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Spatial and temporal variation in seagrass coverage in Southwest Florida: assessing the relative effects of anthropogenic nutrient load reductions and rainfall in four contiguous estuaries

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Abstract

The estuaries of Tampa Bay, Sarasota Bay, Lemon Bay, and Upper Charlotte Harbor are contiguous waterbodies located within the subtropical environment of Southwest Florida. Based on an examination of rainfall data over the period of record (1916–2001) within the watersheds of these estuaries, there is no evidence for spatial differences (at the watershed level) or monotonic trends in annual rainfall. During the 1980s, nitrogen loads into Tampa Bay and Sarasota Bay (generated primarily by domestic wastewater treatment facilities) were reduced by 57% and 46%, respectively. In response, both Tampa Bay and Sarasota Bay have lower phytoplankton concentrations, greater water clarity and more extensive seagrass coverage in 2002 than in the early 1980s. As there is no evidence of a concurrent trend in rainfall during the period of 1982–2001, it is unlikely that variation in rainfall can account for the observed increase in seagrass coverage in these two bays. In contrast, seagrass coverage has remained relatively constant since the mid 1980s in Lemon Bay and Charlotte Harbor. Domestic wastewater treatment facilities are minor sources of nitrogen to Lemon Bay, and water clarity in Charlotte Harbor varies mostly as a function of dissolved organic matter and non-chlorophyll associated turbidity, not phytoplankton levels. Even in estuaries that share boundaries and are within 100 km of each other, varied responses to anthropogenic changes and natural phenomena were observed in water quality and associated seagrass extent. Resource management strategies must take into account system-specific factors—not all strategies will result in similar results in different systems.

Keywords: Seagrass; Water quality; Eutrophication; Florida; Estuaries

1. Introduction

Between 1950 and 2000, Florida's population increased from approximately 2.7 million to nearly 16 million residents, with 3 million new residents moving to the state during the 1990s alone (Letson, 2002). More than 70% of Floridians live in coastal counties, which is where most of the state's building permits are issued, as well (Florida Department of Community Affairs [FDCA] 1996). Waterborne trade, beach-related tourism, and land development are much larger contributors to the economy of Florida's coastal counties than commercial fishing, with annualized net worths of $47 billion, $15 billion, $11 billion, and $202 million, respectively (FDCA, 1996). With an estimated economic impact of just under $3 billion a year (Letson, 2002),

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even the value of recreational fishing activities (both residents and visitors) pales beside that of more intensive uses of Florida's natural resources. As the intrinsic value of a clean environment is difficult to quantify (Pearce, 1993), successful management and protection of Florida's natural resources is often done despite, rather than because of, the economic pressures of continued development. Tools that can be used to track the effectiveness of efforts to restore and/or protect water quality have tremendous value, given this scenario.

In Southwest Florida, a substantial amount of research has focused on the relationships between human population, land use patterns, pollutant loads, estuarine water quality, and seagrass health. In Tampa Bay, historical losses of seagrass coverage have been linked to both direct and indirect impacts (i.e., Lewis et al., 1985; Lewis, 1989; Haddad, 1989). In contrast, recent (1982–1996) increases in seagrass coverage in Tampa Bay have been linked to improved water quality in the bay (i.e., Johansson, 1991; Johansson and Ries, 1997; Lewis et al., 1998; Johansson and Greening, 1999). Improvements in the early 1980s in the treatment and disposal of wastewater discharges by the Cities of Tampa, St. Petersburg and Clearwater have been identified as major causes of improved water quality in Tampa Bay (Johansson and Greening, 1999). A similar situation exists in Sarasota Bay, where recent (1988–1996) increases in seagrass coverage are most likely related to reductions in anthropogenic nitrogen loads to the bay by the City of Sarasota and Manatee and Sarasota Counties (Kurz et al., 1999). However, significant nonpoint sources of nitrogen loads still exist, especially urban stormwater runoff and septic tank systems.

In contrast to Tampa and Sarasota Bays, neither Lemon Bay nor Upper Charlotte Harbor (i.e., waters north of 26°40′ N latitude) seem to have experienced any substantial changes in seagrass coverage during the period of 1982–1996 (Kurz et al., 1999; Tomasko et al., 2001). The relatively stable seagrass coverage in Upper Charlotte Harbor in recent years could be related to the comparatively minor importance of phytoplankton populations to overall light attenuation in the Harbor (McPherson and Miller, 1987). That is, the relationship between pollutant loads and seagrass meadows may not be as recognizable in Upper Charlotte Harbor as in Tampa Bay (Tomasko and Hall, 1999).

