Estuaries Vol. 24, No. 6A, p. 926–938 December 2001

Assessment of Present and Future Nitrogen Loads, Water Quality, and Seagrass (*Thalassia testudinum*) Depth Distribution in Lemon Bay, Florida

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ABSTRACT: Nitrogen loads into Lemon Bay, Florida were modeled to have increased ca. 59% between pre-development (i.e., 1850) estimates (5.3 kg TN ha^{-1} yr^{-1}) and estimates for the year 1995 (8.4 kg TN ha^{-1} yr^{-1}). By the year 2010, nitrogen loads are predicted to increase an additional 45% or 58%, depending upon progress being made toward replacing older septic tank systems with centralized sewerage (nitrogen loads of 12.2 and 13.3 kg TN ha^{-1} yr^{-1}, respectively). Using 1995 estimates, nonpoint sources (stormwater runoff) are thought to be responsible for ca. 76% of the annual nitrogen load, followed by septic tank systems (14%), rainfall (10%), and an insignificant load from baseflow. Based on an empirically-derived nitrogen load:chlorophyll *a* relationship developed for a portion of nearby Tampa Bay, a 45% increase in nitrogen loads into Lemon Bay could result in a 29% increase in annual average chlorophyll *a* concentrations. Using the estimate of a 29% increase in future chlorophyll *a* concentrations, an empirically-derived optical model for Lemon Bay suggests that light attenuation coefficients in the bay would increase ca. 9%, and the average depth limit of *Thalassia testudinum* in Lemon Bay would decrease by ca. 24%.

Introduction

In Tampa Bay, Florida, recent increases in seagrass coverage are thought to be linked to improved water quality, which in turn is related to substantial reductions in anthropogenic nitrogen loads (Johansson 1991; Avery 1997; Johansson and Ries 1997; Johansson and Greening 1999; Kurz et al. 1999). In Sarasota Bay, Florida, the biomass and productivity of seagrass meadows were shown to be better bioindicators of pollutant loads than traditional water quality variables (Tomasko et al. 1996). In response to reductions in point-source nitrogen loads, both water quality and seagrass coverage have improved in Sarasota Bay in recent years (Kurz et al. 1999). Spatial and temporal variation in the productivity and biomass of seagrass meadows in Charlotte Harbor, also in Florida, do not appear to be related to anthropogenic influences (Tomasko and Hall 1999).

Lemon Bay, located south of Sarasota Bay and west of Charlotte Harbor, is experiencing rapid population growth in its watershed (Florida Department of Natural Resources [FDNR] 1991; Charlotte Harbor National Estuary Program [CHNEP] 1997). In contrast to Tampa and Sarasota Bays and Charlotte Harbor, little is known about the susceptibility of Lemon Bay to the increased nitrogen loads that are expected to accompany the ongoing urbanization of its watershed. Unlike Tampa and Sarasota Bays and Charlotte Harbor, Lemon Bay's seagrass coverage appears to have decreased during the past decade (Kurz et al. 1999).

The funding and resources used to develop pollutant loading models, water quality monitoring programs, and seagrass mapping efforts for Tampa and Sarasota Bays and Charlotte Harbor have not been previously available for Lemon Bay. As Lemon Bay may be poised to experience significant losses of seagrass coverage if future nitrogen loads increase as expected, it is imperative that resource-management oriented research efforts be undertaken to aid in the development of strategies to minimize or reverse potential losses of fisheries habitats in Lemon Bay.

In response to the need to provide timely and cost-effective information to ongoing resource management efforts, this study was designed to develop a nitrogen loading model for Lemon Bay, estimate potential changes in phytoplankton populations that might occur under future nitrogen loading scenarios, determine the relationship between phytoplankton populations and water clarity in Lemon Bay, estimate potential changes in water clarity that would occur as a result of changes in

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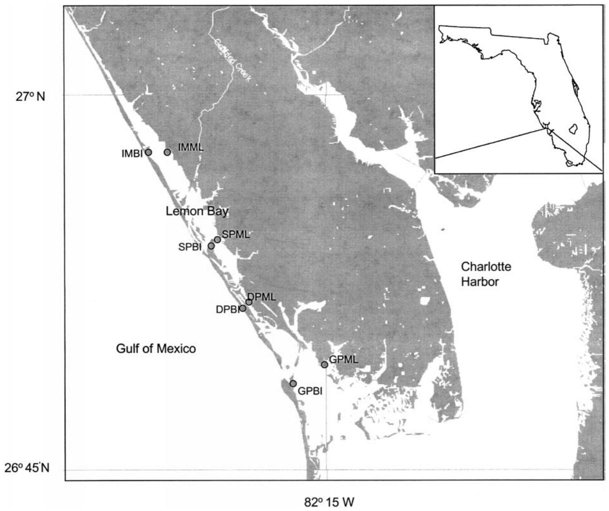

Fig. 1. Location of Lemon Bay in southwestern Florida with study sites. GPBI = Gasparilla Pass Barrier Island, GPML = Gasparilla Pass Mainland, DPBI = Don Pedro Barrier Island, DPML = Don Pedro Mainland, SPBI = Stump Pass Barrier Island, SPML = Stump Pass Mainland, IMBI = Indian Mound Barrier Island, and IMML = Indian Mound Mainland. Lemon Bay is approximately centered at 26°56′N, 82°21′W.

phytoplankton populations, predict the changes (if any) in seagrass depth distributions that would occur with changes in water clarity, and predict the changes (if any) in seagrass productivity that would occur with changes in water clarity.

Materials and Methods

Field Sites

Lemon Bay is located in Southwest Florida (Fig. 1). The bay is 21 km long and 0.2 to 1.9 km wide (FDNR 1991). The open water of the bay is 31 km^2 in size, with an average depth of 2 m (FDNR 1991). Tidal exchange occurs via two inlets, Stump Pass and Gasparilla Pass. The contributing watershed to Lemon Bay is 154 km^2 (CHNEP 1999), which gives Lemon Bay a watershed to open water ratio of 5 to 1, a value similar to those for Tampa Bay (6:1; Squires et al. 1998) and Sarasota Bay (3:1; Heyl 1992) but considerably less than for nearby Charlotte Harbor (12:1; Tomasko and Hall 1999).

Eight locations were chosen within Lemon Bay to represent a variety of conditions. Four pairs of stations were selected, with two pairs being located near passes and two pairs located in anticipated null zones for flushing (Sheng and Peene 1992). For each of the four pairs of stations, individual stations were located either on the barrier island side or on the mainland side of the bay. Stations were named to reflect both their relative degree of influence from flushing passes and their barrier island or mainland location: Gasparilla Pass Barrier Island (GPBI), Gasparilla Pass Mainland (GPML), Don Pedro Barrier Island (DPBI), Don Pedro Mainland (DPML), Stump Pass Barrier Island (SPBI), Stump Pass Mainland (SPML), Indian Mound Barrier Island (IMBI), and Indian Mound Mainland (IMML; see Fig. 1). Water quality, seagrass growth, and biomass data were collected approximately every month between June 1998 and July 1999.

Water Quality

At each location, water temperature and salinity were measured using a YSI water quality meter. Light attenuation coefficients were determined for downwelling irradiance using a planar quantum sensor (LI-COR 192SA) attached to a lowering frame that had depths marked in 20-cm increments. Incident irradiance was simultaneously recorded with an on-deck sensor (LI-COR 190SA) to correct for changing cloud cover. Two light profiles were recorded at each station, with measurements made at depths of 40, 60, and 80 cm below the surface. All light measurements were made as close to solar noon as possible (i.e., between 1000 and 1400, local time).

