest recorded blade epiphyte levels occurred at Siesta Key, which had the highest nitrogen load, this phenomenon varied temporally. Thus, the shading effect of blade epiphytes, which is known to decrease seagrass productivity in areas of nutrient overenrichment (Sand-Jensen 1977; Twilley et al. 1985; Silberstein 1986; Tomasko and Lapointe 1991), might not be the sole causative factor for diminished seagrass vigor at Siesta Key and Raccoon Key.

Lapointe et al. (1994) documented the importance of drift macroalgae as markers of nutrient overenrichment in the Florida Keys. Unfortunately, macroalgal biomass was not quantified in this study. Additionally, Burkholder et al. (1992) have

shown evidence of a direct negative impact of elevated nitrate loads on seagrass productivity.

Likely, the impacts to *Thalassia testudinum* meadows in Sarasota Bay from elevated nitrogen loads are due to a combination of factors—higher epiphyte loads during critical time periods, additional light reduction due to greater macroalgal biomass, and, possibly, direct and indirect impacts of near-dawn hypoxia associated with greater algal biomass (Leverone and Marshall 1992; Tomasko et al. 1992).

Although various studies have shown a positive influence of nutrient availability on the biomass and productivity of subtropical seagrasses (e.g., Short et al. 1985, 1990; Powell et al. 1989, 1991; Fourqurean et al. 1992a, b), these studies have typically taken place in locations referred to as isolated, or unimpacted. Interestingly, in one such area in Florida Bay (USA), a study by Powell et al. (1989) documented the replacement of a mature community of *Thalassia testudinum* by *Halodule wrightii* after 2 yr of nutrient enrichment with bird feces. These data, combined with similar findings from Cancun Lagoon, Mexico (Reyes and Merino 1991), and the Florida Keys (Lapointe et al. 1994), suggest that *T. testudinum* warrants consideration as a truly oligotrophic species, which cannot do well in areas of elevated nutrient loading.

#### SUGGESTIONS FOR ESTUARINE MONITORING PROGRAMS

We suggest that in shallow lagoonal environments such as Sarasota Bay, water-quality monitoring programs can fail to detect differences between locations with dramatically different levels of anthropogenic nutrient enrichment (even with bi-weekly sampling). Consequently, monitoring programs in areas comparable to Sarasota Bay should include the use of seagrasses, and perhaps macroalgae, as "bio-indicators" of system health, along with traditional water-quality parameters. Programs designed to monitor the condition of near coastal waters in Florida should incorporate the status of seagrass habitats (e.g., coverage, species diversity, standing stocks, productivity, epiphyte abundance, etc.). Otherwise, negative trends in estuarine environments are less likely to be recognized and appropriately acted upon.

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