This paper will review some of the information available relating to pollutant loads and water quality for contiguous Southwest Florida estuaries, to determine the relationships (if any) between anthropogenic versus natural phenomena and temporal variation in seagrass coverage. Also, this paper updates a previous summary of the status and trends in seagrass coverage in Southwest Florida (Kurz et al., 1999) by including data from two additional mapping events, in 1999 and 2002.

2. Materials and methods

2.1. General description of locations

For purposes of this paper, the following estuaries will be considered: 1) Tampa Bay, 2) Sarasota Bay, 3) Lemon Bay, and 4) Upper Charlotte Harbor (Fig. 1). The region "Upper Charlotte Harbor" will include only those areas north of 26°40′ N latitude. The climate in this portion of Southwest Florida is subtropical, with warm, wet summers and mild, dry winters. Annual average temperatures range between 21 and 24 °C, depending upon distance from the coast and latitude (Southwest Florida Water Management District [SWFWMD] 1999). Mean annual rainfall usually ranges between 136 and 144 cm year^{-1}, with more than half that amount occurring during the typical wet season of June to September (SWFWMD, 1999).

The different estuaries vary in terms of both the size of the open waters of each system, as well as the size of each system's watershed (Table 1). With a watershed:open water ratio of 24.3:1, Upper Charlotte Harbor experiences a much greater influence of freshwater inflow than Sarasota Bay (ratio of 2.9:1). Tampa Bay and Lemon Bay have similar watershed:open water ratios (6.2:1 and 4.9:1, respectively); values that are intermediate between those for Sarasota Bay and Upper Charlotte Harbor.

2.2. Pollutant loading models

Manipulative studies in Upper Charlotte Harbor have shown nitrogen to be the principal nutrient limiting primary production (Montgomery et al., 1991). In addition, strong empirical evidence points to the importance of nitrogen in controlling phytoplankton biomass in Tampa Bay (e.g., Johansson, 1991; Wang et al., 1999). In Lemon Bay, nitrogen loads appear to be related to water column chlorophyll *a* values (Tomasko et al., 2001). Consequently, pollutant loading models for these four estuaries focus on nitrogen as the primary nutrient of concern.

Nitrogen loading estimates for Tampa Bay, Sarasota Bay, Lemon Bay, and Upper Charlotte Harbor combine both measured nitrogen loads and estimated loads from difficult to quantify sources. For Tampa Bay, approximately 57% of the watershed is gaged for both flow and water quality, allowing for direct estimates of loads. However, much of the developed portions of these same watersheds drain directly to the bay, downstream of any gages. In situations such as this, loads from stormwater runoff are estimated using predictions based on rainfall, land use, soils, and seasonal land-use-specific water quality concentrations (Greening and Janicki, unpublished data; Pribble et al., 2001). In Upper Charlotte Harbor, 87% of the Peace River's watershed is gaged,