Water samples were transported back to the laboratory for quantifying chlorophyll *a* (chl *a*) concentrations ($\mu g\ l^{-1}$), as well as color (Platinum-Cobalt units) and turbidity (Nephelometric Turbidity Units).

For chl *a,* samples were collected in 1 liter opaque containers and brought back to the laboratory on ice. After filtration through 1 μm glass fiber filters, the filters were mechanically ground in 90% acetone, then centrifuged for 20 min at 500 G. After acidification with 0.1 N HCl, absorbance of the supernate was measured at 750 and 665 nm using a Perkin-Elmer UV/VIS Spectrophotometer.

For color, samples were collected in 250 ml containers and brought back to the laboratory on ice. After filtration through 1 μm glass fiber filters, the filtered samples were placed in a Lovibond Colormeter and compared to readings from an adjacent rotating color disc.

Turbidity samples were collected in 250 ml containers and brought back to the laboratory on ice. Unfiltered and unacidified samples were shaken and then measured using a Hach 18900 Ratio Turbidimeter.

Seagrass Sampling

At each location, marker buoys were installed in areas of continuous seagrass (*Thalassia testudinum* Banks ex König) coverage. Water depths ranged from 50 to 70 cm (MTL). Sites were visited approximately every month between June 1998 and July 1999.

Short-shoot density, aboveground standing crop, and blade productivity were determined using standard procedures (i.e., Tomasko and Lapointe 1991). At each station, short shoots were enumerated in ten replicate quadrats (25 × 25 cm). Ten haphazardly chosen short shoots within 2 m of a PVC post were tagged and their blades punched at the blade-sheath junction of the oldest intact blade with a 25-gage hypodermic needle, and after 7–14 d (depending on season) the marked short shoots were collected. After collection of at least five previously marked shoots, and their transport to the laboratory in coolers, blade epiphytes were removed by lightly scraping the leaves with a razor blade. Newly formed blade material was then separated from old material, and epiphytes, new blade material, and old blade material were dried separately for at least 24 h at 65°C. Mean values for biomass and productivity per short shoot at each station were multiplied by average short shoot densities m^{-2} to determine areal blade biomass (g dw m^{-2}) and areal blade productivity (g dw m^{-2} d^{-1}). On occasion, either vandalism or wave energy resulted in the loss of marker buoys. These events did not prevent us from acquiring data on density, biomass, or epiphyte abundance, but did preclude us from determining productivity using standard procedures. On these occasions (9 of 48 occurrences or 19%), blade biomass data were multiplied by the average blade turnover rate calculated from the other study sites, allowing productivity estimates for locations where productivity experiments were disturbed. One-way analysis of variance showed no significant differences in blade turnover rates among stations on dates when data were available from all locations ($p > 0.10$, 7 df). Blade turnover rates of *T. testudinum* did not exhibit evidence of spatial variation in either Sarasota Bay (Tomasko et al. 1996) or Charlotte Harbor (Tomasko and Hall 1999), providing additional support for the use of this method to estimate areal productivity at disturbed sites.

Maximum depth distributions were measured by towing a diver along the sea bed into increasing water depths until no further seagrass shoots were encountered. Local tide tables were used to correct measured water depths for tidal stage to normalize all depth measurements to mean tide level (MTL).

Nitrogen Loading Model Development

Traditionally, nitrogen is thought to be the primary nutrient limiting phytoplankton productivity in estuarine ecosystems (Ryther and Dunstan 1971). Previous monitoring efforts in Lemon Bay have shown that nitrogen to phosphorus ratios (molar) of water samples average less than 12 (Southwest Florida Water Management District [SWFWMD] 1999), suggesting strong nitrogen limitation of phytoplankton biomass (Florida Department of Environmental Protection 1994). Results of manipulative experiments in the adjacent estuaries of Sarasota Bay and Charlotte Harbor have shown nitrogen to be the limiting factor for phytoplankton productivity (Dixon and Kirkpatrick 1999a and Montgomery et al. 1991, respectively). The pollutant loading model for Lemon Bay constructed here was focused on nitrogen as the nutrient of concern.

Estimates were made of the amount of nitrogen loaded into Lemon Bay from nonpoint sources (i.e., stormwater runoff), baseflow (uncontaminated groundwater), direct rainfall onto the waters of Lemon Bay, and septic tank systems. A previous assessment did not identify any significant point source loads in the Lemon Bay watershed (CHNEP 1997). Nitrogen loads were estimated for four different scenarios: expected nitrogen loads from a pre-development time period (i.e., 1850), nitrogen loads in 1995, nitrogen loads in the year 2010 with septic tank loads mostly eliminated, and nitrogen loads in the year 2010 with septic tank loads not eliminated.

The development of a nitrogen loading model for Lemon Bay is based on previous efforts for Sarasota Bay (Heyl 1992) and Charlotte Harbor (Squires et al. 1998), which are themselves similarly constructed as most estuarine nutrient loading models (see reviews by Stanley 2001; Turner et al. 2001). The nonpoint source loading estimates for these models attempt to determine the overall effect of different land use types on stormwater loads of various nutrients. Implicit in these and other nutrient loading models is the concept that nonpoint source nutrient loads associated with particular land use categories (e.g., agriculture, residential, forested, etc.) will be similar throughout the entirety of the watershed. In this manner, land use maps can be used, with locally derived nonpoint source data, to estimate nonpoint source loads on a watershed level. This approach has been used previously to estimate nonpoint source loads for numerous estuaries throughout the U.S. (Stanley 2001; Turner et al. 2001). Also, nutrient fluxes into Lemon Bay from the Gulf of Mexico were not accounted for, as these waters were assumed to be a boundary condition where nutrient-induced water quality degradation was minor or nonexistent. Most nutrient loading models for U.S. estuaries also exclude loads from oceanic sources (Stanley 2001; Turner et al. 2001), as does the U.S. Geological Survey's watershed nitrogen loading model, SPARROW (Spatially Referenced Regression on Watershed attributes; Alexander et al. 2001).

As organic nitrogen has been shown to be biologically available for phytoplankton assimilation (Anita et al. 1991; Peierls and Paerl 1997; Seitzinger and Sanders 1997), loads for all sources are treated without distinguishing between inorganic and organic forms of nitrogen.

Stormwater nitrogen loads were modeled by first estimating land use acreage in the Lemon Bay watershed. The SWFWMD maps land use throughout its jurisdictional area on a five-year schedule. The most recent completed effort was for photography shot in 1995. The SWFWMD's Mapping and Geographic Information Systems program was used to estimate the acreage of land use in the following categories: residential, commercial, industrial, mining, agricultural, forested/uplands (uplands includes non-forested rangeland), wetlands, and other land uses. For estimating land use patterns in 2010, information on expected population growth and land use changes were extracted from CHNEP (1997). For estimating land use patterns in pre-development times (i.e., 1850), it was assumed that the only categories of land use were forested/uplands and wetlands. This is thought to be a safe assumption, as the first documented settlers in the Lemon Bay region did not arrive until 1878 (FDNR 1991). It was assumed that the 50% decline in wetlands coverage that had occurred throughout Florida between 1850 and 1990 (Estevez 1992) had also occurred in Lemon Bay, with the result that Lemon Bay would have had twice as much wetlands acreage in 1850 as in 1995.

For each land use category, runoff coefficients were estimated using values from nitrogen loading models developed for the adjacent estuary Charlotte Harbor (Squires et al. 1998). These values were verified by comparison with values derived from Heyl (1992) in a study from adjacent Sarasota Bay. Runoff coefficient estimates were used for both the dry season (October 1 to June 30) and the wet season (July 1 to September 30). In the Lemon Bay watershed, 0.51 and 0.78 m of rainfall occur in typical dry and wet seasons, respectively (FDNR 1991). By multiplying rainfall amounts by runoff coefficients and summing up for the acreage of land use in each category, estimates were made for the total amount of runoff generated by each land use type for each scenario. Event mean concentrations of nitrogen (i.e., the concentration required to account for a measured load) associated with runoff from each land use type were derived from Squires et al. (1998) to determine the nitrogen load associated with stormwater runoff for each land use type (Table 1). The event mean concentration data used for nutrient loading models for Charlotte Harbor and Sarasota Bay are based on multiple storm event sampling efforts originally conducted by Delwiche and Haith (1983) and Harper (1991).

TABLE 1. Parameters used to develop a pollutant loading estimate for non-point source total nitrogen loads.

Land-Use Type	Run-off Coefficient: Dry Season[1]	Run-off Coefficient: Wet Season[1]	Run-off Coefficient: Heyl[2]	Event Mean Concentration ($mg\ l^{-1}$)
Residential[3]	0.37	0.53	0.45	2.05
Commercial	0.82	0.91	0.87	1.06
Industrial	0.80	0.90	0.87	1.64
Mining	0.50	0.50	nd	1.18
Agricultural[4]	0.22	0.30	0.16	1.24
Forested/Uplands[5]	0.19	0.24	0.16	0.44
Wetlands[6]	0.73	0.80	0.85	1.44
Other[7]	0.52	0.60	0.54	1.29

[1] Data from Squires et al. (1998).
[2] Data derived from Heyl (1992).
[3] Average of all residential land use types.
[4] Average of all agricultural land use types.
[5] Uplands includes non-forested rangeland.
[6] Average of all freshwater wetland types.
[7] Average of all land use types.
nd = no data available.

Baseflow (groundwater flow to Lemon Bay) nitrogen contributions from the entire watershed were estimated using a formula modified from the nutrient loading model for Charlotte Harbor (Squires et al. 1998). When modified, the algorithm for estimating groundwater flow on a watershed level is:

$$Q = (1{,}000) \times T\ I\ L$$

Where, Q = flow of groundwater into Lemon Bay ($l\ d^{-1}$), 1,000 liters = one cubic meter of water, T = transmissivity of the surficial aquifer ($m^2\ d^{-1}$), I = the hydraulic gradient of the watershed ($m\ km^{-1}$), and L = the flow zone (km of shoreline).

For this effort, T was estimated to be 74 $m^2\ d^{-1}$ (Squires et al. 1998), I was estimated to be 0.94 $m\ km^{-1}$ (FDNR 1991), and L was estimated to be 12.9 km (FDNR 1991). The resulting watershed level groundwater inflow estimate was then multiplied by a locally-derived estimate of surficial aquifer nitrate concentrations from the Charlotte Harbor watershed (0.014 $mg\ l^{-1}$; Squires et al. 1998). Nitrate, rather than total nitrogen, was used for calculating groundwater nitrogen loads, as nitrate is the predominant form of nitrogen in local groundwater, and is the only form of nitrogen with an ability to migrate horizontally over any appreciable distance (Florida Department of Health and Rehabilitative Services 1994). Daily loads were then summed for the dry season (273 d) and the wet season (92 d).

The nitrogen load associated with rainfall directly onto the surface of Lemon Bay was calculated by multiplying rainfall amounts by locally-derived rainfall nitrogen concentrations and summing for the area of open water. In the average dry season, 0.51 m of rain falls on the 31 km^2 of open water of Lemon Bay (FDNR 1991). This value was then converted to liters and multiplied by the average dry season total nitrogen concentration measured from adjacent Sarasota Bay (Dixon and Kirkpatrick 1999a). The average wet season rainfall amount of 0.79 m was used to calculate the volume of rain falling on the open waters of Lemon Bay, and this amount was multiplied by the average wet season total nitrogen concentration measured for Sarasota Bay (Dixon and Kirkpatrick 1999a) to calculate a wet season nitrogen load.

For septic tank system nitrogen loads, methodology from Squires et al. (1998) was used. In 1995, about 10,000 septic tank systems were located in the Lemon Bay watershed (Englewood Water District personal communication). A public works program is now in place to remove approximately 90% of these systems by the year 2010, resulting in a planned future septic tank system population of only 1,000 (Englewood Water District personal communication). Septic tank system nitrogen loads were estimated by assuming an average of 2.3 people per household, producing a wastewater stream of approximately 285 liters per person per day (Heyl 1992; Squires et al. 1998). This flow rate was then multiplied by an estimated total nitrogen concentration of septic tank effluent of 39 mg TN l^{-1} (Heyl 1992; Squires et al. 1998). This potential nitrogen load was estimated to be attenuated by approximately 80% as effluent filters through the unsaturated soil layer between the bottom of the septic tank drainfield and the water table, and then to nearby surface water features. While nitrate has been shown to travel significant distances in some geologic settings (Walker et al. 1973; Lapointe et al. 1990), previous work in Southwest Florida has shown that the majority of nitrogen loaded into the surficial aquifer in the Lemon Bay watershed would most probably be denitrified prior to its appearance in the creeks and shoreline emptying into Lemon Bay (Florida Department of Health and Rehabilitative Services 1994). The 80% attenuation value used for septic tank nitrogen loads would result in a reduction in groundwater nitrogen concentrations beyond that due to dilution alone, and is meant to approximate the fate of the average plume of septic tank effluent over the 154 km^2 watershed.

Development of an Optical Model for Lemon Bay Water Quality

An empirically-derived optical model for Lemon Bay was constructed using techniques modified from McPherson and Miller (1987) in their work on Charlotte Harbor. Light attenuation coefficients from each location and date combination (n = 96) were used with concurrently collected data on color, chl *a,* and turbidity to develop a statistical relationship through the use of forward stepwise multiple regression (Statgraphics). Light attenuation coefficients were separated into partial attenuation coefficients as follows:

$$k_{total} = \text{constant} + (k_{color} \times \text{color}) + (k_{chla} \times \text{chla}) + (k_{turb} \times \text{turb})$$

Where, k_{total} = total light attenuation coefficient, constant = value derived from multiple regression, k_{color} = partial attenuation coefficient of color (from multiple regression), color = dissolved organic matter (in Platinum-Cobalt units), k_{chla} = partial attenuation coefficient of chl *a* (from multiple regression), chla = chl *a* concentration ($\mu g\ l^{-1}$), k_{turb} = partial attenuation coefficient of turbidity (from multiple regression), and turb = turbidity (in nephelometric turbidity units; NTU).

The regression equation developed for all site and date combinations was then used with water quality data from individual locations to determine site-specific estimates of the amount of light attenuation associated with phytoplankton biomass, as in McPherson and Miller (1987).