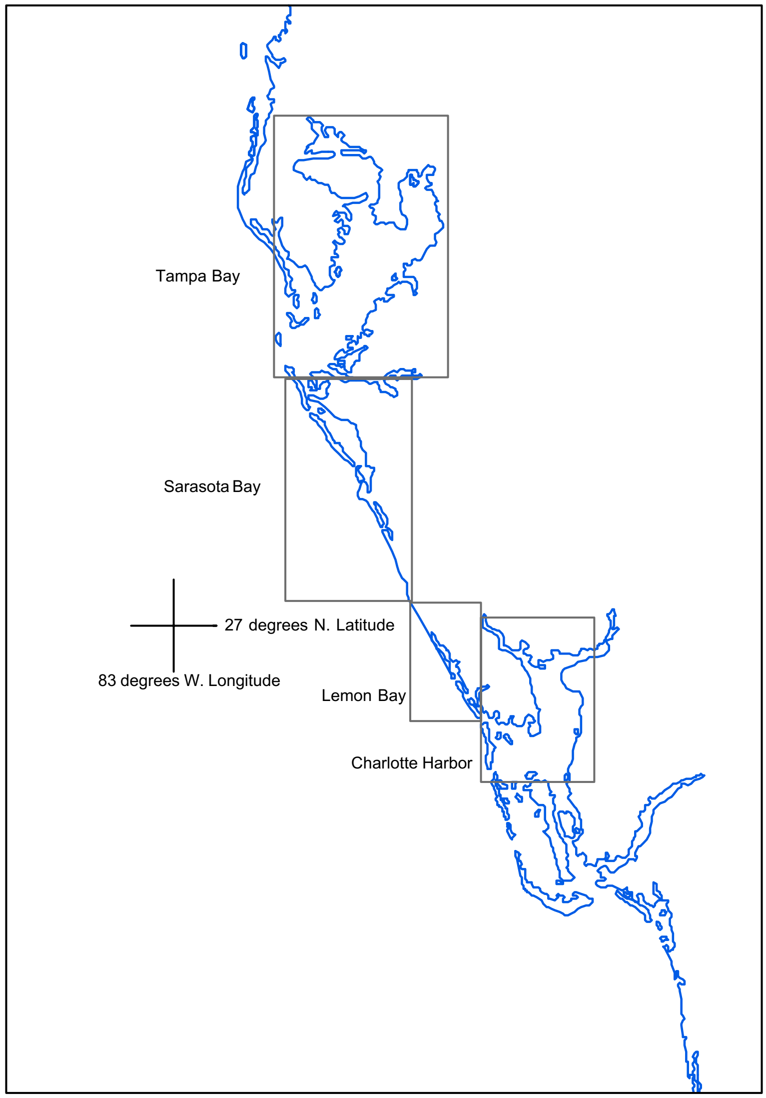

Fig. 1. Location of Tampa Bay, Sarasota Bay, Lemon Bay, and Upper Charlotte Harbor within Southwest Florida. Note that the term "Upper Charlotte Harbor" refers to those areas north of 26°40′ N latitude.

Table 1
Estimates of watershed size (km^2), open water (km^2), and the watershed to open water ratio by estuary

Estuary	Watershed (km^2)	Open water (km^2)	Watershed:open water ratio
Tampa Bay	5896	959	6.2
Sarasota Bay	389	135	2.9
Lemon Bay	154	31	4.9
Charlotte Harbor	8549	337	24.3

Data for Tampa Bay are from Pribble et al. (2001). Data for Sarasota Bay are from Tomasko et al. (1996). Data for Lemon Bay are from Tomasko et al. (2001). Data for Charlotte Harbor are from Squires et al. (1998), with open water estimate from Tomasko (unpublished data).

and approximately 50% of the Myakka River's watershed is gaged (Squires et al., 1998). Consequently, the majority of nitrogen loads to Upper Charlotte Harbor from stormwater runoff are measured, not modeled. In contrast, the entirety of stormwater-related nitrogen load estimates for Sarasota Bay and Lemon Bay are from models of non-point source loads associated with various land use types (e.g., Tomasko et al., 1996; Tomasko et al., 2001, respectively).

In addition to stormwater runoff, other pollutant sources included in these models are point sources (both industrial sites and domestic wastewater treatment plants), atmospheric deposition of nitrogen onto open water, baseflow (i.e., uncontaminated groundwater), septic tanks, and, in the case of Tampa Bay, material losses from the substantial fertilizer processing and transport facilities along the shoreline.

For point source load estimates, values for individual facilities are calculated from measurements of discharge rates and constituent levels required for maintaining permit compliance (Greening and Janicki, unpublished data; Pribble et al., 2001; Squires et al., 1998). These loads are then summed for all point sources for each estuary.

In Tampa Bay, atmospheric deposition of nitrogen to the open waters of the bay were calculated by multiplying the volume of precipitation onto the bay by nitrogen concentrations in rainfall. Recent estimates include direct measurements of both wet and dry deposition (Poor et al., 2001). In Upper Charlotte Harbor, wet deposition load calculations were multiplied by a correction factor to account for dry deposition (Squires et al., 1998). In both Sarasota and Lemon Bays, estimates are for wet deposition only (Tomasko et al., 1996; Tomasko et al., 2001, respectively).

Baseflow (uncontaminated groundwater contributions) and septic tank system loads were estimated using various algorithms to quantify their impacts (e.g., Squires et al., 1998; Tomasko et al., 2001). These techniques all require significant extrapolation to derive watershed-level estimates of these diffuse sources of nutrient loads.

For Tampa Bay, nitrogen load estimates exist for the years 1938, 1976, a 1992–1994 average, and a 1995–1998 average (Pribble et al., 2001). 1938 is considered a relatively unimpacted or "baseline" condition, while 1976 is considered to be indicative of the most degraded condition of the bay (Johansson and Greening, 1999; Greening and Janicki, unpublished data). For Sarasota Bay, nitrogen load estimates are for the year 1990. Additional load estimates have been derived for the years 1890 and 1988. Estimates for 1890 are for a mostly undeveloped watershed, while 1988 is considered to be indicative of the most degraded condition in Sarasota Bay (Kurz et al., 1999). Nitrogen load estimates for Lemon Bay are for the years 1850 and 1995 (Tomasko et al., 2001), while estimates for Upper Charlotte Harbor are for the year 1992 (Squires et al., 1998).

2.3. Rainfall data

The SWFWMD collects and/or compiles rainfall data from 370 rainfall gage sites throughout its approximately 28,000 km^2 jurisdictional area. Data are available for various periods of record, although most regions have one or more rainfall gage sites that date back to at least the 1910s. For purposes of this paper, rainfall data were combined for all stations throughout the entirety of the particular estuary's watershed. For Sarasota and Lemon Bays, rainfall data were combined, as the individual watersheds are relatively small, compared to Tampa Bay and Upper Charlotte Harbor. For Upper Charlotte Harbor, rainfall data were combined from throughout the Peace River watershed, which is the largest ($\approx$6000 km^2) source of freshwater inflow into the Harbor.

2.4. Seagrass mapping efforts

Seagrass maps are produced through a multiple step process. First, aerial photography is obtained, usually in the late fall to early winter. This time of year is typically associated with both good water clarity (i.e., Dixon, 1999; Tomasko and Hall, 1999) and relatively high seagrass biomass (i.e., Tomasko et al., 1996; Dixon, 1999). Flights are not flown unless Secchi disk depths meet or exceed 2 m at sampled locations for each estuary, on the day that photography is shot. In addition, wind speed on the day photography is shot must be at or below 10 knots, and wave heights must be less than 60 cm.

Second, photointerpretation efforts are conducted in the field, to allow for the successful evaluation of distinct photographic signatures. Seagrass signatures are divided into two classes (continuous and patchy coverage), with a minimum mapping unit of 0.2 ha. Continuous and patchy polygons are delimited based on the interpretation of aerial photographs, with continuous polygons characterized as having less than 25% of the delimited polygon visible as unvegetated areas. Patchy polygons have between 25% and 75% of the delimited polygon visible as unvegetated areas.

Third, polygons are integrated into an ARC/Info program. For past efforts (i.e., 1988, 1990, 1992, 1994, and 1996), individual polygons were delineated on Mylar overlays, cartographically transferred using a Zoom Transfer Scope to USGS quadrangles, and then digitally transferred to an ARC/Info database for further characterization. For 1999 and 2002 seagrass maps, analytical stereo plotters were used for photointerpretation, as opposed to stereoscopes. This technique allows for the production of a georeferenced digital file of the photointerpreted images, without the need for additional photo-to-map transfer.

Fourth, hard copy plots are made of photointerpreted seagrass coverage, and randomly chosen points are identified for a post-map-production classification accuracy assessment. A hand-held Global Positioning System unit is used, along with the draft seagrass map and the latitude and longitude of the randomly located stations, to develop an unbiased determination of the map's classification accuracy. A 90% classification accuracy standard is required for these efforts, and 94% accuracy was achieved for 2002 efforts (i.e., 64 of 68 stations that could be visited were accurately described).

Historical (i.e., 1950 and 1982) estimates of seagrass coverage had been developed by the Tampa Bay Regional Planning Council (1986) using 1:24,000 scale true color aerial photographs. A similar approach was used for Sarasota Bay for 1950 coverage estimates.

As an assessment of the "repeatability" of seagrass polygon delineations, an exercise to quantify mapping errors was introduced for the 2002 effort. Each photointerpreter was given five separate aerial photos, and asked to delimit seagrass coverage for each photo three separate times. All photos were interpreted at random before an individual photo was re-interpreted, then this procedure was repeated again. Seagrass coverage per photo varied from approximately 2 ha to >900 ha. Means and standard deviations of coverage were derived for each of the five photos ($n = 3$), for each of two photointerpreters used, and the coefficient of variation was calculated for each of 10 trials (2 photointerpreters × 5 photos). Pooling for all photo and photointerpreter combinations, the mean coefficient of variation was 1.9%.

2.5. Statistical analyses

Rainfall data were tested for potential differences between watersheds and trends over time using annual averages for pooled data by watershed. When testing for differences between watersheds, pooled data from the period of record (1916–2001) were compared using ANOVA, with significance set at $p < 0.05$. Data were tested for normality and homogeneity of variance, prior to using parametric statistical analysis. For testing for potential trends over time, data were analyzed for three time intervals: (1) period of record (1916–2001), (2) 1950–2001, and (3) 1982–2001. The time period of 1950–2001 encompasses the period of rapid population growth in Southwest Florida (Letson, 2002), while the time period of 1982–2001 encompasses the time period associated with significant improvements in water clarity, at least for Tampa Bay and Sarasota Bay (Johansson and Greening, 1999; Kurz et al., 1999, respectively). Data were tested for monotonic trends over time with a linear regression model of annual rainfall vs. years, with significance set at $p < 0.05$.