Nitrogen Loading:Chl *a* Relationships

A problem encountered with the development of a linked pollutant load-water quality model for Lemon Bay is the scarcity of historical water quality data. In contrast to Tampa and Sarasota Bays and Charlotte Harbor, where water quality data sets extend back into the 1970s, there has been a conspicuous absence of organized water quality monitoring efforts in Lemon Bay. There is too little information to test for trends (if any) in water quality in Lemon Bay (CHNEP 1997).

To estimate future chl *a* concentrations in Lemon Bay, based on future nitrogen load estimates, Lemon Bay's nitrogen load estimates and chl *a* concentrations were compared to an empirically-derived relationship developed for Hillsborough Bay, in the northeastern region of Tampa Bay. While no residence time estimates are available for Lemon Bay, estimates are available for three similar lagoonal systems located just north of Lemon Bay, in the Sarasota Bay region. Based on Eulerian plots of residence times, with residence time defined as the period required for initial particle concentrations to be reduced to 1 e^{-1} times the initial concentration, these three systems, Roberts Bay, Little Sarasota Bay, and Blackburn Bay, have residence times between 5 and 10 d (Sheng and Peene 1992). In Hillsborough Bay, much of the bay has a similarly defined residence time (based on Euleri-

TABLE 2. Light attenuation coefficients (k m^{-1}), percent of sub-surface irradiance at the deep edge of seagrass meadows, chlorophyll *a* concentrations (μg l^{-1}), turbidity (NTU), and color (platinum-cobalt values) for all sites. Values are means (SD) of n = 12. Note that the minimum detectable level for color is 5.

	Attenuation Coefficient	Percent Subsurface Irradiance	Chl *a*	Turbidity	Color
Gasparilla Pass—Barrier Island	1.20 (0.59)	45.38 (21.88)	3.66 (1.28)	4.70 (3.69)	5.42 (1.44)
Gasparilla Pass—Mainland	1.52 (0.74)	15.23 (6.60)	10.05 (4.31)	6.87 (7.25)	6.25 (4.33)
Don Pedro—Barrier Island	1.47 (0.43)	20.16 (7.14)	11.31 (4.95)	6.35 (1.87)	5.00 (0.00)
Don Pedro—Mainland	1.51 (0.32)	18.73 (6.02)	10.76 (3.94)	6.43 (1.66)	5.00 (0.00)
Stump Pass—Barrier Island	1.29 (0.28)	17.20 (4.93)	8.32 (4.26)	7.58 (3.33)	5.00 (0.00)
Stump Pass—Mainland	1.46 (0.42)	31.82 (12.01)	9.55 (4.15)	8.23 (6.04)	5.00 (0.00)
Indian Mound—Barrier Island	1.67 (0.68)	21.10 (14.34)	15.84 (9.13)	6.96 (3.34)	6.25 (2.26)
Indian Mound—Mainland	1.76 (0.65)	17.00 (8.68)	16.24 (8.75)	6.87 (3.94)	5.00 (0.00)

an plots) of less than 18 d (Burwell et al. 2000). As residence time estimates were not entirely dissimilar between lagoonal systems adjacent to Lemon Bay and in Hillsborough Bay, the empirical relationship between watershed nitrogen yields and water column chl *a* concentrations derived for Hillsborough Bay was used to predict future chl *a* concentrations in Lemon Bay that might be expected with future nitrogen loading scenarios.

The Hillsborough Bay empirical model compares annual nitrogen loads to annual average chl *a* concentrations for the years 1964 to 1990 (Johansson 1991). By normalizing watershed level nitrogen loads to kg TN ha^{-1} yr^{-1}, Lemon Bay nitrogen load estimates could be directly compared to those from Hillsborough Bay. In Lemon Bay, phytoplankton abundances at sites GPBI and SPBI, the two sites closest to passes, are probably kept low through the flushing action of their respective passes; data from these two locations were not included in this analysis.

Statistical Analyses

Statistical analyses were used to examine spatial and/or temporal differences in seagrass growth parameters and water quality data. Data were tested to ensure compliance with assumptions regarding normality and homogeneity of variance using Statgraphics. If data failed either test, comparisons were made using the non-parametric Kruskal-Wallis test. If significant differences were found ($p < 0.05$) for normally distributed data, Least Significant Differences multiple comparison tests were used. If significant differences were found ($p < 0.05$) for non-normally distributed data, median notch Box and Whisker plots were used to determine where differences occurred between data sets.

Forward stepwise multiple linear regression (Statgraphics) was used to develop an optical model relating light attenuation coefficients (the dependent variable) against potentially significant independent variables (color, chl *a*, turbidity). The F-value to add a variable was 4.00 ($p = 0.050$), and the F-value to remove a variable was 3.90 ($p = 0.053$).

Results

Water Quality

Water temperature varied between 20.3°C and 33.7°C. Values varied among dates ($p < 0.01$; Kruskal-Wallis) but not locations ($p > 0.05$; one-way ANOVA). Salinity ranged between 26.0‰ and 37.6‰, with significant variation between dates ($p < 0.01$; Kruskal-Wallis) but not locations ($p > 0.05$; Kruskal-Wallis). Salinities were lowest in the wet season (July to September) of 1998 and highest in the dry season (October to June) of 1998 to 1999.

Light attenuation coefficients (Table 2) varied between dates ($p < 0.01$; Kruskal-Wallis) but not locations ($p > 0.10$; Kruskal-Wallis). Although not significantly different, mean attenuation coefficients were lowest at sites located closest to flushing passes (GPBI and SPBI).

The mean percent of subsurface irradiance at the deep edges of the sampled seagrass meadows (Table 2) varied between locations ($p < 0.01$; Kruskal-Wallis) but not dates ($p > 0.05$; Kruskal-Wallis). Mean values ranged from 15.2% to 21.1% of subsurface irradiance for sites GPML, DPBI, DPML, SPBI, IMBI, and IMML. Mean values for SPML and GPBI were significantly higher: 31.8% and 45.4% of subsurface irradiance, respectively.

Mean chl *a* concentrations ranged between 3.66 and 16.24 μg l^{-1} (Table 2). Concentrations varied between both dates and locations ($p < 0.01$; Kruskal-Wallis). Mean concentrations were lowest at the two stations located closest to flushing passes (GPBI and SPBI) and highest at the four sites located in null zones for circulation (IMBI, IMML, DPBI, and DPML). Mean turbidity values ranged between 4.70 and 8.23 NTU (Table 2). Values varied between dates ($p < 0.01$; Kruskal-Wallis) but not locations ($p > 0.05$; Kruskal-Wallis). Mean values for color ranged between 5.00 and 6.25 plati-

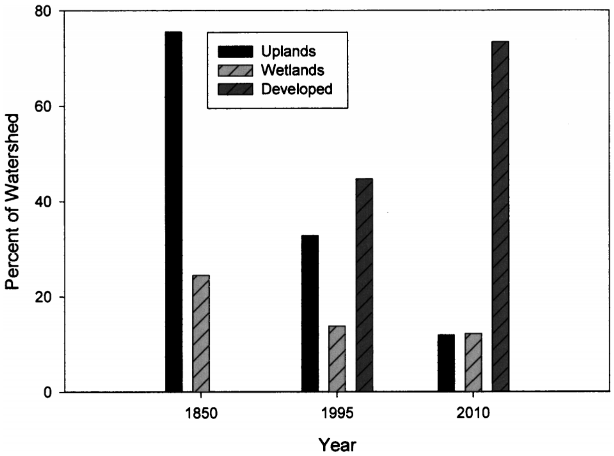

Fig. 2. Percent of Lemon Bay watershed in uplands, wetlands, and developed land use categories (see text for explanation).

num-cobalt units, with many locations having no value recorded other than 5.00 (Table 2). Values did not vary between either dates or locations ($p > 0.05$; one-way ANOVA).