For seagrass coverage, data from individual estuaries were plotted versus years, and a variety of regression techniques were tested to determine if coverage varied over time in any predictable manner. Regression equations tested included the following: (1) linear, (2) power, (3) exponential, (4) second-order polynomial, and (5) third-order polynomial. Lines of best fit, when present, represent the regression equation between seagrass coverage and date with the highest r^2 value, with statistical significance set a priori at $p < 0.05$.

3. Results

3.1. Nitrogen loads

Estimated nitrogen loads into Tampa Bay and Sarasota Bay show evidence for both substantial increases over "baseline" conditions, as well as significant reductions over the past 20 years (Table 2). For Tampa Bay, nitrogen loads are estimated to have increased more than fivefold between 1938 and 1976, while loads into Sarasota Bay increased by nearly fivefold between 1890 and 1988. Between 1976 and 1992–1994, annual average nitrogen loads into Tampa Bay were reduced by approximately 57%. Subsequent increases to 1995–1998 annual average nitrogen load levels are associated with high rainfall amounts and resulting stormwater-related nitrogen load increases during the 1997–1998 El Niño event. Nitrogen loads into Sarasota Bay decreased by 46% between 1988 and 1990. Lemon Bay also shows evidence of substantial increases in nitrogen loads over historical levels (1850–1995), although the magnitude of the increase (59%) is much less than increases in Tampa Bay and Sarasota Bay. Load estimates for Upper Charlotte Harbor are for the year 1992 only.

Table 2
Estimated nitrogen loads by estuary for different time periods

Estuary	Year(s)	Nitrogen load (kg TN year^{-1})
Tampa Bay	1938	1,737,288
	1976	7,984,909
	1992–1994	3,447,360
	1995–1998	4,653,858
Sarasota Bay	1890	191,419
	1988	905,386
	1990	488,981
Lemon Bay	1850	81,345
	1995	129,713
Charlotte Harbor	1992	1,632,960

Data are summarized from existing nitrogen loading models. Data for Tampa Bay are from Pribble et al. (2001). Data for Sarasota Bay are from Tomasko et al. (1996). Data for Lemon Bay are from Tomasko et al. (2001). Data for Charlotte Harbor are from Squires et al. (1998). See Materials and Methods for details.

The substantial nitrogen load reductions for Tampa Bay between 1976 and the early 1990s, and Sarasota Bay between 1988 and 1990, are mostly due to significant improvements in the treatment of domestic wastewater discharges (Johansson and Greening, 1999; Kurz et al., 1999). As a result of these large-scale reductions in point source nitrogen loads, other nitrogen load sources have become proportionally more important (Fig. 2). In 1976, point source nitrogen loads were approximately 67% of the total load into Tampa Bay. By the mid to late 1990s, point sources had declined such that they were only approximately 12–13% of the bay-wide load. For Sarasota Bay, point source nitrogen loads were approximately 51% of the total load into the bay in 1988. By 1990, point sources contributed only approximately 10% of the nitrogen load into Sarasota Bay.

3.2. Rainfall

Annual rainfall amounts for pooled data over the period of record varied between 83.7 and 216.2 cm.

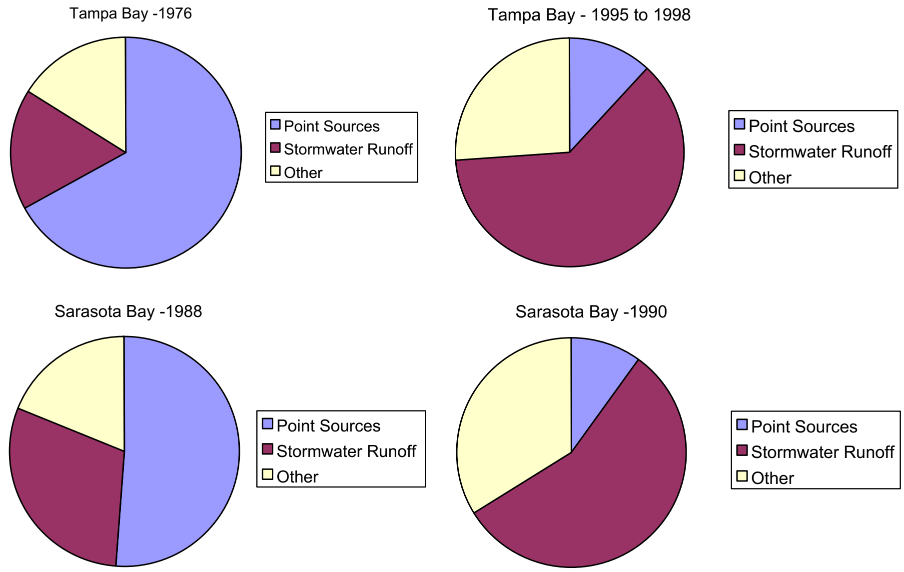

Fig. 2. Sources of nitrogen loads (kg TN year^{-1}) for Tampa Bay in 1976 and the 1995–1998 annual average, and for Sarasota Bay in 1988 and 1990. "Point Sources" includes both industrial sites and domestic wastewater treatment plants. "Stormwater Runoff" includes both gaged and ungaged surface flows. "Other" includes atmospheric deposition, baseflow, septic tank systems, etc. (see materials and methods for details). Data are summarized from Pribble et al. (2001) for Tampa Bay, and summarized from Tomasko et al. (1996) for Sarasota Bay.

Mean annual rainfall over the period of record for the Tampa Bay watershed was 131.4 cm. For the Sarasota Bay and Lemon Bay watershed, mean annual rainfall was 133.4, while the mean annual rainfall for the Upper Charlotte Harbor watershed was 133.0 cm. ANOVA did not detect a difference in mean annual rainfall between the Tampa Bay, Sarasota Bay plus Lemon Bay, and Upper Charlotte Harbor watersheds, when comparing data over the period of record ($p = 0.81$, 2 d.f.; see Fig. 3). In addition, no monotonic trends were found (using linear regression) for any watershed for any of the three chosen time intervals: (1) period of record (1916–2001), (2) 1950–2001, and (3) 1982–2001. Overall,

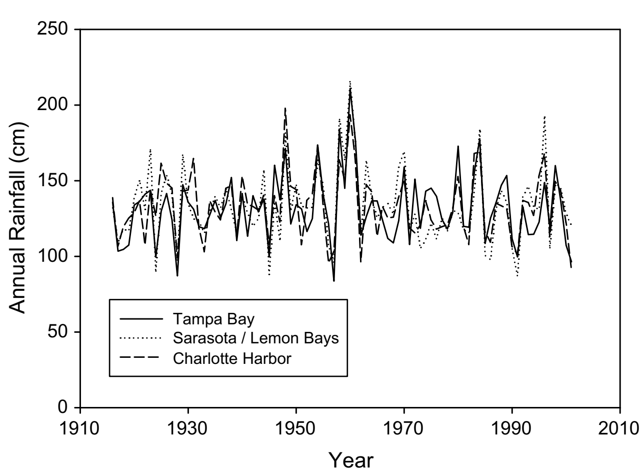

Fig. 3. Annual rainfall (cm year^{-1}) for pooled data from gages within the watersheds of Tampa Bay, Sarasota Bay plus Lemon Bay, and Upper Charlotte Harbor from 1916 to 2001. Data from SWFWMD.

the data suggest no spatial differences or long-term monotonic trends exist for annual rainfall in the watersheds of Tampa Bay, Sarasota and Lemon Bays, and Upper Charlotte Harbor. Using the best-fit forecasting tool for pooled data over the period of record (based on linear regression of the reciprocal of annual rainfall vs. years), the 95% prediction limit for forecasted rainfall ranges between 99 and 187 cm year^{-1}.

3.3. Seagrass coverage

Seagrass coverage varies both spatially and temporally in Southwest Florida. Combined coverage for all estuaries in 2002 was 22,689 ha. Tampa Bay contained 46% of the total mapped seagrass coverage in 2002, followed by Upper Charlotte Harbor, Sarasota Bay and Lemon Bay (33%, 16%, and 5%, respectively). Seagrass coverage in the combined estuaries expanded by 1262 ha between 1988 and 2002, an increase of 6%. The pattern of seagrass coverage over time varied between the different estuaries (see Fig. 4a–d). Additionally, from 1988 through 2002, the "continuous" cover category for all estuaries combined increased by approximately 4212 ha, indicating some areas previously mapped as "patchy" had converted to meadows with a denser photographic signature. Implications of this increased continuous coverage are currently under study.

In Tampa Bay (Fig. 4a), a third-order polynomial regression suggests a pattern of seagrass loss (1950–1970s), followed by recovery (1970s–1990s), followed by a period of little additional change over time (1990s to present). In greater detail, coverage in Tampa Bay decreased by 46% between 1950 and 1982, followed by a