Seagrass Parameters

Shoot density values ranged between 56 and 450 shoots m^{-2}. Densities varied between locations ($p < 0.01$; Kruskal-Wallis) but not dates ($p > 0.05$; one-way ANOVA). Highest mean values were at IMBI and GPML, 347 and 337 shoots m^{-2}, respectively. Lowest mean values were at DPBI and SPML, 55 and 119 shoots m^{-2}, respectively.

Aboveground standing crop estimates ranged between 3.7 and 115.1 gdw m^{-2}. Values varied between locations ($p < 0.01$; Kruskal-Wallis) but not dates ($p > 0.05$; Kruskal-Wallis). Highest mean standing crop values were at IMBI and GPML, 68.3 and 58.9 gdw m^{-2}, respectively. Lowest mean values were at DPBI and SPML, 11.1 and 17.7 gdw m^{-2}, respectively.

Areal blade productivity values ranged between 0.08 and 2.55 gdw m^{-2} d^{-1}. Values varied both between locations ($p < 0.01$; Kruskal-Wallis) and dates ($p < 0.05$; Kruskal-Wallis). Highest mean areal blade productivity values were at IMBI and GPML, 1.42 and 1.37 gdw m^{-2} d^{-1}, respectively. Lowest mean values were at DPBI and SPML, 0.26 and 0.38 gdw m^{-2} d^{-1}, respectively.

Epiphyte abundance ranged between 5.5% and 197.1% of blade weight. Values varied between dates ($p < 0.01$; Kruskal-Wallis) but not locations ($p > 0.05$; Kruskal-Wallis). Highest mean epiphyte abundance values were at SPBI and DPBI, 82.4% and 70.9% of blade weight, respectively. Lowest mean values were at IMBI and GPML, 30.6% and 32.9% of blade weight, respectively.

TABLE 3. Nitrogen load estimates (kg TN) for nonpoint source loads, baseflow, direct rainfall, septic tank systems, and total loads for the dry season (October 1 to June 30), the wet season (July 1 to September 30), and total annual loads. 1850 = pre-development estimates, 1995 = estimates for the year 1995, 2010 A = estimates for the year 2010 with 90% of septic tank system loads removed, 2010 B = estimates for the year 2010 with no septic tank system loads removed.

	1850	1995	2010 A	2010 B
Dry Season				
Nonpoint Source	25,144	33,095	56,838	56,838
Baseflow	4	4	4	4
Rainfall	3,317	3,317	3,317	3,317
Septic Tank Systems	0	13,903	1,390	13,903
Total Loads	28,465	50,319	61,548	74,062
Wet Season				
Nonpoint Source	43,850	65,679	117,649	117,649
Baseflow	1	1	1	1
Rainfall	9,029	9,029	9,029	9,029
Septic Tank Systems	0	4,685	469	4,685
Total Loads	52,880	79,394	127,148	131,365
Annual				
Nonpoint Source	68,994	98,774	174,487	174,487
Baseflow	5	5	5	5
Rainfall	12,346	12,346	12,346	12,346
Septic Tank Systems	0	18,588	1,859	18,588
Total Loads	81,345	129,713	188,696	205,426

Nitrogen Loading Model

Land use mapping estimates using 1995 photography indicate that residential land use accounted for ca. 41% of the watershed, followed by forested/uplands (32%) and wetlands (14%). By 2010, residential land use is expected to account for 67% of the watershed, with forested/uplands declining to 12%, and wetlands down to 12% as well (Fig. 2). Forested/uplands were estimated to have comprised ca. 75% of the watershed in pre-development times, with wetlands comprising the remaining 25%.

The increased urbanization of the watershed is modeled to have caused an increase in freshwater inflow into Lemon Bay associated with the increased impervious nature of the landscape. Average annual runoff into Lemon Bay in pre-development times (1850) is modeled to have been approximately 7.10×10^{10} liters. The 1995 estimates are 19% higher (8.47×10^{10} liters), and estimates for 2010 suggest a further 17% increase (9.93×10^{10} liters).

Dry season nitrogen loads into Lemon Bay are estimated at 50,319 kg TN for 1995 conditions with 66% of the load (33,095 kg TN) coming from stormwater runoff (Table 3). Nitrogen loads from septic tank systems are 28% of the dry season load. Rainfall comprises 7% of 1995 loads with baseflow at < 1%. Estimated dry season nitrogen loads are 77% higher than loads in pre-development times.

Wet season nitrogen loads into Lemon Bay are estimated at 79,394 kg TN for 1995 conditions with 83% of the load (65,679 kg TN) coming from stormwater runoff (Table 3). Nitrogen loads from septic tank systems are ca. 6% of the wet season load. Rainfall is 11% of 1995 loads with baseflow at < 1%. Estimated wet season nitrogen loads are 50% higher than loads in pre-development times.

Total annual nitrogen loads into Lemon Bay are estimated at 129,713 kg TN yr^{-1} for 1995 conditions with 76% of the load (98,774 kg TN) coming from stormwater runoff (Table 3). Nitrogen loads from septic tank systems are estimated at 14% of the annual load, followed by rainfall at 10% of the annual load. Baseflow is < 1% of the annual load. Annual nitrogen loads are estimated to be ca. 59% higher than in pre-development times. With the removal of 90% of the septic tank system nitrogen load, annual nitrogen loads would be expected to increase ca. 45% from 1995 to 2010, to 188,696 kg TN yr^{-1}. If existing septic tank system nitrogen loads are not removed, annual nitrogen loads would be expected to increase ca. 58% from 1995 to 2010 to 205,426 kg TN yr^{-1} (Table 3).

After normalizing for the size of the contributing watershed, modeled nitrogen loads in pre-development times are estimated at 5.3 kg TN ha^{-1} yr^{-1}, versus 8.4 kg TN ha^{-1} yr^{-1} in 1995. Estimates for 2010 are 12.2 and 13.3 kg TN ha^{-1} yr^{-1}, respectively, depending on whether existing nitrogen loads from septic tank systems are removed or allowed to continue.

Optical Model

Linear regression revealed a significant relationship between light attenuation coefficients and chl *a* concentrations in Lemon Bay ($r^2 = 0.37$, $p < 0.01$; Fig. 3a). There was also a significant correlation between light attenuation coefficients and turbidity values ($r^2 = 0.33$, $p < 0.01$; Fig. 3b). No correlation was found between light attenuation coefficients and color ($p > 0.10$).