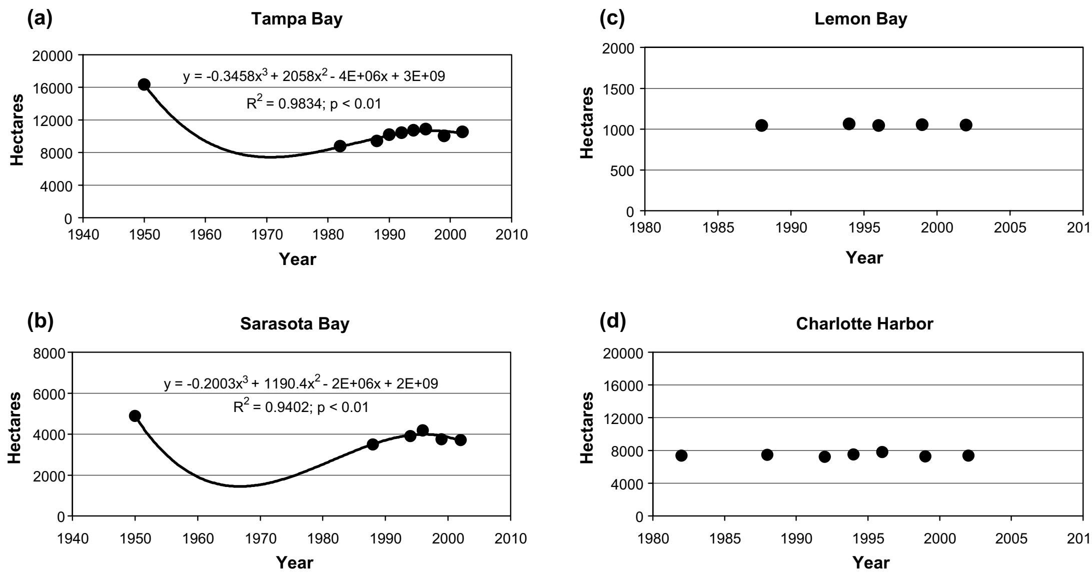

Fig. 4. Seagrass coverage estimates for (a) Tampa Bay, (b) Sarasota Bay, (c) Lemon Bay, and (d) Charlotte Harbor. Note different scales on axes. Lines represent statistically significant ($p < 0.05$) relationships between year and coverage (ha) based on third-order polynomial regression.

24% increase between 1982 and 1996. Between 1996 and 1999, coverage decreased by 8%, followed by a 5% increase between 1999 and 2002. Bay-wide, seagrass coverage in Tampa Bay in 2002 is 65% of 1950 coverage, but 20% higher than in 1982.

In Sarasota Bay (Fig. 4b), a third-order polynomial regression suggests a pattern of seagrass loss (1950–1970s) followed by a period of increased coverage (1970s–1990s), followed by a period of either stable or slightly declining coverage (1990s to present). In greater detail, seagrass coverage decreased by 28% between 1950 and 1988, followed by a 19% increase between 1988 and 1996. Between 1996 and 2002, coverage decreased by 11%. However, seagrass coverage increased by 214 ha between 1988 and 2002, a 6% improvement. In Lemon Bay (Fig. 4c), seagrass coverage in 2002 (1049 ha) is essentially unchanged from coverage in 1988 (1044 ha), and is probably within the margin of error for these mapping efforts (see Materials and Methods). No trends over time were detected for seagrass coverage.

In Upper Charlotte Harbor (Fig. 4d), seagrass coverage varied between a low value of 7216 ha in 1992 and peak coverage of 7781 ha in 1996. Seagrass coverage in 2002 (7381 ha) is essentially unchanged from coverage in 1982 (7369 ha), and also (as in Lemon Bay) is probably within the margin of error for these mapping efforts. As in Lemon Bay, no trends over time were detected for seagrass coverage.

4. Discussion

The estuaries of Tampa Bay, Sarasota Bay, Lemon Bay, and Upper Charlotte Harbor are all located in the subtropical environment of Southwest Florida. In this region, rainfall averages 132 cm year^{-1}, with approximately 60% of rainfall associated with convective storms, which are common phenomena in the afternoon hours of the typical June to September wet season (SWFWMD, 1999). Despite similar rainfall patterns in the watersheds of these four contiguous estuaries, seagrass coverage patterns and trends vary considerably.

In both Tampa and Sarasota Bays, recent increases in seagrass coverage have been linked to sustained improvements in water quality, which in turn have been related to significant reductions in anthropogenic nitrogen loads (e.g., Johansson, 1991; Johansson and Ries, 1997; Johansson and Greening, 1999; Kurz et al., 1999). Although estimated nitrogen loads into Tampa Bay and Sarasota Bay have decreased by 57 and 46%, respectively, in recent years, loads are still substantially higher than estimated values for "baseline" conditions (see Table 2). Most of the reductions in nitrogen loads have been due to improved management of effluent discharged from domestic wastewater treatment plants (Johansson and Greening, 1999; Kurz et al., 1999). In general, load reductions have accompanied the state-mandated effluent limitation requirements of the Grizzle-Figg Act (Section 403.086, Florida Statutes). This legislation, promulgated in 1980, requires effluent discharges from wastewater treatment plants in Southwest Florida to be below the levels of 5, 5, 3 and 1 (mg l^{-1}) for total suspended solids, biological oxygen demand, total nitrogen and total phosphorus, respectively (although Tampa Bay received a waiver for total phosphorus levels). Wastewater treatment plants have achieved the required reductions in effluent nitrogen concentrations through the use of linked nitrification and denitrification processes (Doug Taylor, City of Sarasota, personal communication).

As a result of these point source nitrogen load reductions, phytoplankton levels have decreased, and water clarity has increased, in both Tampa and Sarasota Bays in recent years (e.g., Johansson and Greening, 1999; Kurz et al., 1999, respectively). As there is no evidence for a trend in rainfall during the 1982–2001 time period, improvements in water clarity in these two estuaries do not appear to be due to reductions in nutrient loads due to changes in freshwater inflow. Previously, Johansson (1991) also found that variation in rainfall could not account for the observed positive trends in water clarity in Tampa Bay during the period of 1968–1990.

Increased water clarity in Tampa and Sarasota Bays in recent years has resulted in increased seagrass coverage since the 1980s (Fig. 4a and b). In contrast to Tampa and Sarasota Bays, seagrass coverage in Lemon Bay and Upper Charlotte Harbor appears to be non-trending since the 1980s (Fig. 4c and d). In Lemon Bay, the relatively stable pattern of seagrass coverage since the 1980s may be at risk, if pollutant loads increase and water quality declines in the future, as in a scenario outlined in Tomasko et al. (2001). In Upper Charlotte Harbor, water clarity varies mostly as a function of variation in the levels of dissolved organic matter and non-chlorophyll associated turbidity in the water column (McPherson and Miller, 1987). Seasonal variation in rainfall amounts and streamflow results in considerable spatial and temporal variation in water clarity in Charlotte Harbor, with subsequent impacts on biomass and productivity of the Harbor's seagrass meadows (Tomasko and Hall, 1999). The results presented here (Fig. 4d) suggest that there is no underlying trend of loss or gain of seagrass coverage in Charlotte Harbor, despite the above-mentioned spatial and temporal variability in water clarity.

Phenomena such as the 1997–1998 El Nińo event can result in dramatic increases in freshwater inflow into these systems, as has been documented for Tampa Bay (Schmidt and Luther, 2002). Elevated freshwater inflows associated with El Nińo events can cause substantial increases in nutrient loads and chlorophyll *a* concentrations, at least in Tampa Bay (Greening and Janicki, unpublished data). In Upper Charlotte Harbor, increased freshwater inflows are also associated with reduced water clarity, mostly due to increases in dissolved organic matter (Tomasko and Hall, 1999). Decreased water clarity associated with high inflows of stormwater runoff during the 1997–1998 El Nińo event is a likely cause of the decrease in seagrass coverage seen between 1996 and 1999 in Tampa Bay, Sarasota Bay, and Upper Charlotte Harbor.

Associated with the reductions in point-source nitrogen loads into Tampa and Sarasota Bays in recent years, stormwater runoff has become the largest single source of nutrient loads into these two systems (see Fig. 2). With stormwater runoff as the primary source of nutrient loads, year-to-year variation in nitrogen loads will be strongly associated with year-to-year variation in rainfall. A complicating factor for resource management and assessment purposes is the highly variable annual rainfall in Southwest Florida (see Fig. 3). While future rainfall amounts can be forecasted, the 95% prediction limit ranges between 99 and 187 cm year^{-1}. As water clarity is moderately to strongly related to phytoplankton abundance in Southwest Florida's estuaries (with the exception of Upper Charlotte Harbor), and phytoplankton abundance is linked to nitrogen loads, year to year variation in seagrass coverage needs to be examined in context with variation in rainfall amounts.

Rainfall patterns in the watersheds of these four contiguous subtropical estuaries result in different water quality and seagrass coverage responses in each system, depending on both anthropogenic and natural phenomena. The same resource management strategy (i.e., nutrient reduction) therefore should not be expected to increase seagrass coverage similarly in all systems. Resource managers are typically tasked with documenting the water quality and natural resource benefits (if any) of costly pollution reduction strategies. Consequently, the potential exists in Southwest Florida for both inappropriate pessimism (during times of above-average rainfall) and inappropriate optimism (during times of below-average rainfall). Detecting the effects of anthropogenic pollution load reduction strategies, given highly variable rainfall and the response of water clarity to nutrient reduction, requires careful examination of all relevant data sets and an understanding of the factors driving seagrass growth and expansion.

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