Using values for chl *a* and turbidity alone, attenuation coefficients can be predicted ($r^2 = 0.56$, $p < 0.01$; Fig. 3c). However, the best-fit line tends to overestimate light attenuation at the low range of observed values and underestimate light attenuation at the high range of observed values. Based on the results of the multiple regression analysis, the equation of the fitted model relating light attenuation coefficients to chl *a* concentrations and turbidity values (Fig. 3c) is:

$$k = 0.65 + 0.04 \times \text{chl } a\ [\mu g\ l^{-1}] + 0.06 \times \text{turbidity [NTU]}$$

Using site-specific mean attenuation coefficients and chl *a* concentrations, the percent of total light

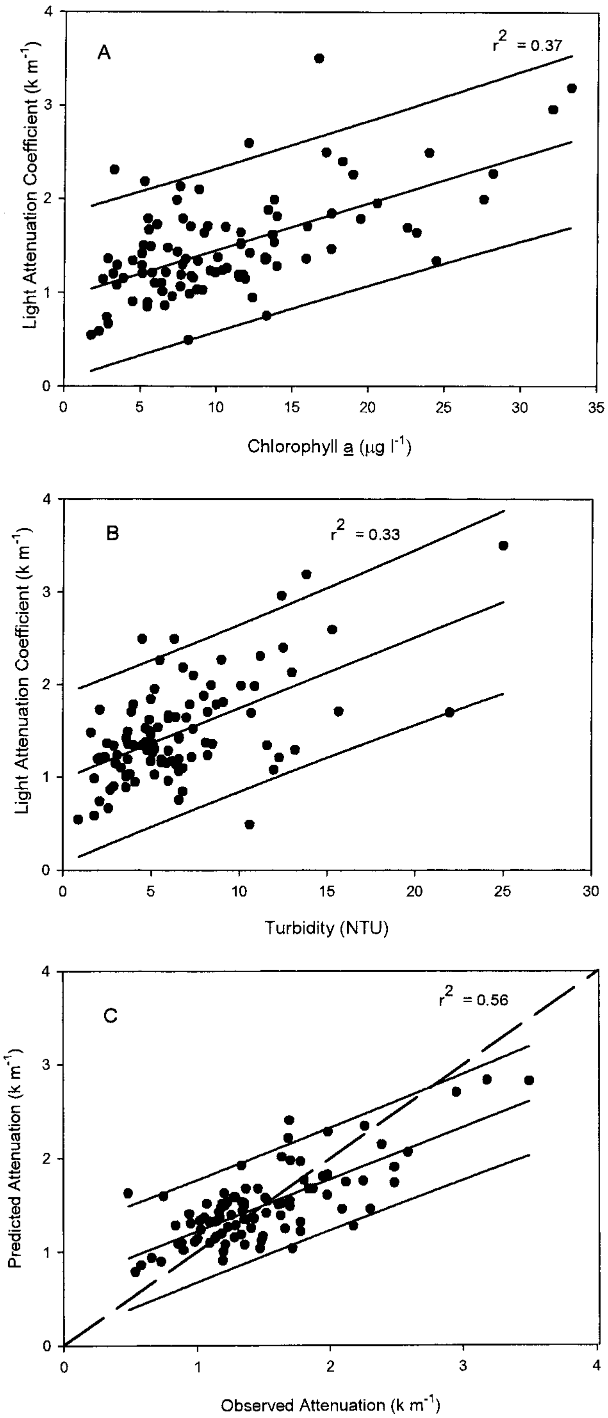

Fig. 3. A) light attenuation coefficients (k m^{-1}) versus chlorophyll *a* ($\mu g\ l^{-1}$) for all site and date combinations (n = 96). Line is best-fit relationship with 95% prediction limits (see text). B) light attenuation coefficients (k m^{-1}) versus turbidity (NTU) for all site and date combinations (n = 96). Line is best-fit relationship with 95% prediction limits (see text). C) predicted versus observed light attenuation coefficients (k m^{-1}) for all site and date combinations (n = 96). Line is best-fit relationship with 95% prediction limits (see text). Dashed line is 1:1 relationship.

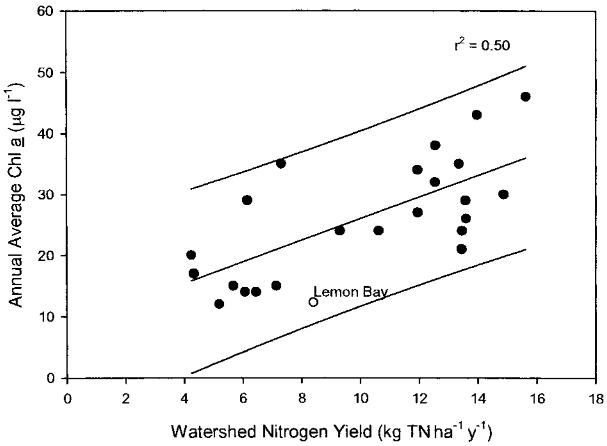

Fig. 4. Annual average chlorophyll *a* ($\mu g\ l^{-1}$) versus annual total nitrogen load (kg TN $ha^{-1}\ yr^{-1}$) for the years 1964 to 1990 in Hillsborough Bay, Florida. Lemon Bay values are represented by open symbol. Line is best-fit relationship with 95% prediction limits (see text).

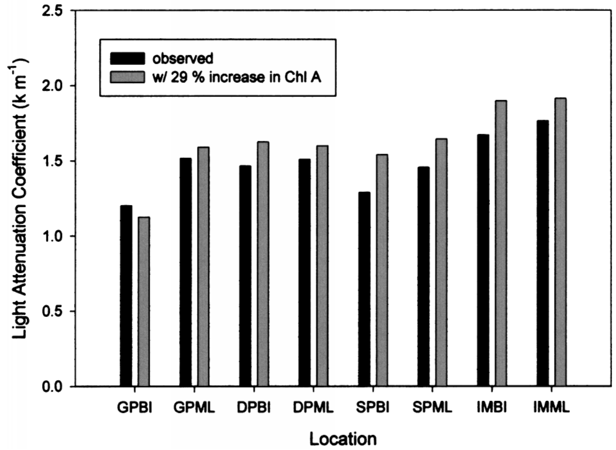

Fig. 5. Observed mean light attenuation coefficients (k m^{-1}) and predicted light attenuation coefficients with a 29% increase in chlorophyll *a* concentrations for all sites (see text). GPBI = Gasparilla Pass Barrier Island, GPML = Gasparilla Pass Mainland, DPBI = Don Pedro Barrier Island, DPML = Don Pedro Mainland, SPBI = Stump Pass Barrier Island, SPML = Stump Pass Mainland, IMBI = Indian Mound Barrier Island, and IMML = Indian Mound Mainland.

attenuation due to phytoplankton biomass was estimated as in McPherson and Miller (1987). Values ranged from 12.4% at GPBI to 38.5% at IMBI. The percent of light attenuation associated with phytoplankton levels for sites located next to the mainland (GPML, DPML, SPML, and IMML) was 27.0%, 29.0%, 26.7%, and 37.4%, respectively. The remaining sites (DPBI and SPBI) had values of 31.3% and 26.2%, respectively.

Nitrogen Loading:Chl *a* Relationship

Nitrogen loading rate estimates from Hillsborough Bay (in the northeastern portion of Tampa Bay) were normalized for watershed size to produce estimates in units of kg TN $ha^{-1}\ yr^{-1}$. Linear regression found a significant correlation between annual nitrogen loads and annual average chl *a* concentrations ($p < 0.01$, $r^2 = 0.50$, 21 df), as has been previously reported by Johansson (1991). Nitrogen loading estimates and mean chl *a* concentrations from Lemon Bay fit within the 95% prediction interval for the empirical relationship developed for Hillsborough Bay (Fig. 4). The relationship between annual average chl *a* concentrations in Hillsborough Bay and annual nitrogen loads can be expressed as:

$$\text{chl } a\ [\mu g\ l^{-1}] = 8.24 + (1.78 \times \text{nitrogen load}\ [\text{kg TN ha}^{-1}\ \text{yr}^{-1}])$$

Using the best-fit equation for linear regression of the Hillsborough Bay data set, a 45% increase in nitrogen loading rates (2010 A scenario nitrogen loads versus 1995 nitrogen loads) would be expected to increase annual average chl *a* concentrations by 29%, as the predicted annual average chl *a* concentration for the 2010 A scenario is 29% higher than the predicted chl *a* concentration for the 1995 nitrogen loading estimate.

A prediction was then made that a future 45% increase in Lemon Bay nitrogen loads would be expected to increase the annual average chl *a* concentration in Lemon Bay by 29%, and future light attenuation coefficients were predicted for individual locations with a 29% increase in the site-specific mean chl *a* concentration (Fig. 5). The mean increase in light attenuation coefficients associated with a 29% increase in chl *a* concentrations is 8.7%. Perhaps due to errors associated with predicting light attenuation coefficients, values at GPBI, where observed phytoplankton levels account for only 12.4% of total light attenuation, are slightly lower for the future scenario than the presently observed value (Fig. 5).

Seagrass Depth Distributions

The calculated percent of subsurface irradiance at the deep edges of the sampled seagrass meadows ranged between 15.2% and 21.1% of subsurface irradiance for sites GPML, DPBI, DPML, SPBI, IMBI, and IMML with a mean value for these six locations of 18.2%. Mean values for SPML and GPBI were significantly higher: 31.8% and 45.4% of subsurface irradiance, respectively. The values for SPML and GPBI are also much higher than previous estimates of 20% to 25% of subsurface irradiance as the minimum light requirement for *T. testudinum* in Tampa Bay (Dixon 1999), Sarasota Bay (Dixon and Kirkpatrick 1995), and Charlotte

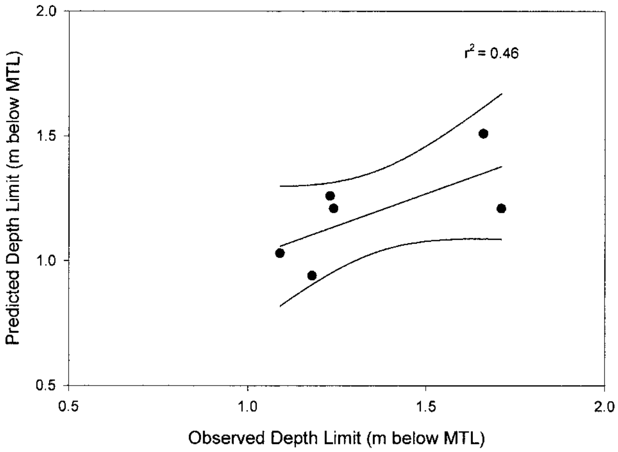

Fig. 6. Predicted versus observed depth limits (m below MTL) for sites GPML = Gasparilla Pass Mainland, DPBI = Don Pedro Barrier Island, DPML = Don Pedro Mainland, SPBI = Stump Pass Barrier Island, IMBI = Indian Mound Barrier Island, and IMML = Indian Mound Mainland. Line is best-fit relationship with 95% confidence intervals (see text).

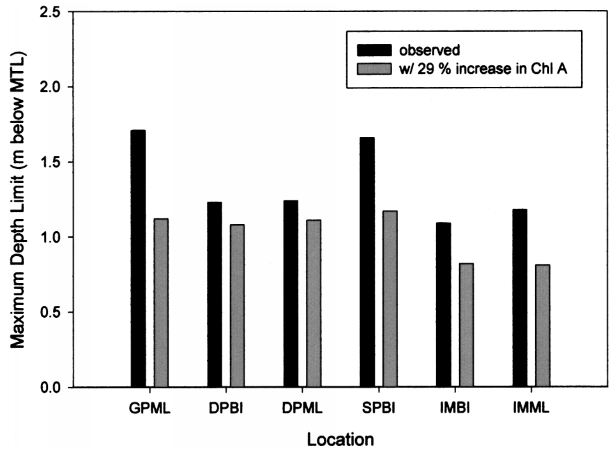

Fig. 7. Observed maximum depth limits (m below MTL) and predicted maximum depth limits with a 29% increase in chlorophyll *a* concentrations for sites GPML = Gasparilla Pass Mainland, DPBI = Don Pedro Barrier Island, DPML = Don Pedro Mainland, SPBI = Stump Pass Barrier Island, IMBI = Indian Mound Barrier Island, and IMML = Indian Mound Mainland.

Harbor (Dixon and Kirkpatrick 1999b). Values for SPML and GPBI were not used for estimating potential changes in depth distribution as related to potential changes in water clarity. By using the mean percent of subsurface irradiance at the deep edge for the six remaining locations (18.2%) as an estimate of minimum light requirements and using mean attenuation coefficients for individual sites, predicted depth limits were found to be significantly correlated with observed depth limits ($p < 0.05$, $r^2 = 0.46$; Fig. 6).

Figure 7 compares observed depth limits with estimated depth limits if light attenuation coefficients were to increase due to a 29% increase in chl *a* concentrations at individual sites. Predicted reductions in depth limits ranged from 10% to 35% (DPML and GPML, respectively), with the average reduction for all light-limited sites (i.e., excluding GPBI and SPML) being 24%.

Areal blade productivity (gdw m^{-2} d^{-1}) was not correlated with the percent of subsurface irradiance reaching the depths at which seagrass sampling efforts took place ($p > 0.05$). Areal blade productivity was weakly yet significantly associated with water temperature ($p < 0.01$, $r^2 = 0.11$).

Discussion

Water Quality

Water clarity in Lemon Bay was most probably influenced by flushing rates, as the lowest mean light attenuation coefficients were at the two sites closest to passes (GPBI and SPBI), and the highest mean attenuation coefficients were at sites (IMBI and IMML) located in a null zone for circulation (i.e., Sheng and Peene 1992). The importance of phytoplankton biomass, as related to water clarity in Lemon Bay, is illustrated both by the correlation between light attenuation and chl *a* concentrations (Fig. 3a) and the finding that locations with the greatest water clarity (GPBI and SPBI) also had the lowest chl *a* concentrations.

Residence time appears to be an important factor relating to phytoplankton biomass in Lemon Bay, as the four sites located in null zones for circulation (DPBI, DPML, IMBI, and IMML) had the highest mean chl *a* concentrations. These findings are consistent with a previous review by Nixon et al. (1996) who examined relationships between nutrient loads, flushing rates, and phytoplankton biomass for various estuaries in the North Atlantic Ocean.

Using an empirical modeling approach developed by McPherson and Miller (1987), the calculated mean percent of light attenuation due to phytoplankton biomass in Lemon Bay ranged from 12% (GPBI) to 39% (IMBI). The mean percent of light attenuation due to phytoplankton for all site and date combinations was 29%. This value is similar to estimates from Tampa Bay (27%; McPherson and Miller 1993) but higher than in Sarasota Bay, where the mean percent of light attenuation due to phytoplankton biomass ranged between 4% and 12% (Dixon and Kirkpatrick 1995). In Charlotte Harbor, the mean percent of light attenuation due to phytoplankton biomass was 4% (Dixon and Kirkpatrick 1999b). In contrast to Lemon Bay, water clarity in adjacent Charlotte Harbor appears to vary mostly in response to rainfall-derived changes in the amount of dissolved organic matter brought into Charlotte Harbor from its substantially larger watershed (see Tomasko and Hall 1999). The partial attenuation coefficient for phytoplankton derived in this study ($k = 0.041 \times$ chl *a* [in $\mu g\ l^{-1}$]) is nearly three times higher than the commonly-cited coefficient developed by Lorenzen (1972), perhaps reflecting the estuarine locations that were the source of data for this study as opposed to the marine locations used by Lorenzen (1972).

Seagrass Parameters

Short shoot densities, aboveground standing crop and areal blade productivity of *T. testudinum* from this study were similar to values reported from Tampa Bay (Dawes and Tomasko 1988; Dixon 1999) and Sarasota Bay (Tomasko et al. 1996). Aboveground standing crop and areal blade productivity values in Lemon Bay were mostly higher than values reported for Charlotte Harbor (Tomasko and Hall 1999), perhaps reflecting the lower light attenuation coefficients of Lemon Bay as compared to Charlotte Harbor. Areal blade productivity was correlated with water temperature, but not percent of subsurface irradiance at sample depths, similar to results from Charlotte Harbor (Tomasko and Hall 1999).

Nitrogen Loading Model

Annual nitrogen loads to Lemon Bay were modeled to have increased by 59% versus pre-development (1850) conditions, a rate of increase lower than estimates from Sarasota Bay (200% increase; Heyl 1992). Nitrogen loads appear to be dominated by nonpoint sources (i.e., stormwater runoff), as has been previously reported for Sarasota Bay (Heyl 1992) and Charlotte Harbor (Squires et al. 1998).

While nitrogen loads from septic tank systems were only 14% of the total annual load, they were estimated to contribute 28% of the dry season nitrogen load (Table 3). The increased importance of septic tank-derived nitrogen loads in the dry season (as compared to annual loads and wet season loads) is mostly due to the decrease in nonpoint source loads that occurs due to decreased rainfall and runoff volumes during the dry season. Although the removal of septic tank nitrogen loads into Lemon Bay may not have a dramatic effect on nitrogen loads on an annual basis, this activity might have a more substantial effect during the 9 mo (October 1 to June 30) of Lemon Bay's typical dry season.

The estimated nitrogen yield for Lemon Bay's watershed (8.4 kg TN $ha^{-1}\ yr^{-1}$) is higher than yields independently calculated for the watersheds of Charlotte Harbor and Tampa Bay (2.6 and 5.9 kg TN $ha^{-1}\ yr^{-1}$, respectively; Stacey et al. 2001), but lower than the yield calculated for Sarasota Bay's watershed (10.9 kg TN $ha^{-1}\ yr^{-1}$; Stacey et al. 2001). Lemon Bay's 1995 watershed nitrogen yield is substantially lower than yields estimated for various mid-Atlantic estuaries (range of 15.1 to 27.2 kg TN $ha^{-1}\ yr^{-1}$; Stacey et al. 2001). The best-case scenario for Lemon Bay's 2010 estimated watershed nitrogen yield (12.2 kg TN $ha^{-1}\ yr^{-1}$) is higher than Sarasota Bay's present-day yield, and just slightly lower than Delaware Bay's estimated present-day yield (15.1 kg TN $ha^{-1}\ yr^{-1}$; Stacey et al. 2001).

Nitrogen Loading:Chl *a*:Seagrass Depth Distribution Relationships

Based on an empirically-derived nitrogen load: chl *a* relationship developed for a portion of nearby Tampa Bay, the expected best-case scenario for future nitrogen load increases (45%) would be expected to result in a 29% increase in mean chl *a* concentrations. If the estimate of a 29% increase in expected chl *a* concentrations is used, the mean light attenuation coefficient for all sites would increase by 9% (Fig. 5).

As was previously found for one of two *T. testudinum* meadows in the Tampa Bay area studied by Dawes and Tomasko (1988), it appears that the seagrass meadows at GPBI and SPML are limited in their depth distribution by factors other than light availability alone. The presence of nearby boating channels is perhaps responsible for preventing seagrass meadows in these two areas from growing down to the depths they could reach if water clarity was the only factor limiting their offshore development. The mean percent of subsurface irradiance at the deep edges at the other six locations (18.2%) matches up fairly well with the estimated minimum light requirement for *T. testudinum* in Tampa Bay (20.1% to 23.4% of subsurface irradiance; Dixon 1999) and Sarasota Bay (21% of subsurface irradiance; Dixon and Kirkpatrick 1995). It should be noted that the role of epiphyte coverage in further reducing light availability was not examined in this study (see Dixon 1999).

Depth limits for the seagrass meadows in Lemon Bay (excluding GPBI and SPML) could be predicted based on water clarity alone (Fig. 5). The reduction in depth limits that would be brought about by a 29% increase in chl *a* concentrations was examined to estimate the reduction in seagrass coverage that might be expected to occur with future water quality degradation. The range of expected reductions was 10% to 35% of present depth limits, with a mean reduction of 24%.

There is a reasonable comfort level with predicting the average percent reduction in depth limits that would be expected to occur for Lemon Bay's light-limited seagrass meadows, based on expected reductions in water clarity in the year 2010. There is more uncertainty when attempting to predict the change in biomass and productivity that might also occur in remaining seagrass meadows with a reduction in water clarity. The inverse relationship found between watershed nitrogen loads and seagrass biomass and productivity in adjacent Sarasota Bay (Tomasko et al. 1996) suggests that Lemon Bay would not only be expected to have fewer acres of seagrass meadows by the year 2010, but remaining seagrass beds would probably become sparser and less productive as well.

Conclusions

Further increases in the degree of urbanization of Lemon Bay's watershed are expected by the year 2010. This increased urbanization is expected to generate increased nitrogen loads into Lemon Bay, mostly as a function of increased nonpoint source pollution. While septic tank systems are not as important a source of nitrogen loads as nonpoint source pollution, they appear to be considerably more important during the 9-mo dry season than the 3-mo wet season. With or without a septic tank system replacement program, Lemon Bay is expected to experience degraded water clarity by the year 2010, due to nitrogen-fueled increases in phytoplankton biomass. Using an estimate of the degree of water clarity degradation expected, the deep edges of light-limited seagrass meadows in Lemon Bay would retreat by 24%. Concurrent with future reductions in seagrass acreage, remaining meadows would be expected to become sparser and less productive. The combination of fewer, sparser, and less productive seagrass meadows would almost certainly result in a decline in the abundance of seagrass-dependent species of finfish and shellfish in Lemon Bay.

Acknowledgments

The authors thank the Governing Board of the Southwest Florida Water Management District and the Florida Department of Environmental Protection for financial and logistical support for this effort. Thanks are given to the members of the Charlotte Harbor Surface Water Improvement and Management (SWIM) Program's Technical Advisory Committee for advice and comment throughout the course of this project. Special thanks are given to Dr. Tom Frazer (University of Florida) for his critical review of light attenuation data, Dr. Penny Hall for logistical support and the use of laboratory space, and to the anonymous reviewers whose comments have resulted in a much improved manuscript.

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SOURCE OF UNPUBLISHED MATERIALS

ENGLEWOOD WATER DISTRICT. Personal Communication. 201 Selma Avenue, Englewood, Florida 34223.

Received for consideration, April 14, 2000
Accepted for publication, May 16, 2001